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SCHOOL OF WATER SCIENCES



Eng. D. THESIS

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Development of a Novel Medium to Improve the Performance
of Biological Aerated Filters (BAFs)

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degree of Doctor in Engineering

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ABSTRACT

A foamed clay, StarLight C, was selected as a potential BAF medium after qualifying as a filter medium according to the BEWA standard tests, and showing the propensity to collect a biofilm. At loading rates $> 10 \text{ kg COD/ m}^3 \text{ working volume/ d}$, two pilot-scale downflow BAFs running in parallel and containing either StarLight C or medium A (a standard expanded clay) achieved a mean of 87% suspended solids removal and 75% soluble COD removal. Both reactors were backwashed daily for ten minutes at a liquid flow rate corresponding to 40% of the minimum fluidisation velocity (V_{mf}) of the media. The approximate experimental V_{mf} of StarLight C was 120 m/h compared to 215 m/h for medium A. Therefore, the BAF using StarLight C exhibited between 11 and 19 % increased productivity at liquid velocities from 1.4 to 2.4 m/h. Lowering the V_{mf} of a BAF medium should also reduce the operating and capital costs of the process. According to representatives of the wastewater treatment companies, high operating costs are the major disadvantage of the BAF process.

Increasing the feed bentonite concentration from 35 g/l to 56 g/l produced a StarLight C variant with improved crush resistance and a lower attrition rate. Meanwhile, a larger extruder nozzle was used to increase the particle size of the medium. Pilot-scale trials showed that the maximum run times were extended by between 8 and 31 hours when using the larger variant (2.5-4.5 mm diameter) compared to the smaller (1.5-3.5 mm diameter). Observing logarithmic head loss development and the cessation of suspended solids removal in the lower region of the reactor, suggested that the smaller StarLight C was an inefficient filter medium.

Using an environmental scanning electron microscope, hydrated biofilms were observed growing on StarLight C's surface. The highly variable biofilm distribution and structure is not accounted for by current biofilm and mechanistic process models. A first-order empirical model described the removal of soluble COD in the top section of the reactors containing medium A, small StarLight C and large StarLight C. The first-order constants were dependant on the liquid velocity and at 1.4 m/h were 61 d^{-1} , 45 d^{-1} and 33 d^{-1} respectively.

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NOTATION

AS	Activated sludge
ADP	Adenosine diphosphate
ANA	Anaerobic processes
ATP	Adenosine triphosphate
BAF	Biological aerated filter
BEWA	British effluent and water association
BOD	Biological oxygen demand
BSE	Back scattered electron
BSED	Back scattered electron detector
CFSTR	Continuous flow stirred tank reactor
COD	Chemical oxygen demand
CSLM	Confocal scanning laser microscopy
CSO	Combined sewer overflow
DNA	Deoxyribonucleic acid
EC	European commission
ECCI	English China Clays International
EM	Electron microscope
EPS	Extracellular polymeric substances
ESEM	Environmental scanning electron microscope
ETS	Electron transport system
EXT	Extensive processes
GSED	Gaseous secondary electron detector
HLR	Hydraulic loading rate
HMIP	Her majesty's inspectorate of pollution
HNDA	High natural dispersion area
HRT	Hydraulic residence time
ICP	Inductively-coupled plasma mass spectrometer
LPS	Lipopolysaccharides
MBBR	Moving bed bioreactor
MBR	Membrane bioreactor
NRA	National rivers authority
OMP	Outer membrane proteins
pe	Population equivalent
pH	Potential hydrogen
PVC	Polyvinyl chloride
RBC	Rotating biological contactor
RQO	River quality objectives
RNA	Ribonucleic acid
sBOD	Soluble biological oxygen demand
sCOD	Soluble chemical oxygen demand
SBR	Sequenced batch reactor
SE	Secondary electron
SEM	Scanning electron microscope
SS	Suspended solids
tBOD	Total biological oxygen demand
tCOD	Total chemical oxygen demand
TF	Trickling filter
tHRT	Theoretical hydraulic residence time

TOC	Total organic carbon
UASB	Upflow anaerobic sludge bed reactor
UV	Ultra violet
UWWTD	Urban wastewater treatment directive
WWT	Wastewater treatment
XRD	X-ray diffraction
XRF	X-ray frequency

NOMENCLATURE

A	Cross sectional area (m^2)
C	Substrate concentration (mg/l)
C_0	Influent substrate concentration (mg/l)
d_m	Media diameter (m)
E_c	Crushing modulus (Pa)
ΔF^{adh}	Change in free energy of adhesion (kJ/ m^2)
g	Gravitational acceleration (m/s^2)
Ga	Galileo number
ΔG	Change in Gibbs free energy (kJ/ mole)
ΔH	Change in systems heat (kJ/mole)
h	Separation distance of two particles (nm)
k	First order rate constant (d^{-1})
k'	Retardant order rate constant (d^{-1})
k''	Kozeny constant
k^*	Mann & Stephenson's rate constant ($\text{kgm}^{-3}\text{d}^{-1}$)
K	Eckenfelder's rate constant (d^{-1})
L	Depth (m)
m	Gradient of $\ln (C/C_0)$ versus L
n	Constant for Eckenfelder's equation
N	Number of tanks in series
N/N_0	Fraction of particles removed from suspension
p	Constant for Eckenfelder's equation
P	Head loss (Pa)
q	Constant for Mann & Stephenson's equation
Q	Volumetric flow rate (l/min)
S	Entropy (kJ)
ΔS	Change in systems entropy (kJ/mole)
SSA	Specific surface area (m^2/m^3)
t	Time (minutes)
T	Temperature ($^{\circ}\text{C}$)
v	Velocity (m/s)
V_a	Van der Waals force between two particles (kJ)
V_{mf}	Minimum fluidisation velocity (m/s)
V_r	Electrostatic energy between two particles (kJ)
V_t	Total interactive energy of two particles (kJ)
α	Required significance level
α_c	Collision efficiency factor
ε	Clean bed porosity
ε_l	Strain
γ_{BS}	Bacterium-substratum interfacial tension (kJ/ m^2)
γ_{BL}	Bacterium-liquid interfacial tension (kJ/ m^2)
γ_{SL}	Substratum-liquid interfacial tension (kJ/ m^2)
η_0	Single collector efficiency for sedimentation and interception
μ	Fluid absolute viscosity ($\text{kg m}^{-1} \text{s}^{-1}$)
ρ_f	Fluid density (kg/m^3)
ρ_m	Medium density (kg/m^3)
σ_c	Compressive stress (Pa)

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CHAPTER 1

INTRODUCTION AND SYNOPSIS

In 1991, the European Community introduced the Urban Wastewater Treatment Directive (UWWTD), which set minimum standards for the treatment of wastewater that had to be reached between 1998 and 2005. The UWWTD made secondary treatment of all municipal wastewater mandatory, unless the receiving water was designated as less sensitive (High Natural Dispersion Areas- HNDAs). Meanwhile, for receiving waters designated as particularly sensitive, the wastewater required additional tertiary treatment to remove nutrients. As a result of the UWWTD and the introduction of stricter consent limits to other European Directives, untreated coastal discharges have required new installations of wastewater treatment plants, and many existing wastewater treatment plants have needed uprating and upgrading.

Trickling filters and activated sludge reactors are the traditional biological treatment processes used for secondary and tertiary sewage treatment. Trickling filters have a large footprint and so activated sludge reactors have often been the process choice for urban sites. However, half of the activated sludge plants in England, France and Germany experience problems involving sludge bulking and subsequent discharge of solids (Rogalla & Sibony, 1992). Additionally, many large sewage treatment facilities are located in densely urbanised areas where land is restricted. This has led to a growing interest in high rate, compact wastewater treatment systems (Ryhiner *et al.*, 1992).

The biological aerated filter (or BAF) is one of the most compact biological wastewater treatment processes. BAFs have achieved 90% BOD removal at loading rates up to 2.8 kg BOD/m³/d. For a comparable performance, the maximum loading rate for a trickling filter or an activated sludge reactor would be seven and four times lower respectively (Stephenson *et al.*, 1993). BAFs can be designed not only to offer

either complete secondary or tertiary treatment, but also to provide combined single unit treatment. More recently the reactors have also proved to be efficient for complete nutrient removal (Goncalves *et al.*, 1994).

BAFs are fixed film reactors whose high rate performance is derived from their ability to maintain four to five times the concentration of biomass found in suspended growth reactors (Smith *et al.*, 1990). They are three phase reactors consisting of submerged beds of media with an aeration source. BAFs have many reactor configurations but the major differentiation is between reactors with high voidage structured packing, and those using granular media. Reactors using granular media remove suspended solids by depth filtration and so negate the need for a separate clarification stage. However, like depth filters these BAFs require regular backwashing to avoid clogging.

In an economic analysis for plants sized for secondary treatment of population equivalents between 10,000 and 100,000, the capital costs for a Biocarbene BAF system were shown to be lower than those for a trickling filter, and higher than those for an activated sludge process. The operating costs for the Biocarbene system were comparable to those for activated sludge and higher than those for trickling filters. Thus, the net present value of the Biocarbene process was uncompetitive for all plant sizes. The relative capital costs were improved when the costs for complete treatment were considered but the high operating costs remain the biggest disadvantage of BAFs (Dillon & Thomas, 1989). To lower the operating costs of the BAF process, the aeration and backwash procedures should be optimised.

The single most important influence on the performance of a BAF reactor is its medium. The surface of the medium acts as a support for the microbial biofilm and so should promote bacterial attachment and propagation. Additionally, the material must have all the physical attributes required to act as an efficient filter medium. The overall objective of this investigation was to use foamed clay technology to produce a BAF medium that improved the properties of the BAF process.

The foamed clays had not been examined previously, and so had to be tested to ensure that they qualified as potential BAF media (Chapter 4). Their physical properties were compared to the BEWA standards (1993) for filter media, whilst their biofilm retention properties were assessed in small-scale reactors. StarLight K incurred a 25% weight loss after attrition testing which is not acceptable for a filter medium. Therefore, it was concluded that only StarLight C, which reached all of the requirements of filter media, had the potential to be used in BAFs.

The performances of StarLight C and a frequently used expanded clay medium (medium A) were compared in pilot scale BAFs treating settled sewage (Chapter 5). Both BAFs achieved a mean of 87 % suspended solids removal, and 75% sCOD removal after four days and with loads over 10 kg COD/m³/d (based on working volumes). Nitrification stabilised after 25 days and peaked at 99% removal. The BAFs were backwashed every 24 hours at a liquid flow rate corresponding to 40% of the minimum fluidisation velocity (V_{mf}) of the media. Due to their different V_{mf} s, the StarLight C BAF required approximately half the backwash water needed by the BAF containing medium A.

Since BAF technology is relatively novel, the standards for filter media have been used to select BAF media, without an assessment of their suitability (Kent *et al.*, 1996). Comparing the results of the standard filter media tests to the performance of BAF media at pilot-scale, indicated that some tests required modification (Chapter 6). For instance, it may be unnecessarily stringent to set the maximum media weight loss after attrition testing at just 3%. Medium A exhibited 5% weight loss after attrition testing, but there were few signs of media attrition during nine months pilot scale operation, possibly due to biofilm protection. The experimental V_{mf} needed to be measured to set backwashing velocities since the assumptions made in the theoretical calculations of V_{mf} did not hold true for the BAF media (Coulson & Richardson, 1978). It was also suggested that new tests needed to be developed to assess potential BAF media. The surface roughness of BAF media appeared to have a significant effect on reactor performance, and measuring a medium's internal porosity allowed its sunken density to be estimated.

The results of tracer studies on empty filter beds were shown to be good approximations to those performed with operating beds (Chapter 6). Thus, potential problems such as channelling and excess liquid hold up could be highlighted prior to operation. For three media beds, which were equivalent to 3, 4 and 8 tanks in series, the nitrification results were superior for the reactors in which the liquid flow was closest to plug flow.

The lower backwash water requirements of StarLight C could reduce the operational and capital costs of the process as well as improving its efficiency. The desired improvement to the BAF process, most frequently requested by representatives of the wastewater treatment companies was a reduction in their operating costs. Other suggested improvements included lowering the attrition of the media and increasing its biological performance (Chapter 7).

Altering the inputs to the clay foaming process allowed a number of variants to be produced with different levels of surface roughness and crush resistance (Chapter 8). Results from small-scale reactors indicated that, contrary to previous work (Baker, 1984), altering the surface roughness of the foamed clay did not effect its biofilm retention properties. It was therefore proposed that surface roughness influences biofilm formation by providing sites for glycocalyx attachment. The rough surface of foamed clay may have an excess number of attachment sites and so, altering the degree of roughness would not effect its biological properties. It was suggested that the variant produced using 56 g/l, rather than 35 g/l, bentonite solution was the most suitable BAF medium. This variant had a lower surface roughness and a higher crush resistance than the other foamed clay variants, which translated into a reduction in weight loss after attrition testing from 3% to 1%.

Headloss development data and suspended solids removal profiles indicated that the BAF using the original StarLight C, with particle diameters ranging from 1.5 to 3.5 mm, performed inefficiently as a depth filter (Ives, 1969). Although the BAF containing larger StarLight C (particle diameters 2.5 to 4.5 mm) acted as an efficient depth filter, its effluent deteriorated for 30 minutes after backwashing at a flow rate of 0.4 l/min. It was concluded that each medium should be combined with a specific maximum reactor height and flow rate and used in different applications (Chapter 10).

Biofilm development on a foamed clay BAF medium was monitored using an environmental scanning electron microscope (ESEM) with a sample chamber saturated with water vapour and maintained at 4 °C and 4 Torr (Chapter 9). Within 24 hours colonies between 5 and 10 µm in diameter were visible. Subsequently, colonies up to 100 µm in diameter were found and after six days the colonies had started to converge. The biofilm images looked similar to those from previous ESEM studies, with cells invisible beneath a canopy of moist ECP (Sutton *et al.*, 1994, Little *et al.*, 1991). The observed growth stages indicated that the biofilm does not form a uniform monolayer across the BAF medium. Concurring with the observations of Stewart *et al.* (1993), thick biofilm was observed next to bare patches of the media surface. Conventional models describe biofilms as uniform layers across the media surfaces, into which the substrate penetrates only by diffusion (Bishop & Rittmann, 1995). Therefore, the ESEM micrographs from this study supply further evidence that conventional models do not describe the structure of all biofilms correctly.

Three empirical BAF models were applied to the sCOD removal profiles for pilot scale BAFs using medium A, small StarLight C and large StarLight C (Chapter 11). The model proposed by Mann & Stephenson (1997), and forms of both a simple first order (Schulze, 1960) and a retardant (Eckenfelder, 1961) model were examined. Removal appeared to follow first-order kinetics through the first part of the reactor. The mean overall first-order constants were 72 for the expanded clay, 55 for the small StarLight C and 38 for the large StarLight C. However, there was less variation in the first-order constant when the results for each flow rate were considered separately. There was no consistent trend in the sCOD concentrations in the bottom part of the reactors. Sometimes substrate removal continued at a reduced rate, and on other occasions the substrate concentration was stable or even increased.

Overall, the requirements of an ideal BAF medium have been identified and an assessment has been made of the methods used to determine the properties of potential new materials. A novel foamed clay has been developed which exhibits some of these ideal properties and two size ranges of this material have proven effective as BAF media at pilot scale.

CHAPTER 2

LITERATURE REVIEW

PART A: BIOLOGICAL AERATED FILTERS (BAFs)

2.1 INTRODUCTION

Over the last hundred years a number of processes have been designed to promote the microbial degradation of the organic materials in wastewaters. The currently available biological wastewater treatment (WWT) reactors can be classified as harnessing either suspended or attached micro-organisms. The activated sludge (AS) process uses suspended growth, whereas most of the micro-organisms in trickling filters (TFs), rotating biological contactors (RBCs), fluidised beds and BAFs, are attached to surfaces of solid supports as biofilms.

2.2 ADVANTAGES OF FIXED FILM SYSTEMS

The main advantage of fixed film systems compared to suspended growth systems is that higher biomass concentrations can be maintained at short hydraulic retention times (HRTs) without sludge recycling. Fixed film systems are also thought to be less vulnerable to shocks or transients than suspended growth systems (Characklis & Marshall, 1990). This is because an inhibitor will only be retained in the reactor for a relatively short time and the inner part of the biofilm is protected by the outer layers (Heitkamp *et al.*, 1993). Additionally, Cao *et al.* (1993) found that the substrate removal properties of biofilm reactors were less temperature dependant than those of suspended biomass reactors.

2.3 ADVANTAGES OF BAFS

Biological aerated filters (BAFs) are three phase reactors consisting of a submerged solid medium with an aeration source. They have been developed for secondary and tertiary WWT in the last twenty years and although they are not in widespread use in the UK, they have a number of advantages.

2.3.1 Small Footprint

BAFs are high rate aerobic reactors, which can be used as a small footprint alternative to conventional WWT processes such as AS and TFs (Stephenson, 1997). The medium in a BAF provides a large surface area for microbial attachment and also retains micro-organisms in the voids between the grains. This allows BAFs to maintain a biomass concentration about four to five times that found in suspended growth reactors (Smith *et al.*, 1990). True BAFs contain granular rather than structured media and so can incorporate depth filtration. This negates the need for a post clarification stage, which reduces the land requirements of the process further. Full-scale reactors throughout the world illustrate that BAF systems can reduce plant footprint by as much as 80% (Stephenson *et al.*, 1993).

2.3.2 Single Unit Treatment

Single BAF units can remove dissolved and particulate organic material, suspended solids, and ammonia. Denitrification and phosphate removal can also be achieved by alternating aerobic and anaerobic conditions (Clarke, 1996, Sagberg *et al.*, 1992). High phosphate uptake is possible since optimal conditions for bacteria can be maintained independently of the HRT (Goncalve & Rogalla, 1992). BAF operation can be automated and their modular design gives treatment flexibility (Smith & Brignal, 1996).

2.3.3 Aesthetic Benefits

Where necessary, BAF plants can be totally covered or even buried (Pujol *et al.*, 1992). This is particularly pertinent in the current environment where WWT companies are under increasing pressure to reduce their plant's aesthetic impact.

2.4 DISADVANTAGES OF BAFS

BAFs can not operate continuously because as the filter blinds, there is short circuiting through the reactor, which decreases the effluent consistency. BAFs therefore have to be backwashed regularly with treated effluent to remove trapped particles and excess biomass. Backwashing can occur at set times, typically once every 24 hours (Bacquet *et al.*, 1991). Alternatively, the filters are backwashed when head loss reaches a predetermined level (Amar *et al.*, 1986, Lilly *et al.*, 1991), or when the effluent quality deteriorates (Gray, 1993).

Many of the problems associated with using BAFs are related to the backwashing procedure. Backwashing uses energy and treated effluent and so increases the operating costs of the process whilst decreasing its efficiency. Since privatisation, the WWT companies have been under pressure to improve treatment standards whilst decreasing costs. Therefore, systems with the lowest energy, maintenance and labour costs are chosen when possible (Klemetson & Rogers, 1983). Backwashing may also cause media break up or washout (Kent *et al.*, 1996) and the backwashing pumps increase the capital costs of the plant.

2.5 CONFIGURATIONS

BAFs are also known as submerged aerated filters, packed bed reactors and immersed aero-filters (Faup *et al.*, 1982, Rusten, 1984). The term “BAF” has been applied to reactors containing structured packing and to those containing a granular medium.

2.5.1 Structured Media

The reactors containing structured media have a high voidage and are completely mixed (Rusten, 1984). This can be advantageous because toxic shocks will be rapidly dispersed and so have a diminished affect on performance. Structured media reactors also have the advantages of negligible pressure drop through the packing and reduced backwash frequency requirements (Ryhiner *et al.*, 1992). However, completely mixed reactors are less favourable for nitrification and there is minimal solids removal. Therefore, a post clarification stage is normally required, increasing the footprint of the reactors.

2.5.2 True BAFs

True BAFs contain granular media, which is either sunken or floating. The wide range of materials currently used as media in BAFs are listed in Table 2.1. Pulverised pumice stone (Fdz-Polanco *et al.*, 1994a), porous stone (Costa Reis & Sant’Anna, 1985) and slag (Dee *et al.*, 1994) have also been utilised as BAF media.

Tracer studies have shown that the hydraulic regime of the reactors approximates to plug flow (Mann *et al.*, 1995). In plug flow reactors the strength of the wastewater decreases through the reactor. Therefore, there are a variety of niches present allowing different species of bacteria to co-exist. In a BAF, the nitrifying bacteria colonise the regions with low organic concentrations yet high oxygen levels (Mann *et al.*, 1998).

Table 2.1: Details of the major proprietary BAFs

Process	Manufacturer	Flow direction	Media	Density
Biobead	Brightwater	Up	Polyethylene	Floating
Biostyr	OTV Birwelco	Up	Polystyrene	Floating
Biofor	Degremont	Up	Biolite	Sunken
COLOX	Capital Controls (Tetra)	Up/Down	Sand	Sunken
SAFe	Thames Water (PWT)	Down	Expanded shale	Sunken
Biocarbone	OTV Birwelco	Down	Expanded shale	Sunken

2.5.3 Upflow Versus Downflow

BAFs can operate in either an upflow co-current or a downflow counter current configuration. Ideal solids filtration is thought to occur in the direction of media compression (Mann *et al.*, 1998, Pujol *et al.*, 1992), so upflow reactors tend to be combined with floating media and downflow with sunken media (Figure 2.1). Downflow BAFs are always backwashed with an upflow liquid stream whereas upflow BAFs are backwashed with either an upflow or downflow liquid stream. A summary of the configurations currently available in commercial systems is given in Table 2.1.

There are many advantages and disadvantages associated with both the upflow and downflow configurations. These can be categorised as aesthetic, operating and performance related and are listed in Table 2.2.

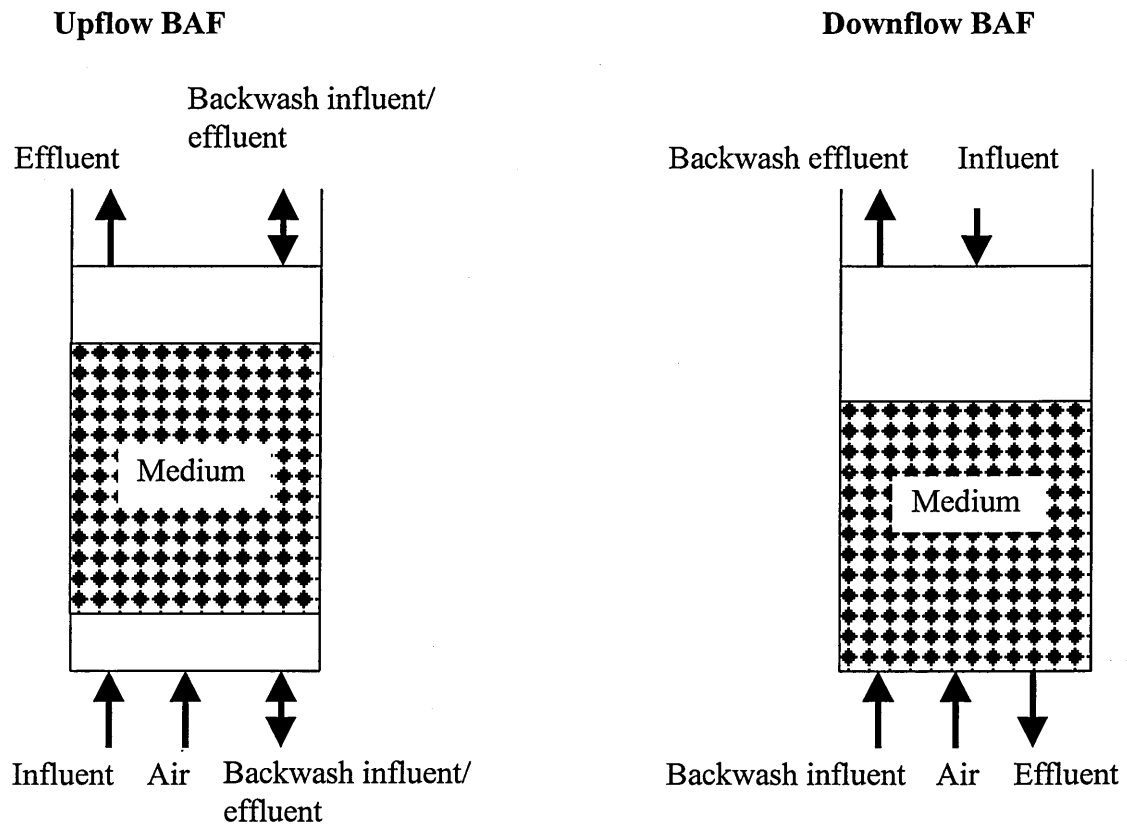


Figure 2.1: Typical BAF configurations

Table 2.2: Advantages of upflow and downflow configurations

Advantage	Upflow	Downflow
Aesthetic	No odours	
	Smallest footprint	
Operational	Lowest backwash water volumes	Easiest maintenance
	Lowest air blowers pressure	Most efficient backwashing
	Longest filter cycles	Highest oxygen transfer
		Lowest feed pump pressure
Performance	Least channelling	Best suspended solids removal
		Most consistent performance
		Better nitrification
		Adaptable for nutrient removal

2.5.3.1 Differences in the aesthetic properties of upflow and downflow BAFs

In terms of aesthetics, when using upflow BAFs the ambient air is only in contact with the treated effluent and so odour problems are less likely (Meaney, 1994). Additionally, upflow BAFs can have smaller footprints since the backwash water can be stored above the reactor (Stephenson *et al.*, 1993). However, there are rarely difficulties with the aesthetics of BAFs since inherently their footprint is small and they are often totally enclosed or covered (Smith & Edwards, 1994).

2.5.3.2 Differences in the operational properties of upflow and downflow BAFs

Despite process automation, the aeration and backwashing procedures in BAFs need regular monitoring and maintenance (Pujol *et al.*, 1992). In this area, downflow BAFs have benefits over upflow BAFs since the media and the influent channels are accessible and the air diffusers are protected from solids (Newbigging *et al.*, 1995). The backwashing procedure in downflow BAFs is thought to be very efficient as the media at the top of the bed experiences the greatest disturbance, and this is the region entrapping the greatest concentration of solids (Newbigging *et al.*, 1995). In downflow BAFs, the feed pumps operate at lower pressures and less air is required due to the high oxygen transfer rate achieved by counter current aeration (Rogalla *et al.*, 1990). However, in upflow BAFs, the air blowers operate at lower pressures and it is claimed that reduced volumes of water are required for backwashing (Visvanathan & Nhien, 1995).

2.5.3.3 Differences in the performance of upflow and downflow BAFs

In terms of performance, downflow filters are thought to show better suspended solids removal. However, removal takes place in the top part of the filter instead of throughout the bed as in upflow filters. As a result downflow BAFs tend to develop head loss prematurely and so require more frequent backwashing than upflow BAFs (Jepsen & Jansen, 1993).

Upflow BAFs can show decreased effluent consistency due to elevated solids concentrations prior to and straight after backwashing (Newbigging *et al.*, 1995). However, the upflow configuration is thought to make better use of the full reactor volume since there is less air pocket development and subsequently less preferential channelling (Faup *et al.*, 1982).

Nitrifying bacteria are located in the parts of the filter bed where the concentrations of organic matter are lowest. In upflow BAFs this is at the top of the column where they are vulnerable to being washed out if an upflow backwash is used (Mendoza-Espinosa & Stephenson, 1997). In downflow BAFs, the nitrifiers are in ideal conditions at the bottom of the filter, where oxygen levels are highest (Akunna *et al.*, 1994). Additionally, there is less intermixing of media in downflow BAFs, as after backwashing the media tends to re-grade with the particles holding the less dense heterotrophic biomass moving to the top of the reactor. Therefore, downflow BAFs offer the most favourable conditions to maintain heterotrophic and autotrophic bacterial populations within the same reactor and are ideal for single step BOD (biological oxygen demand) and ammonia removal (Mann, 1997).

Chemical dosing into a mixing zone at the top of a downflow BAF, facilitates the removal of phosphates. In an upflow BAF, a distribution device that may be sensitive to clogging would be required. However, a denitrification zone can be produced at the base of a single unit upflow BAF by introducing air part way up the filter bed (Rogalla *et al.*, 1992).

2.6 APPLICATIONS

Since their introduction in the late 1970s, BAFs have been successfully employed in many applications (Table 2.3). They are most frequently used to treat domestic wastewaters, either at the secondary or tertiary stage. Additionally, an unaerated, multilayered zone has been introduced at the bottom of an upflow BAF, allowing prefiltration, and so, complete treatment without primary settling (Desbos *et al.*, 1990). The small footprint of the BAF process promotes their use in upgrading established sewage works (Sagberg *et al.*, 1992, Budge & Gorrie, 1996). Meanwhile, their compact design makes them ideal for environmentally sensitive sites, such as coastal areas (Gilles, 1990).

The high effluent qualities achievable using BAFs has allowed their application in the recycling and reuse of both domestic and industrial wastewater (Adachi & Fuchu, 1991). The biomass retention time is longer in a fixed film reactor than a suspended growth reactor. It has therefore been proposed that specific bacteria, with a high capacity for oxidising recalcitrant pollutants, can be selected and maintained in a BAF (Hamoda, 1993). Thus, one of their industrial applications has been to treat wastewaters contaminated by petroleum related substances (Drushlyak & Kirichenko, 1990). Another major application is the treatment of food industry wastewater. This waste has a high but variable concentration of organic matter and an extremely variable volumetric flow rate, which often stops altogether over weekends. Suspended growth reactors do not function well under these conditions and TFs need effluent recycling to keep moist. Meanwhile, BAFs offer effective and economical treatment (Rusten & Thorvaldsen, 1983).

Table 2.3: Applications of the BAF process

APPLICATION	REFERENCE
Full treatment of primary sewage	Carrio <i>et al.</i> , 1995, Desbos <i>et al.</i> , 1990
Treatment of secondary sewage	Amar <i>et al.</i> , 1986, Bacquet <i>et al.</i> , 1991, Canler & Perret, 1994, Janning <i>et al.</i> , 1995, Koutsakos <i>et al.</i> , 1992, Newbigging <i>et al.</i> , 1995, Pujol <i>et al.</i> , 1992, Robinson <i>et al.</i> , 1994, Rogalla <i>et al.</i> , 1994, Rusten, 1984, Sakuma <i>et al.</i> , 1993, Smith & Marsh, 1995, Smith & Hardy, 1992, Stensel <i>et al.</i> , 1988, Wheale <i>et al.</i> , 1992, Young <i>et al.</i> , 1975
Treatment of tertiary sewage	Beg & Hassan, 1987, Carrand <i>et al.</i> , 1990, Chen & Cheng, 1994, Cromphout, 1992, Dillon & Thomas, 1989, Faup <i>et al.</i> , 1982, Fruhen <i>et al.</i> , 1994, Kurbiel, 1987, McHarness <i>et al.</i> , 1975, Paffoni <i>et al.</i> , 1990, Peladan <i>et al.</i> , 1996, Schlegel, 1988, Terayama <i>et al.</i> , 1997, Vedry <i>et al.</i> , 1994, Wheale & Cooper-Smith, 1995,
Compensating for anaerobic processes	Goncalves <i>et al.</i> , 1998, Heijnen <i>et al.</i> , 1991
Phosphorus removal	Goncalves & Rogalla, 1992, Goncalves <i>et al.</i> , 1994, Pujol <i>et al.</i> , 1994, Rogalla <i>et al.</i> , 1991, Sagberg <i>et al.</i> , 1992, Sammut <i>et al.</i> , 1992
Denitrification	Æsøy <i>et al.</i> , 1998, Andersen <i>et al.</i> , 1995, Borregaard, 1997, Cecen & Gonenc, 1995, Chevron <i>et al.</i> , 1997, Chudoba & Pujol, 1998, Chui <i>et al.</i> , 1996, Eichinger <i>et al.</i> , 1993, Iida & Teranishi, 1984, Jepsen & Jansen, 1993, Kleiber <i>et al.</i> , 1994, Meaney & Strickland, 1994, Rogalla & Sibony, 1992, Rogalla & Bourbigot, 1990, Rogalla <i>et al.</i> , 1991, Ryhiner <i>et al.</i> , 1994, Sagberg <i>et al.</i> , 1992, Sanz <i>et al.</i> , 1996
Urban storm water run off	Anderson <i>et al.</i> , 1997
Domestic water recycling	Adachi & Fuchu, 1991, Kuribayashi, 1992
Dairy waste	Odegaard & Rusten, 1990, Ravindran <i>et al.</i> , 1997
Distillery waste	Gerards & Vriens, 1997
Food industry waste	Rusten & Thorvaldsen, 1983, Rusten & Odegaard, 1986
Landfill leachates	Welander <i>et al.</i> , 1997
Paper mill effluents	Kantardjiev & Jones, 1997, Rovel <i>et al.</i> , 1994
Petroleum products/ phenols	Drushlyak & Kirichenko, 1990, Hamoda <i>et al.</i> , 1987, Hamoda & Al-Haddad, 1987
Removing malodorants	Hwang <i>et al.</i> , 1994
Industrial water recycling	Adachi & Fuchu, 1991

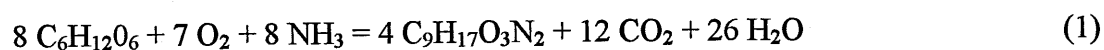
2.7 PERFORMANCE

2.7.1 Removal of BOD

Organic matter in a wastewater represents an oxygen demand, which could reduce the dissolved oxygen (DO) concentration in a receiving water body. The components of total biological oxygen demand (tBOD) in a wastewater are particulates and soluble BOD (sBOD). The main removal mechanism for particulate organic matter is filtration, although some particles may be hydrolysed and then oxidised by bacteria (Sarner, 1986). Meanwhile, the removal of soluble BOD is entirely dependent upon the metabolism of micro-organisms. Organic matter is also measured as chemical oxygen demand (COD).

2.7.1.1 Microbiology of BOD removal

The majority of the micro-organisms in secondary treatment BAFs are heterotrophic, obtaining the bulk of their carbon from organic compounds. They are also chemoorganotrophs and obtain energy from metabolising organic compounds. In BAFs, conditions are generally aerobic so the micro-organisms respire, oxidising sugars to carbon dioxide and water. A simplified expression of biomass production is given by Equation 1.



The bacteria likely to be present in BAFs used for secondary WWT include the genera *Achromobacter*, *Acinetobacter*, *Arthrobacter*, *Bacillus*, *Enterobacteria*, *Aeromonas*, *Flavobacterium*, *Alcaligenes*, *Pseudomonas*, *Microthrix*, *Moraxella* and *Zoogloea* (Zinebi *et al.*, 1994, Pike, 1975). The predominant bacterium changes during biomass acclimation to different substrates and temperatures (Fujie *et al.*, 1994).

Higher organisms, including protozoa, rotifers, nematodes and annelids, may also inhabit BAFs. These organisms influence the reactor performance by feeding on bacteria and particulate organic matter (de Beer & Muyzer, 1995).

2.7.1.2 BOD removal performance

Typical start-up times for AS plants are in the range of months (Carrio *et al.*, 1995). Meanwhile, the start-up period for full-scale secondary treatment BAFs can be as short as 10 days (Pujol *et al.*, 1992). In a pilot biofilter, start-up has been achieved within five days (Bacquet *et al.*, 1991). This is a particularly pertinent advantage for some applications, such as coastal plants, which only need to operate for a few months each year (Pujol *et al.*, 1992).

BAFs need smaller reactor volumes to treat a given mass of substrate, as illustrated by comparing the maximum loading rate to achieve 90% BOD removal for different processes. TFs can handle loads up to 0.4 kg BOD/m³/d; AS reactors can cope with up to 0.6 kg BOD/m³/d; meanwhile BAFs can remove up to 2.8 kg BOD/m³/d (Stephenson *et al.*, 1993).

One of the most frequently used proprietary BAF systems is the Biocarbone process. This has been extensively tested by several major research institutions, for secondary treatment with effluent quality objectives of 15 mg/l BOD and 30 mg/l suspended solids. The maximum loads removed were consistently reported as 4 kg BOD /m³/d or 9 kg COD/m³/d (Dillon & Thomas, 1989, Stensel *et al.*, 1988, Rogalla & Sibony, 1992). Meanwhile, after ten years experience with the Biofor process, the operating limits for carbon removal were set at 10 kg COD/m³/d (Pujol *et al.*, 1994).

Both the Biocarbone and the Biofor systems use clay media. When a plastic medium was used in an upflow reactor, the maximum load removed was 1.2 kg COD/m³/d (Mann *et al.*, 1999). Meanwhile, when using sand as a BAF medium, a loading rate under 6 kg COD/m³/d is recommended (McCoy, 1997). The clay based media have a greater surface roughness which may aid biofilm retention (Baker, 1984). However, plastic and sand media show very low attrition rates (Kent *et al.*, 1996).

The maximum loads treated by BAFs should be compared with caution since the value depends on the volume measurement. If the volumetric loading rates are compared to the influent concentrations and flow rates it appears that Dillon & Thomas (1989) used the active reactor volume for their loading calculations. Other research groups (Mendoza-Espinosa and Stephenson, 1997) have also quoted loading

rates using the working volume of the reactor. Meanwhile, Stensel *et al.* (1988) reported loading rates using the volume of medium. The lack of consistency and clarity in the literature data makes it difficult to compare the performance of BAFs in terms of load removal.

2.7.1.3 Efficiency

BAFs and conventional AS reactors have a similar ratio of average specific power consumption to load removed (Pujol *et al.*, 1992, Stensel & Reiber, 1983). The value varies between 0.7 and 1.5 kWh/kg COD removed (Rogalla & Sibony, 1992, Pujol *et al.*, 1992, Canler & Perret, 1994).

Power costs relate to the inlet pumping, process air supply, backwash water, air scour supply and plant automation and control (Smith & Edwards, 1994). The energy required for process air can represent more than 80% of the energy demand in a BAF system (McCoy, 1997, Pujol *et al.*, 1992). The volume of air required for treatment depends on the pollution load, the endogenous respiration rate and the oxygen transfer efficiency (Robinson *et al.*, 1994). If too little air is supplied, anoxic zones may be created and there will be insufficient substrate removal. Meanwhile, if too much air is supplied, scouring of the biofilm may reduce the solids removal efficiency (Pearce, 1996). Matching the air supply to the influent flow rate can reduce aeration costs by 50% (Smith & Edwards, 1994).

Using different types of aerator does not affect the oxygen transfer efficiency (OTE) in a BAF. This is because the sizes of the air bubbles are determined by the voidage gaps in the packed bed (Harris *et al.* 1996). The media in a BAF increases the detention time of air bubbles passing through the reactor, which increases the OTE. Additionally, there is direct contact between the bubbles and the biofilm, which allows interfacial oxygen transfer, so that BAFs are not limited by bulk liquid oxygen concentrations (Stensel *et al.*, 1984). As a result, the reported oxygen utilisation efficiencies in a BAF have been up to 9.2%, at least double what would be expected in an AS reactor of the same height (Stensel *et al.*, 1988).

The backwash frequency for a secondary treatment BAF depends on the influent concentration, the temperature, the medium and the hydraulic loading rate (Dillon &

Thomas, 1989). Ideally the cycle length should be maximised to reduce both the energy and clean water consumption. Even though the backwashing procedure in secondary treatment BAFs occurs only every 12-48 h it accounts for up to 20% of the daily energy consumption (Mendoza-Espinosa and Stephenson, 1999). Meanwhile, backwashing utilises between 5 and 35% of the effluent treated during a cycle (Dillon & Thomas, 1989, Smith *et al.*, 1990, Canler & Perret, 1994, Stensel *et al.*, 1988). In comparison, sand and anthracite water filters use 0.9-8.3 % of the treated water from a cycle (Ruffer & Rosenwinkel, 1984).

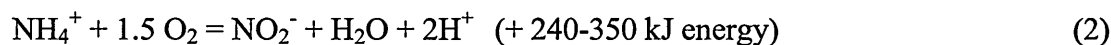
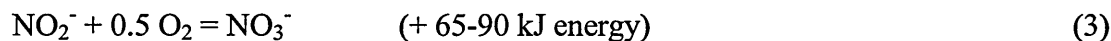
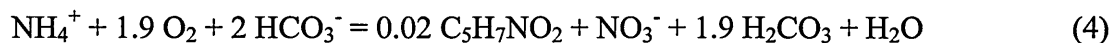
The excess sludge from a BAF has a similar nature to that produced in a high rate AS reactor and settles and thickens well in continuous gravity thickeners (Stephenson *et al.*, 1993). In secondary treatment BAFs, between 0.41 and 1.06 kg of suspended solids are produced for every kg of COD eliminated (Canler & Perret, 1994, Dillon & Thomas, 1989, Amar *et al.*, 1986, Rogalla & Sibony, 1992)

2.7.2 Nitrification

2.7.2.1 Microbiology of nitrification

The bacteria responsible for the nitrification of wastewater are chemolithotrophs and derive energy from the oxidation of inorganic compounds. They are also autotrophs, and so use inorganic carbon sources such as carbon dioxide, bicarbonates or carbonates.

The process is performed by two groups of micro-organisms, the genus *Nitrosomonas* oxidise ammonia to nitrite (Equation 2), whilst the genus *Nitrobacter* oxidise nitrite to nitrate (Equation 3). Overall the nitrifiers use 4.3 g of oxygen and 0.14 equivalent alkalinity per gram ammonia-nitrogen removed (Equation 4). Meanwhile, they produce a small amount of biomass, 0.2g volatile suspended solids per gram of ammonia-nitrogen removed (Henze, 1995).



As the rate limiting step in the nitrification process is the first reaction (Equation 2), the concentration of nitrite is normally low (Kowalski & Lewandowski, 1983, Hao & Chen, 1994).

Nitrifying bacteria have a slower growth rate than most heterotrophic bacteria. Therefore, where the concentration of organic matter in a BAF is high, there is little nitrification. The heterotrophic bacteria multiply to cover the surface of the medium with a thick biofilm, and due to their slow growth rate, nitrifying bacteria will be positioned deep inside the biofilm where there is low oxygen penetration. Thus, for nitrification to occur in a BAF the filtered BOD concentrations should be less than 20 mg/l for 3 mg/l DO (Harremoes, 1982).

The slow growth rate of nitrifiers also makes them particularly vulnerable to predation by higher organisms. When eucaryotic inhibitors were added to fluidised bed reactors the nitrification rate doubled (Lee & Welander, 1994).

Nitrifiers can be inhibited by DO concentrations below 0.2 mg/l, and it has been recommended that bulk DO concentrations should be maintained above 4-5 mg/l in BAFs (Chen & Cheng, 1994). Low concentrations of inorganic carbon can also inhibit nitrification but this is unlikely to occur in BAFs where carbon dioxide is present in excess due to carbonaceous oxidation (Barnes & Bliss, 1983).

The optimal pH range for nitrification is 7.8 to 8.4 and the optimal temperature range between 30 and 36 °C. Nitrifiers are more sensitive to environmental conditions and toxic substances than heterotrophic bacteria (Prosser, 1989).

2.7.2.2 Performance

Nitrification using AS is a well mastered process. However, large aeration tanks are required so that long sludge ages can be maintained (Faup *et al.*, 1982). In BAFs the nitrifying bacteria attach to surfaces and so the biomass retention time is not dependent on the HRT. BAFs have therefore proved themselves particularly efficient in ammonia removal and so have become established as a tertiary treatment process (Rogalla *et al.*, 1990). Ammonia removal rates of up to 1.67 kg NH₃-N/m³/d at 19 °C have been achieved in tertiary nitrifying BAFs (Vedry *et al.*, 1994).

2.7.2.3 Efficiency

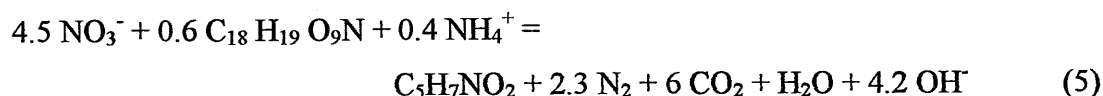
Due to the lower growth rate of autotrophs, tertiary BAFs often have a lower backwash frequency and may only be backwashed weekly (McHarness *et al.*, 1975, Carrand *et al.*, 1990). Correspondingly a lower % of the cleaned effluent is used during backwashing. For instance, Carrand *et al.* (1990) found that wash water was only 3% of the treated water. They also reported that 2.1 kg of suspended solids were produced per kg of ammonia removed, and the energy efficiency was 11 kWh/kg NH₄-N.

2.7.3 Total Nitrogen Removal

Total nitrogen removal can be achieved by converting the nitrates, produced during nitrification, to nitrogen gas by the denitrification process.

2.7.3.1 Microbiology of denitrification

When anoxic* conditions exist, some bacteria can use an alternative terminal electron acceptor such as nitrate or sulphate instead of oxygen. Nitrate is converted into nitrogen gas by denitrification (Equation 5). The process uses 5.5 g COD and produces 0.07 equivalent alkalinity per gram of nitrate-nitrogen removed. The biomass production is 1.8 g volatile suspended solids per gram nitrate-nitrogen removed (Henze, 1995).



Facultative anaerobes capable of nitrate respiration include *Pseudomonas* and *Enterobacteriaceae*. These are heterotrophic bacteria which need a carbon source. Therefore, if the denitrification unit follows secondary treatment an external carbon source is required. Alternatively, by recycling part of the wastewater, the denitrification unit can precede secondary treatment (Dee *et al.*, 1994). Traditionally, post-denitrification systems used deep bed sand filters dosed with methanol, and pre-denitrification systems used AS reactors. However, BAFs can be used for both pre- and post-denitrification (Jansen *et al.*, 1994). In upflow BAFs, the bottom section of the filter can be anoxic allowing pre-denitrification in a single unit (Rogalla *et al.*, 1992).

* Anoxic conditions exist when oxygen is available and utilised in the form of oxy-anions, whereas anaerobic conditions exist when no oxygen is available in any form (Glossary of Terms Used in Water Pollution Control, Institute of Water Pollution & Control, 1975).

2.7.3.2 Performance

By coupling anoxic and aerated filters, total nitrogen removal can be achieved using BAFs. At pilot-scale the volumetric removal rates using fixed film reactors are one order of magnitude higher than achieved with the AS process (Jansen *et al.*, 1994). However, the maximum denitrification rates will decrease in full-scale BAFs for a number of reasons.

Firstly, the degradability of the carbon source is crucial for denitrification. Therefore, in pre-denitrification recycling systems, where incoming wastewater is used as the carbon source, a lower rate will typically be the result of the inferior carbon quality (Jansen *et al.*, 1994).

Additionally, using a fixed film reactor, the required DO concentration in the aerobic unit is higher than in a suspended growth reactor. Therefore, in a pre-denitrification system, the recycled effluent would increase the DO concentration in the anoxic filter and reduce the denitrification rate. However, experiments show that denitrification can take place in a biofilm reactor despite the presence of oxygen in the bulk liquid. This is because nitrate and organic matter can penetrate deeper into a biofilm than oxygen, a phenomenon described as the three component diffusion theory. As a result, denitrification takes place in the anoxic part of the biofilm (Hagedorn-Olsen *et al.*, 1994).

Maximum denitrification efficiency is achieved at an influent COD to NO_x-N ratio of approximately 5, when using pilot-scale structured media reactors and either post-denitrification (Cecen & Gonenc, 1992), or pre-denitrification (Chui *et al.*, 1996). This relationship held true for a full-scale upflow BAF with an anaerobic bottom section and the process achieved a nitrogen removal rate of 1 kg/m³/d (Rogalla *et al.*, 1992).

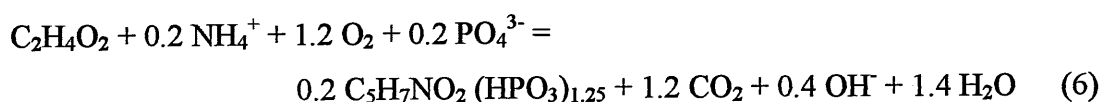
2.7.4 Phosphorus Removal

2.7.4.1 Chemical Phosphorus Removal

BAFs have been combined with pre-precipitation using aluminium salts, iron salts or polyelectrolytes. However, the precipitating agents reduce the carbonaceous load of the settled sewage and subsequently denitrification can be inhibited. This problem is avoided by simultaneously precipitating the phosphorus on the biofilter by chemical addition to the influent (Sagberg *et al.*, 1992, Sammut *et al.*, 1992). Dosing after primary treatment also reduces the reagent requirement and the sludge production. It has been shown at pilot-scale that up to 90% of phosphorus can be removed with chemical dosing without significantly affecting the filter run time or performance (Clark *et al.*, 1997).

2.7.4.2 Biological Phosphorus Removal

In order to avoid the additional cost of reagents, and the increased sludge production associated with physical/chemical phosphorus removal, there has been increasing interest in biological phosphorus removal (Florentz *et al.*, 1986). Certain micro-organisms can accumulate phosphorus beyond their metabolic need for growth, when exposed to varying anaerobic and aerobic conditions, as shown in Equation 6 (Henze, 1995). When the carbon source is acetic acid, 1 g of phosphorus can be removed for each 10 g of acetic acid. The process is neutral to alkalinity and does not alter the pH.



Biological phosphorus removal can be achieved using two BAFs in series. The first column is anoxic and receives settled sewage, whilst the second is aerobic and treats the effluent of the first. At the end of a specified time the aerobic conditions are reversed along with the flow direction through the two reactors. Thus, the biomass is exposed to alternating anaerobic and aerobic conditions and phosphorus can be extracted by backwashing at the end of an aerobic phase. Using upflow pilot-scale BAFs, approximately 60-70% phosphorus removal has been achieved. When the anaerobic cycle time was kept under eight hours, simultaneous nitrification was maintained (Goncalves & Rogalla, 1992, Goncalves *et al.*, 1994).

2.8 MODELLING THE BAF PROCESS

2.8.1 Mechanistic Models

Mechanistic models look at processes on a micro scale, applying kinetics and other reaction engineering principles. To be used accurately mechanistic models require information about the system including the biofilm thickness, the oxygen utilisation rate, and the mass transfer coefficients. These parameters are often estimated since, being difficult to measure, they are usually unavailable. Therefore, mechanistic models assume biofilms are uniform in thickness, density and distribution despite evidence to the contrary (section 2B.2.9). Additionally, the flow through granular beds is characterised as ideal plug flow even though tracer studies suggest mixing does occur (Mann *et al.*, 1995).

Mechanistic models of biofilm reactors combine fundamental biofilm substrate utilisation kinetics with substrate mass transfer. The overall reaction rate can be limited either by the substrate availability or the biomass reaction rate, and this will be discussed further in Chapter 11.

2.8.2 Empirical Models

Empirical models directly relate input variables such as organic loading or substrate concentration, to output variables such as substrate removal rate. These models are easy to interpret and are useful for process design and sizing. There are many empirical models for TFs (Velz, 1948, Schulze, 1960, Eckenfelder, 1961), and Roberts (1973) gives a good account of their development. Recently, an empirical TF model was applied to BAFs. Effluent soluble COD was successfully related to reactor height and influent soluble COD for particular media and hydraulic configurations (Mann & Stephenson, 1997, Mendoza-Espinosa & Stephenson, 1998).

2.9 CONCLUSIONS OF PART A

Due to their unique combination of advantages, BAFs have a great deal of potential as WWT reactors. However, their operational costs, derived from the energy consumed to aerate and backwash the filters, may limit their application to sites with severe land restrictions (Mendoza-Espinosa and Stephenson, 1999).

The aeration and backwash procedures need to be optimised in order to improve the running costs of BAFs. The process will then be more competitive with traditional WWT reactors at all sites.

PART B: BAF MEDIA

2B.1 INTRODUCTION

From their experience gained in BAF operation, Smith & Brignal (1996) identified the two real issues in BAF operation as the hydrodynamics of the packed bed and the characteristics of the attached biomass. The hydrodynamics of the bed govern the head loss across the bed and so effects the flows and wash frequency. They will be effected by the reactor design and the size and shape of the medium. The biomass factors relate to the composition and concentration of the biofilm throughout the bed. These are dependent on the loading rate, the composition of the wastewater, the temperature and the characteristics of the medium. Evidently the medium is one of the most influential parameters on the effectivity of BAFs.

Any novel BAF media must provide a better combination of physical and biological characteristics to increase the efficiency and reduce the costs of the process. The characteristics of BAF media which influence its role as a biofilm support will be reviewed separately from those which influence its role as a filter medium.

2B.2 BAF MEDIA AS A BIOFILM SUPPORT

2B.2.1 Introduction to Biofilms

To understand which media properties enhance BAF performance, the biology of the process needs to be examined. Most of the biomass in a BAF attaches to the medium's surfaces to form a biofilm. Biofilms consist of living and dead micro-organisms embedded in an organic polymer matrix, or glycocalyx, of microbial origin. Biofilms are often variable in time and space and provide heterogeneous niches for microbes, due to gradients existing throughout the biofilm (Characklis & Marshall, 1990). Biofilms are ubiquitous in flowing aqueous environments, existing for instance on river bed rocks, as plaque on teeth, on ships hulls, heat exchange surfaces and in the intestines where they can cause disease.

2B.2.2 Biofilm Formation

Although many researchers describe three main stages in biofilm formation (Tijhuis *et al.*, 1994, Heijnen *et al.* 1992), the process can be best described as the net result of all of the following physical, chemical and biological processes:-

1. Transport and adhesion of soluble compounds
2. Transport of micro-organisms
3. Adhesion of microbes to the surface
4. Desorption of microbes from the surface
5. Biofilm growth
6. Further attachment to the biofilm
7. Detachment from the biofilm

2B.2.3 Transport and Adhesion of Soluble Compounds

Adsorption of an organic monolayer occurs within minutes after exposure of an initially clean surface to a flowing fluid that contains dispersed micro-organisms, nutrients and organic molecules (Trulear & Characklis, 1982). The charges of the components of the monolayer attract ions of the opposite charge, until electrostatic forces and thermal motion restrictions balance. The result is a conditioning film composed of a diffuse electric double layer of ions which contains many of the essential inorganic nutrients needed by bacteria (Annachhatre & Bhamidimarri, 1992).

The composition of the film depends on the contents of the fluid and the potential, charge and critical surface tension of the substrata. In general, high energy and high polarity substrata will adsorb macromolecules strongly with a flattening of the macromolecules across the surface. Meanwhile, low energy, non-polar surfaces will bind macromolecules less strongly although a thicker layer may develop (McEldowney & Fletcher, 1986). The surface energy of clean high energy surfaces is reduced as a result of the film. However microbial adhesion is still related to the initial surface energy even after conditioning (Characklis & Marshall, 1990).

The conditioning film is not static as, with increasing molecular weight, polymers adsorb more strongly due to multiple binding sites. Thus, they may displace molecules of lower molecular weight (Characklis & Marshall, 1990). The conditioning film is presumed to be uniform in both composition and coverage. However, there has been no evidence for this and considering its dynamic nature it is probably patchy.

Conditioning films form prior to microbial adsorption in all aquatic environments and can inhibit or enhance bacterial adsorption. Chamberlain (Personal communication in Characklis & Marshall, 1990) found that increasing the concentration of organic materials decreased the adsorption of microbes and suggested that surface coverage could cause an inhibition of adsorption. However, other evidence suggests that the coating interacts efficiently with cell surfaces, exopolymers or organelles such as pili. The most effective thickness is considered to be between 15-20 nm (Characklis & Marshall, 1990). The role of the conditioning layer in attracting microbes could be the

concentration of nutrients. It has also been suggested that the divalent cations could act as a bridge between the two negatively charged surfaces of the cells and the medium (Ellwood *et al.*, 1982).

2B.2.4 Transport of Micro-organisms

When microbes are present at low dilutions the transport of micro-organisms may be the rate controlling step in biofilm accumulation. The processes involved are poorly understood, however in turbulent flows, molecular diffusion and eddy transport are viable, whilst in quiescent fluid chemotaxis is also possible (Trulear & Characklis, 1982). Brownian motion only applies to small particles and so micro-organisms of 1-4 μm^3 are hardly affected. However, larger cells are more strongly influenced by fluid dynamic forces, such as inertia, lift, drag, drainage and down-sweeps. If the density of the micro-organism differs substantially from the fluid density, gravity may play a significant role. The minimum transport rate occurs when cells are around 0.1 μm in diameter and so too small to be effected fully by fluid dynamic forces and too large to be fully effected by Brownian motion (Characklis & Marshall, 1990).

Transport is also effected by the geometry of the experimental system, which affects the fluid forces. Surface roughness is thought to influence transport rate by increasing mass transport near the substratum (Characklis & Marshall, 1990). Resistance to microbial transport increases as the cells penetrate the viscous boundary layer, and diffusion becomes the controlling mechanism. Cell motility may then become important as microbes exhibit chemotactic responses to the nutrient layer (Annachhatre & Bhamidimarri, 1992).

2B.2.5 Adhesion of Micro-organisms to Surfaces

The adhesion of bacteria to solid surfaces has been explained by two different physico-chemical theories. Firstly, the electrokinetic theory proposes that adhesion depends on the relative surface charge of the bacteria and the solid surface (Ellwood *et al.*, 1982). Secondly, the thermodynamic theory speculates that adhesion is governed by the change in free energy (ΔG) associated in bringing the bacterium and the surface from infinite separation to distances of molecular dimensions (Rosenberg and Kjelleberg, 1986).

McEldowney & Fletcher (1986), tried to evaluate the contribution of both electrostatic and hydrophobic forces to bacterial adhesion. The data indicated that both types of interaction played a role in adhesion, but that neither were dominant in all situations. It was suggested that a bacterium might use different adhesion mechanisms in different conditions.

Both the physico-chemical theories consider bacteria in the same way, as colloidal particles, and attempt to explain their adhesion to surfaces in terms of colloidal adsorption phenomena (Fletcher, 1987). Whether these approaches are valid depends on whether the metabolism of the bacteria is important in determining their adhesion to solid surfaces. Physiologists have presented evidence suggesting that bacteria do have an active role in adhesion to surfaces. This evidence constitutes a third type of adhesion theory. It is probable that all of the forces described by these theories have an influence on adhesion.

2B.2.5.1 The Electrokinetic Theory

Since the electrokinetic theory proposes that adhesion is dependent on charge interactions, the surface charge of bacteria (as the major component of biofilms) and media needs to be assessed.

2B.2.5.1.1 Cell surface charge

The primary polymer found in Gram-positive bacteria cell walls is peptidoglycan, which makes up 40-50% of the cell wall. Teichoic acids are another important constituent, accounting for over 60% of the wall in some species (Ellwood *et al.*,

1982). Teichoic acids are anionic polymers of glycerol and ribitol phosphate, they extend the surface of the peptidoglycan giving a negative charge to the Gram-positive wall (Prescott *et al.*, 1990). The net charge of the teichoic acids is modified by the presence of ester-bound D-alanine groups, the concentration of which is known to vary with the pH of growth. Therefore it can be hypothesised that the charge distribution on the surface walls of Gram-positive bacteria alters reflecting environmental conditions (Ellwood *et al.*, 1982).

The outer membrane of Gram-negative bacteria is relatively complex. It consists of a basal layer of 1-3 nm peptidoglycan and outer layers with 7-8 nm of protein, lipoprotein, phospholipid and lipopolysaccharide moieties. Lipopolysaccharides (LPS) have been well studied and used to characterise Gram-negative bacterial species. They consist of a lipid, which is buried in the outer membrane and linked via a C₈ sugar acid to a core polysaccharide. The polysaccharide can be substituted by side chains of oligosaccharides and 4-aminoarabinose or O-phosphorylethanolamine residues. The overall charge of LPS is derived from the balance of acidic phosphorus diesters, and the two positively charged bases. These could be variable, like the teichoic acid component of Gram-positive bacteria (Ellwood *et al.*, 1982).

The biofilm of BAFs have a greater diversity than AS flocs but the dominant aerobic genera are the same. These tend to be Gram-negative rods including *Zoogloea*, *Pseudomonas*, *Achromobacter*, *Alcaligenes* and *Flavobacterium* (Pike, 1975). Under general conditions, the bacteria in sewage will have a net negative charge.

2B.2.5.1.2 Media surface charge

Most of the materials used as BAF media have a net negative charge in sewage, however the charge density varies between the different materials. Plastics usually have very low charge densities, followed by fired clays and then sand. Since this research examines the use of foamed clays as BAF media, the composition of this material is described in more detail. The main constituent of the foamed clay, StarLight C, is calcined china clay.

Unfired china clay consists primarily of kaolinite ($\text{Al}_2\text{Si}_2\text{O}_5(\text{OH})_4$), a microcrystalline form of hydrated aluminium silicate, thought to have been formed by the breakdown

of feldspathic rocks. Kaolinite crystals are generally thin, hexagonal lamellae with diameters between fractions of a micron to five microns and aspect ratios between 10:1 to 30:1 (McColm, 1983). However, larger aggregated crystals are also common. During calcination, progressive dehydroxylation of the kaolinite occurs and the hydrated alumino-silicate is converted into material containing predominantly alumina and silica. The crystal structure begins to collapse as the hydroxyl groups are driven off as steam. When the temperature is raised, changes in the structure of metakaolinite occur. After 900 °C, formation of a matrix of an alumina/silica complex described as a defect spinel occurs, in which very small amounts of fine crystals of mullite are embedded. As the temperature rises, more alumina forms mullite and the excess silica remains as amorphous silica glass. StarLight C has a mullite structure with some amorphous silica and alumina and a slight negative charge derived from remaining hydroxyl groups (McColm, 1983).

2B.2.5.1.3 Surface charge interactions

The Derjaguin & Landau and the Verwey & Overbeek (D.L.V.O.) theory of colloid stability states that the total interaction energy (V_t) of two particles is determined by the sum of the attractive van der Waals energy (V_a) and the usually repulsive electrostatic energy (V_r). Curves showing the variation of V_t with separation distance (h) show two values of h at which a net attraction occurs (Figure 2.2). These are referred to as the primary minimum when h is very small and the secondary minimum when h is 5-10 nm. The minima are separated by a repulsive maximum.

Bacteria captured by a surface in a weak secondary minimum are in equilibrium with the remaining bacteria suspended in the bulk phase. The number of captured cells depends upon the concentration of the bacteria in the suspension and the depth of the secondary minimum. To move into the strong primary minimum requires the input of energy to overcome the repulsion peak. It has been proposed that while the cells are held at the secondary minimum, their cell surface polymers have the opportunity to adsorb non-specifically to the surface, or interact specifically with complimentary polymers adsorbed to or forming part of the surface (polymer bridging). When enough bridges have been formed the cell is held irreversibly. Attachment may be consolidated by the synthesis of insoluble macromolecules (Ellwood *et al.*, 1982).

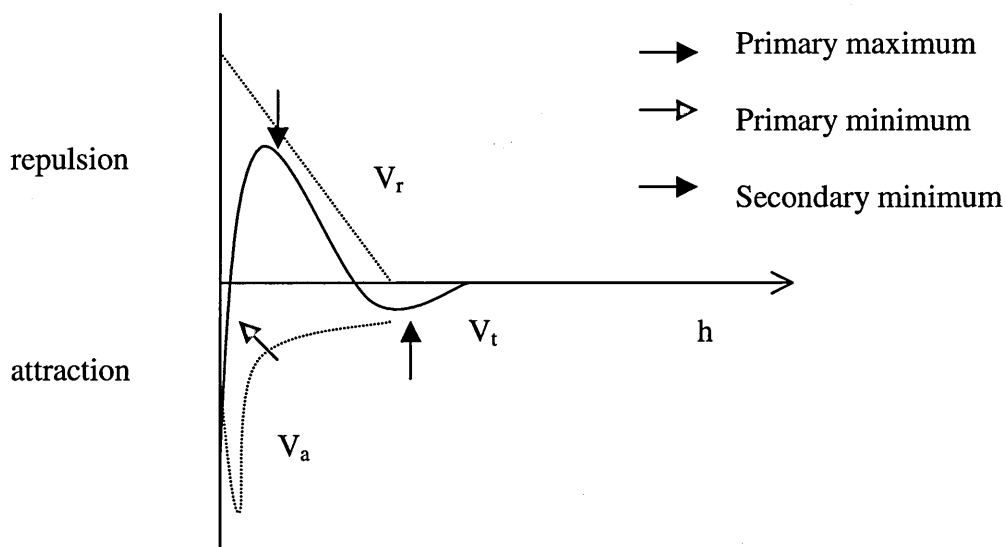


Figure 2.2: Variation of interaction energy of a particle and a surface with distance

The two stages of microbial adhesion described in the DLVO theory are defined as reversible physical adsorption and irreversible chemical adsorption (Characklis & Marshall, 1990). Physical adsorption involves short range forces such as Van der Waals, hydrogen bonds, protonation, co-ordination bonds and water bridging. The heat of adsorption per bond is only 20-50 KJ/mol, there is low specificity, and the adsorption is reversible and instantaneous. The microbes still show Brownian motion and there is a continuous exchange between free and adsorbed cells (Annachhatre & Bhamidimarri, 1992). Chemosorption is time dependant and thought to be due to the bacterial production of secondary fibrous acidic-polysaccharides that further cement attachment. Often both the bacterial cells and the collector surface have a negative charge, thus as their diffusive double layers overlap, cells experience a strong repulsive force. This results in the heat of adsorption per bond being high, 40-400 KJ/mol. The reaction is more specific, irreversible and usually results in only a single layer of microbes (Characklis & Marshall, 1990).

2B.2.5.1.4 Experimental evidence

The influence of the net surface charge of the substratum and the bacterium on adhesion have been tested experimentally. Fletcher and Loeb (1979) reported greater attachment of a negatively charged, marine, *Pseudomonas* species to hydrophilic

metals with a positive (platinum) or neutral (germanium) surface charge, than to hydrophilic negatively charged substrata (glass, mica, oxidised plastics). This appears to fit in well with the theory that electrostatic repulsion forces control adhesion. However, the substrata used may have varied substantially in their active toxicity towards micro-organisms, their surface free energy and surface roughness. All of these factors may influence the adhesive strength and the rate of colonisation of the surface by the bacteria (Marshall, 1992). Additionally, Fletcher and Loeb (1979) reported that the greatest microbial attachment occurred to hydrophobic plastics, which exhibited little or no surface charge. This suggests that electrostatic interactions are not the most important involved.

Altering the concentration of electrolytes in solution, or the pH, can change the surface charge of a particular material. This would appear to be a good way of overcoming the difficulties encountered when comparing different materials. However there are inherent difficulties in interpreting the data since the physiology of the organisms will be profoundly effected by changes in the liquid medium (Fletcher, 1987).

Heckels *et al.* (1976) modified the surface charge on *Neisseria gonorrhoeae* by blocking either the amino or carboxyl groups, and assessed subsequent attachment to human amnion cells grown in culture. They found that blocking amino groups to make the cells more negative, led to less attachment. Meanwhile, blocking carboxyl groups, making the cells more positive, led to more attachment. These results clearly suggest that electrostatic interactions between the cells and the surface are very important to adhesion. Feldner *et al.* (1983) presented a similar result when carboxyl groups were blocked on a mycoplasma membrane and its attachment to a glass surface was measured. However, they found no difference in attachment when amino groups were blocked and the same increase in attachment if both groups were blocked. This suggests that whilst electrostatic repulsion can lead to decreased attachment, electrostatic attraction is not the means by which attachment proceeds. Another explanation is that the methodology for these experiments only allowed the quantification of the chemically-attached cells, and the weakly attached cells in the secondary minimum were rinsed away prior to measurement. As cells need to be physically adsorbed before chemical adsorption can proceed, an increase in physically

adsorbed cells may increase the numbers that attach permanently. However, once a layer of physically-adsorbed cells is present, any further increases in physically adsorbed cells will not increase the overall adsorption rate. Thus for a particular system (i.e. bacteria species, liquid medium and solid surface), whether increased attachment can be achieved by altering the surface charge of the solid surface will depend on the rate at which permanent chemisorption proceeds compared to temporary physical adsorption.

2B.2.5.2 The Thermodynamic Theory

2B.2.5.2.1 Background thermodynamics

Thermodynamic terms used to describe surfaces, their definitions and how they relate to each other are listed in Table 2.4.

Table 2.4: Definitions of thermodynamic terms

Equivalent terms	Definition	Opposite terms
Hydrophobic	Not easily solvated by water, minimise contact with water.	Hydrophilic
Low wettability	Not easily solvated by water, minimise contact with water.	High wettability
Low energy	Low attraction to water means a small amount of energy would be needed displace the water.	High energy
Low polarity	Low numbers of residues which are charged or have strong dipoles.	High polarity
Low surface charge	Low numbers of residues which are charged or have strong dipoles.	High surface charge
Low surface tension (dyne/cm)	Measure of attraction between surface particles.	High surface tension
High contact angle	The angle of contact between small drops of liquid and the surface.	Low contact angle

The first law of thermodynamics states that “the total energy of an isolated system is constant, though within that system it may change its form”. The intrinsic energy of a

system is an attribute that depends on the system's state and is independent of its previous history. When temperature and pressure are constant, the first law assures us that any change in a system's intrinsic energy is accounted for by an equal but opposite change in the intrinsic energy content of its surroundings (Morris, 1976).

The change in enthalpy (ΔH) which occurs during a change in state can be defined as, "the quantity of heat adsorbed by a closed isothermal system when, at constant pressure, it undergoes a change of state without performing any work save that associated with its change in volume". A reaction which involves a loss of enthalpy (e.g. heat to the environment) is called exothermic. Meanwhile a reaction which results in a gain in enthalpy (adsorption of heat) is called endothermic. For a chemical reaction, the sign and magnitude of the ΔH is mainly due to the making and breaking of chemical bonds.

Simplistically, entropy (S) is an index of the stability of a system. It has been described as a measure of the randomness of energy distribution in the system. The second law of thermodynamics states, that systems always progress to a state of maximum disorder. Thus the more chaotic a system, the greater is its entropy and its stability. Meanwhile the more orderly the system is, the smaller is its entropy and its stability (Morris, 1976).

In a closed system the only reactions which occur spontaneously are those that increase the total entropy of the system and its surroundings. The change in entropy (ΔS) of the systems surroundings is equal to the heat adsorbed (ΔH) by the surroundings divided by temperature (T). Thus a reaction will be spontaneous if:

$$\Delta S_{\text{system}} - \Delta H_{\text{system}} / T > 0 \quad (7)$$

$$\text{or } \Delta H_{\text{system}} - T\Delta S_{\text{system}} < 0 \quad (8)$$

Equation 8 is a measure of the change in another function of state named the Gibbs Free Energy (ΔG). When ΔG is negative the reaction is exergonic and can occur spontaneously, whilst reactions which result in a positive change in ΔG are endergonic and will not proceed spontaneously (Morris, 1976).

In summary, if a reaction leads to the system losing enthalpy and / or gaining entropy, it is more likely that the change in Gibbs free energy will be negative and that the reaction will proceed spontaneously. The change in Gibbs free energy depends on the bacterial surface, the substratum and the liquid medium (Fletcher & Pringle, 1983).

2B.2.5.2.2 Hydrophobicity of the bacterial surface

Despite usually having an overall negative charge, microbes often exhibit hydrophobic behaviour. This may be due to the presence of proteins with amino-acids with hydrophobic side groups which act as “polymeric adhesives” (Characklis & Marshall, 1990). Alternatively, the surface tension of charged cells may be lower than that of the aqueous media, meaning that adhesion to hydrophobic surfaces will be favoured.

The surface of bacterial cells varies greatly as a function of growth conditions, so hydrophobicity values are dependent on the environment (McEldowney & Fletcher, 1986). Parameters such as aeration, temperature, feed composition and the age and growth phase of the cell can all affect bacterial cell surface composition (Rosenberg & Kjelleberg, 1986). Often starvation leads to a large increase in hydrophobicity (Rosenberg & Kjelleberg, 1986). Meanwhile, Fletcher (1977) found that a marine pseudomonad showed greater attachment to polystyrene when the micro-organism was in its logarithmic growth phase. These observed changes in adhesiveness are likely to be mediated by changes in the cell wall. Indeed it has been demonstrated that the composition of liquid medium and, to some degree temperature, have a considerable influence on the composition of the major outer membrane proteins (OMP) of *Escherichia coli* (Dalhoff, 1985). Even the molecular size and osmolarity of sugars can influence the synthesis of OMPs (Dalhoff, 1985). Furthermore, there are examples of certain OMPs which are only produced under special growth conditions, e.g. low iron or phosphate conditions (Dalhoff, 1985).

Different species of bacteria have a whole range of hydrophobicity levels and different responses to changes in the environment. This was clearly shown by Pedersen *et al.* (1986) who used three different microbial species that showed different responses to changes in the cell concentrations, nutrient concentrations,

incubation times and substrata. Indeed, cell surface hydrophobicity varies not only from species to species but also from strain to strain (Rosenberg & Kjelleberg, 1986).

The hydrophobicity of a bacterial surface can be measured using many techniques. These include measuring the liquid contact angles on a lawn of cells, using molecular probes, hydrophobic interaction chromatography, phase partition chromatography, the salting-out aggregation test (Rosenberg & Kjelleberg, 1986) and measuring bacterial adhesion to hydrocarbon (Dewanti & Wong, 1995). These methods may give differing results because each measures the hydrophobicity at a different depth in the bacterial surface (Denyer *et al.*, 1993).

2B.2.5.2.3 Hydrophobicity of the solid surface

A conditioning film builds up on a solid surface when it is immersed in fluid. This film changes the hydrophobicity of the substrata, since it is composed of molecules that have both positive and negative charges. Generally the conditioning film will bring the surface tension of the solid nearer to that of the liquid.

Contact angle measurements are usually made to characterise surface hydrophobicity. However, when dealing with real solid surfaces, which are often rough and heterogeneous, the thermodynamic status of the contact angle becomes doubtful (Neumann *et al.*, 1974). For instance, there may be localised adsorption on the high energy sites, but not on the low energy portions of heterogeneous surfaces. Generally, the effects of adsorption become more of a problem as the contact angle decreases.

2B.2.5.2.4 Thermodynamic relationship between cells, medium and solids surface

Hydrophobic interactions can be simplistically explained with the second law of thermodynamics. According to Rosenberg & Kjelleberg (1986) water molecules will surround a non polar moiety introduced into an aqueous phase in the bulk aqueous phase. Such structured water layers are associated with a decrease in entropy related to the increased order of the system. The coming together of two hydrophobic surfaces leads to the release of some of the ordered molecules resulting in a decrease in the free energy.

Experimental evidence suggests that the formation of bonds between two surfaces and the breaking of bonds between the surface and water are negligible compared to the cavity effect (the entropy function). However, this description of hydrophobic interactions was based on extremely non-polar moieties such as molecules of alkane. The surfaces of bacteria and substrata are composed of both polar and non-polar constituents and so a combination of hydrophilic and hydrophobic surface components determine the overall contribution of hydrophobic interactions to the adhesion process.

The outcome of an adhesion process has been predicted by comparing the interfacial tensions between the adhering surfaces, to that of each individual surface with the aqueous medium. Thus, adhesion to hydrophilic surfaces is more pronounced when the bacterial surface tension is larger (more hydrophilic) than that of the surrounding aqueous medium. Conversely, adhesion to increasingly hydrophobic surfaces occurs when the surface tension of the aqueous phase is higher than that of the bacterial cells (Absolom *et al.*, 1983). Due to the highly polar nature of water in most aqueous systems the second situation prevails (Rosenberg & Kjelleberg, 1986). This relationship can be expressed by Equation 9. The interfacial tensions can be related to surface tensions by Young's equation (Absolom *et al.*, 1983).

$$\Delta F^{\text{adh}} = \gamma_{\text{BS}} - \gamma_{\text{BL}} - \gamma_{\text{SL}} \quad (9)$$

Where: ΔF^{adh} = free energy of adhesion
 γ_{BS} = bacterium-substratum interfacial tension
 γ_{BL} = bacterium-liquid interfacial tension
 γ_{SL} = substratum- liquid interfacial tension

2B.2.5.2.5 Experimental evidence

Many researchers have found that bacterial adhesion is greater on hydrophobic rather than hydrophilic surfaces (Fletcher & Loeb, 1979, Dalton *et al.*, 1994, Dahlback *et al.*, 1981, Wanda *et al.*, 1990, Burchard *et al.*, 1990, Shreve *et al.*, 1991, Pedersen *et al.*, 1986). However, Baker (1984) found no differences in the colonisation rate of river bacteria on glass, which has a high surface charge, and polystyrene, which has a low

surface charge. Dexter *et al.* (1975) found increased adhesion to hydrophilic surfaces in the sea. Both these experiments were conducted under natural conditions where a very large number of organisms were available to colonise the surfaces. It was suggested that when such a variety of colonisers are available the influence of the surface charge is reduced, which infers that different bacteria are involved in the initial colonisation of different substrates.

Changing the surface tension of the liquid medium may influence the adhesion of micro-organisms to surfaces as described by the thermodynamic equations (Fletcher & Pringle, 1983, Absolom *et al.*, 1983, Facchini *et al.*, 1988). However, the composition of the liquid medium may also be important. The presence of proteins can impair attachment (Feldner *et al.*, 1983, Fletcher, 1976). Proposed mechanisms include the formation of a protein film on the surface which inhibits attachment through steric exclusion and, a reduction in the hydrophobicity of the substrate and bacterial surfaces due to protein binding to non-polar substances. As previously mentioned in section 2B.2.5.2.2, the composition of growth medium may influence adhesion by affecting the composition of the major OMPs (Dalhoff, 1985).

2B.2.5.3 The Role of Cellular Physiology In Microbial Surface Adhesion

Experiments have suggested that only organisms with an intact energy metabolism can enter the primary energy minimum and attach more firmly to surfaces (Felder *et al.*, 1983). The energy may be necessary for organisms to adopt a more stretched form to overcome the electrical double layer or to produce special binding proteins.

The cells of *Vibrio proteolytica* have been shown to be less adhesive to polystyrene when cell wall synthesis, energy production or protein synthesis is impaired (Paul, 1984). The attachment of the same bacteria to hydrophobic polystyrene was inhibited by over 97% when proteolytic enzymes were present (Paul & Jeffrey, 1985). The enzymes lowered the hydrophobicity of the cells, which may have led to decreased attachment. However, it was also proposed that since some of the attached cells were removed, proteins may have a direct role in adhesion. The proteolytic enzyme treatment had no effect on the attachment of the organism to hydrophilic substrata, like glass, suggesting bacteria may employ different mechanisms to attach to hydrophobic and hydrophilic surfaces.

The adhesiveness of cells can change with different nutrient conditions (Paul, 1984), growth phases (McEldowney & Fletcher, 1986) and salt concentration (Gordon & Millero, 1984). When nutrient availability decreases, bacterial size and membrane fatty acid composition often alters to make the cells more hydrophobic (Dewanti & Wong, 1995). The production of extracellular polymers also appears to be associated with low nutrient levels.

2B.2.6 Influences of the BAF Media Surface on Microbial Adhesion

To increase the start-up rate and backwash recovery times, an ideal BAF medium should promote microbial adhesion. Media surface properties such as asperity, specific surface area (SSA), and surface chemistry may all affect microbial adhesion.

Many researchers have found that increasing the roughness of a surface greatly increases the rate of bacterial colonisation (Baker, 1984, Heijnen *et al.*, 1992). Baker (1984) tested two of the possible reasons for this phenomenon; that roughening increases the available SSA, and that the depressions provide sites which are protected from the fluid flow. Scratches of a known length and depth were made with a diamond marker to increase the SSAs of glass and polystyrene test pieces by up to five times. The corresponding increases in colonisation were two orders of magnitude greater than the increase in SSA indicating it was not the sole influence. However, the bacteria were not concentrated around the grooves in the substrate and there were no significant differences in the colonisation rates at different current velocities. This evidence suggested that the provision of protected sites was not controlling colonisation either. It was hypothesised that asperity may provide sites for the initial glycocalyx attachment.

Often the SSA of structured media reactors, has less influence on biofilm formation than the ability of the medium to redistribute flow (Young and Dahab, 1983, Athanasopoulos & Karadimitris, 1988). This is because BAFs using a medium with a high voidage are prone to short circuiting. A granular medium with a high voidage may exhibit similar tendencies. However, generally increasing the SSA of a granular packing media increases the quantity of biofilm in the reactor (Heitkamp *et al.*, 1993).

Examining the adhesion mechanisms of bacteria suggests that a media's surface chemistry is an influential parameter effecting microbial adhesion. However, sewage contains many different species and strains of bacteria, each of which will preferentially adhere to surfaces with different surface tensions. Altering the media surface chemistry is therefore unlikely to have a great influence on the overall biofilm attachment.

2B.2.7 Desorption of Micro-organisms from Surfaces

Some of the cells that reversibly adsorb, later desorb from surfaces. The process may depend on fluid shear forces (Characklis & Marshall, 1990).

2B.2.8 Biofilm Growth

In BAFs, biofilm accumulation increases as the micro-organisms metabolise organic matter from the surrounding wastewater. Cell numbers increase and significant amounts of extracellular polymeric substances (EPS) are excreted. These products are important in binding the cells to the substrate and to each other.

Some surface-associated organisms grow about twice as fast as their suspended counterparts (Ellwood *et al.*, 1982). This may be because of the higher concentration of nutrient found at the surface. However, unless the mass transfer of the limiting substrate to the surface is continuous and rapid, the increased concentration would be rapidly consumed. Therefore, the high surface-associated growth rate may be a result of other biofilm advantages such as, protection and synergy.

Combining critical point drying and scanning electron microscope (SEM) techniques has revealed fibrils which connect both the cells to surfaces and to each other (Ellwood *et al.*, 1982). These may be involved in a cell to cell transport mechanism. Organisms in dental plaque have shown a distorted morphology to maximise surface contact (Ellwood *et al.* 1982).

A theoretical advantage of the biofilm niche is described in the chemiosmotic hypothesis of energy conservation. Proton motive forces are produced by the translocation of protons out of the cell to generate a membrane potential between the bulk and the negative cell cytoplasm. ATPase in the cell membrane can then catalyse ATP from ADP and inorganic phosphate against the inward flow of protons through the enzyme complex. Experiments have detected pH changes around cells suggesting that some protons are lost during this process. When a cell interacts with a surface the protons may be restricted increasing the probability of their re-uptake. Thus the cell

membrane will be more energised on the side nearest the surface. The resulting polarity may be involved in locating surfaces. The increased energy available to the cell may encourage growth, and reproduction. As the biofilm develops, the proton gradients may be shared within the community (Ellwood *et al.*, 1982).

More specific symbiotic relationships have also been suggested to exist in biofilm communities. For instance, lactate-consumers are often intimately related to lactate-producers (Ellwood *et al.*, 1982).

2B.2.8.1 Environmental influences on biofilm growth

At low temperatures microbial growth rates and EPS synthesis rates are low, and accordingly, biofilm growth and development is hindered. Additionally, adsorption, which is the first stage of microbial adhesion, is strongly enhanced by temperature increases (Annachhatre & Bhamidimarri, 1992).

Nutrients are necessary to maintain optimal growth conditions in the reactor. Phosphorus supply is particularly important for biofilm reactors since it is observed that cells saturated with phosphate have a higher tendency to flocculate and to adhere due to their increased hydrophobicity (Annachhatre & Bhamidimarri, 1992). Calcium and Magnesium are important for the maintenance of the biofilm's intercellular polymer matrix. Certain other metal ions such as Nickel, Cobalt, Iron, Copper, and Molybdenum are known to facilitate biofilm formation though their exact role is not completely understood (Annachhatre & Bhamidimarri, 1992).

Huang & Liu (1993) found that COD removal increased rapidly with biofilm growth until a certain "maximum active thickness" was reached. This thickness corresponded to the minimum biofilm depth to initiate sulphide production. It was essentially equivalent to the thickness of the aerobic layer, beyond which the electron acceptor shifts from molecular oxygen to either NO_3^- , NO_2^- , Fe^{3+} or SO_4^{2-} . As expected, when the DO levels were lowered, the active film thickness also decreased and equations have been formulated to describe this relationship .

2B.2.8.2 Media influences on biofilm growth

Messing *et al.* (1979), found a relationship between the optimum pore size of the media for biofilm accumulation, and the mode of reproduction of the specific microbes. Microbes that reproduced by fission were found to accumulate fastest when the pore diameter was between 1 and 5 times the major dimension of the microbe. Meanwhile for budding microbes the optimum pore diameter was between 1 and 3 times that of the largest dimension. Microbes reproducing by mycelial growth accumulated best when the pores were 16 times the largest dimension of the spores. Thus for a BAF media, pores smaller than about 5 μm will not promote biofilm accumulation.

Growth tests in the presence and absence of suspended clay particles and clay extract have shown that clay releases nutrients that stimulate methanogenic activity. This property is retained when clay is fired. Methanogenic cultures using acetic acid are stimulated markedly by high levels of soluble iron, and analysis found 225 μg iron/g clay (Murray & van den Berg, 1981). It was therefore concluded that the stimulated biofilm formation was due to the release of nutrients such as iron from the clay. Slow release is made possible through the cation exchange capabilities of the clay (Murray & van den Berg, 1981) and would be very beneficial to the slow-growing methanogens. It seems likely that different materials may either stimulate or inhibit aerobic biofilm development.

2B.2.8.3 Reactor influences on biofilm growth

At steady-state, suspended microbes in a continuous flow reactor grow at a rate equal to the hydraulic residence time (HRT). If HRT decreases further when the bacteria have reached their maximum growth rate, cell wash out will no longer be balanced by growth and the numbers of suspended microbes will decrease. As long as media wash out is prevented, microbes in biofilms are not affected directly by HRTs (Characklis & Marshall, 1990). Biofilm growth is best when the HRT is shorter than the inverse of the maximum growth rate of the suspended bacteria. Then, the attached micro-organisms have little competition from suspended microbes (Tijhuis *et al.*, 1994).

The hydrodynamics near the media surface determine the shear stress and the heat and mass transfer rate from bulk liquid to the biofilm (de Beer *et al.*, 1994). Biofilm spatial distribution and thickness, differs as a function of hydrodynamic shear stress conditions (Lazarova *et al.*, 1994). Generally biofilms in low stress environments tend to be thicker and less dense than those in high stress environments (Trinet *et al.*, 1991). Additionally, exopolymer secretion increases with increasing turbulence making the biofilm less vulnerable to detachment. Substrate flux increases as the substrate bulk concentration increases and as the liquid velocity increases. Substrate flux determines the maximum active thickness of a biofilm (Arcangeli & Arvin, 1995).

2B.2.8.4 Measuring biofilm growth

2B.2.8.4.1 Destructive techniques

There are many available methods for directly measuring biofilm growth. Biovolume can be measured using the apparatus illustrated in Figure 2.3 (Characklis *et al.*, 1982). The biomass of biofilms can be calculated by weighing dry samples before and after ashing in a furnace (Gjaltema *et al.*, 1994).

Other techniques require the biofilm to be removed from the substrata prior to analysis, by sonication, acid, or alkaline hydrolysis (Lazarova *et al.*, 1994). The total biofilm amount can then be measured in terms of COD, or total organic carbon (Tijhuis *et al.*, 1994). Chemical methods can also be used to measure specific components of biofilms. There are techniques to assess the protein content (Lowry *et al.*, 1951), the carbohydrate content (Dubois *et al.*, 1956), the extracellular polymer content (Frolund *et al.*, 1996) and the phospholipid content (Findlay *et al.*, 1989) of the samples.

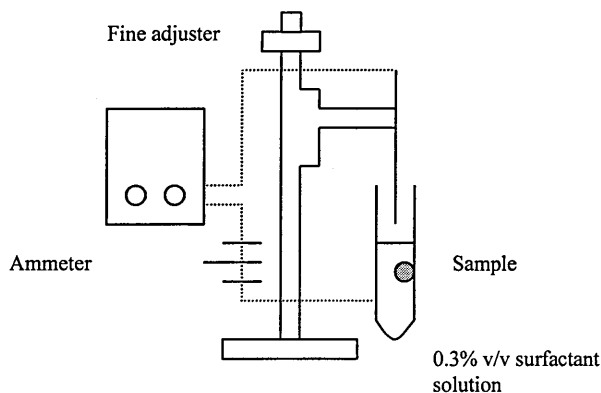


Figure 2.3: Apparatus for measuring biovolume

There are methods that quantify only the viable biomass in a sample. For instance, ATP can be assayed (Jorgensen *et al.*, 1992), DNA synthesis can be measured (Ellenbroek & Cappenberg, 1991) and fixed biomass respirometric activity can be determined by measuring oxygen consumption over three hours (Bacquet *et al.*, 1991). The INT-formazan assay measures the electron transport system activity (ETS) and has the advantage of not requiring an extraction step (Blenkinsopp & Lock, 1990).

2B.2.8.4.2 Non-destructive techniques

Biofilm thickness on a transparent surface can be measured using light microscopy (Bakke & Olsson, 1986). On tubular test sections biofilm thickness can also be assessed by monitoring heat exchange and frictional resistance (Characklis *et al.*, 1982). Light microscopy has been used to make cell counts (Wanda *et al.*, 1990). Biofilm growth can also be assessed indirectly by analysing substrate utilisation (Wolfaardt *et al.*, 1994).

2B.2.8.4.3 Biofilm reactors

Many specialist small-scale reactors have been developed so that representative samples may be repeatedly analysed, and grown in known conditions (Table 2.5).

Table 2.5: Characteristics of Biofilm Test Reactors

Reactor	Reference	Test Surface	Repeat analysis?	Sample represents	Sterile?	Degree of control	Like BAF?	Cost
Robbins device	Nickel <i>et al.</i> , 1985	Flat plugs	Yes	Site	Yes	Complete	No	High
Rectangular open-channel	Ohashi & Harada, 1996	Flat plates	Yes	Site	No	Complete	No	High
Flow cell reactor	Caldwell & Lawrence, 1986	Cover slips	No	Site	Yes	Complete	No	Low
Rotating disc biofilm reactor	Gilbert & Allison, 1993	Flat disc	Yes	Site	No	Part	No	Low
Packed bed reactors	Characklis & Marshall, 1990	Media prills	No	Depends on mixing	No	Part	Yes	Low
Chemostat	Keevil <i>et al.</i> , 1987	Flat tiles	Yes	Reactor	Yes	Part	No	High
Fluidised beds	Tijhuis <i>et al.</i> , 1994	Media prills	No	Reactor	No	Part	Close	High
Radial flow reactor	Peters & Wimpenny, 1988	Flat disc	No	Site	Yes	Complete	No	High
Tubular reactors	Characklis <i>et al.</i> , 1982	Flat plates	No	Site	Yes	Complete	No	High
Annular reactors	Characklis <i>et al.</i> , 1982	Flat slides	Yes	Site	Yes	Part	No	High
Constant depth film fermenter	Peters & Wimpenny, 1988	Flat plugs	Yes	Site	Yes	Part	No	High

Only packed bed reactors represent the conditions found in BAFs. However, they do not allow repeated analysis of the biofilm at a specific site and, in a plug flow reactor, this is necessary to monitor biofilm development. A small sample of media from a completely mixed reactor represents the whole bed. However, as conditions become more plug flow, multiple samples need to be taken as conditions alter throughout the reactor depth .

2B.2.9 Attachment to biofilms

Attachment is the immobilisation of cells and other particulate matter in the biofilm, while adsorption refers to the same processes occurring on the substratum (Characklis & Marshall, 1990). Blenkinsopp & Costerton (1991) suggested that multi-species consortia develop because planktonic cells are attracted to adhere by the metabolic products of a colonised species. This would mean the initial colonisers would have an influence on the resulting biofilm community.

2B.2.10 Detachment from biofilms

Detachment refers to the loss of material from the biofilm, while desorption is loss of cells and other material from the substratum. The detachment rate has the most impact on the slowest growing species and so can increase the proportion of active biomass in the biofilm (Rittmann *et al.*, 1992).

Detachment can be divided into continuous and discontinuous processes. Erosion, caused by liquid shear, and abrasion, caused by collisions, are both continuous processes. Meanwhile sloughing is discontinuous and involves the loss of large patches of biofilm as a result of shear or collisions (Gjaltema *et al.*, 1994).

2B.2.11 Biofilm Structure

Until the early nineties, accepted theory described biofilms as structurally homogeneous, with a constant thickness and density. Numerous optical and SEM studies backed up this supposition. The theory was the basis of biofilm models, which in turn, were utilised to model processes. Good process models allow improved design, understanding and ultimately superior performance to be achieved. New techniques and technologies have exposed some of the complexity of biofilm structure, and so should improve our understanding of biofilm processes (Figure 2.4).

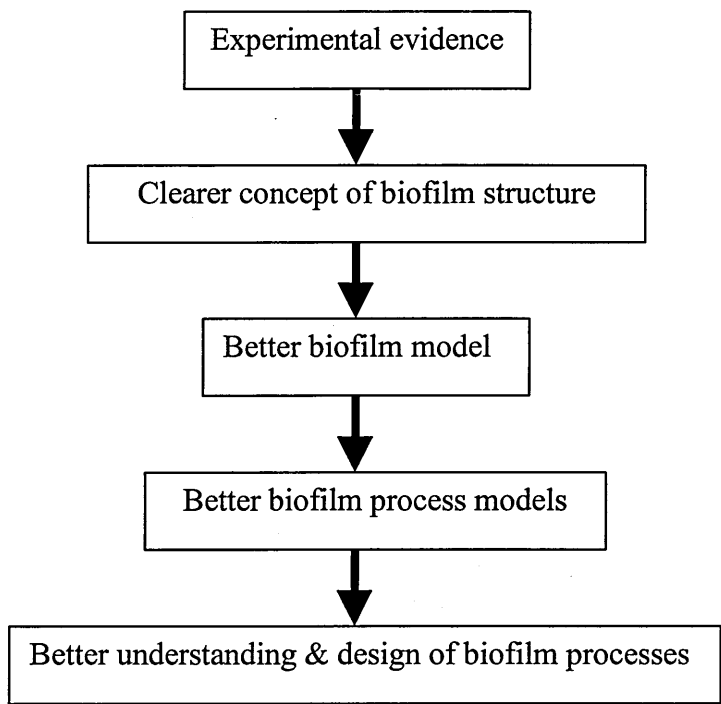


Figure 2.4: Flow chart illustrating the importance of the biofilm structure concept

2B.2.11.1 Heterogeneity with depth

Recent evidence has indicated that biofilms are highly stratified communities. Sectioning with a cryogenic microtome, showed that the density of a biofilm increased towards the its base. Meanwhile, the metabolic activity, porosity, mean pore size and correspondingly the diffusivity for DO, all decreased nearer the biofilm base (Zhang & Bishop, 1994). This stratification has been shown to become more pronounced until the biofilm reaches a critical depth of about 240 µm (Bishop *et al.*,

1995). Then, as the biofilm thickens, internal mass transfer resistance limits the growth of the aerobic bacteria positioned deep inside the biofilm. Facultative organisms and anaerobic bacteria, which are slower growing bacteria, start to increase near the substratum. The slower growth rate combined with the death and lysis of aerobes, leads to a decrease in the density of the biofilm near the substratum surface.

Microelectrodes have been inserted into biofilms to quantify the spatial distributions of pH, DO, ammonium and nitrate (Bishop *et al.*, 1995). The DO probes have confirmed that as the biofilm continues to deepen, two layers develop, one aerobic and the other anaerobic. Previous research suggested that reaching the limiting thickness for oxygen diffusion causes a temporary suppression of substrate removal efficiency. This could be because the biofilm becomes unstable as its structure and composition changes (Hoehn & Ray, 1973).

Microelectrode measurements of ammonium and nitrate profiles, combined with microsectional evidence, showed that most of the nitrification occurred about 100 μm below the biofilm surface. It was suggested that at this position there was reduced competition from the fast growing heterotrophs (Zhang & Bishop, 1994b).

2B.2.11.2 Lateral Heterogeneity

Rather than being planar structures with homogenous cell distributions, there is growing evidence that biofilms are composed of heterogeneous cell clusters and layers with voids, channels, cavities, pores and filaments.

Using an optical method Stewart *et al.* (1993) showed that biofilms vary significantly in thickness over 10 μm distances or less. The biofilm reached 100 μm in some places and yet was bare in others. Meanwhile, oxygen profiles differ between different sites on a biofilm particle and between different particles from the same sample (Loosdrecht *et al.*, 1995).

When fluorescent beads were added to an aerobic reactor they penetrated the biofilm and revealed the complex network of voids (Beer *et al.*, 1994). Voids running perpendicular to the substratum were defined as pores, and those running parallel were called conduits. The difference in the height of the pores and the cell clusters

produced an irregular surface. For biofilm roughness to increase the effective SSA of the biofilm, the surface elements must have a height over half the depth of the mass boundary layer. The mass boundary layer was 40 μm above the cell clusters and 55 μm above the pores. However, at lower velocities, the mass boundary layer will thicken and may not follow the contours of the biofilm. The exchange area will then equal the substratum area.

Beer *et al.* (1994) later showed that liquid was flowing through the voids of their biofilm. This was done qualitatively by micro-injecting fluorescein and quantitatively by tracking fluorescent latex spheres using confocal scanning laser microscopy (CSLM). It was concluded that in voids both diffusion and convection may contribute to mass transfer, whereas in cell clusters diffusion is the dominant factor. The flow velocity in the biofilm depended on the average flow velocity of the bulk liquid.

The formation and closure of cracks and fissures on biofilms has been observed, and may take place continuously (van Benthum *et al.*, 1995). Why these structures are formed is not clear. One theory is that the maintenance of the channels is a general microbial strategy to deal with the problem of limiting diffusive transport in biofilms (Massol-Deya *et al.*, 1995). Alternatively, local shear stresses or local growth rate differences may be involved.

CSLM allows the *in situ* microscopic investigation of the internal structures of biofilms. Biofilms examined in this way have been composed of discrete microbial clusters, in a “mushroom” shape with channels running between them (Bishop & Rittmann, 1995). However, only relatively thin biofilms have been examined because the laser can only penetrate up to 200 μm . More mature biofilms may have more continuous structures (Bishop, 1997).

The dehydration procedure required prior to conventional SEM analysis removes EPS material and so alters the structure of biofilms. However, an environmental scanning electron microscope (ESEM) enables biofilms to be examined directly in their fully hydrated state. These images show cells completely buried under a smooth canopy of EPS (Sutton *et al.*, 1994). Since the samples are uncoated, energy dispersive x-ray spectroscopy data can determine the elemental composition of surface films, as

surface morphology is photographed. It has been suggested this technique could monitor the precipitation and binding of heavy metals on to the EPS (Little *et al.*, 1991).

Further evidence is required to determine whether the “mushroom” shape and the lateral heterogeneity is a transient state in biofilm development. It may only occur in some systems under specific circumstances. However, if it is the dominant type of biofilm structure there are many implications to biofilm processes.

2B.2.11.3 Modelling Biofilms

The first biofilm models (Williamson & McCarty, 1976, Rittmann & McCarty, 1980) considered an idealised biofilm, with uniform cell density and thickness, at steady-state (Figure 2.5a). Thus, the biofilm was not growing or decaying over time. Two basic processes were considered;

1. The limiting substrate must diffuse from the bulk liquid to the biofilm, through a stagnant layer at the biofilm surface. The diffusion process is described by Fick’s Law.
2. The substrate is then transformed within the biofilm following Monod kinetics.

This model was developed for biofilms in nonsteady-state (Rittmann & Brunner, 1984) by considering a further three processes;

3. The molecular diffusion of substrate through the biofilm.
4. The growth of biomass in proportion to substrate utilisation.
5. The loss of biofilm by cell decay and shearing.

Using these assumptions further developments in biofilm modelling included solutions for mixed cultures (Droste & Kennedy, 1986, Namkung & Rittmann, 1987). The positions of autotrophs and heterotrophs were described by Furamai & Rittmann (1994) (Figure 2.5b). This type of model was developed and incorporated into the computer program, AQUASIM (Wanner & Reichert, 1996).

However, the adaptation of the existing models will not be sufficient to describe a “mushroom” biofilm structure consisting of microclusters separated by relatively large channels (Figure 2.5c). All the existing models are essentially one dimensional and so would not account for advective transport to deep parts of the biofilm through the channels. Since the substrate could be supplied laterally from the channels, as well as vertically through the biofilm, a three dimension model would be needed (Bishop, 1997). If this biofilm structure is found to be significant, kinetics will need to be conducted on a macro, rather than a micro scale. For instance, the effective SSA or the integrated diffusion co-efficients could be calculated. It will be unlikely that one model will be useful to all researchers (Bishop & Rittmann, 1995)

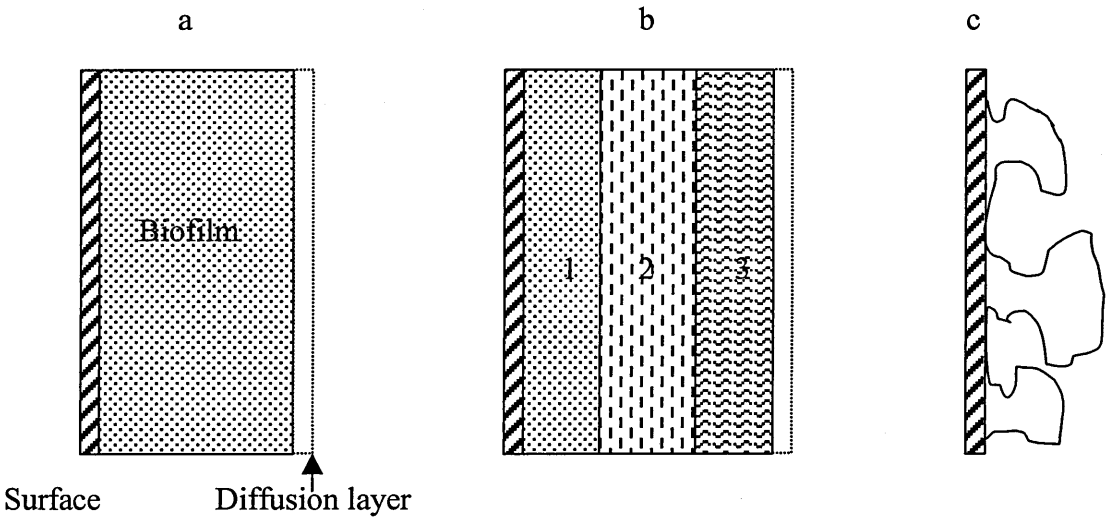


Figure 2.5: Examples of theoretical biofilm structures; a= homogeneous model; b= multispecies model, where 1 is inert or anaerobic biomass, 2 is nitrifying bacteria & 3 is heterotrophic biomass; c= “mushroom” structure

2B.3 THE FILTRATION CHARACTERISTICS OF BAF MEDIA

2B.3.1 Properties of BAF Media to Promote Solids Removal

For granular media filters operated between 5 and 24 m/h, particles are removed throughout the length of the filter by mechanisms other than straining (Goldgrabe *et al.*, 1993). Reported liquid velocities in BAFs vary between 1.5 m/h (Paffoni *et al.*, 1990) and 10 m/h (Vedry *et al.*, 1994). However, the system is normally operated at between 2 and 4 m/h (Rogalla *et al.*, 1990) so there may be a combination of straining, and depth filtration, solids removal mechanisms.

The mechanisms by which solids are transported to the media surfaces in a depth filter are interception, diffusion and sedimentation (Bouwer, 1987). As most particles in settled sewage have diameters greater than 1 μm , the dominant particle removal mechanisms in a BAF will be sedimentation and interception (Goldgrabe *et al.*, 1993). Filtration theory predicts improved filtration when the size and voidage of a medium are decreased (Equation 10). Voidage is lower for media composed of small particles, and decreases as the particle size range, smoothness or sphericity increases.

$$N/N_0 = \exp [1.5 (1 - \varepsilon / d_m) L \alpha_c \eta_0] \quad (10)$$

Where:

- N/N_0 = the fraction of particles removed
- ε = clean bed porosity (%)
- d_m = media diameter (m)
- L = filter media depth (m)
- α_c = collision efficiency factor
- η_0 = single collector efficiency for sedimentation and interception

This expression is only applicable to clean filter beds and does not account for the accumulation of deposits or biofilm formation. In rapid gravity filters, particles are removed from suspension by attaching either to the filter media, or to previously retained particles (Colton *et al.*, 1996). In BAFs particle removal will also be facilitated by the biofilm. Transport and attachment of particles to the biofilm has

been described as a biofilm filtration process that depends on the geometry and structure of the biofilm. A model based on this theory shows that for typical substratum and biofilm structures, particle removal by biofilm filtration exceeds that by other mechanisms (Stolezenbach, 1989). Therefore, media properties that promote biofilm formation will indirectly lead to superior filtration efficiency. In addition, the filtration efficiency changes as the apparent filter grain size increases and the filter porosity decreases (Choo & Tien, 1993)

Non-spherical, rough particles show increased filtration efficiency in experiments and it has been suggested that they do this by inducing local changes in flow direction (Ives, 1990).

2B.3.2 The Influence of BAF Media on Head loss Development

The pressure above a filter bed increases as deposited solids reduce its porosity. The disadvantage in reducing either a medium's size or voidage is a corresponding increase in the head pressure above the filter as approximated by Equation 11. Thus, BAFs using smaller media exhibit shorter filter run times and an increased backwash frequency (Coulson & Richardson, 1978).

$$P / L = k'' (\mu / \rho_f) (v / g) [(1-\epsilon)^2 / \epsilon^3] (6 / d_m)^2 \quad (11)$$

Where: P = change in head loss (N/m^2)

k'' = Kozeny constant

μ = fluid absolute viscosity ($kgm^{-1}s^{-1}$)

ρ_f = fluid density (kg/m^3)

v = approach velocity (m/s)

g = gravitational acceleration (m/s^2)

An ideal media would be composed of particles with a uniform size, just small enough to reach the required effluent standard. Voidage should be maximised by utilising non-spherical, rough particles, so that head loss build-up is minimised.

2B.3.3 Properties of BAF Media that Avoid Backwashing Costs

2B.3.3.1 Influence of the medium's minimum fluidisation velocity (V_{mf})

The minimum fluidisation velocity (V_{mf}) of a medium is often used to determine the backwash velocity required. An air scour between 30-100 m/h combined with a liquid flow at 40-60 % of the V_{mf} stimulates the bed to “collapse-pulse”. These are thought to be the most effective backwashing conditions (Amirtharajah, 1993).

The V_{mf} of a media is effected by the media density, size, shape, voidage, and packing characteristics. Attempts to express these relationships have been made by Carmen & Kozeny (Coulson & Richardson, 1978) and Wen & Yu (1966) (Equations 12 & 13).

$$V_{mf} = 0.0055 (\epsilon^3 / 1 - \epsilon) d_m^2 (\rho_m - \rho_f) g / \mu \quad (12)$$

$$V_{mf} = (R_{mf} \mu) / d_m \rho_f \quad (13)$$

Where: V_{mf} = minimum fluidisation velocity (m/s)

$$R_{mf} = (33.7^2 + 0.0408 Ga)^{0.5} - 33.7$$

$$Ga = [d_m^3 \rho_f (\rho_m - \rho_f) g / \mu^2]$$

Altering the media size, shape and voidage to lower the V_{mf} will decrease the filtration efficiency of the BAF. However, lowering the density of the medium decreases the V_{mf} without effecting substrate removal.

2B.3.3.2 Resistance to attrition

Replacement of media is a costly and complicated procedure. Therefore, BAF media should show minimal attrition in response to either chemical or mechanical attacks. Natural materials exhibit up to 6% weight loss after 100 hours backwashing (Kent *et al.*, 1996) whereas plastic media shows negligible attrition.

The interrelationships between all the BAF media properties and each process characteristics have been summarised in Figures 2.6 and 2.7.

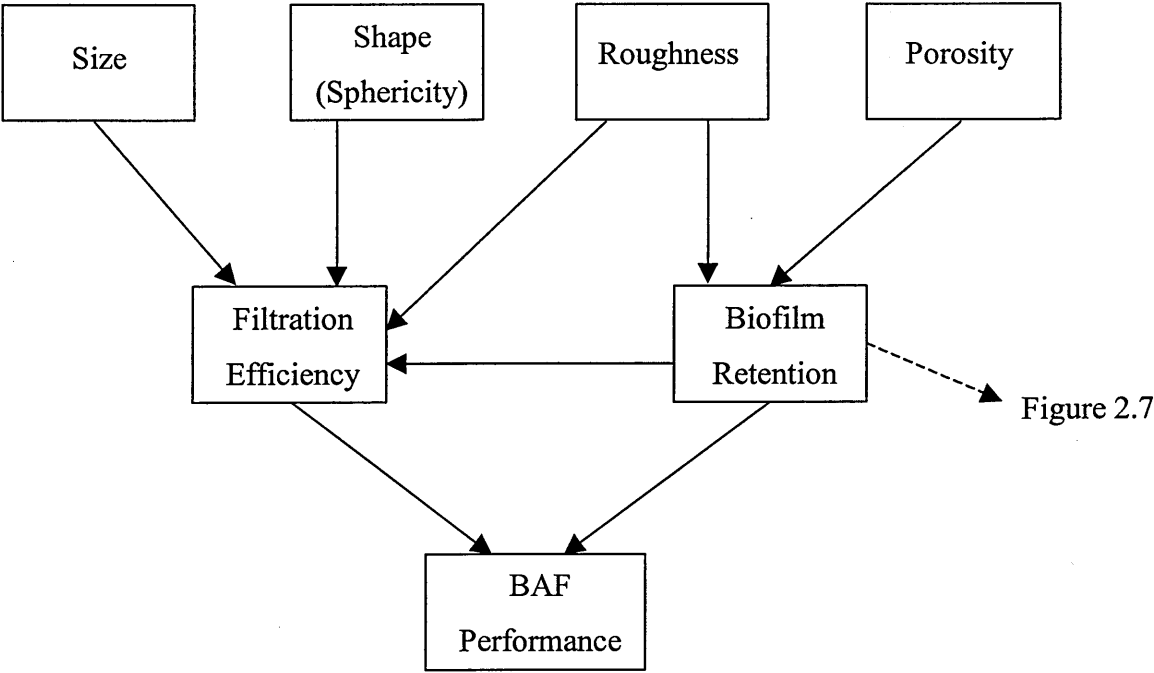


Figure 2.6 Inter- relationships between BAF media properties and reactor performance.

Figure 2.6

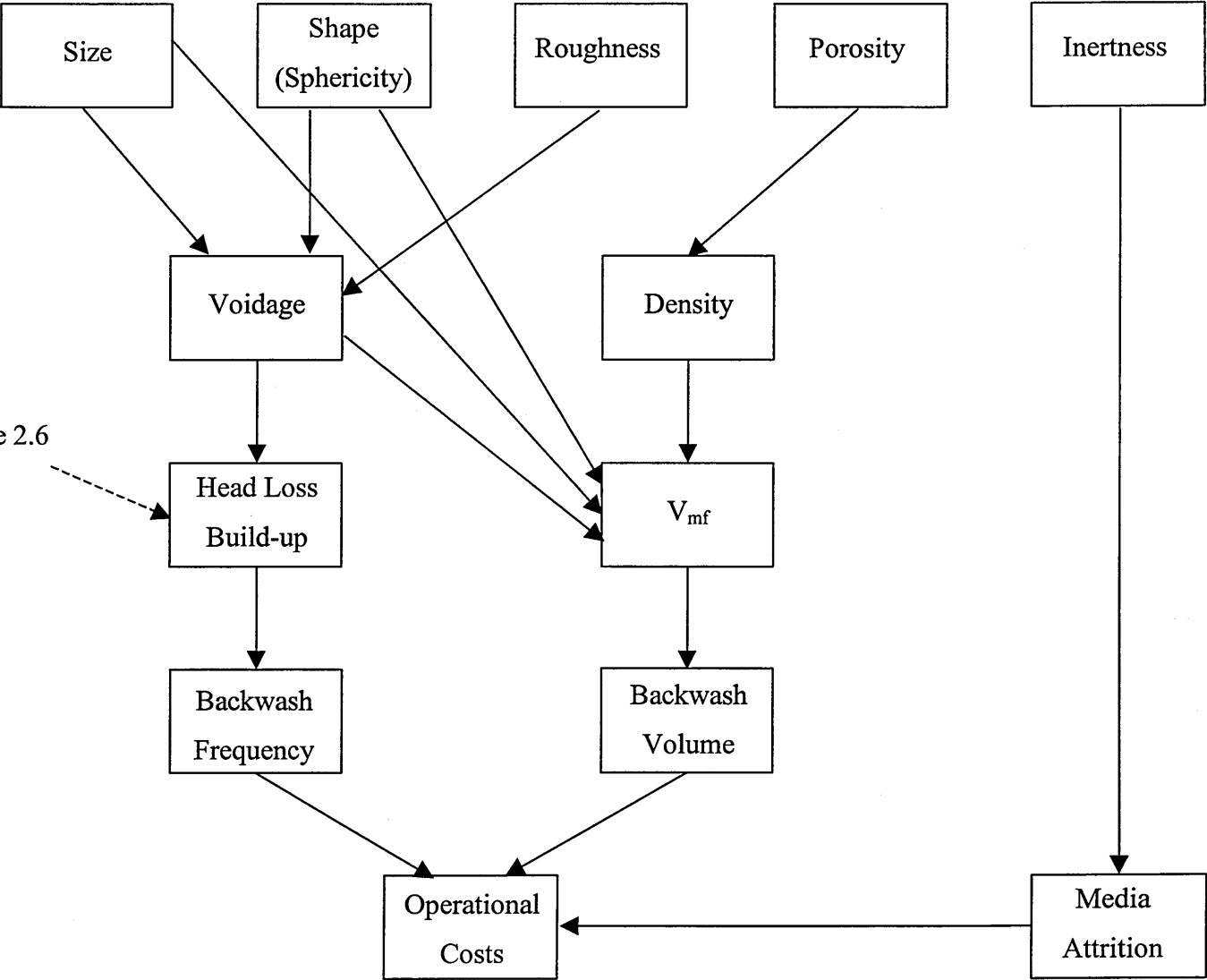


Figure 2.7 Inter-relationships between BAF media properties and operating costs.

2B.4 CONCLUSIONS OF PART B

BAF media can be optimised in terms of:

1. Surface roughness
2. Surface chemistry
3. Pore sizes
4. Inertness
5. Density
6. Shape
7. Size
8. Voidage
9. Hardness

Increasing the surface roughness and the material inertness, whilst selecting the optimal surface chemistry and pore sizes of a medium, will increase its biofilm retention properties. Meanwhile, lowering media density, optimising media shape, size and voidage, and increasing its resistance to attrition will reduce the operating costs of the BAF process.

CHAPTER 3

AIMS AND OBJECTIVES

The overall objective of the research was to develop a novel foamed clay material that could be used as a medium in BAFs and improve the reactor performance. To achieve this objective, aims 1 to 4 were identified. As the research proceeded additional aims, 5 to 7, were introduced. These focused on improving the ease with which potential BAF media could be characterised and assessed by future researchers.

1. Production of a foamed clay medium with the properties to ensure it qualified as a potential BAF medium.
2. Identification of the wastewater treatment industry's required improvements to the BAF process.
3. Development of the properties of the foamed clay media to achieve the process improvements desired by the industry.
4. Comparison of the performance of the foamed clay media with that of a frequently used BAF medium, in pilot scale BAFs.
5. Assessment of the properties required in a BAF medium and how they should be measured.
6. Evaluation of the environmental scanning electron microscope (ESEM) as a tool to observe biofilm structure and development on BAF media.
7. Assessment of the currently used empirical BAF models to identify which has the most potential to produce a meaningful media constant.

CHAPTER 4:

LABORATORY TESTING OF POTENTIAL BAF MEDIA

ABSTRACT

The potential application of two foamed clay materials (StarLight C and StarLight K) as media in biological aerated filters (BAFs) was assessed. The results from biological tests were compared to those for an expanded clay BAF medium (medium X). Meanwhile, the physical properties were compared to the standards for filter media.

StarLight K and medium X exhibited equivalent biofilm retention properties, whilst StarLight C showed slightly inferior biofilm retention properties. StarLight K incurred a 25% weight loss after attrition testing, which is not acceptable for filter media. Meanwhile, since its physical properties reach the required levels for filter media, StarLight C may have the potential to be used as a BAF media.

The methods used to measure biofilm growth were assessed and it was concluded that large samples were necessary to give representative results. It was also noted that the physical properties required by filter media might differ from those required by BAF media.

4.1 INTRODUCTION

The medium is one of the most influential parameters on the efficiency of biological aerated filters (BAFs). Materials used as BAF media include sand, pumice, shale, porous stone, clay, slag and various plastics. Many of the commercially available carriers are expensive, have annual attrition rates greater than 10% and, due to their high densities, use large amounts of energy during backwashing (Heitkamp *et al.*, 1993).

The clay foaming process allows considerable control over the properties of the final product. Therefore, it has been proposed that a foamed clay could be produced which offered improvements over the currently available BAF media. However, before the possible advantages of foamed clay are investigated, it must be proven that the materials qualify as potential BAF media. In this study the physical and biological properties of two foamed clays have been evaluated. The first medium, StarLight K, is made from standard ball clay. Meanwhile, StarLight C is composed of china clay, and was developed specifically in response to doubts about the durability of StarLight K.

The British Effluent and Water Association (BEWA) standards for filter media (1993) have been used to assess the suitability of materials for use as BAF media (Kent *et al.*, 1996). Standard depth filters and BAFs both require an inert medium with high attrition resistance combined with low fluidisation velocities, to reduce the long and short term costs of the backwashing procedure. Therefore, if the StarLight materials fail to reach the minimum standards for a filter medium, it is likely they will also be unsuitable for BAFs.

A BAF medium has a biological role and must be suitable as a biofilm support. There are no standard tests that can be adapted to verify a medium's biofilm retention properties, and the ideal properties to encourage biofilm growth are not known. However, asperity has often been shown to increase the rate of bacterial adhesion (Baker, 1984, Heijnen *et al.*, 1992). Improved biofilm development has also been related to the support surface having large enough pores to allow initial microbial multiplication (Messing *et al.*, 1979).

4.2 MATERIALS AND METHODS

4.2.1 Physical Tests

The BEWA standards for filter media (1993) were used to measure the hydraulic size, dirt content, acid solubility, voidage (poured and packed) and density (bulk and material as defined in Appendix 4A). The size grading was completed manually, since machine sieving can cause media abrasion and increase the proportion of fines (Ives, 1990).

The most significant test is the attrition test which requires a column of medium to be backwashed, in a condition known as collapse pulsing for 100 h. Media samples were washed, dried, sieved and weighed before the test and then dried, sieved and weighed afterwards. Both StarLight C and StarLight K were tested with a combined air and water backwash. With an air flow rate of 2 l/min, StarLight C required a 1 l/min liquid flow rate to reach collapse pulsing conditions. A slightly lower liquid flow rate of 0.75 l/min was required for StarLight K. Additionally, StarLight K was tested using a liquid only backwash, with a liquid flow rate of 2.1 l/min.

4.2.2 Biological Tests

4.2.2.1 Operating conditions

There are many different configurations of test reactors which allow the development of biofilms on different surfaces to be measured (Section 2B.2.8.4.3). However, only in packed bed reactors are media prills used as the test surface and the conditions of a BAF represented. The small-scale packed bed reactors used for the biological tests are illustrated in Figure 4.1, and the operating conditions are summarised in Table 4.1. Variable rate peristaltic pumps (302S-220, Watson Marlow, UK) controlled the influent flow rates. The air flow rates were controlled by glass tube flow meters with a 1-5 l/min capacity (CT Platon Ltd, Basingstoke, UK).

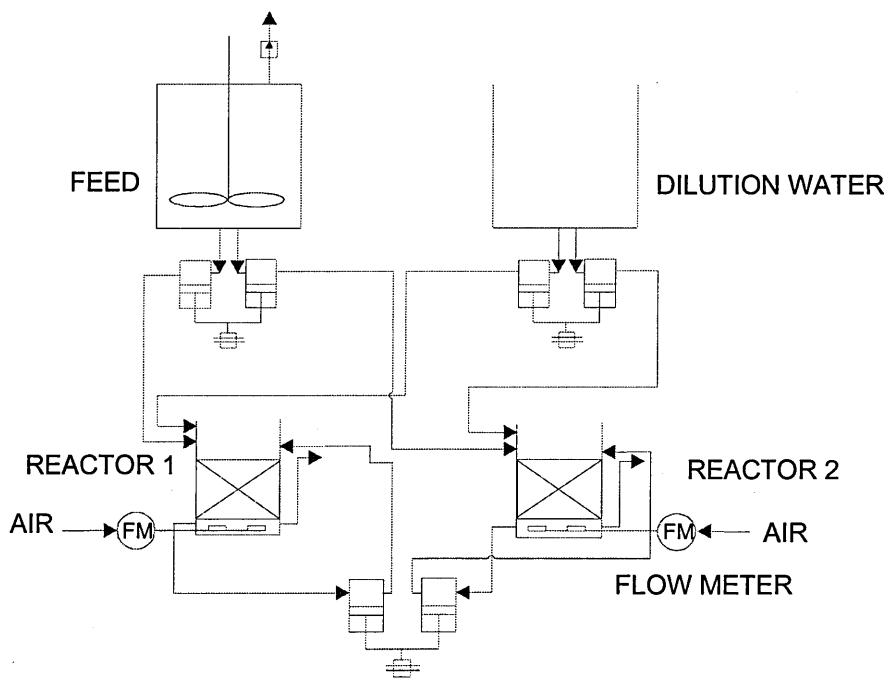


Figure 4.1: Schematic of small-scale reactors

Table 4.1: Operating parameters of the small-scale packed bed reactors

Parameter	Value
Surface Area	0.0176 m ²
Active Volume	0.0016 m ³
Influent flow rate	1.6 l/h
Hydraulic retention time (approx.)	1 h
Re-cycle flow rate	16 l/h
Linear velocity	2.5 m/h
Influent COD concentration	235 mg/l
Organic loading rate	5.6 kg/m ³ /d
Air flow rate	1.5 l/h

The influent was synthetic sewage (Sterritt & Lester, 1981) and represented a weak settled sewage which contains approximately 250 mg COD/l (Metcalf & Eddy, 1991). The reactors were filled with synthetic sewage and aerated for three days, to allow a bacterial culture to develop.

Typically, BAFs have organic loading rates of up to 10 kg COD/m³/d (Bacquet *et al.*, 1991, Dillon & Thomas, 1989, Rogalla *et al.*, 1990). Meanwhile, reported linear velocities are between 1.8 and 4 m/h (Rogalla *et al.*, 1990). Biofilm formation is promoted when the hydraulic retention time (HRT) of the reactor is less than the reciprocal of the maximum growth rate of suspended biomass (Tijhuis *et al.*, 1994). These conditions are found in BAFs where the HRTs range between 0.4 (Rogalla & Bourbigot, 1990) and 1.3 hours (Wheale & Cooper-Smith, 1995).

Since BAFs operate under a regime close to plug flow (Mann *et al.*, 1995), small samples of the medium will not represent the whole reactor. However, it is possible to operate a reactor that would normally be plug flow as a continuous flow stirred tank reactor (CFSTR) by re-cycling part of the effluent flow (Characklis & Marshall, 1990). A re-cycle rate ten times the influent rate was selected to ensure the linear velocity through the reactor was within the range found in BAFs. The re-cycle pump mixed the liquid in the two reactors, so the conditions in each reactor remained identical throughout the experimental period.

Two experiments were undertaken using these small-scale reactors. The first compared the performance of StarLight K to a standard expanded clay medium (medium X). Meanwhile, StarLight K and StarLight C were compared in the second.

4.2.2.2 Monitoring performance in experiment 1

Four media samples, each representing less than 0.05% of the bed, were removed from the reactors at least three times a week. Four analytical techniques for measuring the biofilm growth on these samples were assessed. Firstly, volatile solids were measured as an indication of the mass of accumulated biofilm (Gjaleltema *et al.*, 1994). Secondly, the volume of biofilm present was measured using volumetric displacement (Characklis *et al.*, 1982). Finally, the soluble components of the biofilm were extracted in 15 ml of either 0.1 M NaOH (proteins), or 1 M sulphuric acid (sugars) by boiling the sample in a water bath for 15 minutes (Lazarova *et al.*, 1994). The concentrations of protein (Lowry *et al.*, 1951) and sugar (Dubois *et al.*, 1956) in the solution were then assessed.

At the end of the experiment, when the reactors blocked, larger samples representing approximately 0.1% of the bed, were analysed using the same techniques. Additionally, the viability of the biofilm was tested by measuring the activity of the electron transport system (ETS) using the INT-formazan assay (Blenkinsopp & Lock, 1990).

4.2.2.3 Monitoring performance in experiment 2

After experiment 1, an attempt was made to assess biofilm development indirectly. Effluent samples were taken every day during operation and analysed for soluble COD (sCOD) using the Hach test-n-tube method (Camlab, Nuffield, UK).

As in experiment 1, when the reactors blocked, samples of the media were analysed for volatile solids, protein and sugar levels and ETS activity. The sample sizes were maximised so that each represented between 1 and 5 % of the bed.

4.3 RESULTS

4.3.1 Physical Tests

The results from the physical tests are summarised in Table 4.2. StarLight K has a higher mean hydraulic size than StarLight C, even though both materials use the same extrusion method. StarLight K is fired at 1050°C whereas StarLight C is fired at 1400°C. As the temperature of calcination increases, the degree of shrinkage of the prills also increases.

Table 4.2: Physical properties of StarLight K and StarLight C.

Parameter	StarLight K	StarLight C
Mean hydraulic size (mm)	2.6	2.1
Material density (kg/m ³)	1200	2500
Bulk density (kg/m ³)	250	420
Dirt Content (%)	3.7	0.3
Acid solubility (%)	0.5	0.015
Poured Voidage (%)	44	44
Packed Voidage (%)	39	40
Weight loss after attrition testing (%)	25* 3+	3 [@]

[@] with a water flow rate of 1 l/min, and an air flow rate of 2 l/min.

* with a water flow rate of 0.75 l/min, and an air flow rate of 2 l/min.

+ with a water flow rate of 2.1 l/min, and no air flow.

The media density influences the required backwash water velocity, and so can affect the proportion of cleaned effluent utilised during backwashing. StarLight C has a higher bulk density and material density than StarLight K. However, the BEWA standards have no methods to measure the overall particle density for porous materials or the soaked density of the particles. These would be more relevant tests for BAF media.

The “dirt” removed during the dirt content test was predominantly dust. StarLight C showed weight loss far below the standards maximum weight loss of 1%. Meanwhile, since there were high levels of dust associated with StarLight K, it may require a dust removal stage in its production.

The acid solubility test measures the chemical inertness of the materials. Both calcined clays showed very little weight loss, well below the maximum levels of 5%. Therefore, they are both suitable as BAF media in this respect.

Voidage has a large influence on the reactor operation, and is linked to the size and shape of the medium. Measures of voidage vary by up to 5% depending on the degree of compaction applied to the bed (Lang *et al.*, 1993). Both media had approximately 40% voids once the beds were packed down.

When a combined water and air backwash was applied, StarLight K exhibited a 25% weight loss during attrition testing. Meanwhile, StarLight C showed only a 3% decrease in weight. Ives (1990) stated that for a filter medium a weight loss of less than 1% was satisfactory; 1-3% doubtful; 3-5% undesirable and more than 5% unsatisfactory. However, air scouring was not used by Ives, and StarLight K showed only 3% weight loss when a liquid only backwash was applied.

The media size distributions before and after attrition testing are illustrated in Figures 4.2, 4.3 and 4.4. There were no significant changes for StarLight C, or, for StarLight K with a water only backwash. However, when air was applied, the particles of StarLight K broke up and the proportion of smaller sized particles increased.

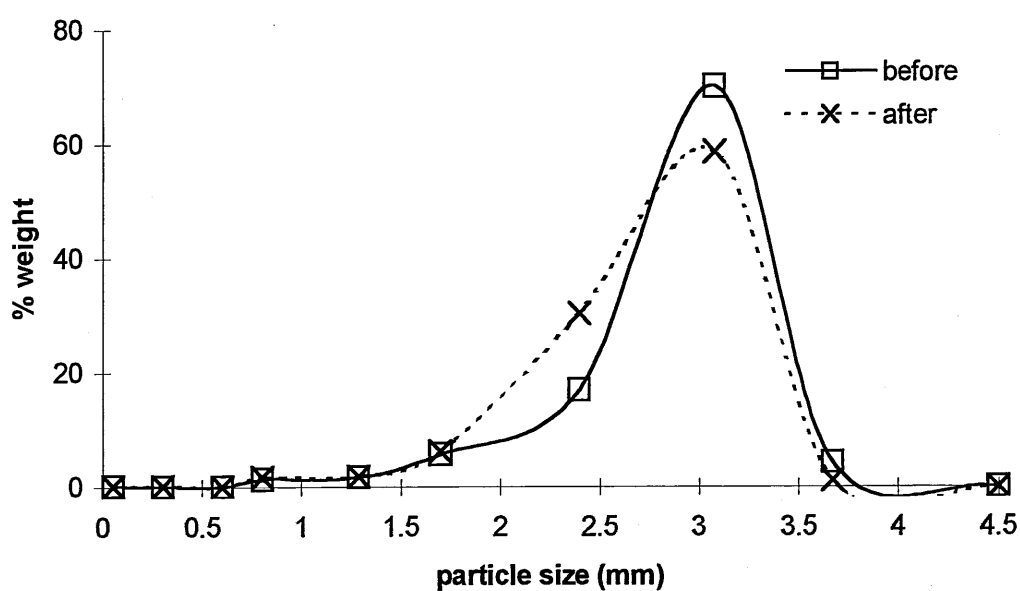


Figure 4.2: StarLight K size distribution before and after air & water attrition test

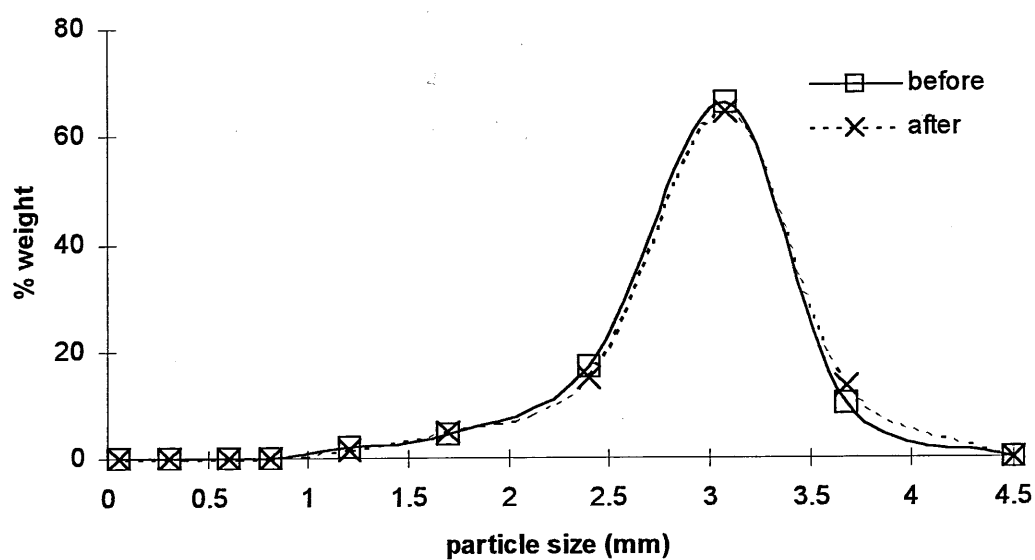


Figure 4.3: StarLight K size distribution before and after water only attrition test

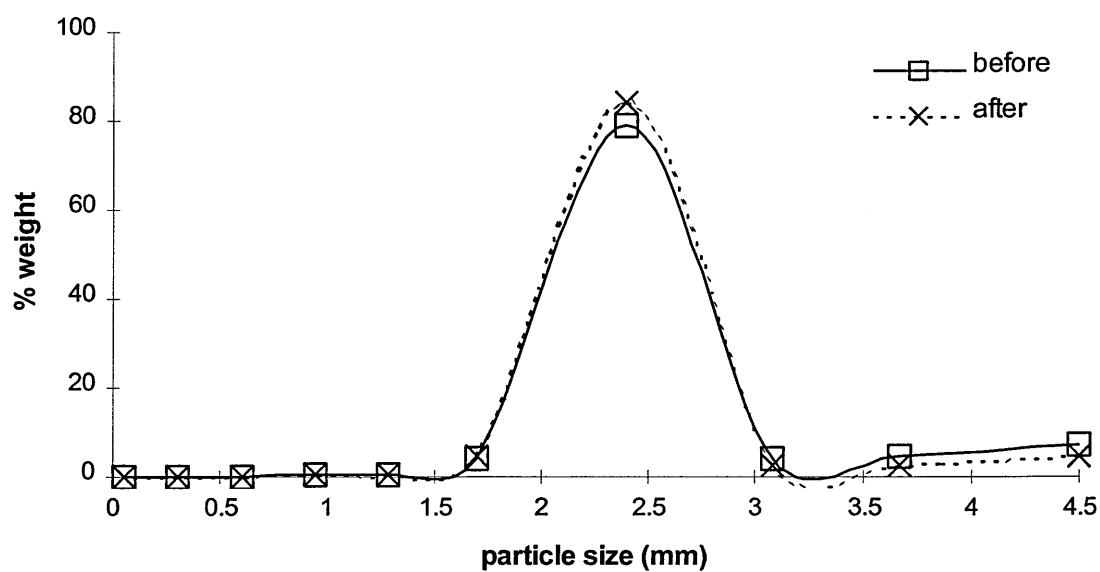


Figure 4.4: StarLight C size distribution before and after air & water attrition test

4.3.2 Biological Tests

The results from experiments 1 and 2 must be considered separately, since without temperature control and an aseptic environment, identical conditions are not ensured.

4.3.2.1 Experiment 1

Biofilm growth was observed on both media types within two days of start-up, and after two weeks the reactors were blocked.

The volumetric displacement method to measure biofilm volume was found to be unsuitable for this application. Both StarLight and medium X are highly porous materials, and it was not possible to ensure all the internal pores of the media prills were filled. Other analytical methods were only useful if the sample of medium represented the whole bed. When the small samples (less than 0.05% of bed) taken during operation were analysed, the results appeared to be unreliable, as shown in Figure 4.5. The larger samples of media analysed at the end of the experiment showed a higher degree of reliability and consistency than the smaller samples (Table 4.3).

Table 4.3: Comparison of Medium X and StarLight K at the end of experiment 1

Parameter (mean per l of reactor)	Samples	Medium X	Starlight K
Volatile solids (mg)	4	2900 ± 1100	3400 ± 700
Protein content (µg/ml)	2*	1800 ± 200	1400 ± 200
Sugar content (µg/ml)	2*	11500 ± 500	5400 ± 400
INT-formazan (µg/ml)	2*	600 ± 200	200 ± 100

* each chemical analysis was triplicated

The results indicate that StarLight K had equivalent biofilm retention properties to medium X. There were only small differences in the levels of media associated volatile solids, and the concentrations of the extracted protein solutions for the two media. However, there were greater differences in the concentrations of extracted sugars and INT-formazan.

4.3.2.2 Experiment 2

Over 80% of the sCOD in the influent was removed between days 2 and 5. Subsequently, the sCOD in the effluents started to rise again probably due to leaching from sloughed pieces of biofilm (Figure 4.6).

The results from analysing media samples at the end of the experiment are shown in Table 4.4. StarLight C appears to have inferior biofilm retention properties to StarLight K. The volatile solids associated with the StarLight C samples were over 50% lower than those for StarLight K. However, there was still good biofilm growth on the StarLight C medium as shown by the high levels of extracted protein. As in experiment 1, the biofilm on StarLight K had a low sugar content and viability.

Table 4.4: Comparison of StarLight C and StarLight K at the end of experiment 2

Parameter (mean per l of reactor)	Samples	StarLight C	Starlight K
Volatile solids (mg)	5	1350 ± 150	3600 ± 700
Protein content (µg/ml)	4*	850 ± 100	900 ± 150
Sugar content (µg/ml)	2*	10300 ± 1200	5600 ± 200
INT-formazan (µg/ml)	2*	200 ± 150	50 ± 6

* each chemical analysis was triplicated

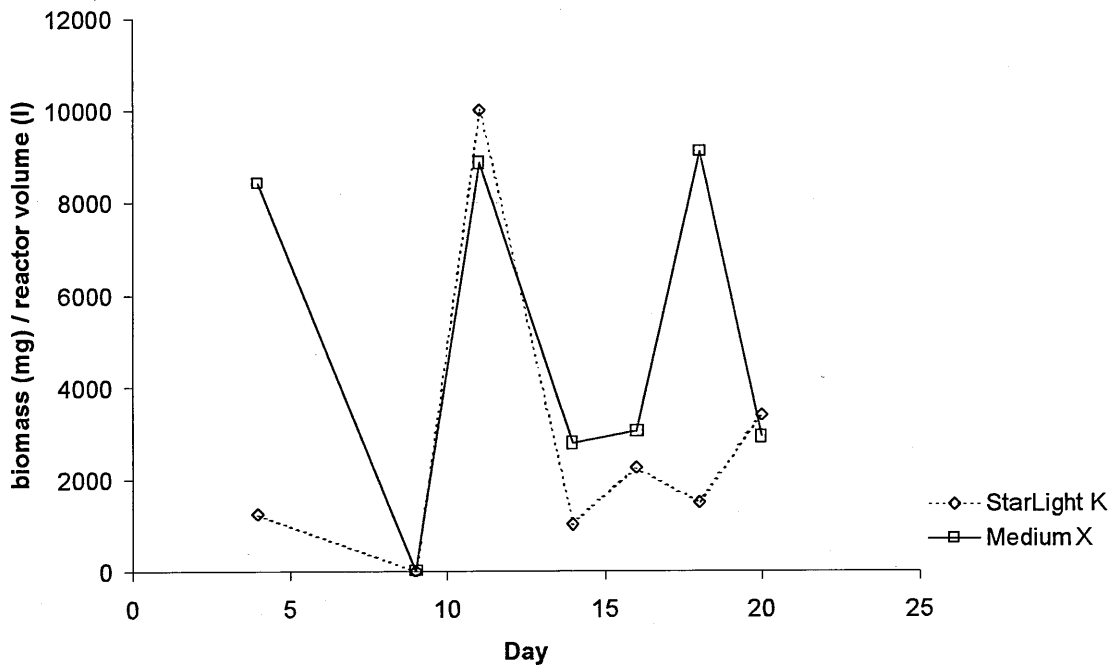


Figure 4.5: Attached biomass per reactor volume during experiment 1

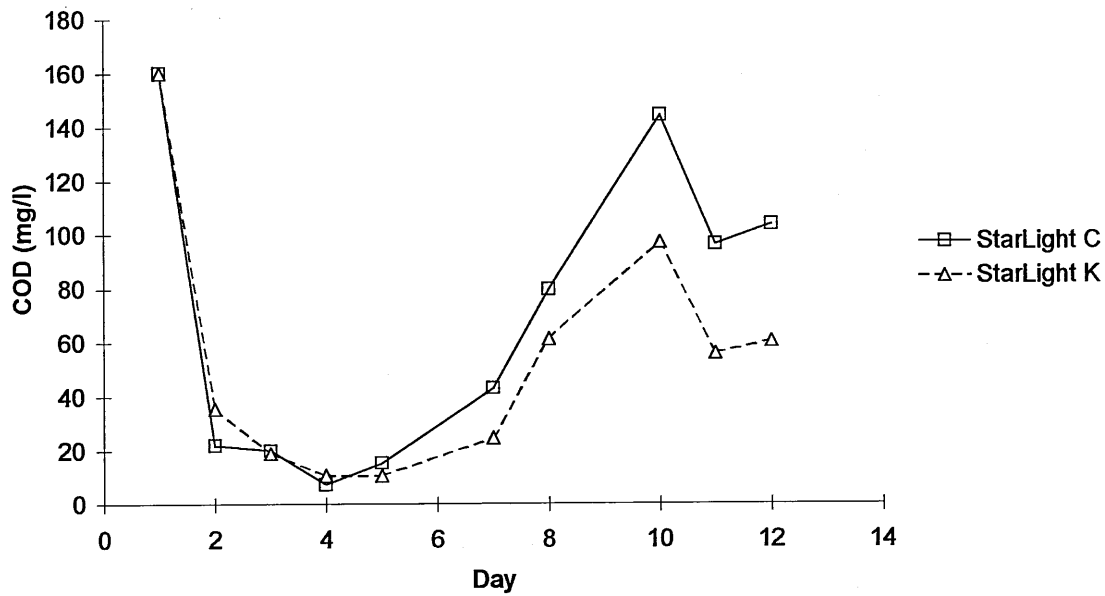


Figure 4.6: COD levels in the effluents of reactors during experiment 2

4.4 DISCUSSION

The results from the tests for dirt, acid solubility and attrition, showed that StarLight C meets the qualification levels for a water filter medium. Therefore, it is assumed that its physical characteristics are suitable for a BAF medium. Meanwhile, StarLight K showed high levels of dirt (dust) and unacceptable levels of weight loss after a combined air and water attrition test. Previous researchers have found that BAF media tends to be prone to attrition and have suggested the standard should be revised (Kent *et al.*, 1996). Indeed, it is possible that the attrition rate of a BAF medium will be lowered when the biofilm grows and encapsulates the material. However, a BAF medium is backwashed every 24 to 48 hours and is continually abraded by the air supply. Since resistance to attrition is imperative and StarLight K showed very high weight loss, it would be prudent to consider it unsuitable as a BAF medium.

When considering the size, density and voidage of BAF media there are no determined minimum or maximum levels. However, the values obtained for the StarLight media can be compared to the published data of Kent *et al.* (1996) for seven other BAF media. Both StarLight media showed typical voidage levels. The size of BAF media ranges between 1.8 and 6.2 mm. The optimal size depends on the desired filtrate quality, the influent composition, the flow rate, the length of filter run and the backwashing conditions (Ives, 1990). Since both of the StarLight media are at the lower end of the range, they will probably be most suitable when a high filtrate quality is required.

Both StarLight C and K have a low density compared to many of the currently available BAF media. As a result they are likely to fluidise at a lower liquid velocity (low V_{mf}). The most efficient backwash conditions are a combination of air scouring with liquid flows between 40 and 60% of the medium's V_{mf} (Amirtharajah, 1993). Thus, the volume of effluent used during the backwash procedure is directly related to the density of the medium. The efficiency of the BAF process could therefore be improved by using a medium with a lower density.

The surface associated volatile matter has been shown to gradually increase during start-up in airlift suspension reactors (Tijhuis *et al.*, 1994), trickling filters (Thorn *et al.*, 1996), and anaerobic fluidised beds (Stronach *et al.*, 1986). The same pattern is shown when either the protein (Shreve *et al.*, 1991) or the polysaccharide (Wanda *et al.*, 1990) content is used to monitor biofilm development. Sampling error is the likely cause of the absence of the normal biofilm growth pattern in experiment 1. It is very difficult to take small representative samples from a packed bed reactor, even when it is run as a CFSTR. This problem can be overcome by taking large samples (30% of the weight of material) and compensating by replacing the medium with that from a second reactor running in parallel (Nouvion *et al.*, 1987). Alternatively, measuring the substrate in the effluent proved to be reliable in experiment 2 and, has been shown to relate to the initial biofilm development (Hoehn & Ray, 1973).

The dry solids of bacteria consist of 50% protein, 15% cell wall components, 10-20% RNA, 10% lipids and 3% DNA (Lazarova & Manem, 1995). However, micro-organisms only make up between 2 and 27% of biofilms, with the remainder being composed of extracellular polymeric substances (EPS) and space (Lawrence *et al.*, 1991). Carbohydrates are usually the main component, making up 28 to 45%, of the EPS for a diverse collection of marine and freshwater isolates. Meanwhile, only 0 to 22% of EPS was composed of proteins (Ford *et al.*, 1991).

After experiment 1, there were only small differences in the media-associated volatile matter or the extracted protein concentration from medium X or StarLight K. This suggests a similar number of cells were attached to both media. However, the lower sugar content and viability levels of the biofilm attached to StarLight K, suggests the biofilm had reached a transitional stage in its development. As biofilms grow, their thickness and density increase until oxygen becomes limiting in the inner region. There follows a transitional period of instability when the aerobic micro-organisms at the base of the film die, lyse and are consumed by surviving facultative and anaerobic organisms. This causes a decline in film density, which may decrease the availability of carbohydrates (Hoehn & Ray, 1973). Additionally, the proportion of INT active cells decreases and the number of dormant or inert cells increases (Zhang & Bishop, 1994b).

After experiment 2, StarLight C had a lower level of associated volatile solids than StarLight K, indicating that StarLight C may have slightly inferior biofilm retention properties. However, as shown by the extracted protein results there was still good biofilm growth on StarLight C. The extracted sugar content and viability results indicated that once again the biofilm on StarLight K may have entered its transitional phase.

The high attrition rate exhibited by StarLight K may be related to its high biomass retention properties. Rough, porous surfaces are thought to promote biofilm development. However, the particles of a rough surface would be vulnerable to gradual erosion and pores may increase particle fragmentation. Overall the results obtained from the biological tests indicated that both types of StarLight have good biofilm retention properties. Meanwhile, the results from the physical tests indicated only StarLight C was a suitable BAF medium.

4.5 CONCLUSIONS

1. In terms of their biological properties, both StarLight C and StarLight K are suitable for use as BAF media.
2. The physical properties of StarLight C make it a suitable BAF medium.
3. StarLight K exhibits unacceptable levels of attrition and so is not recommended for use as a BAF medium.
4. In small-scale reactors where media samples may not represent the whole bed, measuring substrate levels in the effluent is more reliable than assessing the levels of biomass associated with media samples.
5. An evaluation of the suitability of using the BEWA standards for water filter media to assess BAF media is required. Some additional physical tests such as surface roughness, porosity, particle density and soaked density, would also be useful to assess potential BAF media.

CHAPTER 5

PILOT-SCALE TESTING OF A POTENTIAL BAF MEDIUM

ABSTRACT

The main drawback of using biological aerated filters (BAFs) is that they need periodic backwashing. This procedure uses up to 35% of the treated effluent and consequently lowers the reactor productivity. Additionally, the associated energy requirements increase the operating costs of the process.

StarLight C is a lightweight foamed clay medium with a lower minimum fluidisation velocity (V_{mf}) than comparable expanded clays. This chapter reports the performance of StarLight C and standard expanded clay (medium A) in pilot-scale BAFs treating settled sewage. The BAFs were backwashed every 24 hours at a liquid flow rate corresponding to 40% of the V_{mf} of the media. The BAF using StarLight C required approximately half the volume of backwash water needed by the BAF containing medium A.

The BAFs performed consistently with hydraulic residence times between 58 and 96 minutes. They both achieved an average suspended solids removal of 87% after four days and an average sCOD removal of approximately 75% after three days. Ammonia removal stabilised after 25 days and peaked at 99% removal. When the backwash velocity of the medium A reactor was reduced to that of the StarLight C reactor there was incomplete solids removal during backwashing.

5.1 INTRODUCTION

BAFs (biological aerated filters) are three phase reactors consisting of submerged beds of media with an applied air source. Their main advantage over traditional secondary wastewater treatment processes, such as activated sludge reactors and trickling filters, is their reduced plant footprint. This property is derived from the BAF medium, which retains biomass both on its surface and in the voids between the media grains, allowing BAFs to maintain a biomass concentration about four to five times that found in suspended growth reactors (Smith *et al.*, 1990). True BAFs, as defined by Stephenson (1997), contain granular rather than structured media. The granular medium acts like a depth filter medium, and removes solids from suspension. Thus, the need for a final clarification stage is negated and the reactor footprint is minimised further.

There are both upflow and downflow BAFs and a full analysis of the advantages and disadvantages of each configuration can be found in section 2.5. The most significant advantage of using upflow BAFs is their decreased backwash water requirement. Backwash water requirements of less than 1% of the treated effluent have been reported for upflow BAFs with separate units for biological treatment and suspended solids removal (Meany, 1994). However, 6.7% is a more realistic figure for a single unit upflow BAF (Carrand *et al.*, 1990). By comparison, downflow reactors use between 5 and 35% of the effluent to backwash the filter (Smith *et al.*, 1990, Canler & Perret, 1994, Dillon & Thomas, 1989, Stensel *et al.*, 1988). Backwash water is usually treated effluent so lowering its utilisation increases productivity as well as decreasing operating costs. Reducing the backwash water requirements of all BAFs, but particularly those with a downflow configuration, is therefore a research priority.

The percentage of treated effluent used for backwashing can be reduced by extending the filter run time or decreasing the volume of water required at each backwash. The volume of backwash water necessary to clean a downflow BAF filter is related to the minimum fluidisation velocity (V_{mf}) for the medium. The most efficient backwash conditions occur when the bed is collapse-pulsing (Amirtharajah, 1993). The grains then exhibit velocities as high as those measured in a fully fluidised bed. The

collapse-pulsing condition can be reached by applying an air scour together with a sub-fluidisation water wash at a velocity between 40-60% of the V_{mf} . The V_{mf} of a medium is related to its grain size and density, and the bed voidage (Coulson & Richardson, 1978). The density of a medium can be altered to reduce the V_{mf} without effecting the substrate removal characteristics of the filter.

The aim of this study was to compare the start-up and operational performance of two pilot BAFs in terms of suspended solids, oxygen demand and ammonia removal. One BAF contained standard expanded clay BAF medium (medium A) and the other a novel lightweight foamed clay (StarLight C).

5.2 MATERIALS AND METHODS

5.2.1 Pilot Plant

Two identical pilot-scale BAFs were run in parallel over a nine month period. Settled sewage was supplied in a downflow configuration under the conditions summarised in Table 5.1.

Table 5.1: Operating parameters of BAFs with feed flows of 0.3, 0.4 and 0.5 l/min.

Feed flow rate (l/min) Parameter	0.3 (Day 0-55)	0.4 (Day 56-146)	0.5 (Day 147-222)
Mean total COD load* (kg/m ³ /d)	4.5	6.3	8.5
Mean susp. solids load* (kg/m ³ /d)	1.9	2.1	3.5
Mean ammonia load* (kg/m ³ /d)	0.4	0.5	0.7
Hydraulic residence time (minutes)	96	72	58
Air flow rate (l/min)	3	4	5
Liquid velocity (m/h)	1.4	1.9	2.4

* based on original active bed volume

The reactors (Figure 5.1) were PVC cylinders with a diameter of 0.2 m and a height of 3.15 m. They contained 0.3 m of gravel beneath 2 m of media and were supplied with air via three 5 cm nozzles at the base (Degremont, UK). During operation, the flow of compressed air at 1 atmosphere was controlled with a Platon glass tube VA flowmeter with a 1-10 l/min capacity. An identical flowmeter with a 2-25 l/min capacity controlled the backwash air scour (CT Platon Ltd, Basingstoke, UK). The influent flow rate was controlled using a variable rate peristaltic pump (302S-220, Watson Marlow, UK). The flow from the backwash pump (CR2/20, Grundfoss, UK) was controlled with ball valves (Durapipe, UK).

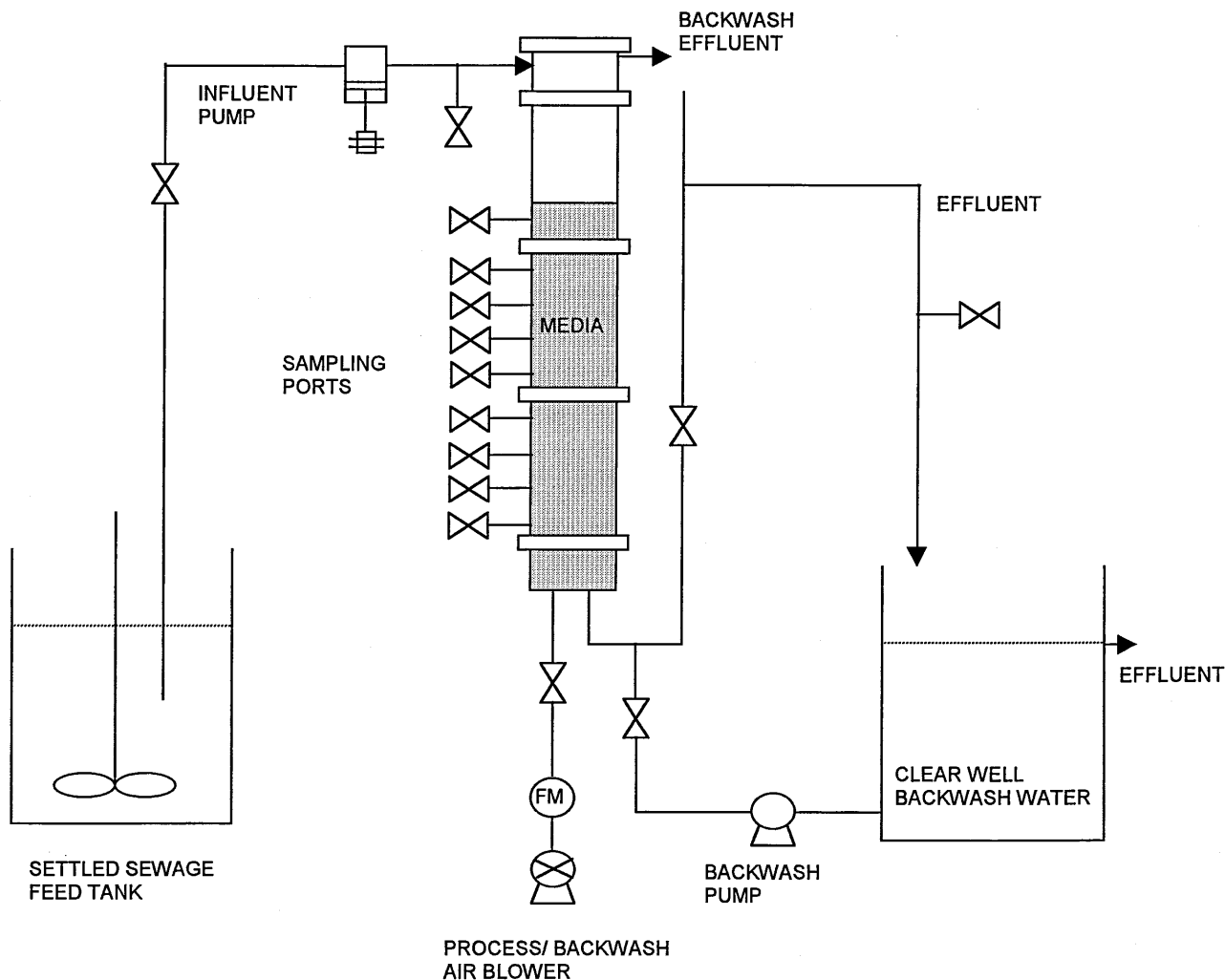


Figure 5.1: Plan of pilot-scale BAFs

Table 5.2: Particle density (defined in Appendix 4A) of BAF media

Material	Density (kg/m ³)	Reference
StarLight C	800-900	-
Medium A	1400	-
Expanded fire clay grog	1720	Kent <i>et al.</i> , 1996
Expanded clay	1350-1600	Kent <i>et al.</i> , 1996, Koutsakos <i>et al.</i> , 1992
Expanded schist	1400-2000	Bacquet <i>et al.</i> , 1991
Expanded shale	1680-1900	Kent <i>et al.</i> , 1996, Harris <i>et al.</i> , 1996
Pulverised fuel ash	1950	Kent <i>et al.</i> , 1996
Pozzolana	1500	Pujol <i>et al.</i> , 1992
Sand	2500	Mendoza-Espinosa & Stephenson, 1998

One reactor contained StarLight C, a lightweight foamed clay and the other a standard expanded clay (medium A) used in BAFs. A complete physical analysis of these two media can be found in Chapter 6. Measuring the experimental V_{mf} and the sunken density of media can help set suitable reactor backwash velocities. Despite this, the parameters have rarely been quoted in previously published data. Therefore, the two media used in this study have been compared to the media used previously in downflow BAFs in terms of particle density (Table 5.2).

5.2.2 Start-up

Settled sewage was re-circulated at the first operating flow rate (0.3 l/min) for 48 hours. This was to allow as many species as possible to colonise the media without the risk of suspended solids blocking the reactors. The nominal flow rate was used immediately to promote the growth of a stable biofilm (Mann *et al.*, 1999). Normal operation followed this inoculation period except that backwashing was performed only every 48 hours for the proceeding four days. Subsequently, the reactors were backwashed every 24 hours.

5.2.3 Monitoring Performance

Influent and effluent samples were taken three times a week and analysed using standard methods (APHA, 1992) for suspended solids, soluble and total chemical oxygen demand (sCOD and tCOD), ammonia (Hach test-n-tube method, Camlab, UK) temperature, pH (Russel, UK) and dissolved oxygen (Jenway, UK). Soluble and total biological oxygen demands (sBOD and tBOD) were also measured weekly, using the five day BOD test, to establish the BOD:COD ratio for the influent and the effluents. Two tailed, paired Student's t-tests were performed to compare the substrate removal results for the two reactors. The observed significance levels, or p-values, for the t-tests were calculated with a null hypothesis that the two reactors were performing equally well. The null hypothesis was rejected if the p-value for a specific substrate comparison was less than the required significance level (α), which was set at 0.05 (Mendenhall & Sincich, 1992). Pressure transducers (VEGA, Germany) were attached to the BAFs to monitor the head build up over filter cycles.

5.2.4 Backwash Procedure

The reactors were backwashed manually with treated effluent every 24 hours. The procedure took thirty minutes and consisted of three stages. The first stage was five minutes without influent and air flow reduced to just 1 l/min. Next, an air spurge at 20 l/min was applied for five minutes. The third stage consisted of ten minutes combined air scour at 20 l/min, and liquid flow at 40 % of the experimental V_{mf} of the medium. This was 10 l/min for StarLight C and 18 l/min for medium A. Finally, stage two and then stage one were repeated before returning to normal operation. The backwash velocity of the medium A column was reduced to 11 l/min for three days to justify setting the backwash velocity from the media experimental V_{mf} .

A backwash profile was conducted to measure the tCOD and suspended solids concentration of the backwash effluent after two, four and eight minutes of backwashing. Additionally, the backwash fluid was collected once a week, mixed, and samples taken for suspended solids analysis. Any washed out media was also collected, dried and weighed.

5.3 RESULTS

5.3.1 Start-up

Both reactors showed similar start-up characteristics, with suspended solids removal stabilising by day four (Figure 5.2), and sCOD by day three (Figure 5.3). With influent temperatures ranging from 15 °C to 20 °C, nitrification did not reach stable operation until day 25 (Figure 5.4).

5.3.2 Operating Performance

Since there were no mechanisms to prevent it, throughout the period of operation both reactors lost media during the backwashing process. Independent of the influent flow rate, a mean of 25 ml of medium A was lost per day. At the end of the run over 5 l, equivalent to 7% of the bed, had been lost. Meanwhile, a mean of 55 ml of StarLight C was lost per day until the influent flow rate reached 0.5 l/min, when the mean washout volume rose to 106 ml/d. At the end of the run over 16 l, equivalent to 22% of the bed, had been lost. As the reactor volumes decreased, the substrate loading rates increased. Since the StarLight C reactor incurred more washout, it also received an increasingly higher substrate load than the medium A reactor.

At each flow rate and for both reactors, the mean suspended solids removal was over 84% and the mean effluent concentration was under 20 mg/l (Table 5.3). Over the whole period two tailed, paired t-tests, showed that although there was no significant difference in the % removal results for the two reactors (p-value 0.32), the difference in the load removed by each reactor was significant (p-value <0.01). Therefore, the StarLight C reactor removed an equal mass of solids with a smaller volume.

The mean sCOD removal for both reactors was over 70% at all the applied flow rates (Table 5.3). There was a reasonably stable relationship between COD and BOD. Approximately 40% of the tCOD in the influent, and 15% in the effluent, consisted of BOD. Meanwhile, approximately 40% of the sCOD in the influent, and 10% in the effluent consisted of BOD. The mean effluent concentrations were below 45 mg/l for

sCOD, 70 mg/l for tCOD, 5 mg/l for sBOD and 12 mg/l for tBOD. Again, the reactor using StarLight C removed a significantly greater load of sCOD than the reactor using medium A (p-value <0.01).

Up to 99% ammonia removal was achieved but the results were inconsistent and, for the StarLight C reactor, deteriorated as the flow rate increased. Although there was no significant difference in the ammonia load removal between the two reactors (p-value 0.44), there was a significant difference in the % removal (p-value <0.01). At 0.5 l/min in the StarLight C BAF, only 50% ammonia removal took place and effluent concentrations rose to around 15 mg/l (Table 5.3).

There were no significant decreases in the % removal of suspended solids and sCOD as the influent flow rate increased. Additionally, there was good stability in effluent sCOD concentrations throughout the day as shown in Figure 5.5. The filters were backwashed at hour 11 and there was no prolonged recovery period.

Table 5.3: Summary of the performance by the two reactors throughout the study

Flow rate	Mean $\pm$ standard deviation	Suspended solids		sCOD		Ammonia	
		MA	StL	MA	StL	MA	StL
0.3 (l/min)	Influent load*	1.75 $\pm$	2.00 $\pm$	2.18 $\pm$	2.25 $\pm$	0.39 $\pm$	0.41 $\pm$
	(kg/m ³ /d)	0.93	0.97	0.53	0.54	0.12	0.12
	Load removed*	1.52 $\pm$	1.72 $\pm$	1.66 $\pm$	1.67 $\pm$	0.25 $\pm$	0.27 $\pm$
	(kg/m ³ /d)	0.81	0.92	0.46	0.48	0.12	0.12
	% removed	87 $\pm$ 12	86 $\pm$ 10	76 $\pm$ 12	74 $\pm$ 12	64 $\pm$ 27	66 $\pm$ 26
	Effluent range (mg/l)	4 - 22	5 - 24	14 - 48	14 - 54	1 - 19	5 - 24
0.4 (l/min)	Influent load*	1.97 $\pm$	2.5 $\pm$	3.37 $\pm$	3.81 $\pm$	0.54 $\pm$	0.62 $\pm$
	(kg/m ³ /d)	0.59	0.74	0.72	0.83	0.1	0.12
	Load removed*	1.67 $\pm$	2.1 $\pm$	2.49 $\pm$	2.74 $\pm$	0.42 $\pm$	0.41 $\pm$
	(kg/m ³ /d)	0.59	0.61	0.61	0.69	0.08	0.11
	% removed	85 $\pm$ 7	84 $\pm$ 7	74 $\pm$ 6	72 $\pm$ 6	78 $\pm$ 8	66 $\pm$ 21
	Effluent range (mg/l)	4 - 36	7 - 42	14 - 56	14 - 73	1 - 10	8 - 42
0.5 (l/min)	Influent load*	3.58 $\pm$	6.27 $\pm$	3.83 $\pm$	5.88 $\pm$	0.75 $\pm$	1.13 $\pm$
	(kg/m ³ /d)	1.21	2.13	0.85	1.05	0.24	0.28
	Load removed*	3.22 $\pm$	5.7 $\pm$	2.72 $\pm$	4.41 $\pm$	0.55 $\pm$	0.59 $\pm$
	(kg/m ³ /d)	1.25	2.22	0.62	0.91	0.17	0.18
	% removed	90 $\pm$ 5	91 $\pm$ 5	71 $\pm$ 18	75 $\pm$ 7	75 $\pm$ 12	52 $\pm$ 21
	Effluent range (mg/l)	5 - 23	1 - 29	10 - 54	16 - 52	1 - 13	1 - 27

* all the loading rates and loads removed have been calculated using the active volume of the reactors.

MA represents the BAF containing medium A and StL represents the BAF containing StarLight C.

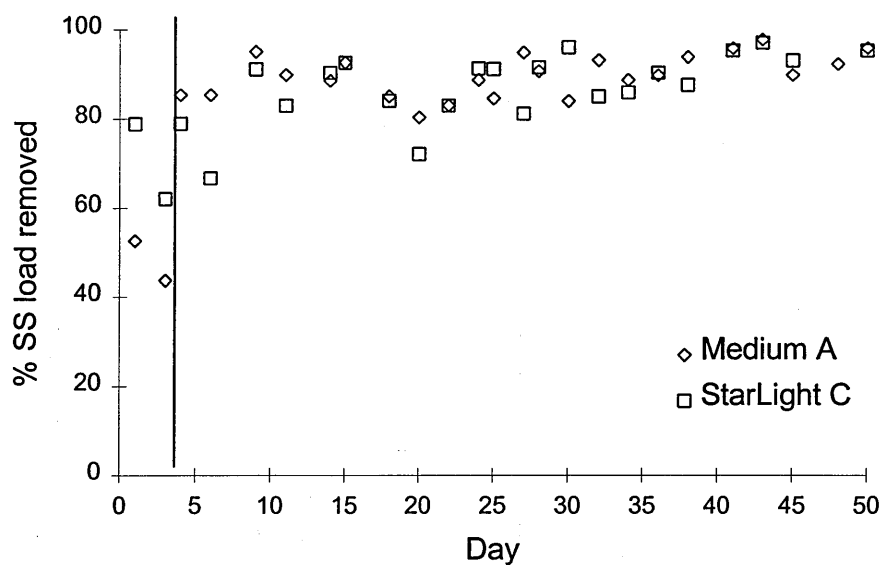


Figure 5.2: Start-up of suspended solids removal (line shows stable operation reached by day 4)

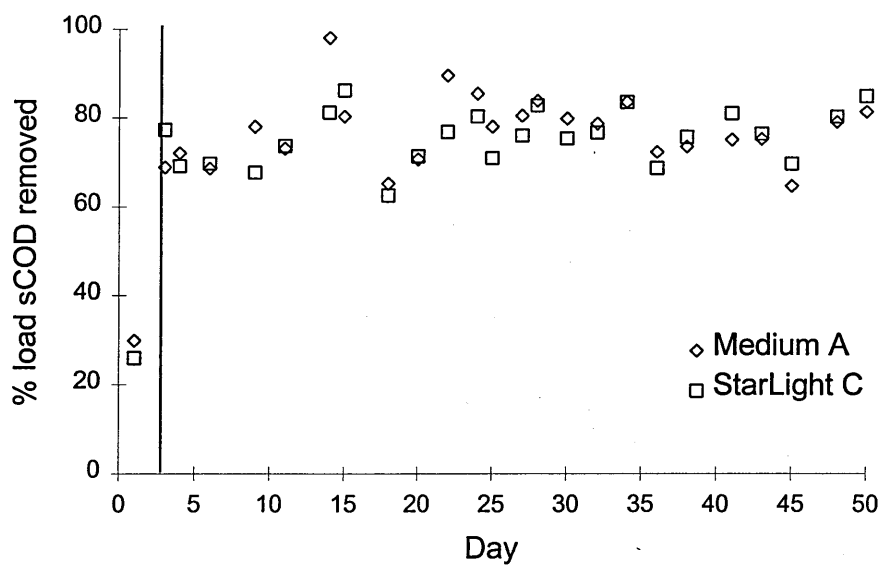


Figure 5.3: Start-up of sCOD removal (line shows stable operation reached by day 3)

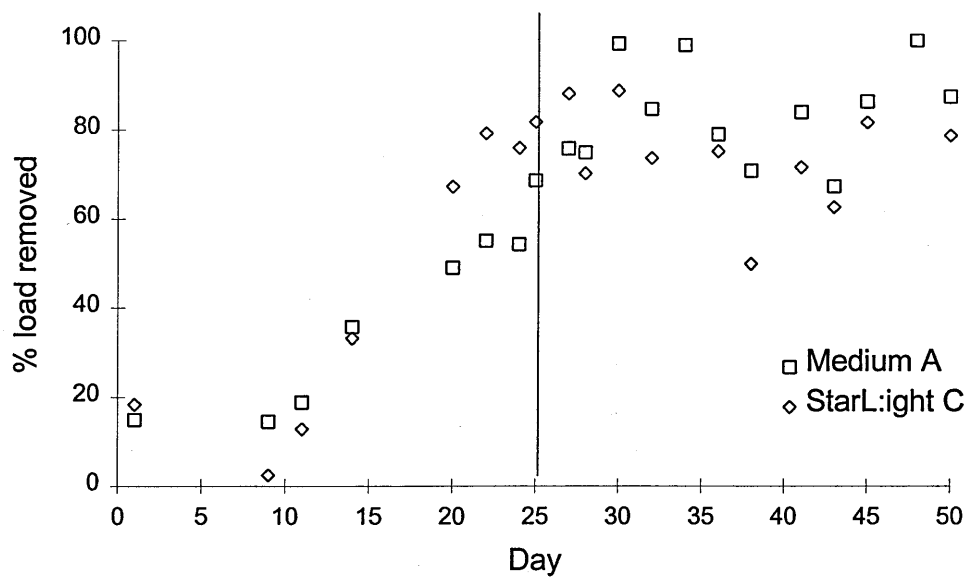


Figure 5.4: Start-up of ammonia removal (line shows stable operation reached by day 25)

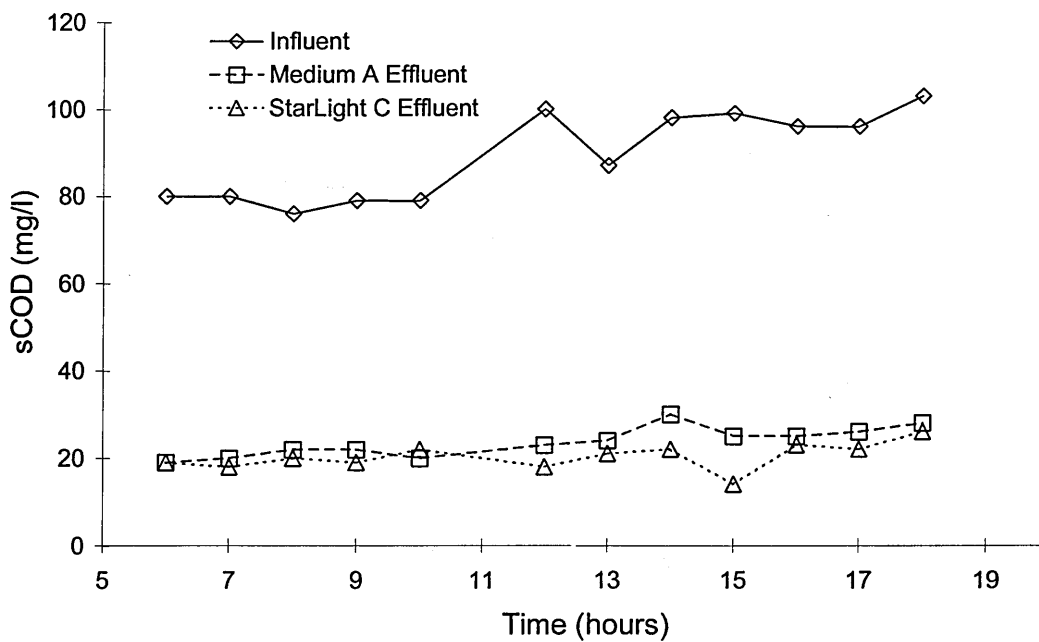


Figure 5.5: Removal of sCOD during a cycle.

5.3.3 Backwash Water Requirements

The profile of backwash effluent illustrated in Figure 5.6 indicates that when the two media were washed at a liquid flow rate of 40% of their V_{mf} , a similar cleaning time was required for each. After eight minutes backwashing, comparable concentrations of suspended solids and tCOD were being removed from the two reactors. However, more backwash profiles should be done to confirm this and optimise the time required.

After the liquid velocity used to backwash the medium A reactor was reduced to 11 l/min, the head loss build-up during the filter run increased as illustrated in Figure 5.7. This indicates that the filter medium was not being cleaned sufficiently at the reduced velocity. This is confirmed by examining the amount of solids removed during backwashing. Since both reactors received the same influent and produced similar effluents, they should have entrapped the same amount of solids. These solids would have been removed during backwashing if the procedure was effective. Before the backwash velocity of the medium A BAF was altered, there was no significant difference between the mass of solids removed from the two reactors during backwashing. However, when the liquid backwash velocity of the medium A reactor was reduced, the difference in the mass of solids removed became significant (Table 5.4). Solids remaining in the reactor can lead to premature blocking and eventually a decrease in effluent quality.

Table 5.4: Mean suspended solids of backwash effluent

Influent flow rate (l/min)	Mean solids removal at backwash (kg ± st.dev.)		p-value for paired t-test
	Medium A reactor	StarLight C reactor	
0.5	0.082 ± 0.031	0.073 ± 0.021	0.75
0.5*	0.051 ± 0.013	0.075 ± 0.006	0.01

* lowered backwash velocity

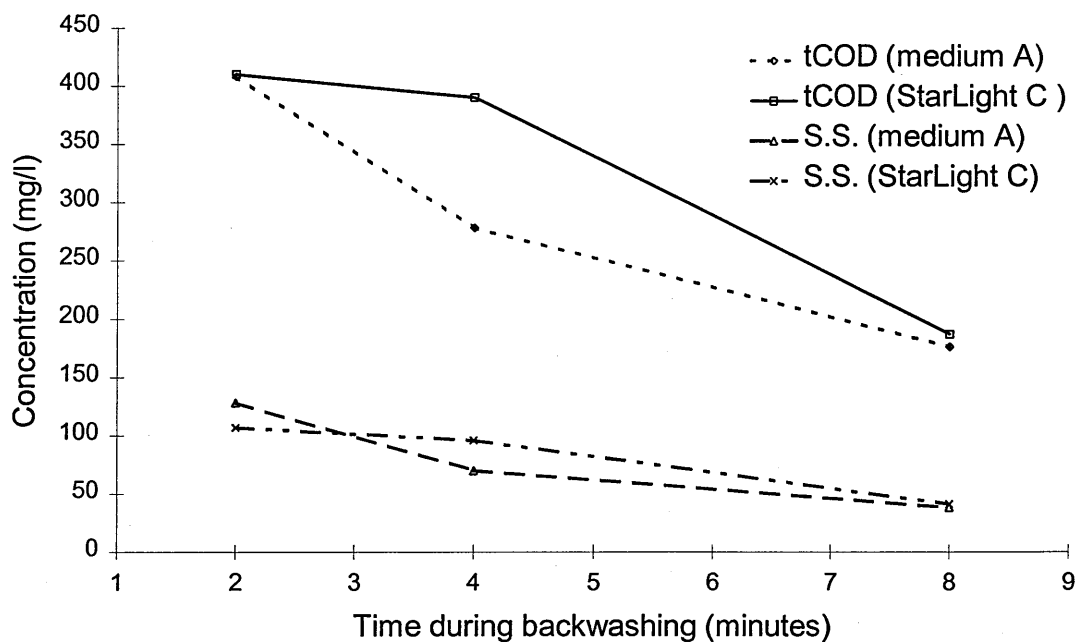


Figure 5.6: *Suspended solids and tCOD concentration of backwash effluent during backwashing*

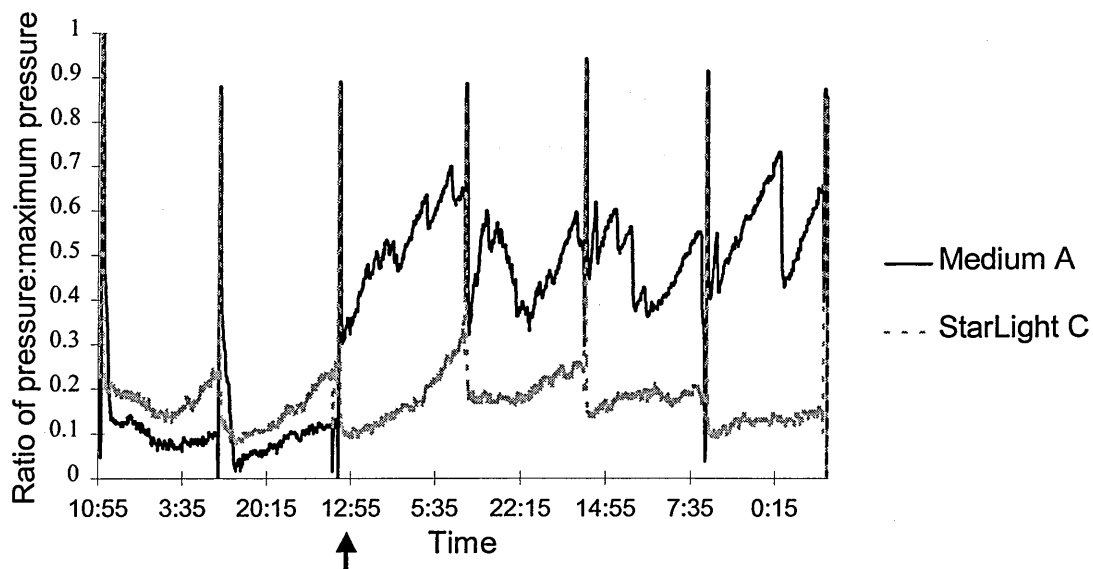


Figure 5.7: *Ratio of pressure to the maximum pressure recorded above the filter beds. The arrow indicates when the backwash velocity of the medium A reactor was reduced.*

5.4 DISCUSSION

Previously, start-up methods for BAFs have included seeding them with activated sludge or gradually increasing the flow rate from a low initial velocity. Using both these methods, previous researchers have reported start-up times for sCOD and suspended solids removal of around one month (Hamoda & Abd-el-Bary, 1987, Mann *et al.*, 1999, Smith *et al.*, 1990). By comparison, the method described in this paper resulted in an extremely rapid start-up. This is partly because applying the nominal flow rate immediately meant that the biofilm did not have to adapt continually to changing conditions.

Additionally, the biomass support materials used previously were plastics (Mann *et al.*, 1999) and sand (Smith *et al.*, 1990), both of which have relatively smooth surfaces. Many researchers have found rough surfaces are more efficient at retaining biofilms (Baker, 1984, Heijnen *et al.*, 1992). The asperity not only increases the available surface area, but may also provide sites for glycocalyx attachment (Baker, 1984). The media surface roughness will be particularly influential during start-up when initial colonisation is occurring.

It should be noted that although substrate removal stabilised very rapidly, bacterial biomass accumulation may take much longer. Bacquet *et al.* (1991), using expanded schist and starting at the nominal flow rate, found the tCOD removal rate was steady after a few days. However, the fixed biomass levels continued to increase for five months, without appearing to contribute to the reactor performance. Previously, confusion over this distinction has led to the rejection of starting up at the nominal flow rate (Mann *et al.*, 1999). The apparent disparity between the biofilm growing without increased nutrient removal can be explained by the classical theory of biofilms. This suggests biofilms are composed of both active and inactive cells and there is a limit to the thickness of the active part of the biofilm (Hoehn & Ray, 1973).

The suspended solids and sCOD removal results compare extremely well with those of Mann *et al.* (1999) who used identical reactors at the same facility, but utilised a sunken plastic medium. They report only 49% suspended solid removal and 73%

sCOD removal at flow rates of 0.3 l/min. When Mann *et al.* (1999) increased the influent flow rate, the removal efficiencies decreased to 40% suspended solids and 62% sCOD removal at 0.4 l/min, and just 35% suspended solids and 30% sCOD removal at 0.5 l/min. Previously the best performances of true BAFs, under normal conditions, have shown no major effluent deterioration in terms of COD and suspended solids concentrations up to loads of 9 kg COD/m³ active volume/d (Dillon & Thomas, 1989). At 0.5 l/min flow rate, the medium A reactor received 10 kg tCOD/m³ active volume/d and the reactor containing StarLight C received 15 kg tCOD/m³ active volume/d. Their performance under these conditions indicated that both media have extremely good biological properties. The surface roughness and porosity of the clay materials may have promoted the rapid formation of a stable biofilm.

Nitrifying bacteria are autotrophic and have a lower growth rate than heterotrophic bacteria. The long start-up period before stable ammonia removal was achieved, and the variations in the % ammonia removed, were therefore expected (Prosser, 1989). The reductions in ammonia removal from the StarLight C reactor as influent flow rates were increased can be also be explained. Due to their relative growth rates, autotrophic bacteria can only proliferate when COD concentrations are low enough to prevent the heterotrophic bacteria dominating. Thus, in a downflow BAF, nitrifying bacteria tend to colonise the bottom part of the reactor. As the influent flow rate was increased, COD may have penetrated deeper into the bed. This would have combined with the washout of medium from the StarLight C reactor to reduce the volume of the filter where conditions were appropriate for nitrifying bacteria.

The observed increase in the washout rate of StarLight C at 0.5 l/min, could have been a result of particles becoming entrained due to the high solids concentration in the filter. Washout of media can be reduced by ensuring a medium has a uniform density. However, the results from this pilot study over emphasise the problem. In a full scale BAF, washout can be reduced by using a filter or winged launder to separate the medium from the backwash water. Therefore, it has been assumed that this problem can be resolved.

The medium A BAF needed to be backwashed at a higher liquid velocity and consequently used more backwash water than the StarLight C reactor. Since the backwashing water was treated effluent from the reactors, the StarLight C reactor treated an extra 80 l of sewage per day. This increased the reactor's productivity by 19 % at 0.3 l/min, 14 % at 0.4 l/min and 11 % at 0.5 l/min. Additionally, reducing the backwash water velocity can reduce the operating and capital costs of the plant by lowering the required specification of the backwash pumping system. The volume of backwash water required by BAFs can also be decreased by backwashing when the pressure in the reactors reaches pre-set levels, rather than at set times (Stephenson *et al.*, 1993). BAFs containing smooth media require backwashing less frequently, but as previously described, often show inferior performance (Mann *et al.*, 1999, Mendoza-Espinosa & Stephenson, 1998).

Further optimisation of BAF media is feasible with the technology used to produce StarLight C. It may be possible to increase the size of the BAF media and still treat the sewage to an acceptable standard. The EC treatment standards often set maximum effluent suspended solids concentrations at 30 mg/l and BOD levels at 20 mg/l (Brewer, 1996). The pilot reactors in this study produced effluents with mean suspended solids concentrations below 20 mg/l and BOD levels below 12 mg/l. Increasing the media size should reduce the pressure build up above the filters and so increase the maximum filter run time.

5.5 CONCLUSIONS

1. Rapid start-up of COD and suspended solids removal can be achieved with BAFs containing expanded or foamed clay media by using nominal flow rates, two days of re-circulation and reduced backwash frequency for a further four days.
2. Both materials performed very well as BAF media, with the reactors showing superior performance to those previously operated with plastic or sand media (Mann *et al.*, 1999, Mendoza-Espinosa & Stephenson, 1998).
3. It was confirmed that when a media is backwashed at below 40% of its V_{mf} there can be incomplete solids removal leading to increases in head pressure build up above the filter.
4. The lightweight StarLight C medium has a lower V_{mf} than medium A. Consequently the BAF using StarLight C used roughly half the volume of treated effluent during backwashing than the medium A BAF. Increased productivity was 19, 14 and 11%, at influent flow rates of 0.3, 0.4 and 0.5 l/minute respectively.

CHAPTER 6

ASSESSMENT OF THE METHODS USED TO CHARACTERISE BAF MEDIA

ABSTRACT

Results from standard tests for BAF media have been compared to data from pilot-scale operation, indicating that some tests may require modifications. Materials that showed 3.0% and 5.2% weight loss after attrition testing exhibited little attrition during nine months operation, possibly due to biofilm protection.

Comparing the performance of BAFs containing media with similar dimensions, but different surfaces, showed the importance of measuring surface roughness. With an influent of 0.3 l/min settled sewage, media with high surface roughness achieved an extra 29%, 15% and 70% removal of suspended solids, soluble chemical oxygen demand and ammonia respectively.

The theoretical minimum fluidisation velocities (V_{mf}) of BAF media did not approximate the experimental V_{mf} s, as many of the assumptions made in the theoretical equations do not hold true for BAF media.

Tracer studies on empty beds gave a good approximation to operating beds, highlighting potential problems such as channelling and excess liquid hold-up. For three media beds which were equivalent to 3, 4 and 8 tanks in series, the nitrification results were superior for the reactors in which the flow was closest to plug flow.

6.1 INTRODUCTION

Biological aerated filters (BAFs) are three phase reactors composed of submerged beds of media with an air supply. BAFs usually treat settled sewage removing biological oxygen demand, suspended solids and ammonia. When operated under alternating anaerobic and aerobic conditions, phosphate removal can also be achieved (Stephenson & Mann, 1995).

The main advantage BAFs have over other secondary wastewater treatment (WWT) processes is their small footprint. The land required for BAFs is approximately one-fifth of that needed for plastic media trickling filters and one-tenth of that needed for activated sludge plants (Stensel & Reiber, 1983). This is achieved through two mechanisms. Firstly, the medium in the reactor acts as a substratum supporting the growth of a microbial biofilm. The medium has a large surface area allowing a high concentration of microbes, typically four to five times that in conventional sewage treatment processes, to be maintained within the reactor (Smith & Hardy, 1992). This facilitates a reduction in the required hydraulic retention time, and a greater load can be treated per unit volume per day. Secondly, the medium acts as a depth filter negating the need for a separate solids removal stage. Evidently the medium used in a BAF has a large influence over the properties of the process.

Many different materials have been used in BAFs in combination with a variety of reactor configurations (Stephenson, 1997). Several tests have been used to determine the suitability of materials for BAFs, including the British Effluent and Water Association (BEWA) standards (1993). These standards were originally designed to test media for potable water treatment, and it has been suggested that further work is required to determine which BEWA standards are not applicable to BAF media (Kent *et al.*, 1996).

Tracer studies have been conducted on empty BAF beds to examine the hydraulic characteristics of reactors containing different media (Mann *et al.*, 1995). It was suggested that media which increased the amount of mixing, making the hydraulic

conditions in the reactor less plug flow, would show better organic but worse suspended solids and ammonia removal characteristics.

This chapter uses two granular sunken media as the main examples to compare the data from physical tests to the results collected during reactor operation and hence evaluates the application of these tests for BAF media.

6.2 MATERIALS AND METHODS

6.2.1 Physical Tests

All of the following tests were carried out on a standard expanded clay BAF medium (medium A) and a novel foamed clay called StarLight C.

The physical properties of the media including its size, acid solubility, dirt content, attrition resistance, voidage (poured and packed) and density (particle and bulk as defined in Appendix 4A), were determined using standard techniques (BEWA, 1993). The size grading was completed manually, since machine sieving can cause media abrasion and increase the proportion of fines (Ives, 1990).

The minimum fluidisation velocities (V_{mf} s) were determined experimentally (Kent *et al.*, 1996) and calculated from theory using the equations of Carmen-Kozeny (Coulson & Richardson, 1978) and Wen & Yu (1966) (Equations 12 and 13, Chapter 2).

A 140 and 240 Pascal mercury porosimeter (CE Instruments, Italy) was used as a more reliable measure of grain density, and to measure the internal porosity of the media. The specific surface area (SSA) was measured by both mercury porosimetry and nitrogen adsorption (Sorptomatic 1990, CE Instruments, Italy).

6.2.2 Empty Bed Tracer Studies

Prior to WWT, tracer studies were performed to examine the hydraulic flow behaviour through the empty media beds. A pulse of molar sodium chloride (GPR, Merck) solution was applied for one minute at a flow rate of 0.4 or 0.8 l/min. The output of the salt was measured using a conductivity meter (LTH Electronics Ltd., UK) with a reading taken every minute for 300 minutes. Two or three repeats were made both with and without an air supply (2 l/min).

6.2.3 Operating Conditions

Two identical pilot-scale BAFs (full description in Chapter 5) were run in parallel for nine months. The reactors were backwashed every 24 hours at 40 % of the experimental V_{mf} of the medium. This was 75 m/h for medium A and 45 m/h for StarLight C. The backwash liquid was collected, mixed and then assessed for suspended, total and non-volatile solids. Influent and effluent samples were analysed three times a week for suspended solids (SS), non-volatile solids, total and soluble chemical oxygen demand (tCOD & sCOD), ammonia, temperature, pH (Russel, UK) and dissolved oxygen (Jenway, UK) (APHA, 1992). Pressure transducers (VEGA, Germany) were used to monitor the head build up during the filter cycle.

6.2.4 Tracer Studies During Operation

When the BAFs had stabilised at 0.4 l/min, further tracer studies were carried out using 1000 mg/l lithium ion solution. At this concentration, temporary exposure of a plug flow column to lithium has no inhibitory effect (Anderson *et al.*, 1991). Effluent samples (30 ml) were taken regularly for 240 minutes, drawn through 0.2 μ m filters and analysed with an ICP (Thermal Jarrel Ash Corporation, USA). Both sets of tracer study results were analysed to obtain the mean hydraulic residence time (HRT), peak tracer output, and number of tanks in series (N) (Levenspiel, 1962). An example of the calculation made to find N is given in Appendix 6A.

6.2.5 Measuring Media Attrition During Operation

When the columns were emptied at the end of the experiment, grab samples of between 0.75 and 1.5 l of medium were taken from sites throughout the reactors. These samples were assessed for volatile solids, and then sieved so that any media break-up could be quantified. The hydraulic size (BEWA, 1993), Hazen effective size and Hazen uniformity co-efficient (Ives, 1990) were calculated and compared to the values obtained before and after the standard attrition test.

6.3 RESULTS

6.3.1 Results from Physical Tests

The physical characteristics of the two media are summarised in Tables 6.1, 6.2 and 6.3. The main differences between the two media, are that StarLight C has a lower density, and correspondingly a lower V_{mf} . The maximum soaked density (defined in Appendix 4A) was calculated by assuming that all of the pores become completely filled with water.

Table 6.1: Physical characteristics of StarLight C and Medium A

Parameter	StarLight C	Medium A
Acid solubility (% weight loss)	0.015	0.990
Dirt content (% weight loss)	0.30	0.19
Weight loss after attrition testing (%)	3.0	5.2
Packed voidage (%)	40 (44)*	44 (45)*
Bulk density (kg/m ³)	460	800
Prill density (kg/m ³)	787	1423
Estimated experimental V_{mf} (m/h)	46-122	176-214
Theoretical V_{mf} using Carmen-Kozeny (m/h)	18.2 (40)+	354.5
Theoretical V_{mf} using Wen & Yu (m/h)	26	116

* indicates poured voidage results, +indicates value after adjusting the surface area

Table 6.2: Nitrogen adsorption and mercury porosimetry results

Parameter	StarLight C	Medium A
SSA by N ₂ (m ² m ⁻³)	1.27×10^6	3.42×10^6
<i>Using mercury porosimetry:</i>		
Total pore area (m ² m ⁻³)	2.8×10^6	8.0×10^6
Mean pore diameter (µm)	3.5	0.5
Total pore area of pores >2.5 µm (m ² m ⁻³)	4.9×10^5	6.8×10^4
Internal porosity (m ³ m ⁻³)	0.76	0.53
Max. soaked density (kg m ⁻³)	1500	2000

The hydraulic sizes of the media were not altered after the pilot study indicating that there was no particle break up (Table 6.3).

Table 6.3: Sieve sizing results before & after attrition testing and after the pilot study

Sample	Hydraulic size (mm)	Hazen effective size d ₁₀ (mm)	Hazen uniformity coefficient d ₆₀ /d ₁₀
Medium A			
Before attrition	4.10	3.50	1.21
After attrition	4.10	3.50	1.22
After pilot study	4.10	3.50	1.23
StarLight C			
Before attrition	2.10	2.10	1.19
After attrition	1.70	2.10	1.20
After pilot study	2.20	2.10	1.16

There were few incidences when the non-volatile solids concentrations in the filter effluents were greater than those of the influent (Figure 6.1). Although solid balances were not determined for specific cycles, the mean figures illustrate that the mass of non-volatile solids removed by the filter approximates the additional mass found in the backwash effluent (Table 6.4).

Table 6.4: Comparison of non-volatile solids in influents and effluents

Non-volatile solids	Medium A reactor	StarLight C reactor
Influent – effluent (mg/l mean ± stdev)	23 ± 43	14 ± 41
Removed / day @ 0.3 l/min (g)	9.9	6.0
Removed / day @ 0.4 l/min (g)	13.2	8.1
Removed / day @ 0.5 l/min (g)	16.6	10.1
Backwash effluent (mg/l mean ± stdev)	450 ± 90	470 ± 140
Mean mass / day in backwash effluent – influent (g)	12.0	7.6

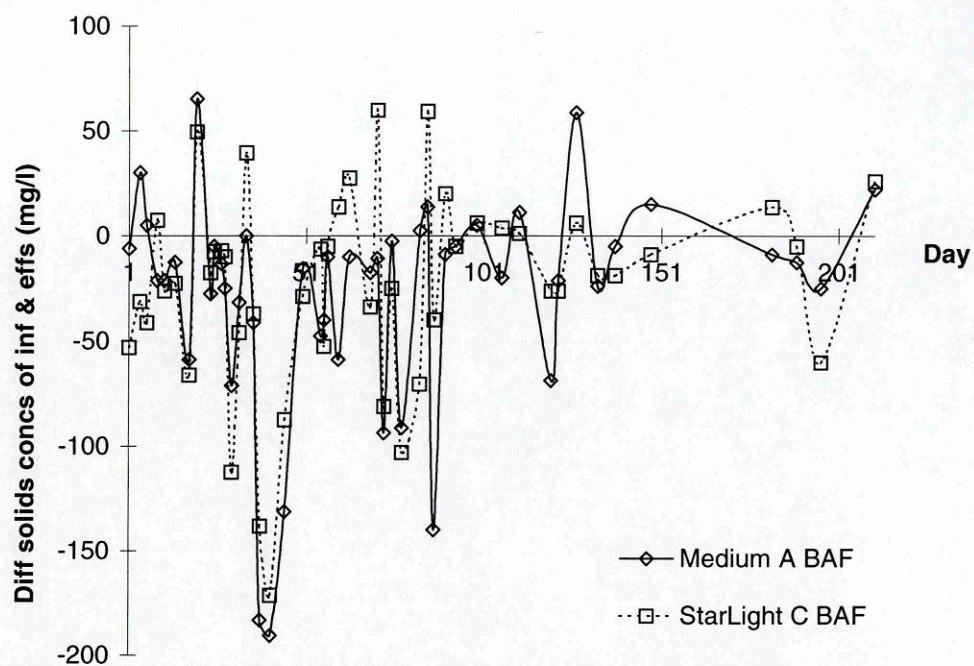


Figure 6.1: Difference between non-volatile solids levels in the effluents and the influent of the BAFs

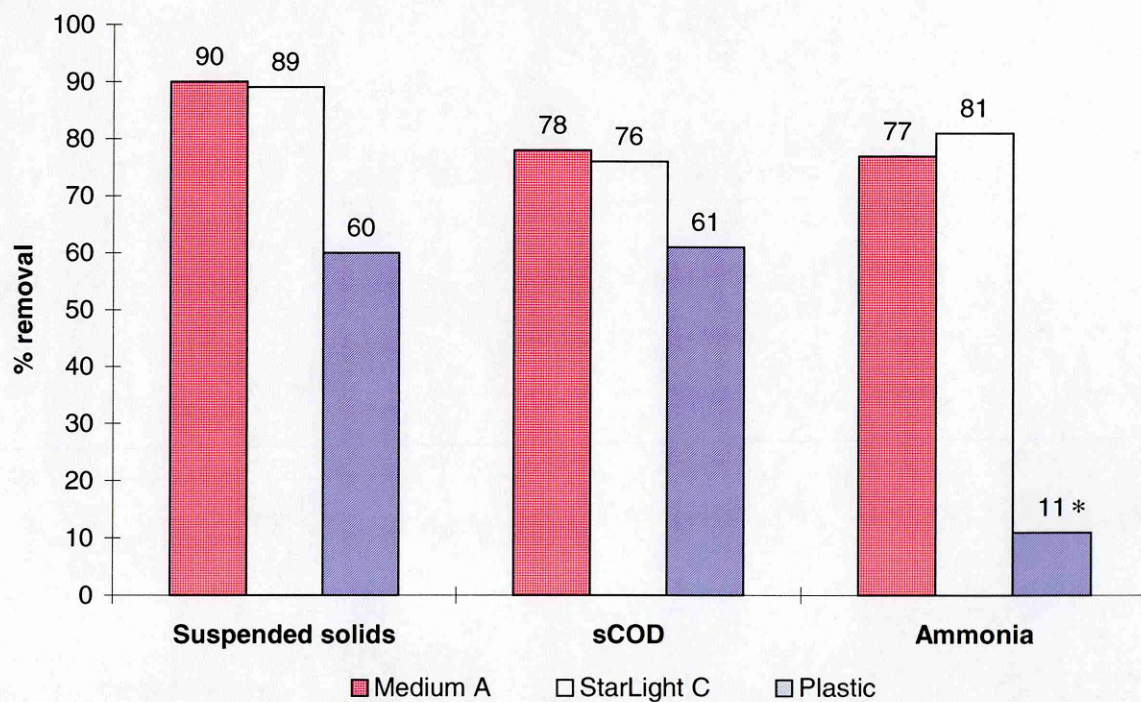


Figure 6.2: Mean % removal of suspended solids, sCOD and ammonia by identical BAFs containing different media at 0.3 l/min.

6.3.2 Operational Results

Medium A consists of irregular particles, whilst StarLight C is composed of cylinders with lengths between 3 and 7 mm and diameters between 1.5 and 3 mm. Both of these media have voidages close to the 42% reported for the sunken, plastic media used by Mann *et al.* (1999). This plastic media consisted of cylindrical particles with lengths of 4-6 mm and diameters of 2.3-2.7 mm. The performance of these three media can be compared since identical reactors were used, at the same WWT facility (Figure 6.2). Inferior removal of SS, sCOD and ammonia was shown by the BAF containing the plastic medium compared to those containing the mineral media.

6.3.3 Results from Tracer Studies

The empty bed tracer study results have been compared to those conducted in identical BAFs containing sand (unpublished data) and plastic media (Mann *et al.*, 1995) (Table 6.5).

Table 6.5: Empty bed tracer study results for different media under various conditions

Material	Voidage %	Liquid Q l/min	Air Q l/min	tHRT (mins)	Mean HRT (mins)	Tracer peak (mins)	N
Medium A	0.44	0.80	2.0	40	67	40	3
		0.40	2.0	80	123	70	4
StarLight C	0.40	0.80	2.0	36	83	54	4
		0.40	2.0	73	153	122	8
Sand	0.38	0.80	2.0	34	48	40	523
		0.40	2.0	68	88	70	1065
Plastic	0.42	0.35	3.5	72	46	25	3

The theoretical HRT (tHRT) can be calculated from the reactor’s working volume divided by the volumetric flow rate. For an empty bed, the void volume is assumed to be a good approximation to the working volume. In an operating BAF, although the working volume is lowered as the accumulation of solids reduces the apparent voidage, it is simultaneously increased, by the build up of fluid above the filter bed.

An attempt has been made to measure these factors in order to calculate the maximum and minimum tHRTs (Table 6.6). Bacterial dry weights can be converted into biovolume using average buoyant density values of 1.09 g/cm³ with 30% dry matter (Bakken & Olsen, 1983). The maximum solids volume can be calculated by combining the mean dry weight of solids removed during backwashing with the residual solids associated with the media. Maximum head pressure for each reactor operating at 0.4 l/min, was recorded by pressure transducers. The tracer studies conducted at 0.4 l/min, before and during operation can be compared after the concentrations have been normalised (Figures 6.3 and 6.4).

Table 6.6: Mean, peak and theoretical HRTs during operation at 0.4 l/min liquid flow

	StarLight C		Medium A	
Before/after backwashing	Before	After	Before	After
Mean HRT (mins)	137	138	108	98
Peak HRT (mins)	110	108	80	58
Theoretical HRT	Volume (l)	tHRT (mins)	Volume (l)	tHRT (mins)
Before operation	29	73	32	80
After media washout	22	55	29	73
Maximum solids capture	20	51	28	70
Maximum head pressure	64	160	67	168

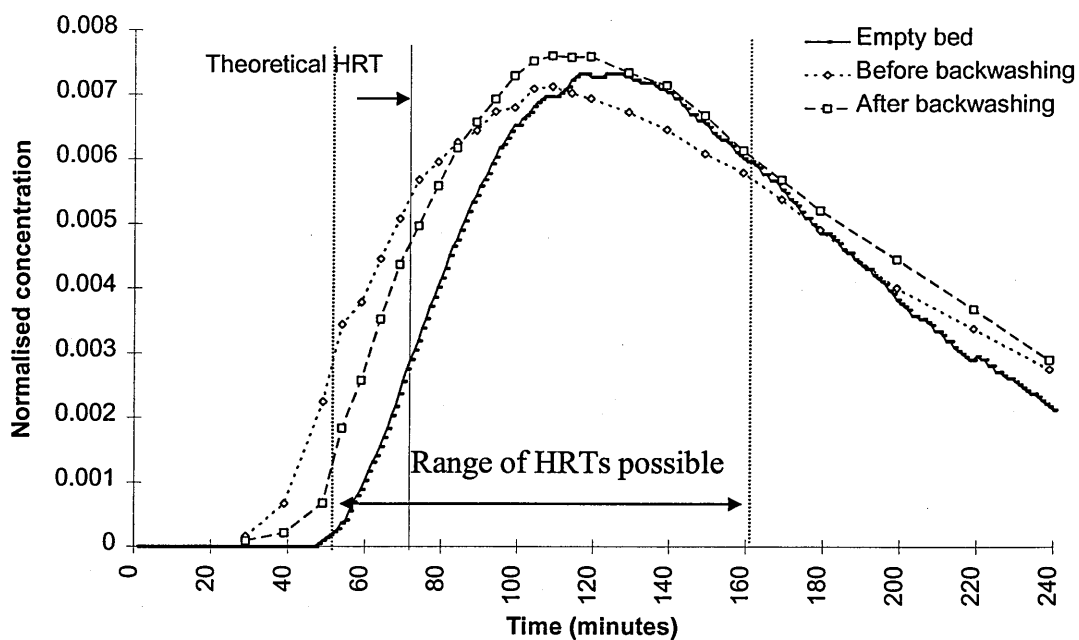


Figure 6.3: Tracer studies before and during operation of StarLight C BAF

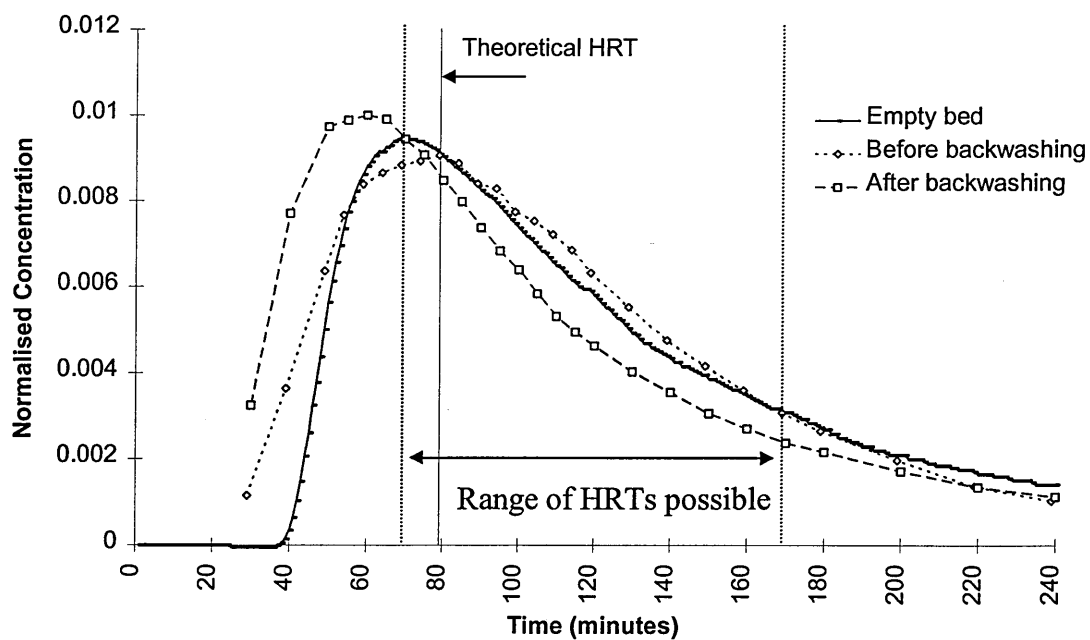


Figure 6.4: Tracer studies before and during operation of Medium A BAF

6.4 DISCUSSION

The BEWA (1993) acid solubility test quantifies the chemical inertness of a medium. It exposes the material to 50% HCl for 30 minutes, and requires over 95% of the initial weight to be maintained. BAF influents may be more variable than inputs to potable water filters. However, during nine months operation, the pilot BAFs influents and effluents were consistently between pH 6.8 and 8.2. Therefore the acid test seems stringent enough under normal conditions. Ives (1990) suggested that the BEWA acid solubility test (1993) was unnecessarily severe for potable filter media and suggested using 20% HCl for 24 hours but reducing the acceptable weight loss to 2%. Using a lower concentration, whilst increasing the exposure time, would also seem sensible for BAF media. The materials passed the acid solubility test, and did not deteriorate during operation. Therefore, the acid solubility method appears to be appropriate for measuring inertness.

The “dirt” removed from BAF media is predominantly dust, and falls well below the 1% BEWA (1993) weight loss standard. It is an appropriate test for BAF media only to indicate how much washing is required both prior to use and other tests.

The BEWA (1993) standards for water filter media, designate a 3% weight loss as the maximum allowable after attrition testing. Previous researchers have found BAF media tends to be prone to attrition, and have suggested that the standard should be revised (Kent *et al.*, 1996). However, like potable water filters, BAFs are backwashed periodically usually with liquid and air scours, so resistance to abrasion and particle break-up is required. Additionally, BAFs are supplied with air, which provides a continual abrasive action on the medium. The results of the attrition tests indicated that medium A was an unsuitable filter medium, due to unacceptable weight loss. StarLight C exhibited borderline weight loss and some reduction in hydraulic size. However, the same two media showed few signs of attrition during the pilot study. One possible explanation for the discrepancy between the BEWA test (1993) and the operational results is that the biofilm, which surrounds the media, affords it some protection against abrasion. This would allow the accepted standard for attrition levels during the BEWA test (1993) to be lowered slightly when it is applied to BAF media.

The relationships between media size, shape and voidage, and the reactors filtration characteristics and head loss development have been discussed in section 2.B3.

Despite the physical similarities between the media used in this study and those used by Mann *et al.* (1999), the suspended solids removal was greatly superior using either of the mineral media compared with the plastic media. This may be because the increased surface roughness of the mineral media has an additional indirect effect on filtration efficiency. As described in section 2.B.3.1, particle removal in BAFs is facilitated by the biofilm. Therefore, media properties that promote biofilm formation will indirectly lead to superior filtration efficiency. Rough surfaces have been found to promote biofilm formation (Baker, 1984). It is also proposed that internal pores may provide protected sites, which can initiate biofilm regrowth after backwashing. The improved biofilm retention of rougher, porous, mineral media was also demonstrated by a greater removal of sCOD and ammonia.

Both the Carmen & Kozeny (Coulson & Richardson, 1978) and the Wen & Yu (1966) equations, assume that media particles are spherical and uniform in size. Additionally, the Wen & Yu (1966) equation has no voidage term. Being non-spherical both StarLight C and medium A have larger voidage values than spherical media of the same size. As increases in voidage lead to increases in V_{mf} , the calculated values using the Wen & Yu (1966) equation are lower than those found experimentally.

Although the Carmen-Kozeny equation (Coulson & Richardson, 1978) contains a voidage term it assumes, $SSA = 6/d_m$ and $K'' = 5$. Cylinders are likely to have a lower SSA than spheres with the same hydraulic diameter, whilst angular materials may have a higher SSA. Increasing the SSA decreases the V_{mf} . Thus the calculated values using the Carmen-Kozeny equation are less than the experimental values for StarLight C and more than the experimental values for medium A. When the SSA value for StarLight C is corrected, the values obtained from the Carmen-Kozeny equation are nearer the experimental values. Meanwhile, K'' depends on the tortuosity and shape of the channels of the filter bed. For most spherical media it is close to five, but it can vary significantly for media of different shapes (Coulson & Richardson, 1978). Coulson (1949) attempted to relate shape changes with K'' but only concluded that

when voidage ranged between 0.3 and 0.5, K'' for most particle shapes, lies between 3.2 and 5.8. As many of the assumptions made in the equations do not hold true for BAF media, it is recommended that the V_{mf} is determined experimentally.

Due to non-uniformity in the size and density of the media, complete fluidisation did not occur at a single velocity. An ideal BAF medium would have a low V_{mf} like StarLight C to reduce the required backwashing velocity and so lower backwash liquid volumes. However, an ideal medium would also have a very narrow band of V_{mf} values, so that all the particles can be cleaned thoroughly during backwashing, without risking media washout.

Many of the parameters, which influence the medium's V_{mf} also determine the filter efficiency and head loss characteristics. Only the media density can be altered without affecting other aspects of reactor performance. For a porous medium there are three relevant density expressions. Bulk density (or packing density) gives the weight of media filling a set volume, whilst particle density is the density of the dry, air filled particles. However, soaked density assumes the pores have filled with water and is most relevant to BAF operation. To calculate the soaked density, the internal porosity of a medium needs to be measured.

Tracer studies performed on BAFs show that the reactors have flow patterns somewhere between plug flow and completely mixed (Mann *et al.*, 1995). However, these results are only relevant during initial start-up since the retention of biomass may alter liquid and air flows. A comparison of the tracer study results carried out prior to, and during operation, indicate that empty bed tracer studies provide a good indication of the flow regime in an operating BAF. In all cases the mean HRT was greater than the tHRT because the reactors were not completely plug flow and some mixing was taking place. Excess mixing can inhibit biofilm growth by applying greater shear stresses. However, a degree of mixing is thought to increase the removal of organic materials by improving nutrient transport (Mann *et al.*, 1995). Meanwhile, suspended solids removal and nitrification are thought to be more efficient under plug flow conditions (Mann *et al.*, 1995, Fdz-Polanco *et al.*, 1994b).

The appearance of the peak tracer concentration before the tHRT occurred in the medium A empty bed test at 0.4 l/min liquid flow and then, after backwashing in the operating medium A reactor. This indicated that there were preferential channels through the medium. The channelling was reduced in the tracer studies conducted before backwashing, indicating that re-compaction of the bed and biofilm growth closes the channels. If channels do not become established there may be no evidence of the negative effects that the increased turbulence has on the biofilm. This would explain why the performance of medium A as a BAF medium was not compromised.

In the StarLight C empty bed test at 0.4 l/min liquid flow, the peak tracer concentration appeared after the tHRT. This indicated that hold-up of liquid was occurring even before biofilm accumulation took place. An ideal media would exhibit low hold-up and show little or no channelling.

The results of tracer studies are thought to be strongly related to the voidage of a medium. As voidage is increased, the diffusional resistance is lowered leading to increased mixing (Fdz-Polanco *et al.*, 1994b). This theory can be tested by comparing the tracer study results with the physical characteristics of the media. Higher values of N , indicating better plug flow conditions, were shown in the reactors containing the media with lower voidages. It is recognised that other media parameters such as size distribution and shape are equally important and therefore continual pooling of information is required.

The results from the tracer studies do not fully explain the differences in the biological performance of the reactors using different media, because other parameters also have an effect. However as expected, ammonia removal was best in the StarLight C reactor ($N = 8$), followed by media A ($N = 4$), and then the plastic media reactor ($N = 3$).

6.5 CONCLUSIONS

1. The BEWA standards for filter media are appropriate for assessing media inertness, size, voidage and bulk density.
2. Media that showed borderline and unacceptable weight loss in the BEWA (1993) standard attrition test, exhibited few signs of attrition during operation. If the biofilm surrounding the medium protects it from attrition, then the standard could be lowered.
3. Comparing the performance of BAFs containing media with similar dimensions but different surfaces showed the importance of measuring surface roughness. With an influent of 0.3 l/min settled sewage, media with high surface roughness achieved an extra 29%, 15% and 70% removal of SS, sCOD and ammonia respectively.
4. The V_{mf} of BAF media should be determined experimentally, as many of the assumptions made in the theoretical equations do not hold true for BAF media. Lowering a medium's density lowers its V_{mf} without affecting other characteristics, and so could reduce operating costs. It is most pertinent to know the medium's sunken density, which often requires internal porosity measurements.
5. Tracer studies on empty media beds provide a good approximation to studies on operational filters and can highlight potential problems such as channelling or excess liquid hold-up.

CHAPTER 7

ASSESSMENT OF THE POTENTIAL MARKET FOR BIOLOGICAL AERATED FILTER (BAF) MEDIA

ABSTRACT

A marketing study was conducted to collect and analyse information about the markets for BAFs and their media. Interviews were conducted with representatives from all of the wastewater treatment (WWT) companies, the environment agency and ECCI. The high operating costs of the BAF process were identified as its major disadvantage. Comparing the required properties of current BAF media with those of StarLight C indicated that the product had good technical attributes. The interviewees from the WWT companies marked eight secondary WWT processes on ten features. The total mark for BAFs was only exceeded by that for sequenced batch reactors (SBRs). When the marks for the required properties of eight different applications were considered, BAFs were technically the most suitable process for urban sites with either a small footprint or a variable wastewater. To reduce the risks of product failure it is recommended that ECCI sell their media to an existing BAF supplier.

Depending whether the traditional WWT processes could be used at all of their treatment works, some of the WWT companies had a technical focus and others had an industrial purchase focus. BAFs have been badly, or under designed several times in the UK during the last decade and subsequently some of the WWT companies have a negative opinion about the process. To reform their opinions it is recommended that ECCI conducts full-scale trials with South West Water. Until OFWAT set the price limits it will be difficult to accurately assess how many BAFs will be built during AMP 3. The views from the WWT companies are that between 50 and 250 new secondary WWT plants will be commissioned.

7.1 INTRODUCTION

7.1.1 Aims of the Marketing Study

When developing a new product, as well as proving and advancing its technical attributes, it is important to conduct a marketing study. It has been observed that 70% of new product launches fail (Millier, 1989). Collecting and analysing information about the products market can help companies reduce the risks of failure.

English China Clays International (ECCI) has no experience of the BAF industry. When entering a new market it is important to identify potential customers and competitors. By characterising the behavioural differences of potential customers it may be possible to select and target those which are most promising first. Additionally, identifying the needs of potential customers allows sources of competitive advantage to be developed. Meanwhile, analysing the competition may highlight groups where the barriers to entry are too high. Thus, the specific aims of this marketing study were to:

1. Evaluate the current status of BAF usage in the wastewater treatment (WWT) industry. For instance, how many BAFs are used by each of the WWT companies, what sizes and type they are and how they have performed.
2. Examine attitudes of the WWT companies towards BAFs and the competing biological WWT processes. Identify the perceived advantages, disadvantages, and the anticipated future application of BAFs.
3. Segment the market on a technical and behavioural basis, and identify the segments with the lowest associated risks
4. Evaluate the potential market size by obtaining estimates of the number of WWT sites requiring capital expenditure for the next five years.
5. Identify the leading BAF suppliers and their sources of competitive advantage.

7.1.2 The Competitive Environment

The competitive environment can be examined on a number of levels as shown in Figure 7.1 (Bolt, 1981). The increasing international pressure to improve environmental standards has undoubtedly led to the recent introduction of national and European standards for WWT.

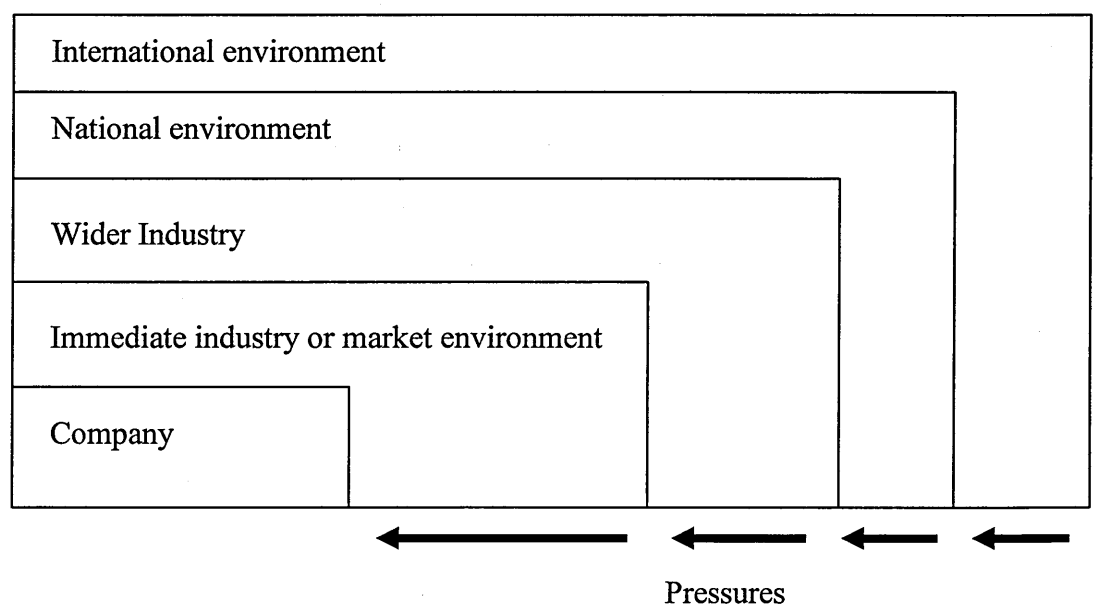


Figure 7.1: Environmental influences on a company

The United Kingdom is the fifth largest environmental market in the World and is in the process of implementing major environmental reforms, which are certain to boost demand for environmental technologies. In April 1996, the UK Environmental Agency was established. This has amalgamated several former agencies, most significantly Her Majesty's Inspectorate of Pollution (HMIP), the National Rivers Authority (NRA) and 83 local waste authorities (ISI, 1997).

7.1.3 The Wastewater Treatment (WWT) Industry

The wastewater systems market, a sub-sector of the environmental technology market, has been very dynamic. The Water Act of 1989, which privatized the water and wastewater systems of England and Wales, created several new regulatory agencies and set strict standards for water quality. Ten water and wastewater companies were formed that provide 99% of the population with wastewater services. These companies are responsible for all aspects of design, construction and operation of sewage treatment plants (ISI, 1997).

The UK market for water and wastewater treatment equipment has grown substantially throughout the 1990s and was estimated to be worth \$538 million in 1997 (Frost & Sullivan, 1997). It was estimated that \$33-45 billion would be spent in England and Wales between 1997 and 2007 to meet new regulations, with a particular focus on municipal wastewater discharges and sludge disposal at sea (ISI, 1997).

7.1.3.1 Market Drivers

The primary market driver in the WWT market is legislation. The European Commission (EC) is increasingly setting the regulatory agenda for all of Europe. There are currently several important pieces of legislation influencing investment in secondary sewage treatment.

The EC Urban Wastewater Treatment Directive (UWWTD, 91/271/EEC) has set minimum standards for treatment of wastewater that must be reached between 1998 and 2005. The primary requirement is that all municipal wastewater goes through secondary treatment unless the area is designated as less sensitive (high natural dispersion areas- HNDAs). As a direct consequence of the Directive, sea outfalls will be phased out and sewage treatment works will be improved. In 1998, the WWT companies estimated that a capital cost of £1,320 million would be needed to meet this requirement. Improvements to the preliminary, primary or secondary treatment facilities were thought to be required at 1,670 sites (OFWAT, 1998a).

Additionally, Ministers have now set their expectations for the quality enhancement program after the year 2000. They plan to remove the HNDA status from all receiving waters so that secondary treatment is provided for all significant discharges (OFWAT, 1998b). The estimated capital cost of providing this additional treatment is £360 million* (OFWAT, 1998a). This investment will be concentrated in coastal regions (Bolt, 1995) so it will have a greater impact on some WWT companies, such as Anglian Water, Northwest Water and Northumbrian Water, than others, such as Severn Trent Water and Thames Water (OFWAT, 1998a).

The EC Bathing Water Directive (76/160/EEC) sets standards for 19 physical, chemical and microbiological parameters. The most widely used parameters are total and faecal coliform bacteria, which are indicators of sewage pollution. Although this directive has been in force for over 15 years, its original scope was very limited and it was only applied to a small number of designated bathing waters. The standards imposed by the directive are now being enforced more rigorously, and many European member states have been taken to the European Court for non-compliance. The main implication of the Directive is that WWT works will need to be installed where there are none, and existing WWT works will need upgrading. The EC recently criticized Britain for failing to designate any inland bathing waters and it is currently investigating which inland waters are used for swimming (ISI, 1997).

Most of the improvements concerning the Bathing Water Directive to date have installed UV disinfection and storm storage. However, the WWT companies predict that investment in secondary and tertiary treatment, as well as UV disinfection and storm storage, will be needed to meet the Revised Bathing Water Directive (estimated capital expenditure of £1,860 million). These costs are likely to be incurred between 2005 and 2010 (OFWAT, 1998a).

Between 2000-2005, an estimated capital investment of £140 to £540 million will be necessary to reach the standards required by the Freshwater Fish Directive (78/659/EEC). Improvements to secondary treatment, generally involving nitrification are required at over 550 sites (OFWAT, 1998a).

* all estimated costs are at 1997-98 prices

Investment in tertiary treatment between 2000-2005, will be driven by the Habitats Directive (estimated capital expenditure of £60 million), the Surface Water Abstraction Directive (estimated capital expenditure of £60 million), an increase in the number of designated sensitive areas by UWWTD (estimated capital expenditure of £710 million) and the Shellfish Waters Directive (estimated capital expenditure of £50 million).

As well as the statutory requirements to comply with European regulations, Ministers have improved some of the additional recommendations of the Environment Agency. A programme will be implemented to maintain and improve the quality of rivers and significantly raise the compliance with river quality objectives (RQOs). The combined sewer overflow (CSO) improvement programme will be accelerated so that at least 66% of the intermittent discharges which remain unsatisfactory in 2000 are remedied by 2005 (OFWAT, 1998b). The estimated costs for each of these initiative are £560-1160 million and, £860 million (OFWAT, 1998a).

The WWT companies have predicted that their total investment for the period between 2000-2005 will be approximately equal to that in the period between 1995-1999. However, the engineering requirements are expected to be different, as the emphasis shifts from above ground assets to below ground assets. More schemes will involve asset maintenance and refurbishment, and less will focus on providing new plant. Additionally, more of the new plant required will be for smaller sites and for site improvements (British Water Document No: BW421, 1999).

7.1.4 The Supply Industry

There is a well-established supply industry comprised of consultants, contractors and equipment suppliers. Past practice has shown a distinction between these groups however, there is a trend towards turnkey contracting for major plants. In these cases, for a given influent and desired effluent the contractor agrees a price. Then they select and design the processes and plant, but face penalties if it fails to reach its objectives. Most engineering companies have a portfolio of processes to offer the customer, but there are also several dominant proprietary systems for many of the processes including BAF reactors.

7.1.4.1 Competing Secondary WWT Technologies

The purpose of secondary WWT is to remove most of the organic substances and the suspended solids from the wastewater. Biological processes, which harness micro-organisms, usually provide the most cost-effective method of doing this. The traditional secondary treatment processes are trickling filters (TFs) and activated sludge (AS) reactors. TFs have low operating costs but require large amounts of land. Meanwhile, AS reactors use less land and have higher operating costs. According to an independent consultant there are likely to be between 100 and 1000 new secondary treatment units installed within the period 2000-2010. The assessment of new secondary WWT technologies, and their scope for reducing costs compared to AS systems has been summarised in Table 7.1 (Babtie Environmental, 1998).

The possible savings associated with using BAFs are derived from their small footprint. This makes for more economical buildings and odour control if the works is to be enclosed. Both the operating and the capital costs of BAFs are slightly higher than those for AS reactors. However, the capital costs have been declining significantly since 1991 (Babtie Environmental, 1998).

Table 7.1: Cost benefits of alternative secondary WWT processes compared to AS

Process	Unit cost	Current development	Financial impact			Reason for saving
			Current	2005	2010	
BAF	high	demonstrated	some	some	some	Small footprint
Sequenced batch reactors (SBR)	medium	demonstrated	little	little	reasonable	No separate clarification tank
Reed beds (EXT)	medium	emergent	little	some	some	Low operating costs
Anaerobic digestion (ANA)	medium	developed	little	little	some	Lower operating costs
Oxygen AS	high	developed	little	little	reasonable	Retrofit & reduce civil costs
TF	high	demonstrated	little	some	reasonable	Low operating costs
Membrane bioreactor (MBR)	high	demonstrated	little	little	reasonable	Small footprint & very good effluent

BAFs were developed in the late 1970s with patents being issued in the USA and France in 1978 (Condren, 1990). During the 1980s, BAFs were developed specifically for secondary and tertiary sewage treatment, as an alternative to conventional processes such as AS reactors and TFs. BAFs are compact, high rate, robust biological reactors. They can achieve complete treatment, including nitrification, denitrification and phosphate removal, with one unit. Despite these advantages the application of the process has been limited (Mendoza-Espinosa, 1999). This is because the traditional techniques are tried, tested and most importantly trusted by the WWT companies. Thus, although the process has existed for almost twenty years the market is relatively new. As there is little existing knowledge about the BAF market, data must be collected directly from a sample of potential clients.

7.2 METHODOLOGY

7.2.1 Data Collection

Data can be collected by face-to-face interviewing, telephone interviewing or a self-administered questionnaire. Each of these methods has its advantages and disadvantages.

A self-administered questionnaire is the most neutral instrument for gathering facts (Rogers, 1976). This method also has cost and time advantages (Fink & Kosecoff, 1985) and should be used if the respondent has to look up any information. However, the response rates to written questionnaires can be low, particularly when there has been no initial contact. Additionally, since the questionnaire can be read through completely before being answered, bias can result from knowing the overall structure of the questioning (Chisnall, 1986).

Written surveys can be faxed rather than posted to save time and money. It has been suggested that a faxed survey has a degree of novelty and may imply importance. It may therefore, achieve higher response rates (Dickson & MacLachlan, 1996). However, faxed surveys may have lower image quality and there is no way to include a pre-paid envelop or incentives. Equipment failure and busy signals for the return message can also deter responses (Dickson & MacLachlan, 1996). E-mailing surveys has similar advantages to faxing them, but may restrict selection of the surveyed sample.

The quality of data obtained by telephone can be comparable to that collected in person (Rogers, 1976). Generally, good response rates are shown and data is collected fairly quickly. However, people can be wary of telephone interviews since many sales schemes use phoney surveys as an entering wedge (Payne, 1956). They are an unsuitable option when asking complex questions.

A face-to-face interview is the best way to find out people's motivations, attitudes, feelings and prejudices. It is also the best method to gather complex information.

However, depth interviews can not usually be carried out on a representative sample due to time and cost restrictions. In this study, since complex and attitudinal questions needed to be asked, it was decided that depth interviews were the most suitable data collection method. However, once contact had been made during the interviews, it was proposed the interviewees could be e-mailed, telephoned or mailed to answer further questions or verify conclusions.

At least one representative from all the WWT companies of England and Wales (Anglian Water, Northumbrian Water, North West Water, Severn Trent Water, Southern Water, South West Water, Thames Water, Welsh Water, Wessex Water and Yorkshire Water) was interviewed. The representatives were WWT process engineers or worked within research and development. Most were already contacts of the School of Water Sciences at Cranfield. In addition to the representatives of the WWT companies, a representative of the Environment Agency and a representative of ECCI were interviewed.

7.2.2 The Interview Questions

The interview questions used during the main study are in Appendix 7A. There are three types of question; Classification, behavioural and attitudinal. Classification questions included the interviewees job title and background. However, care was taken not to ask unnecessary personal questions. Indeed, the relevance and importance of every question was analysed, before being accepted into the interview (Chisnall, 1986).

Behavioural questions asked for information about the company's current status, such as the number, capacity and performance of the operational BAFs. Meanwhile, attitudinal questions probed the opinions of the respondents, for instance asking them to rate their opinion about the various secondary treatment processes. The funnel technique was used to prevent sequential questions influencing the responses (Chisnall, 1986). Thus, the more general questions were asked at the beginning of the interview, and the more specific questions about BAFs, at the end. The first questions were designed to be easy to answer, to put the interviewee at ease. The most sensitive

and difficult questions were asked in the middle of the interview, with the final questions being easy. Two practise interviews were performed with delegates from the part-time Water and Wastewater Treatment MSc at Cranfield University. Feedback from these interviews indicated that the questions were clear and easy to answer.

The interview questions for the representatives of the Environment Agency and ECCI can be found in Appendices 7B and 7C respectively.

7.2.3 Analysis of Responses

The data collected from the interviews was collated and then analysed using the methods detailed by Millier (1989, 1999).

7.3 RESULTS: ANALYSING THE MARKETING SITUATION

7.3.1 Market Structure

The market structure must be characterised at the beginning of a marketing study so that further investigations are in the correct areas (Figure 7.2). For ECCI, arriving in the market as a BAF media supplier, the engineering companies that supply BAFs are the immediate customers. However, the client’s client’s will ultimately use the product, so the priority of this research will be to examine their demands. Some of the larger industrial companies may treat their effluent on site, but the most significant customers for WWT process are the WWT companies.

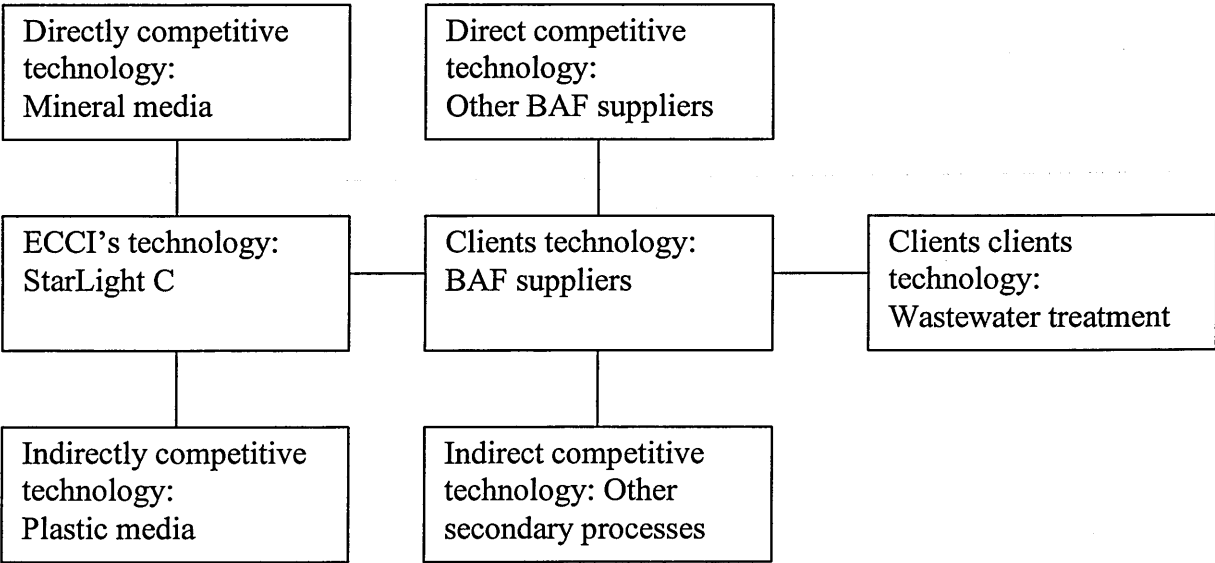
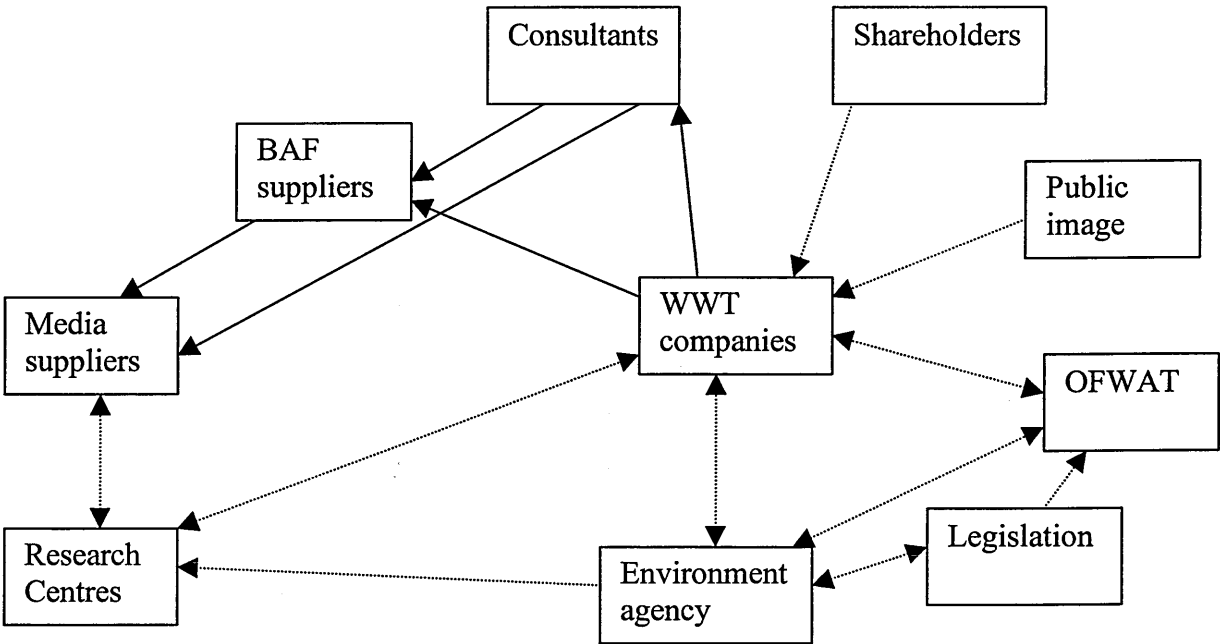


Figure 7.2: Structural Analysis of the Market

Looking beyond the supply chain there are many external influences on the market (Figure 7.3). The WWT companies have obligations to their shareholders and their customers, the general public. Additionally, there are close negotiations between the WWT companies, the Environment Agency and OFWAT. The Environment Agency advises the UK government Ministers on relevant European legislation and the benefits of specific quality programmes. After the Ministers issue their guidelines on future quality obligations, OFWAT set limits on the WWT companies prices. Both bodies are also involved in the evaluation of specific site improvements.



-----> = influence, ———> = supply chain

Figure 7.3: Functional analysis of the market showing connections between participants of the market

7.3.2 Technical Analysis of the Product

7.3.2.1 Analytical Breakdown of the Technology

ECCI have the required technology to produce a foamed clay material, which they call StarLight. The elements of this technology can be broken down into four groups; the components, the equipment, the theoretical expertise and the know-how (Table 7.2).

Table 7.2: Analytical breakdown of foamed clay technology

Technology components	Technical Elements
Components	China clay Surfactant Bentonite Water & air
Equipment	Foaming unit Extruder Kiln
Theoretical knowledge	Mineralogy Firing regime
Know-how	Patented process

The components, equipment and to a lesser extent the theoretical knowledge to produce foamed clay are easy to obtain. However, the know-how is a patented process so, ECCI are in control of the dissemination of the technology to other companies. Therefore, the foamed clay market has high barriers to entry but, it is the market for BAF media not foamed clay that needs to be analysed.

7.3.2.2 The Functions of BAF Media

When customers buy products, they are actually buying the functions of that product. Thus, when buying a medium for a BAF reactor they are not buying foamed clay produced by a patented process. They are in fact buying a product which; retains a biofilm, filters suspended solids from suspension, and is easily cleaned. As well as considering the functions that a product performs, customers also consider unfavourable properties. For a BAF media, unfavourable properties include having a high attrition or washout rate.

The representatives of the WWT companies were asked what improvements they would need to see in a new BAF medium to induce switching (Question 40). These properties could be considered key elements, with those most frequently mentioned having the most strategic importance. Improved biomass performance, lower attrition and washout rates, and lower power consumption were the most frequently mentioned key elements, and therefore have high strategic importance.

Identical pilot-scale conditions have been applied when testing the performance of StarLight and three other major media types; expanded clay, sand and plastics. The results have been used to rank the materials (Table 7.3). The StarLight media has a better than average position overall, for power consumption, and for all aspects of performance. It has an average position in terms of attrition rates, washout rates and media costs. Overall, on a technical basis for the requirements identified by the WWT representatives, StarLight appears to be better than the competition. However, lowering the attrition and washout rates of StarLight are areas for improvement.

Table 7.3: Requirements and Performance of BAF Media

Requirement	No. WWT companies	Performance ranking				Position of StarLight
		St.L.	Ex.Cl	Sand	Plastic	
Better biomass attachment/performance	8	1=	1=	2	3	+
Lower attrition	7	2=	3	2=	1	=
Reduced washout / easily separable	5	2=	2=	1	2=	=
Lower power consumption / reduced backwash velocity	5	1	3	4	2	++
Lower cost of medium	4	1=	1=	1=	2	=
Consistent removal / biomass stability	2	1=	1=	2	3	+
Better solids removal	1	1=	1=	2	3	+
	TOTAL:	9	12	14	16	++

St.L. = StarLight & Ex.Cl = Expanded clay media

For the positions of StarLight; -- worst, - below average, = average, + above average, ++ top

7.3.2.3 The Functions of Secondary Treatment BAF Reactors

BAFs are used as either secondary or tertiary treatment processes. Their potential as a secondary treatment process will be examined primarily. The representatives of the WWT companies were asked to rate the importance of the main characteristics of secondary WWT reactors in general circumstances (Question 2). These parameters can now be ranked from those with the highest average mark to those with the lowest:

1. Good performance in terms of BOD, suspended solids and ammonia removal
2. Low operating costs
3. Low capital costs
4. Good stability to toxicity, and flexibility to volume changes
5. Easy to operate
6. Good aesthetic properties
7. Low footprint

Other important characteristics mentioned by some of the interviewees were:

- Low sludge production / easy sludge treatment
- High COD removal
- Good flexibility for upgrading
- High degree of previous experience

It was explained that the importance of many of the characteristics was very site specific. Many of the WWT representatives said the importance of footprint and aesthetics varied from one to five depending on the site.

The competing secondary treatment technologies were marked for their performance in each of the previously mentioned categories. A summary of the mean marks from the WWT company representatives and the positions of the BAF process compared to other secondary treatment processes are given in Table 7.4.

Table 7.4: Technology and application matrix for WWT processes

Process Feature	AS	ANAs	BAF	EXTs	Fluidised beds	RBC	TF	SBR	BAF position
SS	4.2	2.1	4.3	4.1	3.3	3.4	3.7	4.7	+
BOD	4.6	2.9	4.5	3.2	3.8	3.5	3.8	4.7	+
Ammonia	3.9	0.7	4.4	2.7	4.0	2.8	3.8	3.7	++
Stability	3.4	2.1	3.8	3.8	3.5	2.8	3.5	3.5	++
Flexibility	3.5	3.5	3.5	2.6	2.9	2.8	3.2	3.7	+
Land usage	3.5	3.8	4.5	1.4	4.2	3.4	2.0	3.7	++
Aesthetics	3.4	3.1	4.0	4.5	3.3	3.8	2.5	3.3	+
Op. costs	2.8	4.5	2.6	4.3	2.3	3.6	4.4	3.2	-
Cap. costs	3.7	3.5	3.0	3.0	3.0	2.3	2.4	4.5	=
Operability	3.4	2.3	2.6	4.5	2.0	3.4	4.6	3.5	-
TOTAL	36.4	28.5	37.1	34.1	32.3	31.8	33.9	38.5	+

Note: For the position of BAFs; -- worst, - below average, = average, + above average, ++ top

The highest priority functions, for example good performance and low costs, are veto criteria. However high its other marks are, if scores are very low for these functions the process will never be selected. Intuitively, scores for a veto criterion should be greater than two out of five. Specific examples supply evidence to back up this proposal. For instance, anaerobic processes (ANAs) scored less than two for performance, and none of the WWT companies used the process for sewage treatment. Meanwhile, rotating biological contactors (RBCs) scored just 2.3 for capital costs and are in widespread use. Under certain circumstances where there are additional high priority functions there will be additional veto criteria. For instance, TFs and extensive processes (EXTs) scored below two for land usage. On some sites land usage will be a veto criteria and these processes will not be considered. The veto level for land usage can be as high as 4 out of five on very compact sites. Then, only high rate processes such as BAFs or fluidised beds can be used.

Generally, in terms of substrate removal BAFs and sequenced batch reactors (SBRs) were rated most highly. SBRs were considered the best process in terms of suspended solids removal and BAFs were rated most highly for ammonia removal. The AS process also received above average marks for performance. Fluidised beds and TFs were rated as average, whereas ANAs, EXTs and RBCs were rated as below average, in terms of substrate removal.

The resilience of a process to toxic shocks or changing volumes indicates the overall robustness of a process. The most highly rated process in this area was the BAF process, with AS, TF and SBRs also receiving above average marks. Anaerobic and EXTs were allotted average marks. Meanwhile, Fluidised beds and RBCs were rated as the least robust processes. There was a narrower range of results for this parameter than for any others and none of the processes received an average mark above four out of five. If a process could improve in these areas it would have a significant technical advantage.

Except in the case of EXTs, the processes with lower land usage had higher ratings for their aesthetic properties. The BAF process appeared to be the clear leader in this area. Fluidised beds, SBRs and RBCs were rated as slightly better than the average

processes, AS and anaerobic reactors. TFs were rated as the worst process in terms of land usage.

Only ANAs were rated as having better than average operating and capital costs. The EXTs, TFs and RBCs had good scores for operating costs but below average ratings for their capital costs. Meanwhile, AS and SBRs were scored below average in terms of operating costs, but above average in terms of capital costs. Similarly, BAFs and fluidised beds were scored below average in terms of operating costs, but received average marks for their capital costs.

EXTs and TFs were rated as the most easily operated processes. Meanwhile, ANAs, BAFs and fluidised beds were given below average operability marks.

Processes proposed as competing technology by less than two of the WWT company representatives included membrane bioreactors (MBRs), moving bed bioreactors (MBBRs) and submerged contactors. From these limited results, it was noted that MBRs score very highly in all areas except costs and operability. In fact, the process is disqualified since its score for capital cost was 1.8. It is anticipated that if the capital cost could be lowered, MBRs would be a very competitive process.

It could be argued that the processes which were less familiar to the interviewees, would get lower ratings. The responses to question one showed that on average the WWT company representatives knew most about AS, then TFs, SBRs, BAFs, RBCs, EXTs, ANAs and finally fluidised beds. There seems to be no pattern with the ratings, but this argument should be considered for the fluidised bed results.

In conclusion, from the marks issued by the WWT representatives, BAFs appear to be a technically very competitive process. The total of the marks was only beaten by the total for SBRs. Increasing the operability, and decreasing the operating costs of the BAF process should be research priorities. However, none of the processes were rated highly for both performance and costs. Therefore, it is proposed that at different types of WWT site, the technically most competitive process will alter. This will be examined further in the section 7.3.3.

7.3.2.4 The Functions of Tertiary Treatment BAF Reactors

Looking at the answers from questions 31 to 33, there appear to be three different functions for tertiary treatment. At some sites, the tertiary treatment polishes the effluent, removes solids and increases confidence of meeting consents. At these sites the tertiary treatment may ensure consents are met during storm flows. Sand filters, microstrainers, lagoons, humus tanks or EXTs such as reed beds or grass plots are used. Process choice is often size dependent, since it will often be impossible to treat large flows with EXTs. Operating costs are also important in determining process choice.

Meanwhile, at other sites tertiary treatment is used for further nitrification or nutrient removal. At most of these sites TFs tend to be used. BAFs may be used on compact sites with relatively large flows. As consents tighten there may more opportunities for BAFs to be applied as a tertiary treatment.

Finally, at some coastal sites tertiary treatment is used for the disinfection of effluent released into bathing waters. UV treatment tends to be used in this application.

7.3.3 Technical Segmentation of Secondary WWT Market

Although the general purpose of all secondary sewage treatment plants is the same, there are different constraints and restrictions at different types of site. The technical problem faced by the treatment process therefore changes. After analysing the responses from questions four to nine, the market has been segmented into eight applications, each of which has a different technical problem to solve (Figure 7.4). For each segment the functional priorities, and thus the veto criteria, are different.

Urban				Rural			
Special area		Normal urban area		Special area		Normal rural area	
Site restricted	Site not restricted	Site restricted	Site not restricted	Site restricted	Site not restricted	Site restricted	Site not restricted
A	B	C	D	E	F	G	H

Figure 7.4: Segmentation tree for secondary WWT

The first clear difference, identified by WWT company representatives, was between rural and urban works. Urban sewage treatment works tend to be larger than rural sewage treatment works. This effects process choice since certain processes are more suited to being used on a large scale than others, and smaller sewage treatment works need to be simple and easy to operate without frequent supervision.

It was also noted that at “special” sewage treatment sites, the processes need to be particularly robust and have low sensitivity to toxicity or changing volumes. These sites may have a significant proportion of industrial effluent, or suffer from salinity infiltration. The volume of influent may change due to seasonal population shifts. Alternatively, the secondary reactor may feed a sensitive tertiary process such as an UV disinfection unit.

Some sites are very compact and only certain processes have a small enough footprint to be applied. Additionally, sites that are located in densely populated areas tend to have huge aesthetic constraints. When a totally covered process is preferable, low footprint processes lower the capital costs significantly. Combining these different characteristics produces eight applications, A to H, where;

- A:** Urban site requiring low footprint process with minimal sensitivity to varying volumes/ toxicity.
- B:** Urban site requiring process with minimal sensitivity to varying volumes/ toxicity.
- C:** Urban site that requires a low footprint process.
- D:** Urban site (without particular process requirement).
- E:** Rural site requiring low footprint process with minimal sensitivity to varying volumes/ toxicity.
- F:** Rural site requiring process with minimal sensitivity to varying volumes/ toxicity.
- G:** Rural site that requires a low footprint process.
- H:** Rural site (without particular process requirement).

Identifying the functions and drawbacks to be avoided for processes at each type of site proves that each of the applications has a different combination of constraints (Table 7.5). The segmentation has been verified with the WWT company representatives (Appendix 7D). However, it was noted that the classification of rural sites varied from below 2,000 - 10,000 population equivalency. Also, although many processes would only be used in specific applications, AS was always an option. The type of processes used by each WWT company for the different applications are summarised in Table 7.6.

Table 7.5: Function/ application matrix for secondary WWT processes*

Applications / Functions	A	B	C	D	E	F	G	H
Solids separation	✓	✓	✓	✓	✓	✓	✓	✓
BOD removal	✓	✓	✓	✓	✓	✓	✓	✓
Ammonia removal	✓	✓	✓	✓	✓	✓	✓	✓
Sensitivity to toxicity	✓	✓			✓	✓		
Sensitivity to volumes	✓	✓			✓	✓		
Large footprint	✓		✓		✓		✓	
Aesthetic problems	✓		✓		✓		✓	
High running costs					✓	✓	✓	✓
High supervision					✓	✓	✓	✓

* functions are above, and drawbacks are below the double line

Table 7.6: Processes used for different applications

WWT company	Processes used for different applications A-H							
	A	B	C	D	E	F	G	H
Anglian	AS, MBBR, BAF	AS	AS, MBBR, BAF	AS	BAF	TF	BAF	TF, RBC
Northumbrian	AS, SBR	AS, SBR	AS, SBR	AS, TF, SBR	BAF	TF	BAF	TF, RBC
North West	AS, SBR, BAF	AS	AS, SBR, BAF	AS, TF	AS	TF	AS	TF
Severn Trent	AS	AS, TF	AS	AS, TF	AS	TF, RBC	AS	TF, RBC
Southern	AS	AS	AS	AS	BAF	AS	BAF	TF
South West	AS, BAF		AS, BAF		BAF	AS	BAF	TF, AS, RBC, BAF
Thames	AS	AS	AS	AS	AS, BAF	TF	AS, BAF	TF
Welsh	AS, BAF, SBR	AS, SBR	AS, BAF, SBR	AS, SBR	AS	TF	AS	TF, RBC
Wessex	AS, SBR, MBR	AS, SBR	AS, SBR, BAF	AS, SBR, TF	SBR, AS	AS	SBR, RBC, MBR	TF, AS
Yorkshire	AS, BAF	AS, BAF	AS, BAF	AS	AS, BAF	TF	AS, BAF	TF

If the process marks (Table 7.4) for the priority functions in each application are totalled, an application / technology matrix can be constructed (Table 7.7). All of the selected functions are veto criteria, so a process which receives a mark below 2 will not be considered an option.

Table 7.7: Application/ technology matrix

Applications Process	A	B	C	D	E	F	G	H
AS	26.5	19.6	19.6	12.7	32.7	25.8	25.8	18.9
ANA	V	V	V	V	V	V	V	V
BAF	29	20.5	21.7	13.2	34.2	25.7	26.9	18.4
EXT	V	16.4	V	10	V	25.2	V	18.8
Fluidised bed	23	17.5	16.6	11.1	V	V	V	V
RBC	22.5	15.3	16.9	9.7	29.5	22.3	23.9	16.7
TF	V	18	V	11.3	V	27	V	20.3
SBR	27.3	20.3	20.1	13.1	34	27	26.8	19.8

Note: V= veto criteria, and the top score has been highlighted in bold text.

Comparing Tables 7.6 and 7.7 validates the methodology for assessing processes since the processes with the top functional scores, are also the processes selected by the WWT companies for a particular application. For applications F and H the most suitable process, as determined by the functional scores, and the most frequently used process is the trickling filter. For all applications which have land restrictions (A, C, E & G), BAFs, AS and SBRs are normally considered by the WWT companies as the process options, and these have the top functional scores for those applications. For the remaining applications (B & D) AS, TF, SBR and BAF reactors are used, and these have the highest functional scores. In conclusion, BAFs have the technical properties to make them a highly competitive process particularly for land restricted sites. However, the potential market for the process also depends on the behaviour of the WWT companies.

7.3.4 Customer Analysis

A summary has been made of the processes currently used by each of the WWT companies. Additionally, the attitudes of each of the ten WWT companies towards granular and fixed media BAFs have been reported. Following this customer analysis, the companies can be divided into groups showing similar behaviours.

7.3.4.1 Anglian Water

Approximately 50% of the secondary treatment processes are AS systems, and 50% are TFs. However, TFs are only built now as additions to existing works. There are a few different systems including, RBCs, MBRs, BAFs and MBBRs. The processes selected currently are AS reactors, MBBRs and for smaller works, BAFs (<500 population equivalence or pe). However, these BAF reactors all contain a structured plastic rather than a granular medium.

There are three granular media BAFs in operation each serving 15,000 to 20,000 pe for tertiary treatment. Generally the plants run well but there has been some media loss. The process is considered expensive and difficult to operate and it was thought unlikely that BAFs would be used greatly in the future. MBBRs are being developed by Purac, which is owned by Anglian Water, so they are likely to be the preferred high rate system.

7.3.4.2 North West Water

The AS process tends to be used on large urban treatment works and TFs are used on small, rural works. However, fixed film processes are thought to respond better to toxicity than the AS process. This is very important since about 30% of sites have a significant proportion of trade waste. Many sites that require expansion have restricted land area and are located in highly populated areas. BAFs are already being selected under these conditions and it was predicted their use would increase by 20-30%. Indeed, it was thought that at most new sites BAFs would be considered against other technology. It was suggested that TFs and AS systems would be “used less and less as sewage works look less like sewage works”.

There are currently seven operational BAFs ranging from small to large and being used for both secondary and tertiary treatment. These are running well and the advantages of BAFs were identified as; their flexibility, low capital cost, easy maintenance, resistance to toxicity and small footprint. When asked to identify their weak characteristics it was thought that they were in no way worse than any other process.

7.3.4.3 Northumberland Water

Currently, TFs are the most common secondary treatment process being used in about 80% of the sewage treatment works. AS systems are used for about 5% of the sites, mainly in the larger works. There are also a few RBCs at small sites, a few small package BAFs and reed beds.

AS tanks can be easily converted to SBRs, which are reported to be a promising future process as they are easy to control and optimise. It was also suggested that using a mixture of processes could be beneficial. Already the company has an AS system followed by a BAF, an AS system followed by TFs and TFs followed by reed beds.

There is one large nitrifying BAF (100,000 pe) which, after a few problems during start-up, is working well. Additionally, there are ten small secondary treatment BAFs (<1000 pe) which now perform well for the company. However, there were problems at first since the reactors had been sized for mean rather than peak flows. The initial reason BAFs were selected for these small works was their easy installation. Other advantages of BAF systems were identified as their easy control, their good aesthetic properties and their low land usage. It was considered that natural media allowed a more varied biofilm to develop than plastic media. Media loss and noise problems were the disadvantages associated with using BAFs. Since SBRs are the preferred option for large sites, it was considered that BAFs would continue to be used on small sites with restricted land.

7.3.4.4 Severn Trent Water

Severn Trent Water uses a matrix to select the appropriate process for each site. The matrix plots the pe of the site against the required standard. It has been constructed using the company's years of experience and is adhered to unless the conditions are very unusual.

A large proportion of the sewage works use TFs. However, most of the new large works have AS plants and some small plants use RBCs. They have two Tetra BAFs, which have been built for political reasons, since very few sites in the Severn Trent region have land restrictions. BAFs are considered to have high running costs and to be difficult to operate. Therefore, with the current technology there are likely to be few BAFs built in this region in the future.

7.3.4.5 Southern Water

75% of the sewage treatment sites are small and rural treating less than 5,000 pe and using TFs or fixed media BAFs. The remaining 25% of sites are larger works treating over 5,000 pe usually with AS systems. There are flow variation problems at about 5% of the sites, due either to high level of industrial waste or seasonal population changes in rural sites.

There are two granular media BAFs operating in this region, and the interviewee thought that they may be used more (approx. 0-10%) in the future on compact sites. Up to 50% of the sewage treatment sites that require expanding are reported to have land restrictions. As well as being compact, BAFs are considered to be good at dealing with saline influents. The weaker characteristics of the process are considered to be their operational expense, which is associated mainly with the backwashing procedure. This may explain their stated preference for fixed media rather than granular media BAFs (British Water Document No: MU420, 1998).

7.3.4.6 South West Water

There are about 600 small sites treating less than 1,000 pe (approximately 90% of sites). Many of these use TFs, but very few new TFs are being built as package AS systems are thought to have lower life-time costs. Additionally, the applicability of membrane systems where there are land restrictions on small works is being

investigated. Urban sites make up the remaining 10% of sites and tend to use either AS or BAFs. SBRs are now considered a good alternative to AS to reduce the process footprint.

There are approximately 15 BAFs operating within the region for both secondary and tertiary applications. The secondary BAFs tend to be large (over 10,000 pe) and are currently working well. However, there have been major problems due to the BAFs being incorrectly sized. Indeed, two of the largest BAFs had to be expanded. They found that when operating at loadings around 6 kg BOD/m³/d although the daily average met consents, spot sampling did not meet the 95 percentile levels. They reported high suspended solids levels in the effluent after backwashing. Due to these experiences the company would now prefer to use AS systems instead of BAFs unless a very small footprint process is required.

The main advantage sought from a new BAF medium would be enhanced solids retention. Although the company has not yet changed the medium in any of their BAFs, they would consider doing so if they were convinced of the advantages of a particular material.

7.3.4.7 Thames Water

Thames Water uses AS reactors for most urban sites and TFs for most rural sites. Land restrictions rarely influence process selection and there are few sites receiving significant levels of industrial effluent, since their trade effluent policy encourages companies to have their own effluent treatment. The traditional processes have operator acceptance, whereas many novel processes such as BAFs are considered difficult to operate.

There is currently one BAF operating at Silchester, which treats 26,000 m³/d and two containerised BAFs have been used temporarily in the past. Start-up problems have included excess media loss, which required the launder design to be upgraded. Additionally, it had been noted that due to operator inexperience small plant failures were frequently transformed into big problems. Since the major advantages of the process were not required within the region it was predicted that they would not be used frequently in the future.

7.3.4.8 Welsh Water

90% of the WWT works are small, rural sites treating a total of 10% of the pe and using TFs or RBCs. The remaining 10% of sites are large treating a total of 90% of the pe. Industrial wastes are significant at between 10 and 20% of the large urban works and 90% suffer from land constraints. Most of the urban sites use the AS process. Process selection is generally likely to follow this pattern however, it was mentioned that there are “fashions”. Three years ago BAFs were in vogue and now SBRs are very popular. It was predicted membrane processes would gather attention in the next few years.

There are two operational BAFs in the region, treating 10,000 and 20,000 pe. The benefits of using BAFs were identified as their small footprint and aesthetic properties. However, Welsh Water is well aware that there have been problems with badly designed BAFs particularly in the South West Water region. Although BAFs were currently unpopular, it was thought the process might be revisited in the future for compact sites.

7.3.4.9 Wessex Water

Currently, SBRs are the preferred secondary treatment process for large sewage treatment sites. Meanwhile, for smaller sewage treatment works TFs with a plastic medium are the preferred option.

Wessex water has one large BAF in operation in Poole. This site has major land restrictions and aesthetic concerns, and the BAF performs well under the conditions. However, the plant has high power consumption and is complicated to run. It was considered that in the future, BAFs are only likely to be used when the site conditions are as restricted as the Poole works.

7.3.4.10 Yorkshire Water

Currently, AS and its variations are the most frequently used secondary treatment processes. TFs are considered “old technology” and very few are being built due to their high capital cost.

BAFs are often used instead of TFs and Yorkshire Water has seven operational BAFs ranging in size from 5,000 to 400,000 pe. They are running well and it was predicted that BAFs will be used approximately 20% more in the future, particularly at sites with land restrictions. The benefits identified for BAFs were that solids removal can be controlled, the reactor can be compartmentalised and high loading rates reduce the reactor footprint making it an aesthetically pleasing process. Meanwhile, the weakest characteristics of the BAF process were considered to be the attrition rate and cost of the media. A 6% loss of medium was reported in the first year's operation from a Biocarbene BAF. Other improvements required from altering BAF media, were lower backwashing velocities, increased capacity and reduced washout.

7.3.5 Process Selection

Most of the WWT company representatives estimated that from 1995 to 1999, between 10 and 30 new secondary treatment plants were commissioned. However, they predicted that between 2000 and 2005, only 1 to 5 new secondary treatment plants would be introduced.

The need for new plant is identified by either a capital planning group that monitors sites and consents, or directly by the operational managers. An economics group normally monitors cost guidelines and there can be conflicts due to different groups trying to reduce operational and capital costs. Options studies are frequently used to assess the costs and operation of three or more alternatives in detail before process type is selected. Many people have an influence on process selection including research and development, the operators and the capital group. The final decision makers usually include the operational or production managers, and may even go to the board. There are between 3-10 operational managers in each company who are each responsible for a specific region. The whole decision process usually takes about 12 months, although very large works may need up to 5 years planning.

There are risks associated with changing from the well established AS and TF processes. These risks are that process performance, sludge production and operation are all unknown. Often, risks could be reduced if there were time for a pilot-scale plant to test the process performance at a specific site. Another method of reducing process selection risks is by using turn key contracts.

7.3.6 Behavioural Segmentation

Generally two types of customer can be distinguished according to their behaviour. Firstly, technically focused clients have a technical problem to solve and are willing to co-operate with suppliers whose product has the potential to solve it. Meanwhile, clients with an industrial purchase focus are satisfied with their existing technology and have fewer incentives to aid process development. These are also the companies that have invested more heavily in research and development (Table 7.8).

Following their UK introduction in the last decade, there have been mixed reports on granular media BAFs, which now influence the process selection behaviour of the WWT companies. Most of the negative experiences were associated with badly designed plants and were costly and difficult to rectify. Thus, despite acknowledging the advantages of well designed BAFs, some WWT companies are reluctant to use them again. The technically and industrially focused companies can be sub-divided into those who hold favourable opinions about granular media BAFs, and those who have had difficulties with the process. Using these differences, the WWT companies have been categorised into four groups (Figure 7.5).

Table 7.8: Comparison of R&D investment and the influence of consultants

WWT Company	R&D* £'000	R&D* % of profit	R&D* % of sales	Consultants influence on process	
				selection	design
Thames	8400	1.9	0.7	High/medium	High
Anglian	6600	2.3	0.8	Low	Medium
Severn Trent	6300	1.4	0.5	Very low	Very low
Northumbrian	3400	1.7	0.7	Medium/low	High
North West	4700	1.1	0.2	Medium	High/medium
South West	2000	1.1	0.6	High/medium	Medium
Yorkshire	1100	0.4	0.2	High/medium	High/medium
Welsh	900	0.3	0.1	High/medium	High/medium
Wessex	400	0.3	0.2	Low	Medium/low
Southern	N/A	N/A	N/A	Low	High

* 1997 figures from DTI: The UK R&D Scoreboard 1998

Opinion about granular media BAFs		
	Negative	Positive
Industrial purchase focus	Anglian Water Severn Trent Water	Northumberland Water Thames Water
	Southern Water South West Water Welsh Water Wessex Water	North West Water Yorkshire Water

Note: *Red = group 1, yellow = group 2, blue = group 3 & green = group 4*

Figure 7.5: Segregation of WWT companies by their behaviour as a BAF customer.

The customers with an industrial purchase focus reason in terms of risk. The risks associated with selecting a novel process are inhibitory unless the process has been proven to provide a technical or financial benefit. However, it should be noted that the companies that currently have an industrial purchase focus, might have a technical focus if regulations introduce new technical problems. Technically focused customers reason in terms of motivations. The types of motivation which influence process selection for these customers are; statutory, to comply with regulations; commercial, to increase profits; industrial, to implement a policy of quality and sometimes fashion, to promote the company image.

The technical problem faced by North West Water and Yorkshire Water (Group 4) is that stable robust processes with a small footprint and good aesthetic properties are required to cope with high concentrations of industrial effluent at urban sewage works. The other technically focused clients also have a problem with process footprint but have fewer concerns with industrial effluents (Group 3). These customers have a high proportion of coastal sites with land restrictions because until recently, WWT was not required prior to using sea outfalls.

The industrial purchase focused clients (Groups 1 and 2) tend to have fewer land restricted sites because sea outfalls have never been predominant. They are also located in regions which produce less industrial effluent, or like Thames Water, have

an effective trade effluent policy which forces large companies to install their own treatment plants.

Some of the customers have a negative opinion about the prospects for the granular media BAF process because they favour another high rate WWT process. Preferences may sometimes be based on political influences as well as process merits, such as Anglian Water's selection of the MBBR process. However, Southern Water perceives that granular media BAFs have high operating costs and operational difficulties and therefore, prefers to use fixed media BAFs. Other negative opinions have been derived from unfavourable past experiences with the process. South West Water in particular has suffered from BAFs being under designed to meet regulations when using UK testing methods. South West Water may have the highest requirement for high rate processes but they will have to be convinced that the problems they experienced previously have been overcome. In conclusion, the companies in segment 4 should be targeted first, followed by those in segments 3, 2 and 1.

7.3.7 Segmentation Matrix

The segmentation matrix combines each of the eight applications with each of the four customer groups (Figure 7.6). The size of each of the resulting segments can then be estimated using the data acquired from the representatives of the WWT companies with the e-mailed questionnaire (Appendix 7D).

Applications Customers	A	B	C	D	E	F	G	H
Group 4								2
Group 3	2.25	1.05	5.2	4.35	0.425	0.775	1.25	4.7
Group 2	0.5		0.5	3.5			0.5	1.75
Group 1				7.5				2.5

Figure 7.6: Segmentation matrix for secondary WWT (numbers in millions of pe)

In segment 1 (4A-D highlighted in purple), BAFs are a process option for all the urban sewage works of the companies in group 4. It is estimated that approximately 8 million pe is treated by this segment. Half of this pe is treated on sites suffering from either land restrictions, significant levels of industrial waste or both. Therefore, the stability, robustness and small footprint of the BAF process should be emphasised to sell into this segment.

In segment 2 (3A-C highlighted in pink), BAFs are a process option for some of the urban sewage works which treat approximately 8.5 million pe for the companies in group 3. In segment 3 (3E-G highlighted in burgundy), BAFs are a process option for some of the rural sewage works which treat approximately 2.45 million pe for the companies in group 3. All the sewage works in both these segments have either land restrictions, a particular requirement for a robust process or both. Again, the small footprint and robustness of the process should be emphasised, but most importantly the problems and disadvantages previously reported for BAFs must be avoided.

In segment 4 (2A-C highlighted in turquoise), BAFs are a process option for some of the urban sewage works which treat approximately 1 million pe for the companies in

group 2. All the sewage works in both these segments have either land restrictions, a particular requirement for a robust process or both. The footprint is an important factor for this segment but the whole-life costs of the process should also be minimised.

In segment 5 (3-1D highlighted in grey), BAFs are a process option for the standard urban sewage works of the companies in groups 1, 2 and 3. This segment treats approximately 15.35 million pe, however, process selection will be based on the whole-life costs of equivalent processes. Similarly, in segment 6 (4-1H highlighted in olive) BAFs could be selected for the standard rural sites which treat approximately 11 million pe for the companies in groups 1, 2, 3 and 4, but costs would dominate process selection.

Although Greenfield sites are still needed in some regions, most new process plant is required at existing sites to extend treatment capacity or to replace ageing equipment (British Water Document No BW421, 1999). The existing segment sizes will therefore be used to predict opportunities in each segment.

7.3.8 Quantifying the Potential in Each Segment

A review is currently in progress to plan and prioritise the WWT companies capital investment during the period between 2000-2005, which is referred to as AMP 3. In March 1999 the government made their final decision on the quality programmes and in November 1999 OFWAT will set their price limits. It has been forecast that by March 2000, the WWT companies will have submitted their plans, including timings and dates for major projects, and that OFWAT will publish the quality submissions at the end of May 2000 (British Water Document No: MU442, 1998). British Water has planned a series of seminars at the end of 1999 to explore potential new ways of working in AMP 3. The results will be published in a report to be launched at a conference in January 2000 (British Water Document No: BW 421, 1999).

Until these documents are available it will be difficult to accurately assess the potential of each segment. However, the WWT representatives have made predictions about the type of sites which may require capital expenditure during the next five years. Since the interviews were conducted between June 1998 and February 1999, these predictions were likely to have been made based on the WWT companies preliminary quality strategy proposals. OFWAT (1998b) have reviewed these preliminary plans in the light of government guidance and suggested where additional investment may be required.

7.3.8.1 Group 4

The representative from Yorkshire Water predicted approximately 10 sites with land restrictions, would require capital expenditure on their WWT processes, within the next 5 years.

Meanwhile, the representative from North West Water anticipated moderate expansion with some sites having land restrictions. When assessing the company strategy of North West Water, OFWAT (1998b) concluded that it was likely that more investments in WWT processes than planned were necessary. They identified the north west region as having the highest proportion of rivers with poor water quality, and the lowest compliance with the bathing water directive (OFWAT, 1998b).

7.3.8.2 Group 3

The priority for Southern Water was identified by both the interviewee and OFWAT (1998b), as introducing secondary treatment to all coastal sites that currently have HNDA status. It was estimated that approximately half of these sites have land restrictions. Additionally, further plans for the shell fish waters directive and to meet river quality objectives (RQOs) may be necessary (OFWAT, 1998b).

The representative from Welsh Water predicted that between 60 and 70% of their smaller treatment plants would need upgrading and that 70% of these could suffer from land restrictions.

The representative from South West Water estimated that plants treating 10-15% of the population equivalency would require investment in the next five years. Some of these would be urban sites and others rural, but most of the urban sites would have land restrictions. OFWAT (1998b) suggested further provisions may be necessary to meet the fresh water fish directive.

The representative from Wessex Water predicted investment may be necessary at 20% of their larger plants (>10,000 pe) and half of these would have land restrictions. OFWAT (1998b) suggested that further expenditure may be needed at coastal sites to meet government expectations, and pointed out that no plans had been made to improve compliance with RQOs.

7.3.8.3 Group 2

The representative from Northumberland Water predicted that 5% of sites would be expanded, but all these would be the large, urban sites, and most would have restricted footprint. OFWAT (1998b) suggested that further investment in WWT processes may be required in the light of government guidance on bathing waters and RQO.

In the Thames region, it was expected that 70 sites would be upgraded mainly to ensure compliance with the UWWTD. It was predicted that small footprint may effect process selection on 20% of these sites.

7.3.8.4 Group 1

The representative from Severn Trent Water predicted that between 20 and 30 WWT sites may need investment but that very few would suffer from land restrictions.

The representative from Anglian Water expected 4 or 5 large sites would require expansion. However, OFWAT (1998b) reported that the company had not anticipated the withdrawal of HNDA status and that the company’s plans for improving RQO may be insufficient. Further investment in WWT processes may therefore be required.

7.3.8.5 Quantification of media requirements in a BAF

The pe of a reactor must be converted to quantify the weight and volume of media required in a BAF. Water consumption has been estimated at between 100-150 l/ capita/ d and approximately 95% of this used water returns to the sewer for treatment (Butler 1996). Thus, it is reasonable to suggest that wastewater production is approximately 125 l/ capita/ d. A medium strength sewage contains 220 mg BOD/ l, so BOD production is approximately 27.5 g BOD/ capita/ day.

Maximum loading rates to BAFs vary not only with performance but also with the value selected to represent the reactor volume. Using StarLight C effective treatment has been proved at pilot-scale at loading rates of up to 15 kg COD/ m³ working volume/ d. This corresponds to 6 kg COD/ m³ filter volume/ d and 2.4 kg BOD/ m³ filter volume/ d. Most biological WWT processes are designed to treat three times the mean loading rates so that they are able to handle diurnal variations. Using these estimates the mass of media required for three different sizes of BAFs has been calculated (Table 7.9).

Table 7.9: Quantification of the media requirements in three different sizes of BAF

BAF pe	kg BOD/ day	kg BOD/ day design limit	Media volume required (m ³)	Media mass required (tonnes)
1,000	27.5	82.5	34	15.3
10,000	275	825	344	154.8
50,000	1375	4125	1719	773.6

7.3.9 Analysis of Current BAF Suppliers

Many civil engineering companies are now able to offer their own version of a BAF, including Lurgi Bamag Ltd, Kvaerner plc, and Mowlem Engineering Projects. However, there are several generic BAF systems which have been proven in full-scale applications, and details of the companies involved are given in Table 7.10.

The representatives of the WWT companies were asked to recall the names of as many BAF suppliers as possible and to comment on any advantages or disadvantages associated with each company (Table 7.11). Degremont UK Ltd was the most frequently mentioned company followed by OTV Birwelco Ltd., Brightwater Engineering Ltd., Patterson Candy Ltd (or PWT Projects Ltd), Capital Controls Ltd. and then Purac Ltd. Degremont UK Ltd and OTV Birwelco Ltd are French companies and both have over 40 operational BAFs throughout Europe. Brightwater Engineering Ltd has over 25 operational plants in the UK. Both Capital Controls Ltd and Binnie, Black and Veatch Ltd are American companies and Capital Controls Ltd has over 30 plants in Europe and over 100 world-wide.

Table 7.10: Details of granular media BAF suppliers

Company name	Process	Material	Density	Flow direction	Subsidiary	Turnover £ '000s*	Employees*
Brightwater Engineering Ltd	Biobead	Plastic	Floating	Upflow	N/A.	1182	15
Degremont UK Ltd.	Biofor	Biolite	Sunken	Upflow	Suez-Lyonnaise des Eaux	22516	95
OTV Birwelco Ltd.	Biostyr	Plastic	Floating	Upflow	Compagnie Generale des Eaux	28979	56
Patterson Candy Ltd. (formerly PWT Projects Ltd., a subsidiary of Thames Water)	Biocarbene	Expanded Shale	Sunken	Downflow			
	SAFe	Expanded Shale	Sunken	Downflow	Binnie, Black & Veatch Ltd.	5840	223
Purac Ltd.	Stereau	Pumice	Sunken	Downflow	Anglian Water plc	51477	143
Capital Controls Ltd.	ColOX	Sand	Sunken	Up/downflow	Joint venture with Severn Trent plc for Europe	3570	4

* Source: Business & Industry, Volumes 1 & 2, September 1997-1998

Table 7.11: Company reputations

Company name	Comments from WWT companies		Known by the WWT companies below:									
	Favourable	Unfavourable	A	N	NW	ST	S	SW	T	WL	W	Y
Brightwater Engineering Ltd	Experience with small plants & cheaper	Less robust process	✓				✓	✓	✓	✓	✓	
Degremont UK Ltd.	Good research & experience with large plants		✓		✓	✓	✓	✓	✓	✓	✓	
OTV Birwelco Ltd.	Experience with large plants, good research & trustworthy		✓		✓	✓		✓		✓		✓
PWT Projects Ltd.			✓			✓		✓	✓	✓		
Purac Ltd.			✓					✓				
Capital Controls Ltd.			✓			✓		✓				

7.4 DIAGNOSIS OF THE MARKETING SITUATION

Diagnosing a marketing situation attempts to measure the risks of failing in each segment, and so should be used as a guide to the order in which different segments are attacked. There are two parts to diagnosing a marketing situation. Firstly, the technical diagnosis assesses the technical risks associated with the project, and then the commercial diagnosis evaluates the commercial risks. A method of systematically evaluating these risks has been developed by Millier (1989, 1999).

The technical diagnosis assigns the project with marks from 0 to 4 for five parameters of technology related risk, and five parameters of risk related to the technological environment. Low marks are given to criteria that slow down product diffusion, and high marks to criteria that promote it. When the average marks in each risk category are plotted against each other, the trajectory length is the overall technical risk (technical index) for that segment. As in section 7.3.2.3, there are some veto criteria and if they receive a mark below two, the average does not need to be calculated since the maximum risk then applies. The commercial diagnosis is similar and involves assigning the project with marks from 0 to 4 for eight parameters of the company's advantages and shortcomings, and eight parameters of market appeal and constraints. Finally, the commercial index is plotted against the technical index to obtain the overall strategic index. The completed spreadsheet for segments 1 to 6 with all the criteria listed can be found in Appendix 7E

The marks were assigned intuitively using the information gathered during the interviews with the representatives of the WWT companies, the Environment Agency and ECCI. Under different circumstances the marks will change and thus strategies which minimise the risks associated with the project can be recommended.

If ECCI were to select one or more of the BAF suppliers as either a partner or as an intermediate client, the strategic index for all segments decreases significantly (Figures 7.7 and 7.8). Segments 1 and 2 move from the medium risk area into the lower risk area and segments 3 and 6 move from the high risk area to the medium risk area.

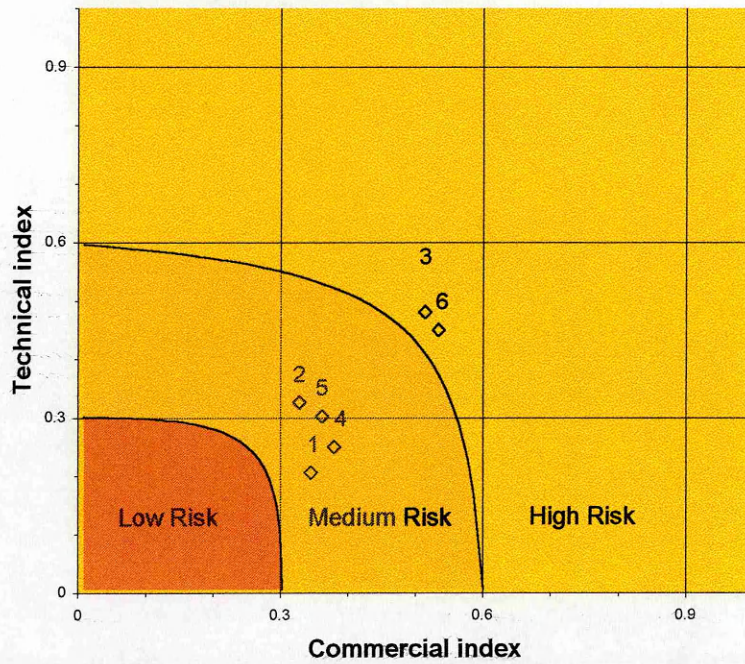


Figure 7.7: Diagnosis if ECCI worked alone under present circumstances

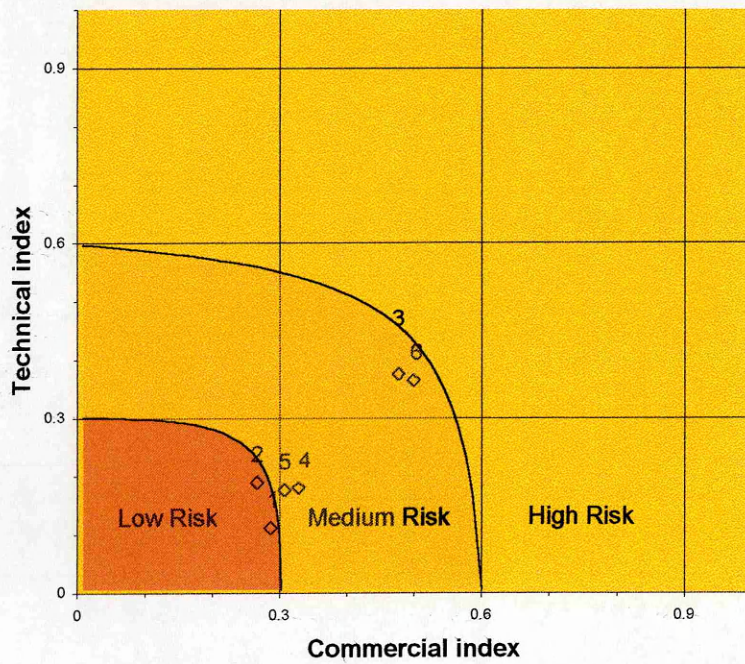


Figure 7.8: Diagnosis if ECCI worked with or sold media to, a BAF supplier, under present circumstances

Segments 1 and 2 have the lowest associated risks and are urban applications. If these were to be targeted first, it would be advisable to select a partner which has experience with larger BAF plants. Although it has not been examined specifically by this study, the international markets also need to be considered. The two French companies are dominant throughout Europe whereas the American markets may be more accessible if either Capital Controls Ltd or Patterson Candy Ltd were selected as partners.

The customers in group 3 are in close geographical proximity in the south of England and have similar WWT problems and constraints. They have all been influenced by unfavourable experiences with BAFs in the last decade. An entry into group 3 may be gained, by testing StarLight C at full-scale in one of their existing sub-optimal plants. South West Water has approximately 15 BAFs treating population equivalencies between 10,000 and 100,000 at either secondary or tertiary stages. They use a mixture of upflow and downflow configurations combined with either plastic or mineral media. The representative for South West Water said that they had never swapped a BAF media, but that they would consider doing so.

If the customers in group 3 were convinced that the problems they experienced previously with BAFs have been solved, the technology's image may be restored. The environment agency would also have to be re-convinced about the BAF process, as they evaluate all plant proposals (British Water Document No: MU397). Altering their opinions would change the marketing diagnosis, as shown in Figures 7.9 and 7.10. The strategic risk in attacking segments 2, 5 and 3 would drop below that for segments 1, 4 and 6 respectively.

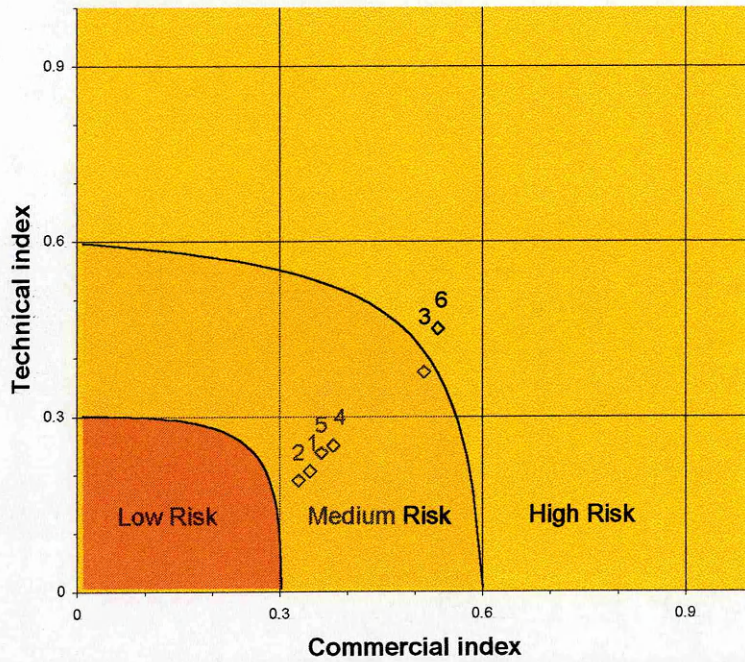


Figure 7.9: Diagnosis if ECCI worked alone but restored the image of BAFs with the companies in group 3

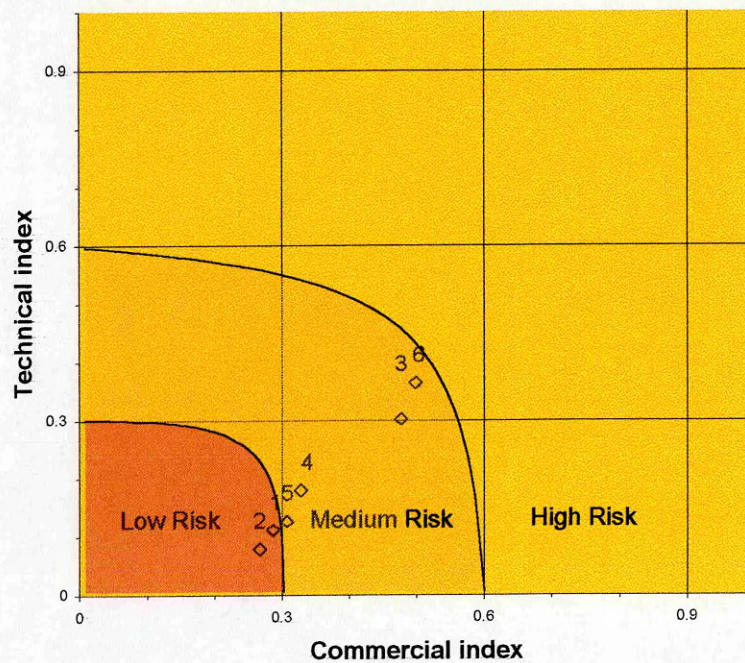


Figure 7.10: Diagnosis if ECCI worked with a partner and restored the image of BAFs with the companies in group 3.

7.5 CONCLUSIONS

1. The high operating costs of the BAF process are its major disadvantage, therefore StarLight C has favourable BAF media properties. The technical properties of the BAF process make it most suitable for urban sites with either a small footprint or a variable wastewater.
2. Some of the WWT companies have a technical focus because the traditional WWT processes are unsuitable for small footprint sites. Meanwhile, other WWT companies have an industrial purchase focus and are generally satisfied with the traditional WWT processes.
3. BAFs are considered to have good technical attributes but they have been badly or under designed several times in the UK during the last decade. On these occasions the plants experienced many operational difficulties and subsequently some of the WWT companies have a negative opinion about the BAF process.
4. Until OFWAT set the price limits, it will be difficult to accurately assess how many BAFs will be built during AMP 3. The views from the WWT companies are that between 50 and 250 new secondary WWT plants will be commissioned.
5. To reduce the risks of product failure ECCI should sell their media to an existing BAF supplier. They should also arrange to conduct full-scale trials with South West Water.

CHAPTER 8

IMPROVING THE PROPERTIES OF FOAMED CLAY BAF MEDIA

ABSTRACT

Materials used previously as biological aerated filter (BAF) media have not combined optimal biofilm supporting properties with optimal wear characteristics. The friability of foamed clay variants depends on the proportions of bentonite, surfactant, alumina and sawdust used during production and the firing temperature. A foamed clay medium made with 56 g/l bentonite solution and fired to 1400 °C, exhibited 1% weight loss after attrition testing.

A method was developed to measure the surface roughness of the foamed clay media. The biofilm retention properties of variants exhibiting reduced surface roughness were not compromised, under the conditions tested. It is proposed that surface roughness influences biofilm formation by providing sites for glycocalyx attachment. The rough surface of foamed clay may have an excess number of attachment sites and so, altering the degree of surface roughness does not effect the biological properties of the medium.

8.1 INTRODUCTION

A biological aerated filter (BAF) is a three phase, wastewater treatment (WWT) reactor consisting of a submerged particulate medium bed with applied air. A biofilm grows on the surface of the BAF medium and metabolises the organic matter in the sewage. When autotrophic bacteria are present, ammonia is also removed. Additionally the medium acts as a depth filter by removing solids from suspension. BAFs are backwashed periodically to remove both entrapped solids and excess biofilm.

Materials used currently as BAF media fall into two categories. Mineral media such as expanded clays and shales show excellent biological performance. They have rough surfaces which promote biofilm retention (Baker, 1984) but tend to be prone to attrition (Kent *et al.*, 1996). Meanwhile, although very low attrition levels are associated with sand and plastic materials, inferior reactor performances have been reported using these media as demonstrated in Table 8.1.

Table 8.1: Performance of three media in identical reactors during stable operation with an influent of settled sewage at a flow rate of 0.3 l/min.

Medium	% Removal		
	Suspended solids	Soluble COD	Ammonia
Expanded clay ⁺	90	78	77
Sand*	76	76	27
Plastic [#]	60	61	11

⁺ chapter 6, * Mendoza-Espinosa and Stephenson, 1998, [#] Mann *et al.*, 1999.

The clay foaming process has been developed by ECC International Ltd., as described in British Patent Application no. 9321617.4. The ceramic foam can be extruded and calcined to produce porous granules. The potential of foamed clay, in applications exploiting the material’s insulating properties, is currently being investigated. Meanwhile, as the granules have a low density and their surfaces are rough, they are highly suitable as a BAF medium. Since the production of foamed clay is controllable, the characteristics can be altered specifically for this application.

Representatives of the WWT companies identified seven properties that were required to improve BAF media (Chapter 7). In the order that they were most frequently mentioned these were; better biomass retention; lower attrition rates; easier separation of particles from backwash effluent; reduced backwash velocity; lower cost; consistent substrate removal during the filter cycle; and improved solids removal.

The price of foamed clay has not yet been precisely determined and will partly depend on the total demand for the material in all its applications. It has been assumed that price will not be the differentiating parameter. Results from pilot-scale studies (Chapters 5 and 6) indicate that the foamed clay StarLight C has an above average position, compared to currently available BAF media, for most of the other requirements. However, it has an average attrition rate and is not particularly easy to separate from the backwash effluent. Therefore, these parameters should be improved without impairing existing properties to give StarLight C a competitive advantage as a BAF medium.

8.2 MATERIALS AND METHODS

8.2.1 Foamed Clay Variants

The main constituents of the foamed clay are surfactant (cationic), air, water and china clay. However, bentonite or alumina can also be added. Bentonite increases the proportion of the glassy phase, whereas alumina increases the proportion of mullite in the fired clay. The ratio of these additives may alter the strength of the fired medium. Sawdust added either to the feed clay or the unfired medium surface will be burnt off during firing, which may increase the surface roughness of the medium. Changing the proportions of these constituents allowed seven variants of foamed clay to be manufactured (Table 8.2).

Table 8.2: Constituents of unfired foamed clay variants

Variant (V)	Bentonite solution (g/l)	Surfactant % w/w	Alumina solution (g/l)	Water % w/w	Sawdust %w/w	Sawdust coated
1	35	2	0	51	0	No
2	56	2	0	51	0	No
3	35	5	0	48	0	No
4	56	2	155	43	0	No
5	35	2	0	57	0	No
6	35	2	0	51	10	No
7	35	2	0	46	0	Yes

Mixing under vacuum, 100 g of V1 and 1 g silane (SILQUEST A-137) in 250 ml of acetone, produced an eighth variant (V8). The vacuum was released slowly forcing the liquid into the pores of the foamed clay. Excess acetone was then allowed to evaporate.

Variants 1 to 7 were calcined at maximum temperatures of 1500°C and 1300°C. The kiln temperature was raised at 1°C per minute, and then maintained at the target level for 90 minutes. Additionally, V1 was calcined at a target temperatures of 1400 °C.

8.2.2 Physical Tests

The prill density and porosity were analysed for all the media variants by mercury porosity (140 & 240 Pa, CE Instruments, Italy). The proportions of crystalline substances in V1 samples fired at 1300 °C, 1400 °C and 1500 °C were analysed using X-ray diffraction, XRD (pw1825 and 1729, Philips, Cambridge, UK). Elemental analysis was carried out on the V1 samples by measuring X-ray frequencies, XRF (pw1480, Philips, Cambridge, UK).

A crush resistance test was used instead of the standard friability test (Humby *et al.*, 1996) since it was thought to be more sensitive to the relatively small changes in media strength. Approximately 20 g of each sample was constrained within a stainless steel piston (50 mm diameter) and subjected to a compressive force using a laboratory tensometer (Monsanto 10) with a 10 kN loadcell. As the compressive stress (σ_c) increased, the strain (ϵ_l) resulting from the fracture of the prills was plotted. The crushing modulus (E_c) was derived from the gradient of the linear portion of the graph and was used to compare variants.

The standard attrition test was performed on variants showing increased crush strength. Media samples were sieved before and after attrition testing to allow any changes in the mean hydraulic diameter to be observed (BEWA, 1993).

Standard methods for measuring surface roughness require a base line corresponding to the overall shape of the object (Dagnall, 1980). This proved to be very difficult to provide for granular prills, which have no standard form. The comparative roughness of the foamed clay variants was therefore measured using image analysis (Quantimet 520, Cambridge Instruments, UK) of sections of the prills embedded in resin (resolution 0.011 mm/pixel). The measured surface areas of the prill sections were used to calculate the equivalent perimeter of a circle. Ratios of the actual perimeter to the equivalent perimeter were plotted against the corresponding shape factors (maximum diameter/ minimum diameter).

8.2.3 Biological Tests

The foamed clay with a high bentonite content (V2), the variant which was coated in sawdust prior to calcination (V7) and standard prills coated in hydrophobic silane (V8) were all compared to the standard foamed clay (V1) as media in small-scale reactors (Figure 8.1). In all cases the variants were calcined to a maximum temperature of 1500 °C.

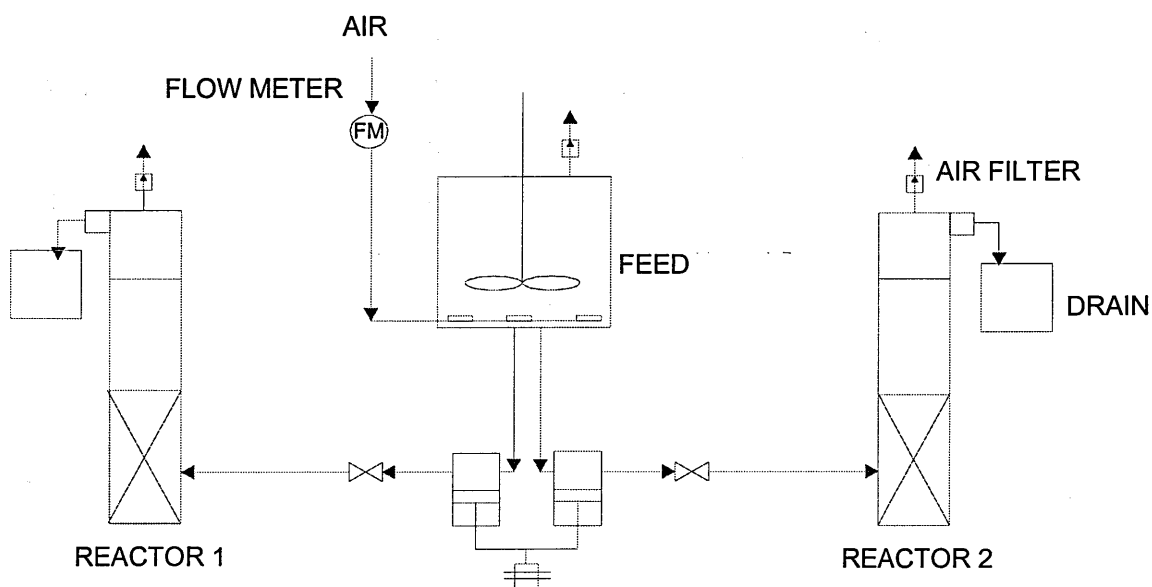


Figure 8.1: Schematic of small-scale reactors

Two identical sterilised reactors were supplied with oxygenated synthetic sewage (Sterritt & Lester, 1981) containing 330 mg/l chemical oxygen demand (COD). An average feed rate of $7.8 \times 10^{-5} \text{ m}^3/\text{h}$ was maintained with variable rate peristaltic pumps (302S-220, Watson Marlow, UK). The working volume of the reactors was 50 ml, the hydraulic residence time was 40 minutes and the liquid velocity ranged between 0.2 - 2.0 m/h. A medium strength sewage contains approximately 500 mg/l COD before settling (Metcalf & Eddy, 1991). Typical BAFs have residence times between 20-90 minutes and liquid velocities between 1.7-3.5 m/h (Mann *et al.*, 1999). The conditions were therefore a good representation of a secondary treatment BAF.

The lower part of the reactors contained loose particles of the medium whilst the middle part contained media attached to stainless steel slides to allow environmental scanning electron microscopy (ESEM) of the biofilm growth (results in Chapter 9). To reduce preferential channelling through the reactor, the top section was constructed to act as an overflow weir.

During start-up, an inoculated feed was supplied at the nominal flow rate for 24 hours. This consisted of a 200 ml culture of *Pseudomonas aeruginosa* (Chemviron), grown over 24 hours to an optical density of approximately 1.0 at 600nm, in 10 l of synthetic sewage. Influent and effluent samples were collected for the first 10 days of operation, and the concentration of soluble COD was determined to observe differences in the biomass retention properties of the media. It was necessary to compare each variant against the standard directly since there was no temperature control and the effects of ESEM analysis on biofilm viability are unknown.

8.3 RESULTS

The prill density, porosity and surface roughness results for variants 1 to 7 when calcined to 1500°C are summarised in Table 8.3. Additionally, E_c for variants 1 to 7 fired at both 1300°C and 1500°C are displayed. All the variants exhibited a greater E_c when calcined at 1500°C instead of 1300°C. However, under both firing regimes V2 showed the highest E_c followed by V1 and V5.

Table 8.3: Prill density, porosity and surface roughness for the seven variants calcined at 1500°C and E_c for variants calcined at both 1300°C and 1500°C

Variant	Prill density (kg/m ³)	Prill porosity %	E_c		Surface roughness	
			1300°C	1500°C	Gradient	Intercept
1	906	66.4	10.5	13.9	-0.084	0.86
2	945	65.0	11.4	14.3	-0.080	0.95
3	1347	50.1	6.6	9.6	-0.086	0.85
4	885	67.0	8.3	11.1	-0.086	0.88
5	815	69.8	10.5	12.7	-0.084	0.94
6	986	59.2	6.9	9.1	-0.079	0.71
7	940	60.1	9.5	9.8	-0.088	0.78

The method used to measure the E_c of the variants was simple and effective. For each plotted graph, most of the data showed a consistent straight-line gradient. This indicates that prill breakage was the major cause of strain, and prill recompaction during the test was of minor importance. V2, which contained high levels of bentonite, showed slightly better resistance to crushing than V1. Correspondingly, in the attrition test, V2 exhibited a 1% weight loss compared to the 3% shown previously by V1 (chapter 6). The mean hydraulic diameter of V1 was reduced from 2.1 mm to 1.7 mm after the attrition test, indicating that the prills were breaking up. Meanwhile, the mean hydraulic diameter of V2 remained 2.1 mm before and after attrition testing.

The method used to measure surface roughness requires further testing and validation but is proposed to be sufficient to identify the order of roughness within the variants. The plot of the perimeter ratio against the shape produced straight lines with similar

gradients (-0.084 ± 0.003) for all variants (Table 8.3 and Figure 8.2). This confirms that the relationship between shape and surface area is maintained for all seven variants. However, the intercept of the line varied between 0.71 and 0.95 and so is proposed, here, to be a measure of surface roughness, with high intercepts indicating low surface roughness.

The prill density of V1 increased with the maximum firing temperature (Table 8.4). The crystalline structure and the strength modulus were not effected by reducing the maximum firing temperature from 1500°C to 1400°C. However, when fired at 1300°C, V1 exhibited a lower E_c and 10% of the structure was composed of cristobalite (Table 8.4). XRF analysis indicated that the molecular composition was not dependent on temperature between 1300°C and 1500°C (Table 8.5)

Table 8.4: Physical properties of StarLight C (V1) fired at different temperatures

Firing temp. (°C)	XRD analysis			Prill density (kg/m ³)	Mean E_c
	% cristobalite	% mullite	% amorphous		
1300	10	45	45	700	10.4
1400	0	50	50	800	13.7
1500	0	50	50	900	13.9

Table 8.5: XRF analysis of V1 fired at 1300°C and 1500°C

Molecules	V1 fired at 1300°C	V2 fired at 1400°C
SiO ₂	55.3	55.6
Al ₂ O ₃	40.4	40.3
Fe ₂ O ₃	0.99	1.01
TiO ₂	0.1	0.06
CaO	0.17	0.17
MgO	0.37	0.37
K ₂ O	1.9	1.91
Na ₂ O	0.12	0.12
Loss on ignition	0.6	0.5

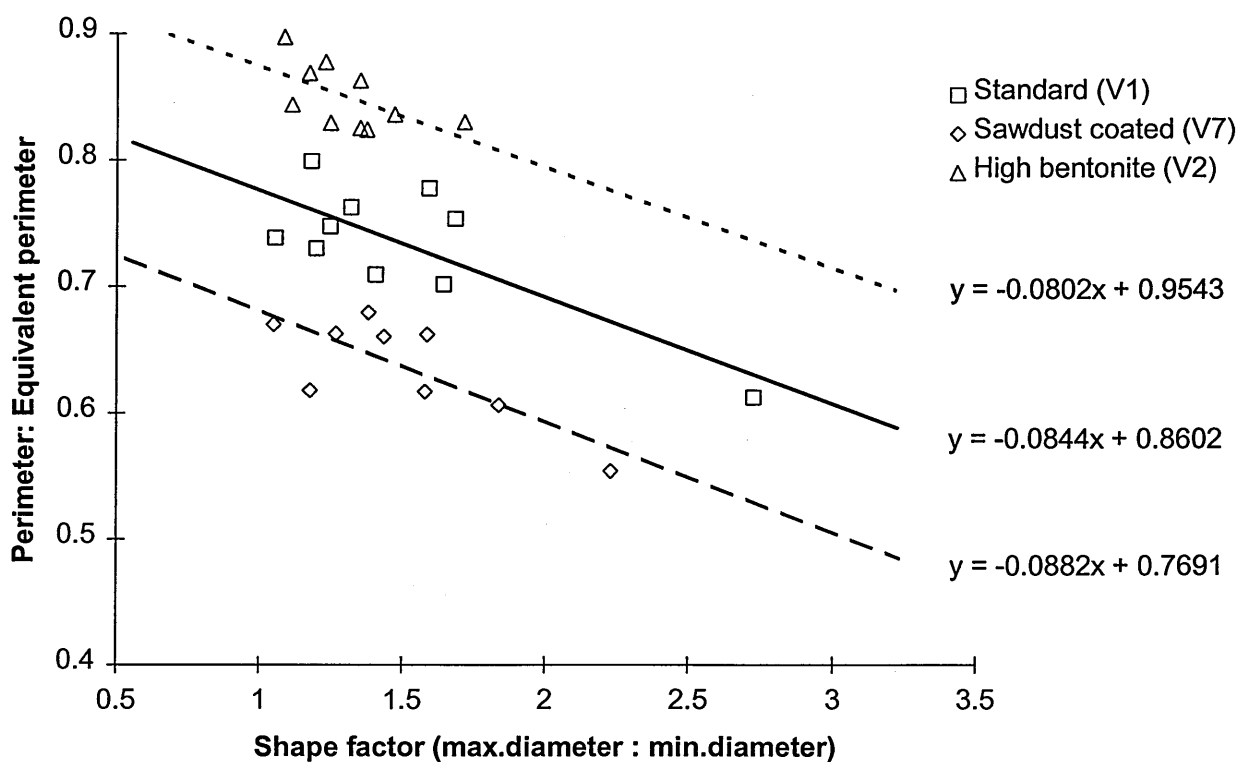


Figure 8.2: Plot of perimeter ratio against shape for variants 1, 2 and 7.

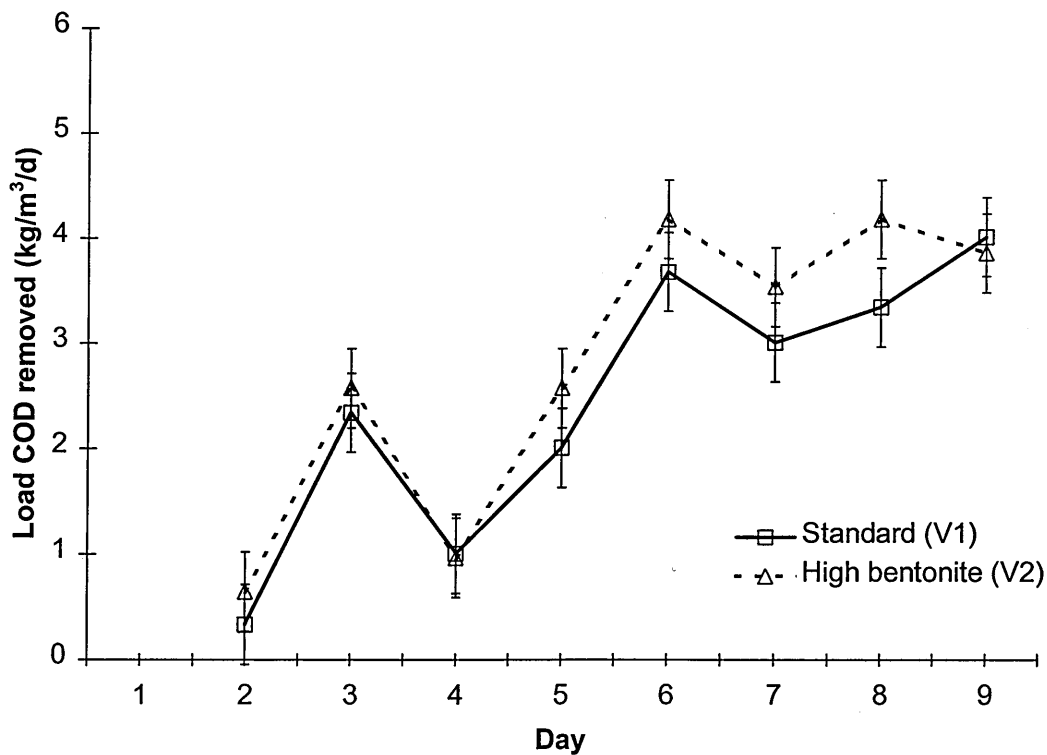


Figure 8.3: Load of soluble COD removed during start-up of small-scale reactors containing V1 and V2.

Comparing V1, V2 and V7 in small-scale reactors showed no clear correlation between surface roughness and substrate removal during start-up (Figures 8.3 & 8.4). However, the silane-coated hydrophobic foamed clay (V8), did show some improvements in biofilm retention compared to V1 (Figure 8.5).

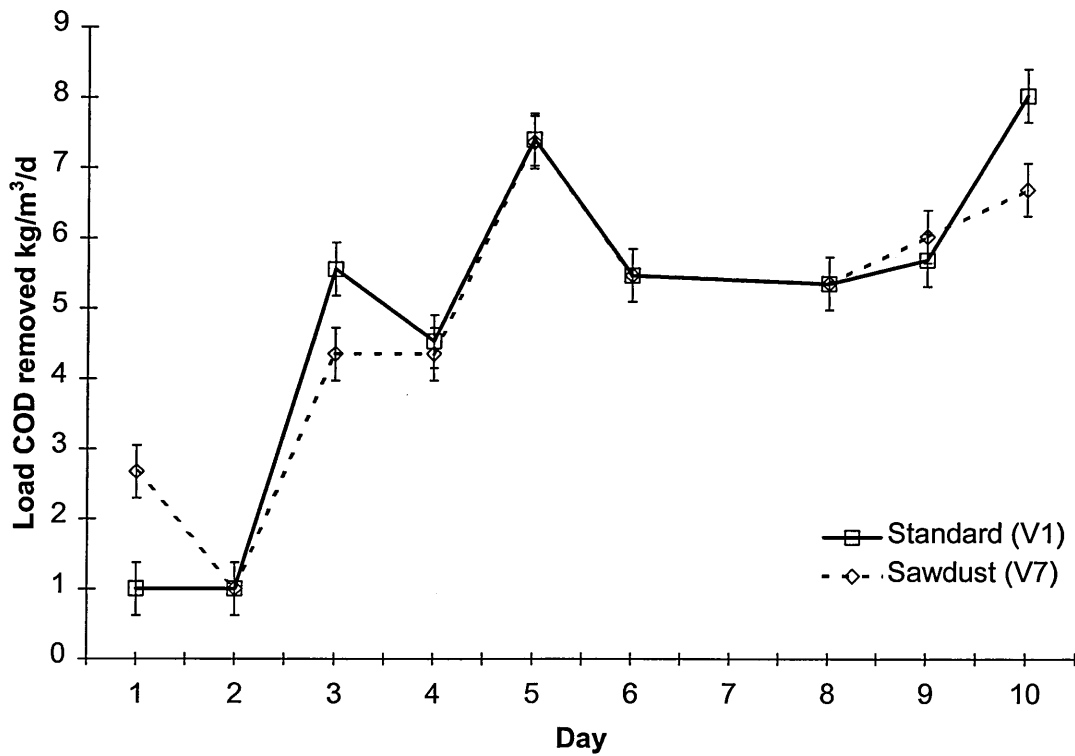


Figure 8.4: Load of soluble COD removed during start-up of small-scale reactors containing V1 and V7.

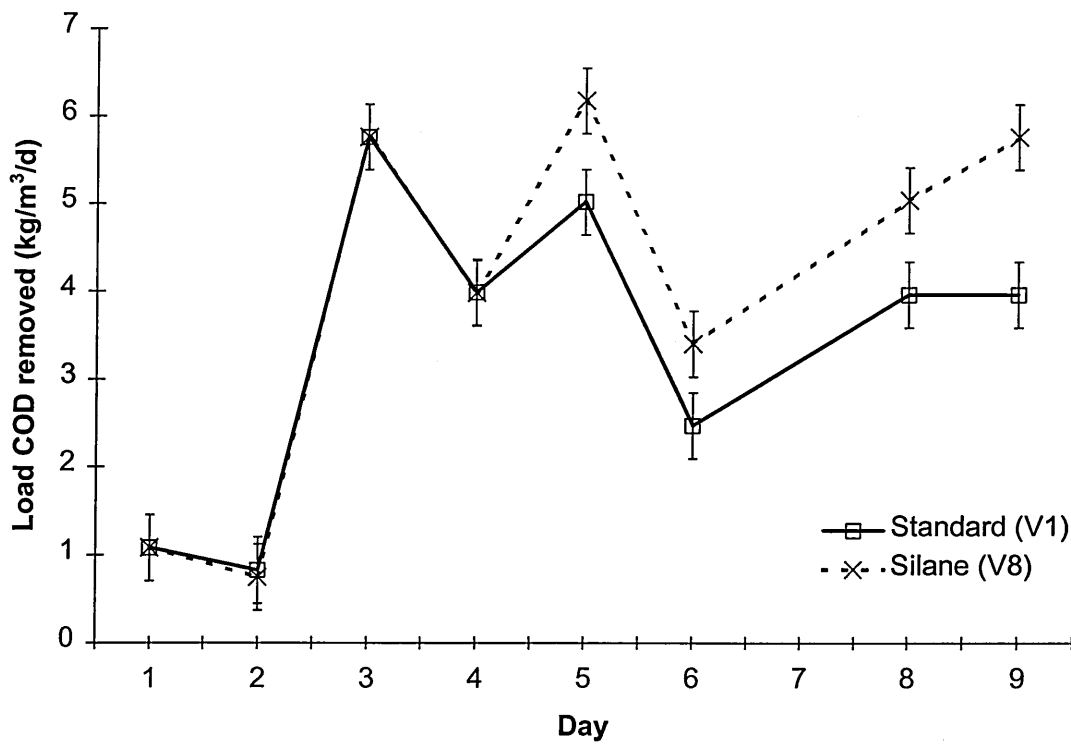


Figure 8.5 Load of soluble COD removed during start-up of small-scale reactors containing V1 and V8.

8.4 DISCUSSION

There was no simple relationship between prill density and resistance to crushing, indicating that the change in structure of the variants was not just related to changes in porosity. The variant V2, which had a high bentonite content exhibited the greatest resistance to crushing. Altering the proportions of the other constituents resulted in decreases in the media crush strength. Future work could investigate if further increases in the proportion of bentonite improves media strength, and determine when the optimal ratio of the glassy phase to mullite is reached. The results from the crush resistance tests correlated with those from the attrition tests. This is because both measure how particles break up when a force is applied (Humby & Fitzpatrick, 1996).

The firing tests indicated that StarLight C should be brought to a maximum temperature of at least 1400°C during firing so that E_c is maximised and cristobalite is not formed. Cristobalite is undesirable because it consists of needle shaped crystals that are harmful on inhalation.

Higher crush resistance appeared to correlate with a decrease in surface roughness. The biological tests indicated that, under the test conditions, there were no differences in the biofilm retention properties of variants with different degrees of roughness. Previously, increases in surface roughness have been positively linked to improved biofilm retention. Three possible explanations for this were that roughening, increases the surface area, provides sites which are protected from fluid flow, and improves glycocalyx attachment (Baker, 1984). However, Baker (1984) compared bacterial attachment to smooth and roughened glass and polystyrene surfaces. All of the foamed clay surfaces are rough compared to glass and polystyrene and only the degree of roughness was altered between the variants. Therefore, it is proposed that increasing surface roughness and thus, the surface area, only increases bacterial attachment when the initial surface is smooth. This infers that the mechanism by which roughness influences attachment is not through increasing the surface area, but through providing particularly suitable anchoring sites for the biofilm. Once these anchors have formed, further bacterial adhesion will be less dependent on the

substrate surface. However, repeated experiments at different liquid velocities are required to confirm this theory.

Research groups have reported improved biofilm formation on hydrophobic, silane-coated surfaces (Burchard *et al.*, 1990). This may be due to water exclusion mechanisms with hydrophobic bacterial surface components making initial contact with the substrata. The silane-coated hydrophobic foamed clay (V8), did show some improvements in biofilm retention compared to V1. However, this variant is unlikely to show improved biological characteristics when in contact with sewage, since the influence of the surface chemistry of the material seems to decrease when a variety of colonisers are available in a natural environment (Baker, 1984). This may be because each species and strain of bacterium will adhere preferentially to surfaces with different surface tensions. Therefore, surface chemistry is unlikely to have a significant effect on heterotrophic biofilm formation in WWT reactors. However, since only two genera of nitrifying bacteria are found in sewage (*Nitrosomonas* and *Nitrobacter*), it may be possible to select a surface that would encourage the adhesion of nitrifiers. If this could decrease the start-up time for nitrification, it would be very beneficial.

8.5 CONCLUSIONS

1. Increasing the levels of bentonite in the feed clay can produce a foamed clay (V2) which exhibits lower levels of attrition. Altering the proportion of the other constituents increased the observed media friability.
2. StarLight C should be fired to a maximum temperature of at least 1400°C to prevent the formation of cristobalite.
3. Although the surface roughness of V2 was lower than that of V1, there was no evidence of lowered biofilm retention properties. Variant 2 is therefore likely to show the best overall performance as a BAF medium in terms of performance and operating costs.

CHAPTER 9

USING AN ENVIRONMENTAL SCANNING ELECTRON MICROSCOPE (ESEM) TO OBSERVE BIOFILM STRUCTURE AND DEVELOPMENT ON BAF MEDIA

ABSTRACT

Further information about biofilm structure at different growth stages, would aid the development of both biofilm and wastewater treatment process models. The environmental scanning electron microscope (ESEM) allows biofilms to be viewed in their fully hydrated state. This provides a realistic view of biofilms with cells completely covered in extracellular polymeric substances (EPS).

Biofilm development on a foamed clay BAF medium was monitored using an ESEM with a sample chamber saturated with water vapour and maintained at 4 °C and 4 Torr. The backscattered electron detector was shown to facilitate the detection of biological material attached to the rough surface of the foamed clay. Within 24 hours colonies between 5 and 10 µm in diameter were visible. Subsequently, colonies up to 100 µm in diameter were found and after six days the colonies had started to converge. Exact areas of biofilm could be repeatedly viewed by attaching media grains to stainless steel plates and constructing a specially designed Peltier stand.

The effects of the sample chamber conditions and the electron beam on cell viability were assessed. Biofilms grown on agar incubated for 2 hours showed no significant change in viability when exposed to electron beams below 10 kV for 10 minutes. However, significant reductions in viability resulted after 10 minutes exposure to a 20 kV beam. It was proposed that EPS could help prevent such losses in viability because biofilms incubated for over 18 hours showed no significant reduction in viability if exposed to a 20 kV beam for less than 30 minutes.

9.1 INTRODUCTION

In all fixed film wastewater treatment systems the biofilm structure effects reactor performance by influencing the transport of substrates and reaction products (Cao & Alaerts, 1995). In biological aerated filters (BAFs), biofilm structure will also affect the suspended solids removal properties, and the run time of the filter between backwashes. Further information describing biofilm structure and development under various conditions would therefore allow the development of more complete biofilm and process models (Bishop & Rittmann, 1995).

Despite the introduction of numerous spectroscopic techniques, which can measure the concentration of various cellular components, microscopy is still preferred to view biofilm morphology (Clark, 1990). Electron microscopy has better magnification and resolution limits than light microscopy. However, since samples are viewed under vacuum in electron microscopes, they must be fully dehydrated. All of the electron microscope (EM) preparation techniques available, including critical point drying and freeze-drying, cause disruption and distortion to the biofilm. Despite confocal laser scanning microscopy showing that biofilms are composed of between 73 and 98% extracellular polymeric substances (EPS) and space (Lawrence *et al.*, 1991), very little EPS remains after EM preparation (Sutton *et al.*, 1994).

The environmental scanning electron microscope (ESEM) allows the fully hydrated topography of the wet biofilm to be revealed. This is possible because multiple pressure limiting apertures separate the sample chamber from the column, and so allow chamber pressures up to approximately 20 Torr to be reached. When water vapour is used as the chamber gas, and a Peltier cooling stage controls the temperature of the sample, a saturated water vapour atmosphere can be maintained (Danilatos, 1993). The gas is ionised by both the secondary (SE) and the back scattered electrons (BSE) derived from the sample. The gaseous secondary electron detector (GSED) can produce images based on surface morphology. Meanwhile, the back scattered electron detector (BSED) produces images based on surface composition. The number of back scattered electrons originating from a specimen increases with the atomic number of

elements in the sample. Specimen charging is dissipated into the gaseous environment, enabling direct observation of uncoated, non-conductive specimens.

Research has shown the ESEM can provide fast accurate images of biofilms (Little *et al.*, 1991, Sutton *et al.*, 1994). Additionally, living plants and insects have survived both the conditions of the sample chamber and the irradiation of the electron beam in the ESEM (Danilatos, 1981, Danilatos & Postle, 1982, Elder, 1995). The aims of this study were firstly, to examine the effects of the ESEM on the viability of bacterial cells growing as a biofilm. Then, once appropriate conditions were found, the development of a biofilm on BAF media was monitored.

9.2 MATERIALS AND METHODS

9.2.1 Testing the Viability of Biofilms Exposed to Various Conditions in the ESEM

Bacterial films were made by spreading 100 μm of *Pseudomonas aeruginosa* (Chemviron, UK) culture on nutrient agar plates (Oxoid, UK). Incubation for between 2 and 22 hours took place at 35°C before 5 mm² blocks were extracted. These blocks were exposed to various conditions in an ESEM 2020 (Electroscan Corporation, USA), before being resuspended in 5 ml Ringers solution. The Miles and Misra viable cell test was then conducted on each sample (Singleton & Sainsbury, 1981). The observed significance level (p-value) was calculated for t-tests where the null hypothesis was that there was no difference between the viability of the test and control samples. The null hypothesis was rejected for p-values less than the selected significance level of 0.05.

For biofilms of various ages, the controls received no treatment, whilst the test samples were exposed to the chamber conditions both with and without the electron beam. The sample chamber was brought to 4°C and 4 Torr using a procedure to minimise sample evaporation and dehydration (Cameron & Donald, 1994). The chamber was evacuated and flooded with water vapour in three stages whilst the sample was maintained at 4°C. Electron beams with accelerating voltages of 2, 5, 10, and 20 kV were scanned over the whole of the agar surface for various times.

9.2.2 Examining Biofilm Growth on a Foamed Clay BAF Medium

Particles of a foamed clay BAF medium were fixed onto stainless steel plates (26 mm × 19 mm). These were stacked inside novel small-scale reactors, which were based on expanded flow cells (details in Section 8.2.3). The reactors received a continuous flow ($7.8 \times 10^{-5} \text{ m}^3/\text{h}$) of fully oxygenated synthetic sewage containing 330 mg/l COD (Sterritt & Lester, 1981). The feed was inoculated with *Pseudomonas aeruginosa* (Chemviron, UK) for the first 24 hours. *Pseudomonas aeruginosa* was selected because the genus is commonly found in biological wastewater treatment reactors, and it is frequently used by biofilm research groups.

The stainless steel plates could be removed from the reactors and placed in a specially constructed tray fixed to the Peltier stand of the ESEM. This facilitated the relocation of exact areas on the surface of the medium. The sample chamber conditions were 4°C and 4 Torr, whilst an electron beam with an accelerating voltage of 20kV was applied. After the results of the experiments on the effects of the ESEM on cell viability, samples were examined for less than 15 minutes. Both an environmental GSED and a BSED (K. E. Developments, Cambridge, UK) were used. Additionally, there was an X-ray analysis facility (PGT, Imix, Peterborough, UK) to determine the elemental composition of specific areas.

9.3 RESULTS

9.3.1 Effects of ESEM on Biofilm Viability

None of the *Pseudomonas aeruginosa* films showed significant reductions in viability after being brought to the conditions in the sample chamber for five minutes. However, the viability of the films which had been incubated for two hours or less, was significantly effected when the samples were exposed to a 20 kV electron beam for 10 minutes (Table 9.1).

Table 9.1: Effect of film age on viability changes after ESEM exposure

Film age (hours)	p-values (mean of 9) for t-tests between control & test samples after	
	5 minutes (no beam)	10 minutes (20 kV beam)
0.25	0.365	0.049
2	0.486	0.013
2	0.302	0.024
18	0.486	0.306
19	0.305	0.160
21	0.412	0.310

However, reducing the accelerating voltage of the electron beam from 20 kV to 10, 5 and 2 kV resulted in there being no significant change in the population viability of the young films after 10 minutes ESEM exposure (Figure 9.1).

When biofilms incubated for over 19 hours, were exposed to the ESEM for longer than ten minutes, there was a decrease in cell viability that became highly significant after 30 minutes. The effect was the same with and without exposure to the electron beam (Figure 9.2).

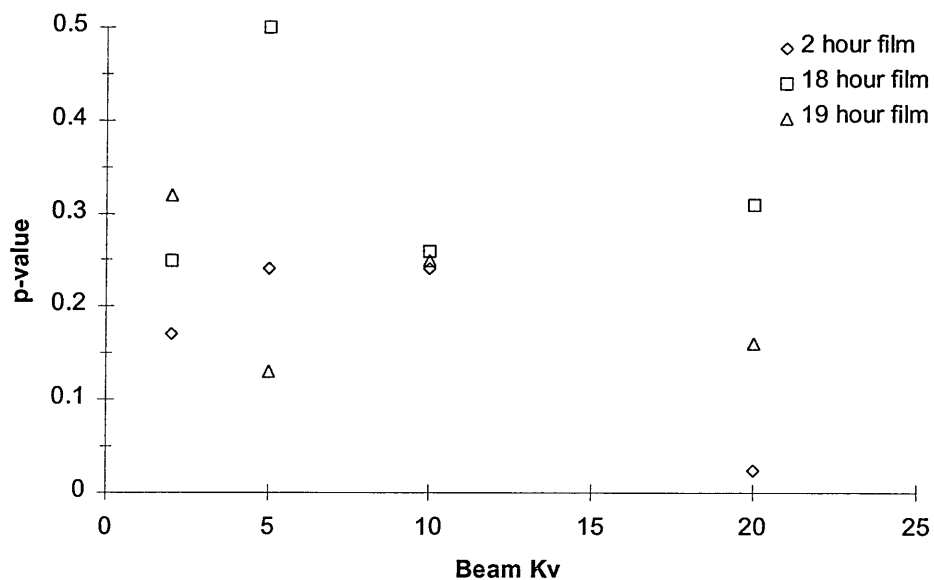


Figure 9.1: The effect of the electron beam accelerating voltage on the change in viability for films incubated for 2, 18 and 19 hours.

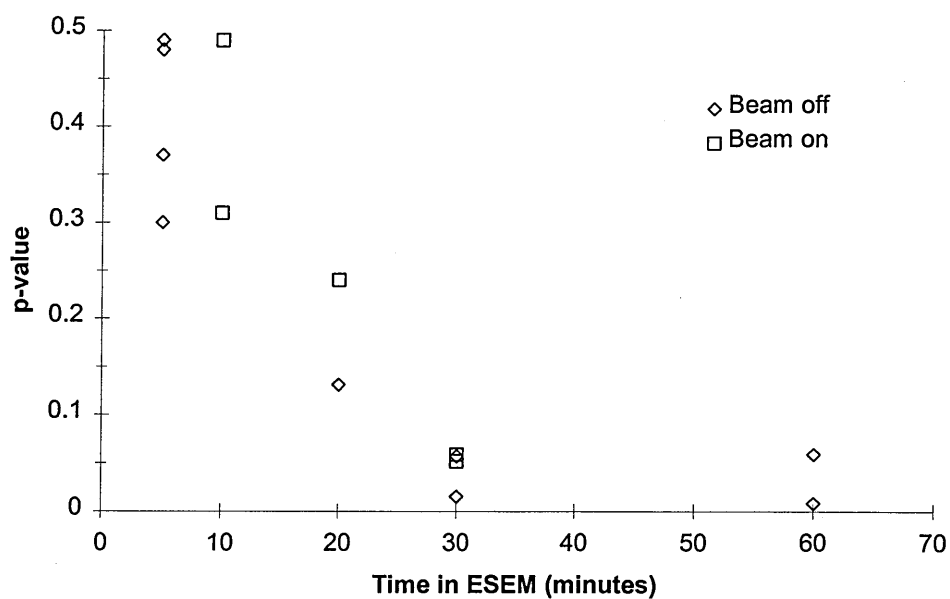


Figure 9.2: The effect of length of time in the ESEM on the viability of biofilms

9.3.2 Observed Stages of Biofilm Growth

The surface of an expanded clay is very rough (Figure 9.3), which made it more difficult to locate biological growth. It was found that biofilm areas could often be located more easily using the BSED. Biofilms are composed of substances with lower atomic numbers than the foamed clay materials. Therefore, biofilm areas were revealed as dark patches on the media surface. When highly hydrated biological material was magnified over 2300 times for approximately five minutes, bubbles started to form (as shown in Figures 9.4 and 9.5). This aided the recognition of biological material initially.

After the small-scale reactors had been operated for 24 hours, there was evidence of biofilm growth on the foamed clay BAF medium. Smooth nodules 5-10µm in diameter and composed of substances with a low atomic number appeared on the surface of the medium (Figures 9.6 and 9.7). As previously noted, the thick layer of EPS made it impossible to image individual cells. During the proceeding days the biofilm developed, with the colonies first growing up to 100 µm in diameter (Figures 9.8 and 9.9) and then converging (Figures 9.10 and 9.11). X-ray analysis showed that the biofilm areas (runa3) contained carbon, phosphorus and sulphur, whilst areas of bare foamed clay (runa4) contained greater proportions of aluminium, silicon and oxygen (Figure 9.12).

Using the specially designed stand, the same areas on the medium could be revisited. When the samples were viewed for less than 15 minutes, there was evidence of biofilm growth (Figures 9.13 to 9.20).

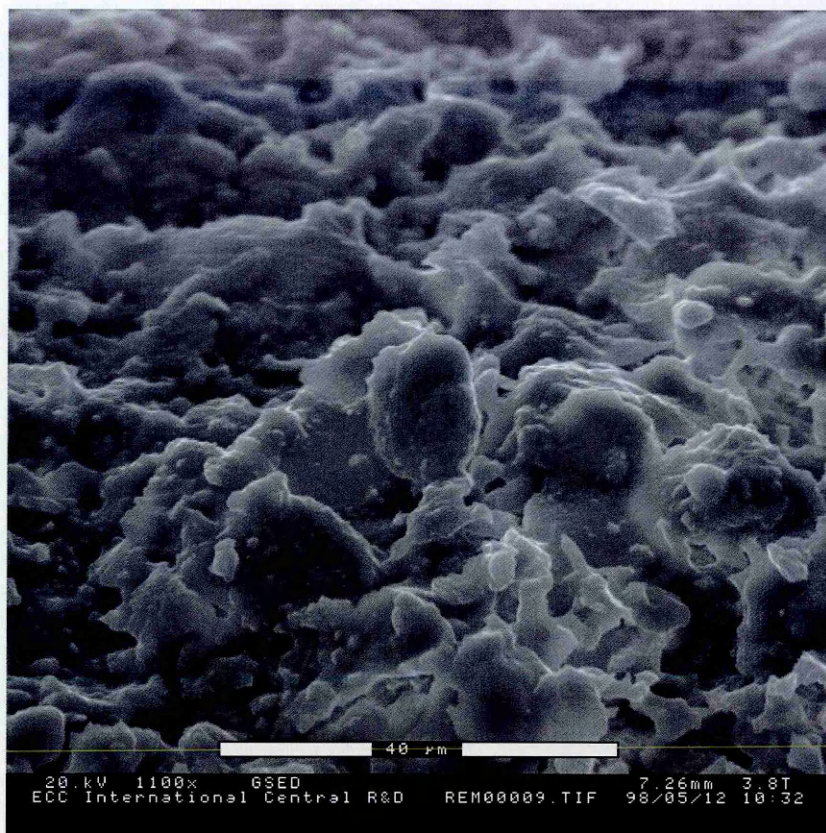


Figure 9.3: Micrograph of the surface of StarLight C

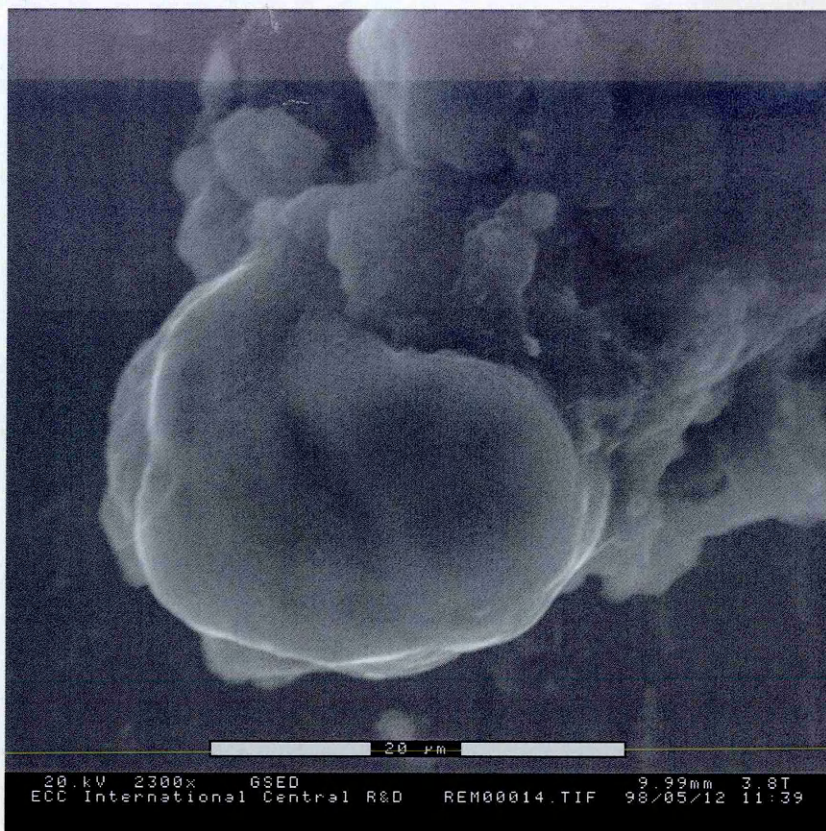


Figure 9.4: Micrograph of biological material initially

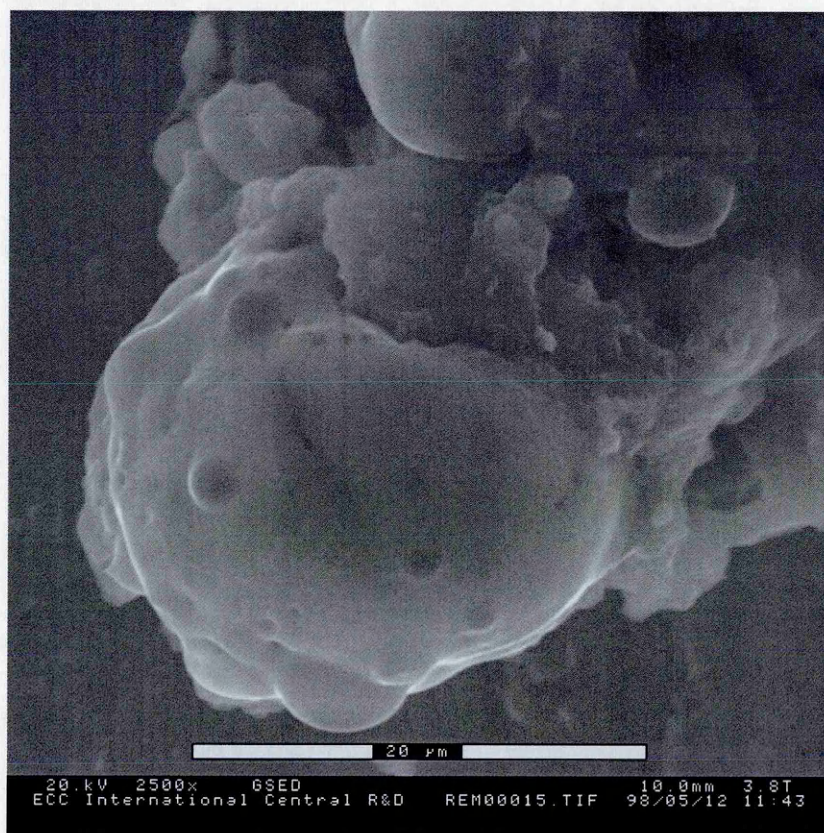


Figure 9.5: Micrograph of biological material after approximately 5 minutes

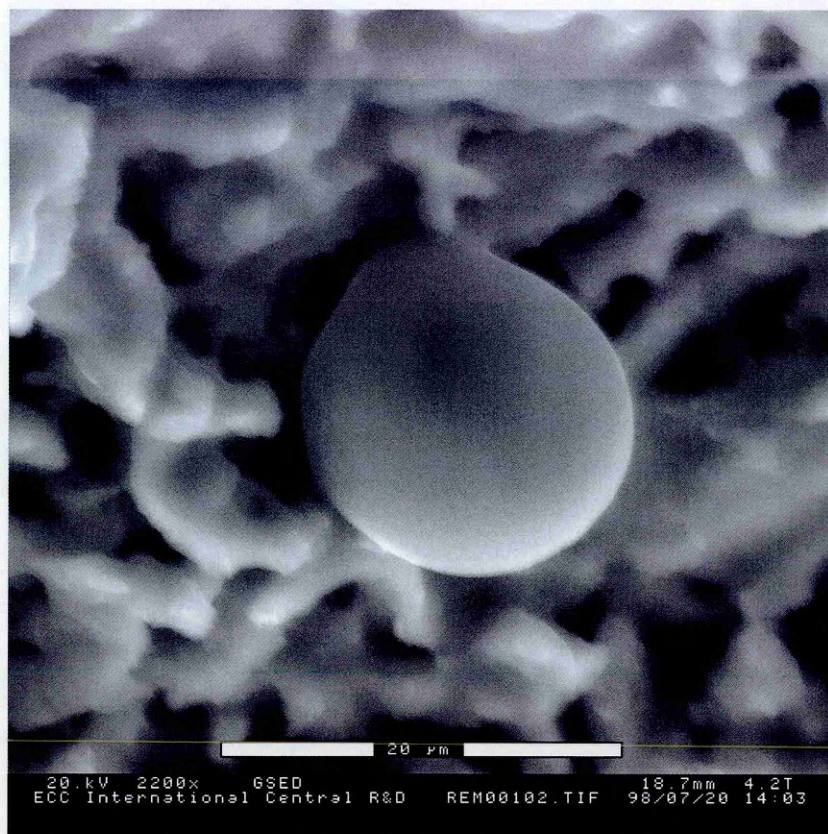


Figure 9.6: Micrograph showing biological nodule (GSED)

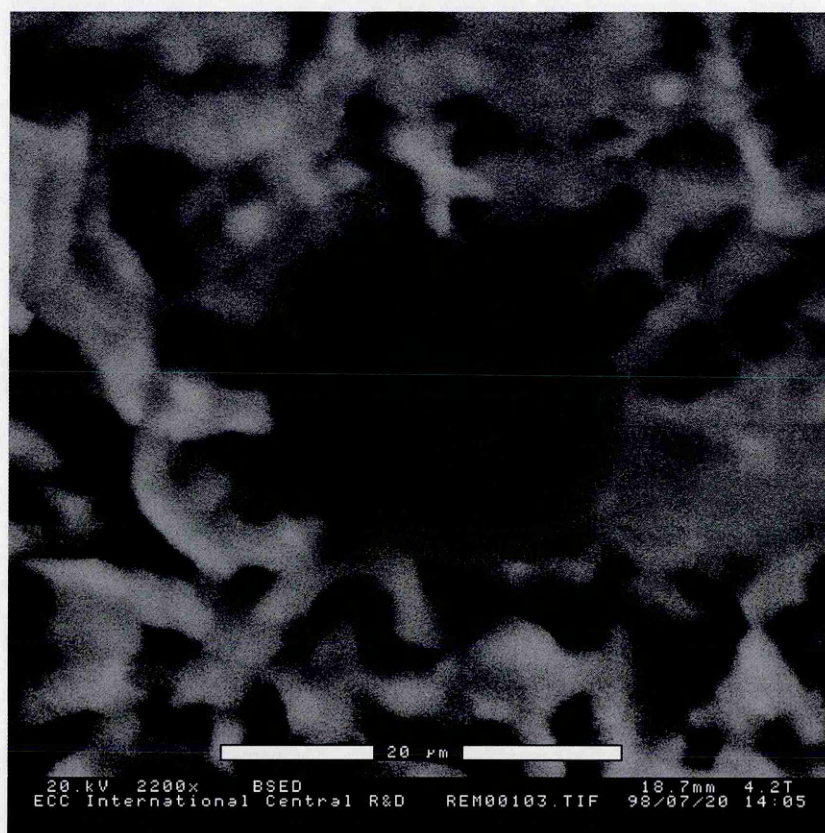


Figure 9.7: Micrograph showing biological nodule (BSED)

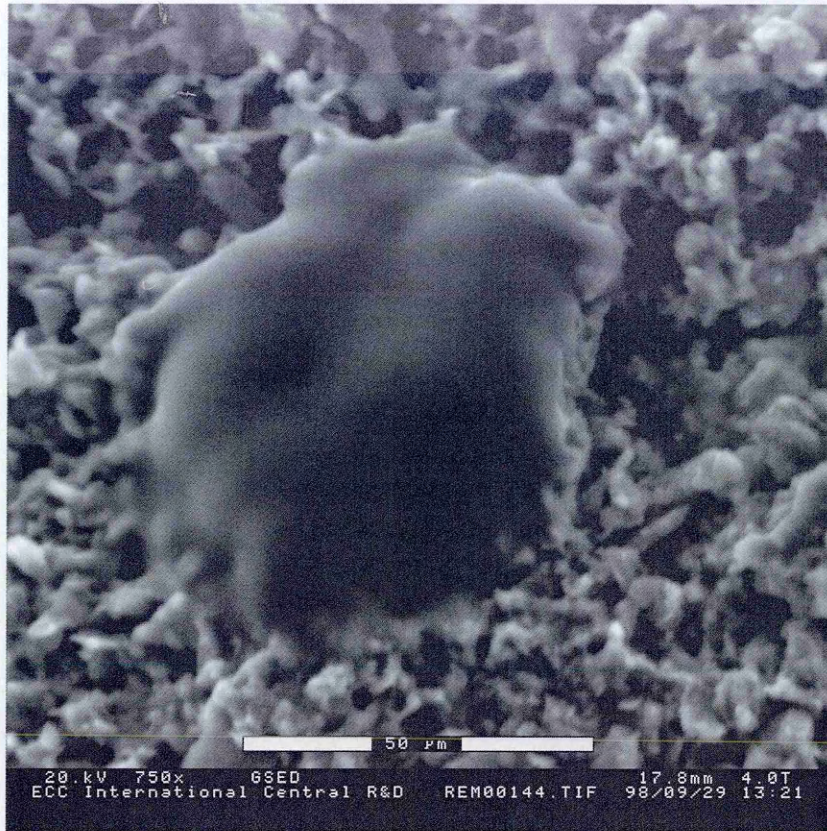


Figure 9.8: Micrograph showing biofilm growing on StarLight C (GSED)

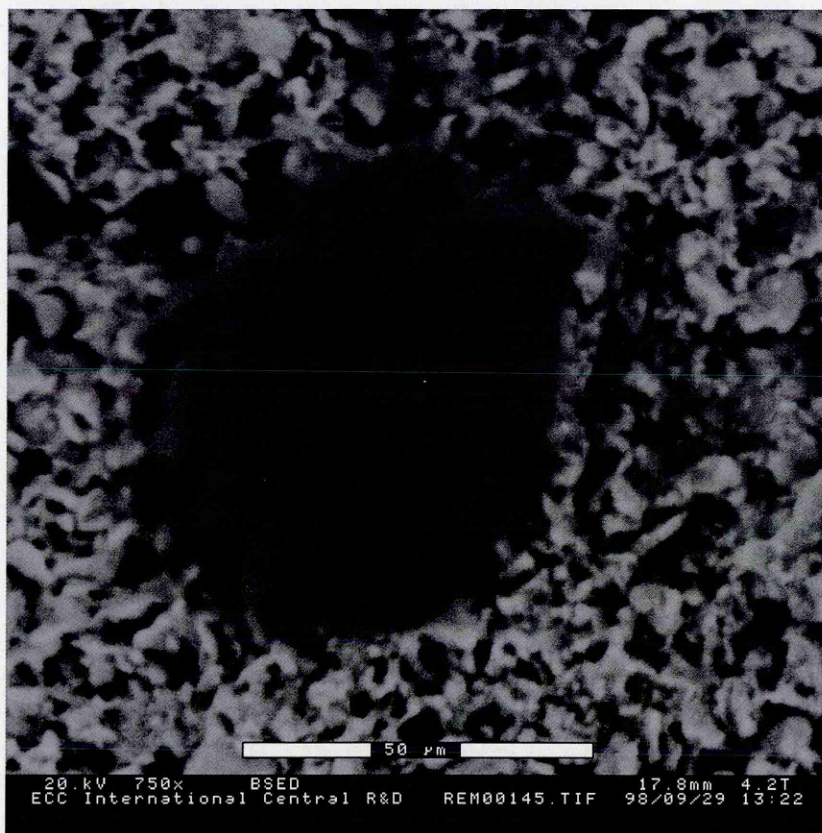


Figure 9.9: Micrograph showing biofilm growing on StarLight C (BSED)

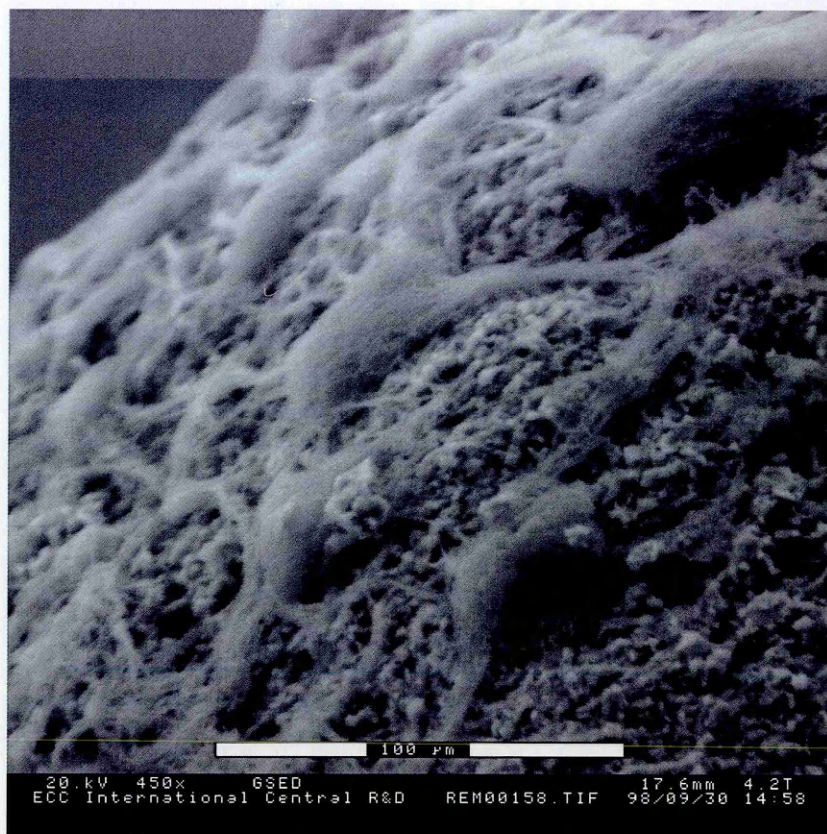


Figure 9.10: Micrograph showing the biofilm spreading across the surface (GSED)



Figure 9.11: Micrograph showing the biofilm spreading across the surface (BSED)

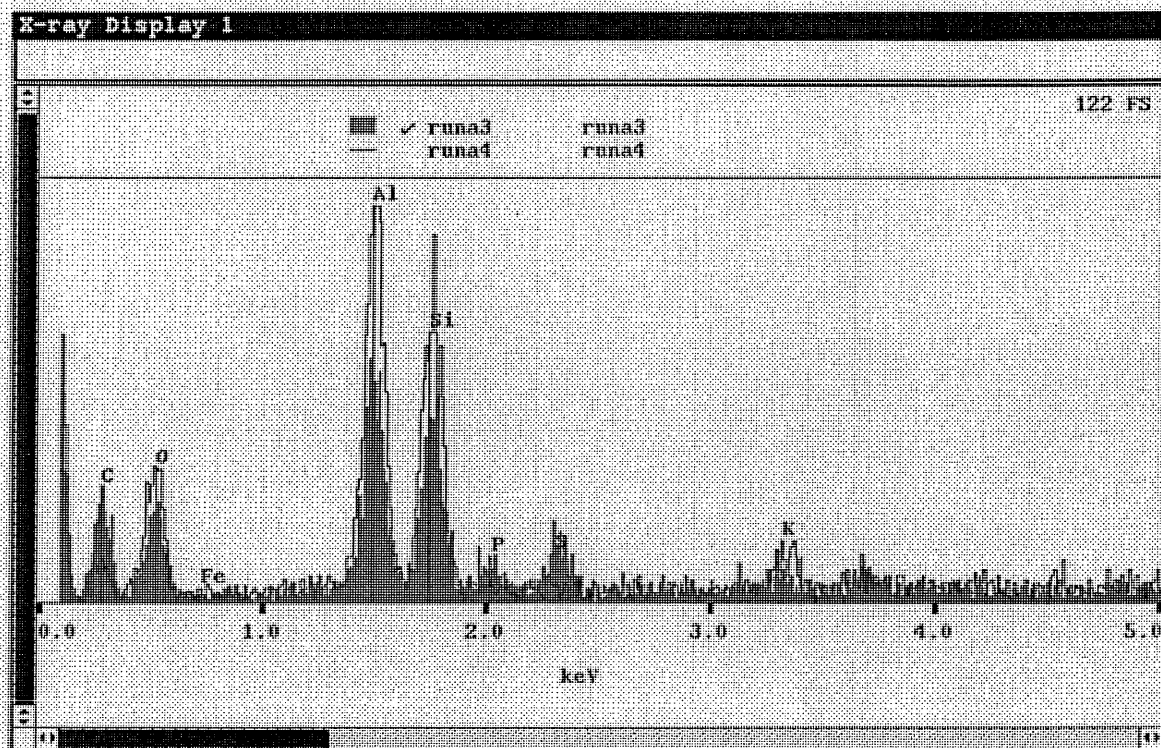


Figure 9.12: X-ray analysis of bare foamed clay (runa4) and area with attached biofilm (runa3).

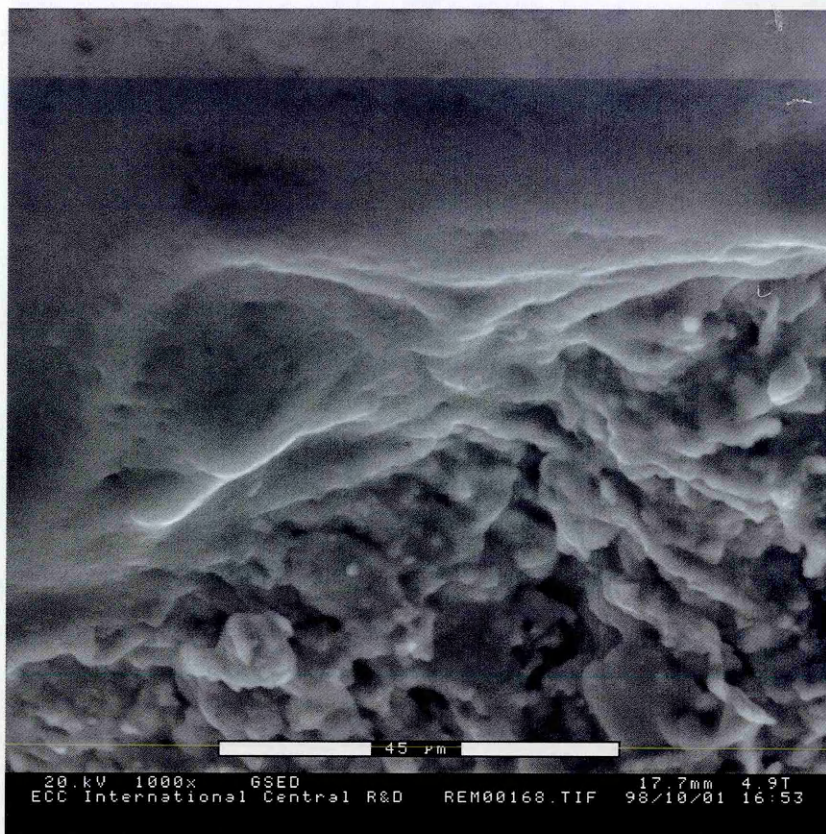


Figure 9.13: Biofilm area 1 on day 7 (GSED)

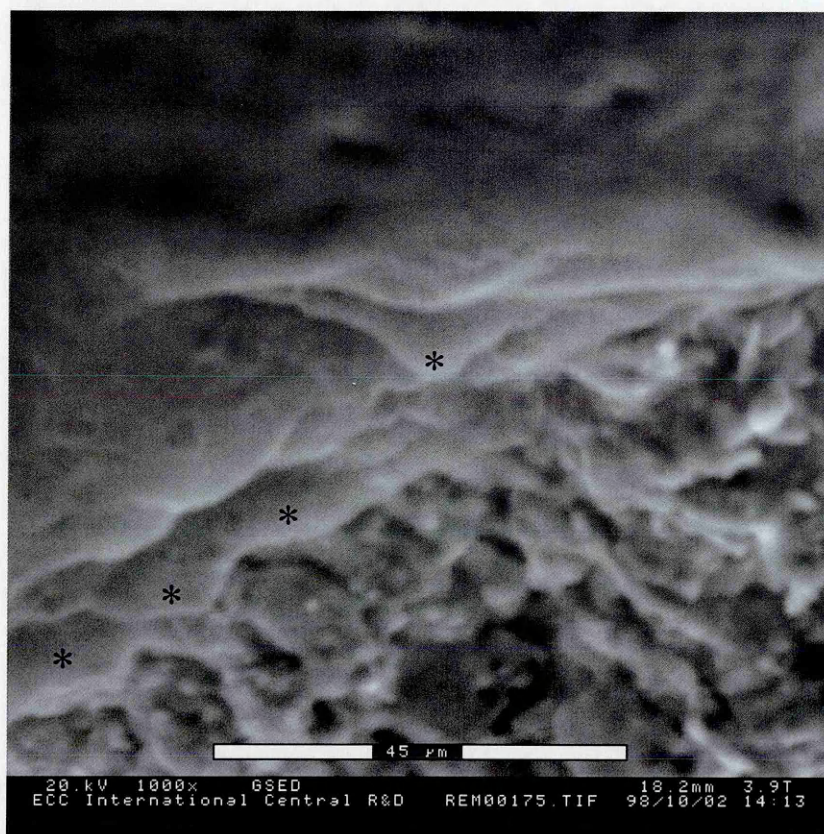


Figure 9.14: Biofilm area 1 on day 8 (GSED)* indicates new growth

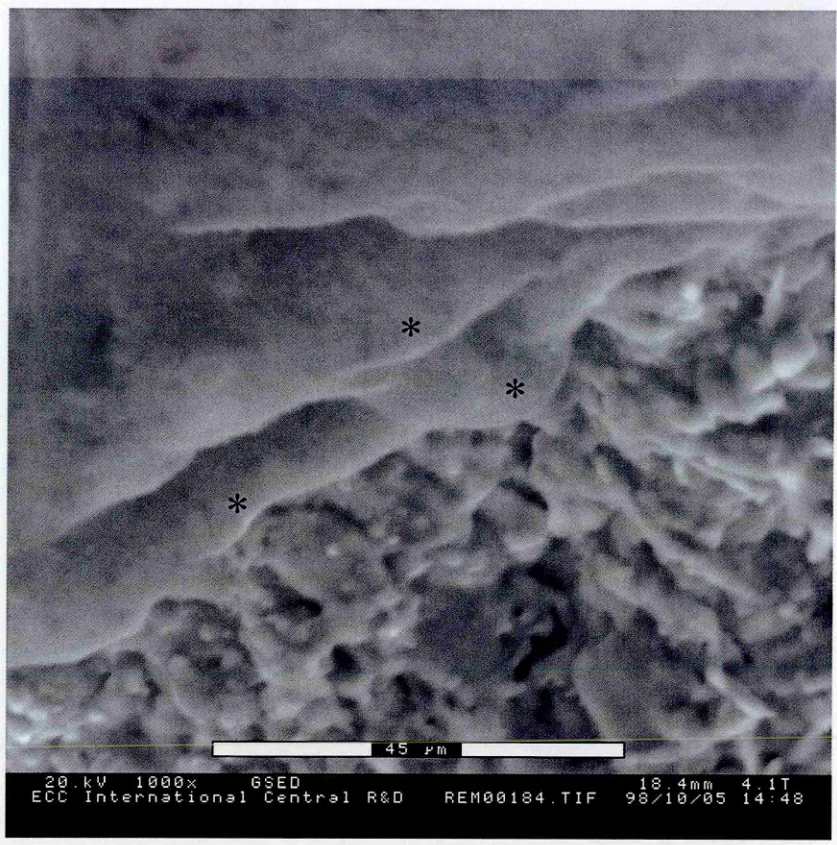


Figure 9.15: Biofilm area 1 on day 11 (GSED) indicates new growth*

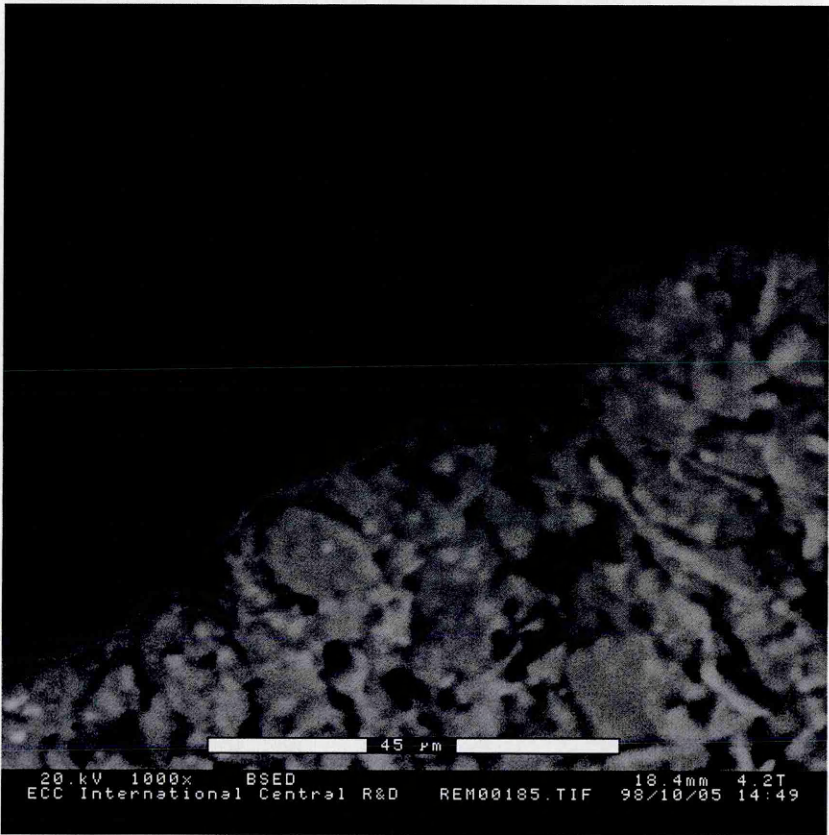


Figure 9.16: Biofilm area 1 on day 11 (BSED)

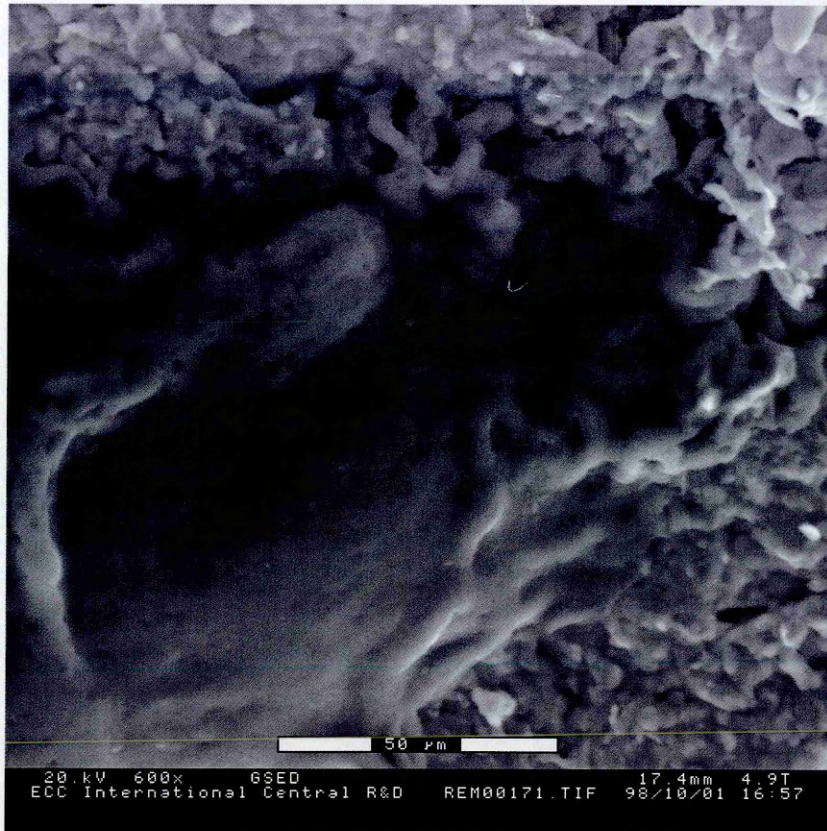


Figure 9.17: Biofilm area 2 on day 7 (GSED)

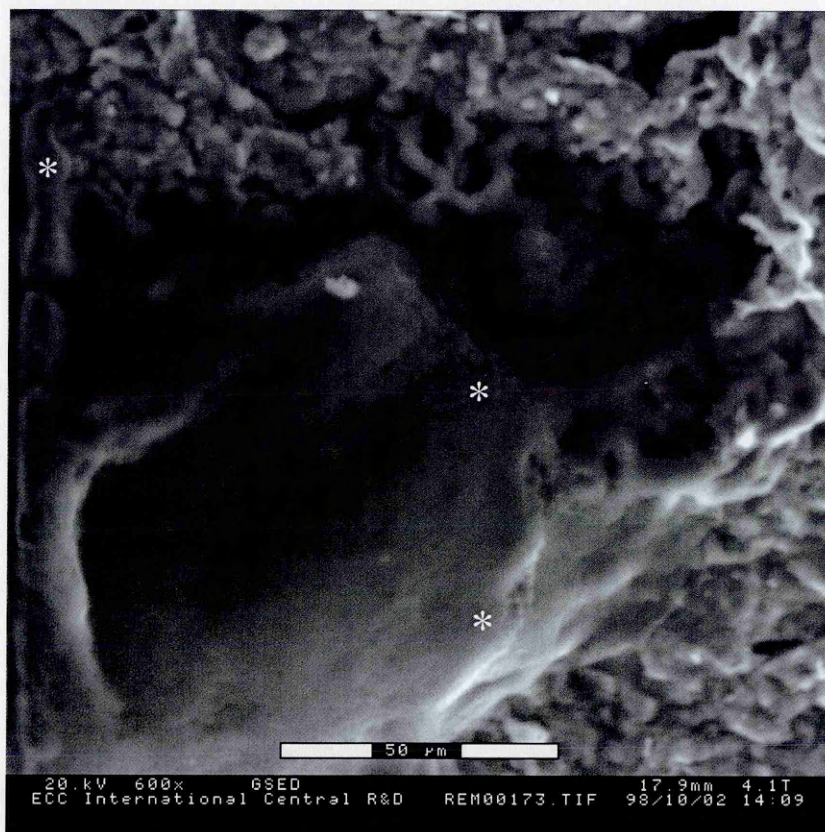


Figure 9.18: Biofilm area 2 on day 8 (GSED) indicates new growth*

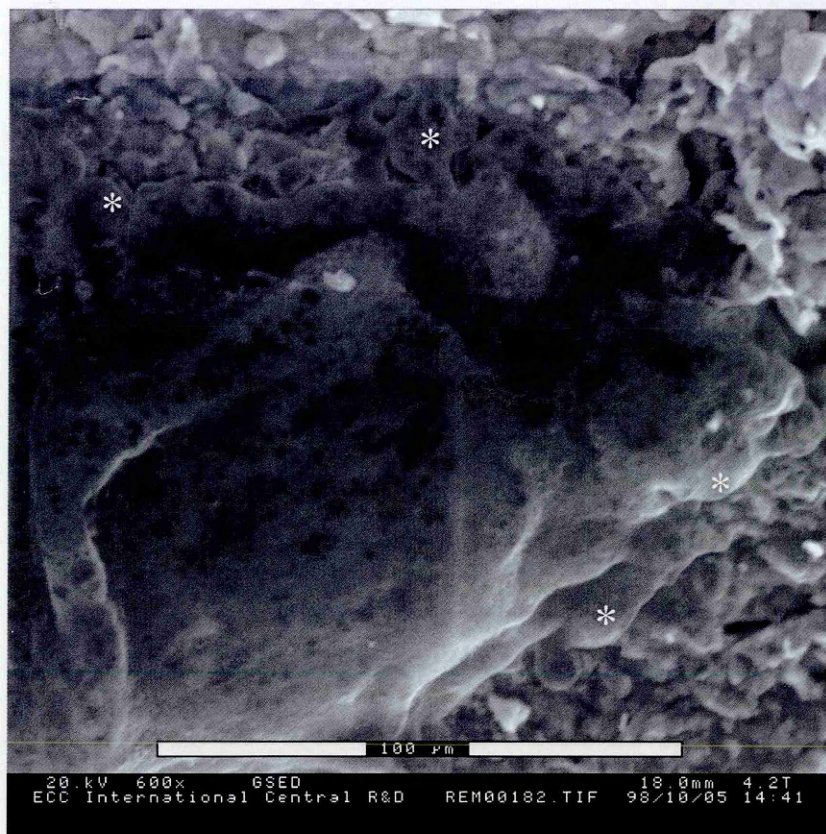


Figure 9.19: Biofilm area 2 on day 11 (GSED) indicates new growth*

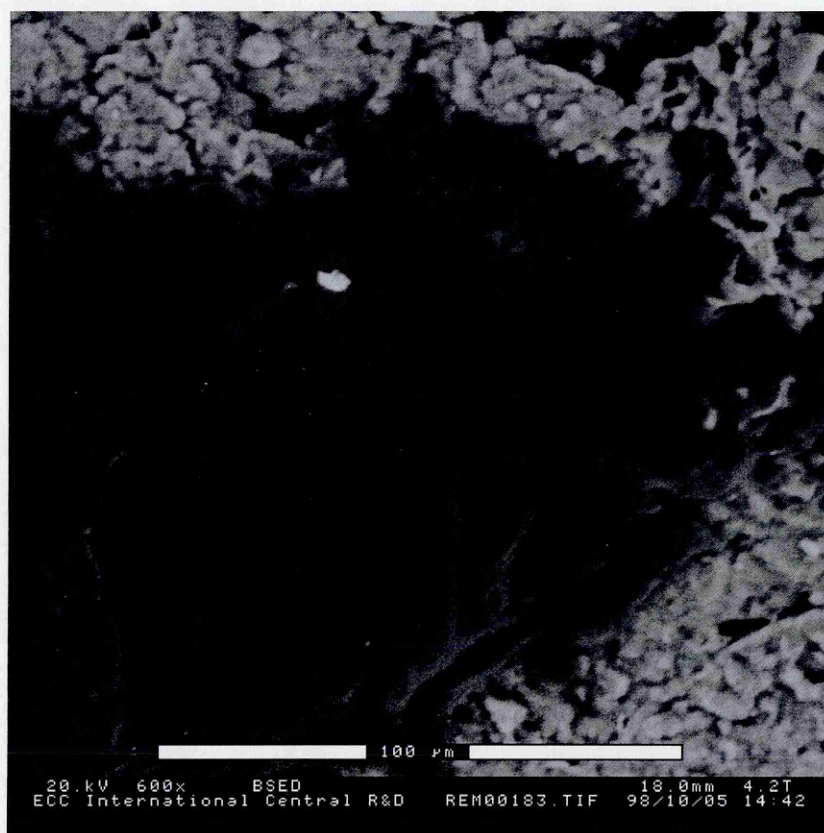


Figure 9.20: Biofilm area 2 on day 11 (BSED)

9.4 DISCUSSION

The results from the viability tests indicate that evaporation from, and condensation on, the sample was minimised during the pump-down sequence to reach 4 Torr and 4 °C in the sample chamber. However, all the biofilms tested showed significantly reduced viability after exposure to the conditions of the sample chamber for longer than 30 minutes. This suggests that the biofilms were gradually dehydrated whilst in the sample chamber, which infers the atmosphere was not completely saturated with water vapour. The pump-down procedure could be improved by increasing the number of times the chamber is flooded with water vapour. This would ensure the initial removal of all the air and prevent conditions in the sample chamber gradually changing (Cameron & Donald, 1994).

Biofilms incubated for under two hours showed less reduction in viability when the accelerating voltage of the electron beam was lowered. This may be because reducing the beam energy confines the radiation effects to a very thin surface layer of the specimen (Danilatos & Postle, 1982). Unfortunately, it also reduces the maximum resolution of the ESEM and so decreases image quality.

The viability of biofilms that were incubated for over two hours was not significantly effected by ten minutes exposure to electron beams up to 20 kV. *Pseudomonas aeruginosa* forms dense heterogeneous biofilms and produces copious quantities of EPS (Sich & Rijn, 1997). Therefore, the films incubated for over two hours were probably completely covered in EPS. An EPS coating would protect cells from both radiation and dehydration.

The electron beam in the ESEM may dehydrate samples, as well as irradiating them (Cameron & Donald, 1994). The observed bubbling of biological material was probably an effect of the localised heating caused by the electron beam. This can be avoided by minimising the amount of time spent focused on the biofilm at close range.

Biofilms growing on a foamed clay BAF medium were viewed in their hydrated state utilising an ESEM, with either an environmental GSED, or a BSED. When using the BSED, the biofilm areas were easily recognisable as dark patches. The GSED biofilm images looked similar to those from previous ESEM studies, with cells invisible beneath a canopy of moist EPS (Sutton *et al.*, 1994, Little *et al.*, 1991). The observed growth stages indicate that the biofilm does not form a uniform monolayer across the medium. Concurring with the observations of Stewart *et al.* (1993), thick biofilm was observed next to bare patches of the media surface.

Previous work has suggested that clusters of bacteria in a biofilm are separated by channels (Beer *et al.*, 1994, van Benthum *et al.*, 1995, Massol-Deya *et al.*, 1995). Pits in the surface of the biofilm were observed in the later images of biofilm development (Figures 9.13-9.20) and these may have been the entrances to such channels. Conventional biofilm models are one dimensional and do not account for substrate transport to deep parts of the biofilm through channels (Bishop & Rittmann, 1995). Therefore, the ESEM micrographs from this study supply further evidence that conventional models do not describe the structure of all biofilms correctly.

Using a specially designed Peltier stand, exact areas of the biofilm growing on foamed clay particles, were relocated. When biofilms over five days old were examined for under 15 minutes, there were some signs of growth after 24 h. However, further work is needed to optimise the operating conditions required to view biofilm growth on BAF media with minimal disturbance.

9.5 CONCLUSIONS

1. The main advantage of using an ESEM instead of a conventional EM is that hydrated biofilm morphology and elemental composition can be viewed.
2. In addition, it may be possible to observe biofilm growth and development in an exact location. Initial tests indicate that changes in cell viability after exposure to various conditions in the ESEM depend on the age of the biofilm, the accelerating voltage of the electron beam and the observation time.
3. The homogeneous biofilm model may not be a suitable representation of biofilm growth on BAF media.

CHAPTER 10

THE EFFECTS OF MEDIA SIZE ON THE PERFORMANCE AND EFFICIENCY OF BIOLOGICAL AERATED FILTERS

ABSTRACT

Biological aerated filters (BAFs) are an attractive process option, particularly when low land usage is required. They can combine BOD, solids and ammonia removal and can be utilised at both secondary and tertiary stages of WWT. Media selection is critical in the design and operation of BAFs to achieve effluent quality requirements. Two size ranges, 1.5-3.5 mm and 2.5-4.5 mm, of a foamed clay called StarLight C were used in pilot-scale reactors. The BAFs, using both media, produced effluents that would meet normal consents above loads of 12 kg COD/m³/d and 5 kg suspended solids/m³/d (based on working volumes).

The most consistent effluent was obtained using the smaller medium since, at flow rates above 0.4 l/min, the BAF using the larger medium produced an effluent containing more than 20 mg/l of suspended solids for over 30 minutes after backwashing. Up to 70% longer run times, as determined by reaching a set head loss, were recorded for the BAF containing the larger rather than the smaller medium. Additionally, the development of pressure above the smaller medium filter bed tended to be logarithmic rather than linear. Reactor profiles indicated that suspended solids removal did not occur over the full 2.3 m depth of the columns. The BAF containing the smaller medium utilised a mean depth of 1.7 ± 0.3 m, whereas a mean depth of 2.1 ± 0.3 m was used by the larger medium BAF. Both the head loss development data and the suspended solids removal profiles indicated that the smaller medium BAF was under-performing as a filter.

10.1 INTRODUCTION

Biological aerated filters (BAFs) are flexible reactors which provide a small footprint process option at various stages of wastewater treatment (WWT). True BAFs, as defined by Stephenson (1997), contain a granular medium that provides a large surface area for biofilm development. The medium also allows the reactors to act as depth filters and incorporate suspended solids removal. As a fixed film process, optimal conditions for the relevant micro-organisms can be maintained independently of hydraulic retention times. The process has therefore achieved high levels of nitrification, denitrification and phosphate uptake (Goncalves & Rogalla, 1992).

The selection of a suitable BAF medium is critical in the design and operation of the process, to enable the required effluent standards to be reached. Superior substrate removal has been shown by BAFs containing mineral media, such as expanded clay, compared to those using sand or plastic media with similar dimensions (Chapter 5). The size of a BAF medium also has a strong influence on process performance. Consequently, different sized media have been recommended for different applications (Mendoza-Espinosa & Stephenson, 1999). A medium larger than 6 mm may be preferable for a roughing stage BAF prior to full secondary treatment. Meanwhile, it has been suggested that a tertiary treatment BAF should use a medium smaller than 3 mm. The intermediate size range of 3 to 6 mm has been designated suitable for secondary treatment BAFs. However, within this mid-range, media with different mean hydraulic sizes will infer different operational constraints on the BAF process.

The performances of pilot-scale downflow BAFs, using four size ranges of the mineral medium Lytag, have been compared (Smith *et al.*, 1999). All except one of the samples of Lytag had a mean hydraulic size between 3 and 6 mm and so would have been considered suitable for a secondary treatment BAF. The results indicated that generally, for a given reactor to produce a particular effluent quality, media size dictates the maximum loading rate. The explanation is that a smaller medium offers a greater specific surface area for biofilm development, and so minimises the required reactor volume.

Using three variants of a vitrified clay medium with mean particle sizes between 2.8 and 4.4 mm, it was found that decreasing the size of a BAF medium increased its backwashing requirements (Stensel *et al.*, 1988). This would have a negative effect on the reactor's operating costs and productivity. The backwashing procedure in a BAF is normally initiated when a limiting head loss has developed, as a result of deposited solids reducing the filter porosity.

Therefore, previous work indicates that decreasing BAF media size improves the reactor's substrate removal performance but increases its backwashing requirements. However, the mechanisms which cause these changes remain unclear. In this study, to achieve a better understanding of the influences of media size, the performance of two size ranges of foamed clay have been analysed in terms of substrate removal, suspended solids removal kinetics, and maximum run times.

10.2 MATERIALS AND METHODS

10.2.1 Operating Parameters

Two identical PVC reactors were built, with a diameter of 0.2 m. This diameter was selected because wall effects are limited if a filter’s diameter is more than fifty times the mean particle size. If the wall effects were significant there would be a greater variance in head loss build up (Lang *et al.*, 1993). Full details of the reactor set up and start-up procedure are given in Section 5.2, and the operating conditions are summarised in Table 10.1.

Table 10.1: Summary of the operating parameters for the two pilot-scale BAFs

Flow rate (l/min)	Day	Medium size range (mm)	HRT (mins)	HLR (m/h)	Organic Loading Rates (kg/m ³ /d)*			
					tCOD	sCOD	SS	NH ₃
0.3	1-40	1.5-3.5	96	1.4	4.9	2.1	1.8	0.4
		2.5-4.5	108	1.3	4.4	1.9	1.6	0.4
0.4	40-61	1.5-3.5	72	1.9	8.1	2.9	2.6	0.6
		2.5-4.5	81	1.7	7.2	2.6	2.3	0.5
0.5	61-69	1.5-3.5	58	2.4	7.9	2.7	2.6	0.7
		2.5-4.5	65	2.1	7.1	2.4	2.3	0.6
0.6	69-76	1.5-3.5	48	2.9	13.7	5.7	4.6	1.1
		2.5-4.5	54	2.5	12.2	5.0	4.1	0.9

* based on working volume of reactors

One reactor contained the cylindrical foamed clay StarLight C, with particle diameters ranging between 1.5 and 3.5 mm. Details of the characteristics of this medium and previous performance results can be found in Section 6.3. The other reactor used a larger StarLight C with particle diameters ranging from 2.5 to 4.5 mm. Since the cylinders were up to three times longer than they were wide, it was assumed both would be categorised as suitable for secondary treatment BAFs. The two media have similar physical properties in terms of acid solubility, dirt content and attrition levels. The larger version has an estimated sunken density of 1650 kg/m³ rather than 1500

kg/m³, and a packed voidage of 45% rather than 40%. However, since their estimated experimental minimum fluidisation velocities were similar, both media were backwashed with a liquid velocity of 45 m/h. Prior to operation, tracer studies were conducted on the BAF containing the larger medium, following the methods, conditions and analysis used previously (Table 10.2).

Table 10.2: Empty bed tracer study results for the larger StarLight (2.5-4.5 mm)

Liquid flow rate (l/min)	Air flow rate (l/min)	Hydraulic residence times (min)			Equivalent no. tanks (N)
		theoretical	mean	peak	
0.4	0	81	98	78	9
0.4	2	81	102	55	5
0.8	2	41	74	38	3

10.2.2 Monitoring Performance

During the 76 days of operation, influent and effluent samples were taken regularly and the concentrations of suspended solids (SS), soluble and total chemical oxygen demand (sCOD and tCOD) and ammonia were analysed using standard methods (APHA, 1992). Portable probes were used to measure the temperature, pH (Russel, UK) and dissolved oxygen concentrations (Jenway, UK). Ports at intervals of 0.2 m along the columns allowed samples to be drawn at specific intervals so that substrate removal profiles could be examined. The BAFs were backwashed every 24 hours using the procedure reported in Section 5.2.4. Generally, samples were drawn at the end of the filter run, but occasionally, samples were also taken immediately after backwashing to examine the recovery of effluent quality.

Pressure transducers (VEGA, Germany) were attached to the BAFs and the pressure build up above the filter beds was recorded using a data logger (Grant Instruments, UK). At each flow rate the reactors were left without backwashing for up to 72 hours to find the maximum run time before filter blockage.

10.3 RESULTS

Start-up was rapid for both BAFs and steady SS removal was shown after just three days (Figure 10.1). For the smaller medium reactor, mean SS removal was over 90% for the whole period (Table 10.3). This performance was consistent following backwashing and increases in the flow rate (Q), and only fell below 80% on day 69 immediately after the flow rate was increased to 0.6 l/min (Figure 10.1).

Table 10.3: Summary of the substrate removal performance for the two reactors

<div>Medium</div> <div>Substrate</div>	Small (mean ± stdev @ Q l/min):					Large (mean ± stdev @ Q l/min):				
	0.3	0.4	0.5	0.6	Overall	0.3	0.4	0.5	0.6	Overall
tCOD: % rem.	78	88	87	86	83 ± 10	71	84	85	80	77 ± 14
Eff. (mg/l)	62	44	41	63	56 ± 22	86	60	49	96	77 ± 36
sCOD: % rem.	65	78	73	76	70 ± 12	64	74	69	70	68 ± 12
Eff. (mg/l)	48	30	29	46	41 ± 16	50	35	34	56	45 ± 17
SS: % rem.	91	93	95	89	92 ± 7	82	87	94	83	85 ± 13
Eff. (mg/l)	10	10	5	16	10 ± 7	20	17	8	22	19 ± 13
NH ₃ : % rem.	33	63	62	58	47 ± 32	28	50	62	38	39 ± 26
Eff. (mg/l)	19	12	11	16	16 ± 9	20	15	12	22	18 ± 8

Note: Eff. Represents effluent

By contrast, the effluent from the larger medium reactor had less consistent levels of solids. Although the effluent SS concentrations were generally below 20 mg/l at all flow rates, there were seven peaks above 25 mg/l, and the effluent contained over 30 mg/l of SS on days 16, 48, 53 and 76. Most of these troughs can be explained, since the effluents were sampled immediately after backwashing on days 9, 16, 33, 40, 48, 53, 61 and 69. The larger medium exhibited reduced SS removal after backwashing and the recovery time appeared to depend on the flow rate (Figure 10.2). At 0.3 l/min the effluent recovered within 10 minutes whereas at 0.4 l/min, recovery took over 30 minutes.

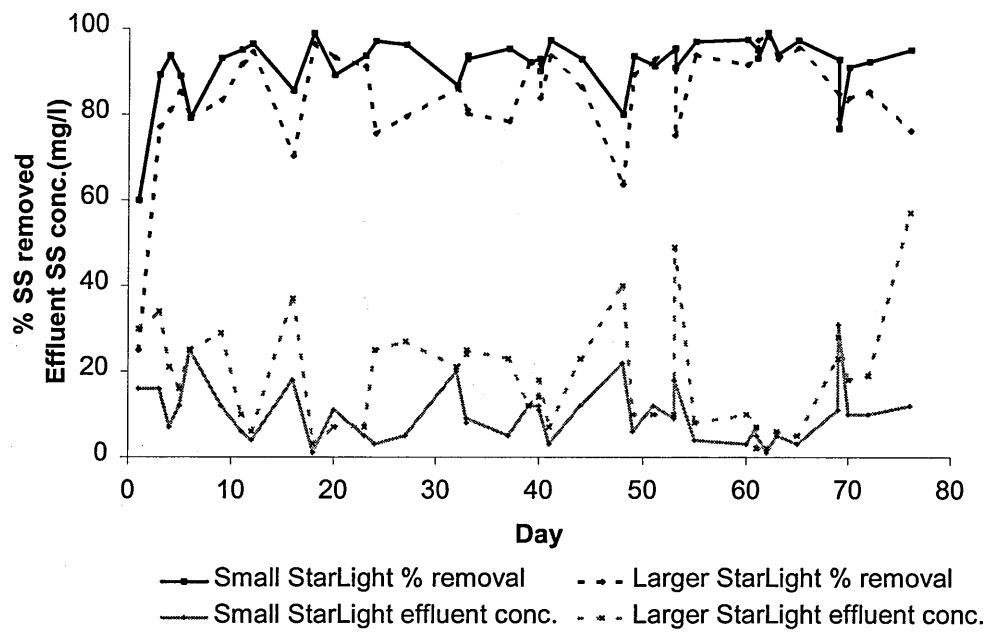


Figure 10.1: Suspended solids (SS) removal performance during operation

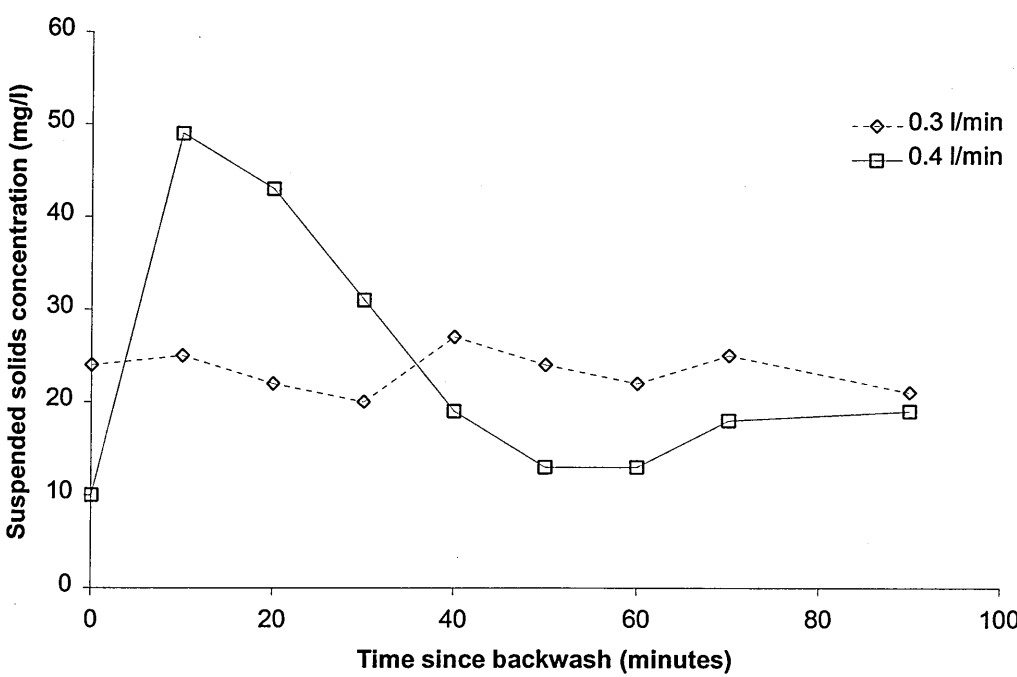


Figure 10.2: Effluent recovery after backwash in the BAF containing 2.5-4.5 mm StarLight at flow rates 0.3 and 0.4 l/min

Both reactors reached stable operation sCOD removal on the third day of operation (Figure 10.3) and subsequently achieved an overall mean of approximately 70% removal (Table 10.3). The mean percentages of sBOD in the sCOD were 80% in the influent and 12% in the effluents. Therefore, the BAFs achieved a mean sBOD removal of 96%. In both BAFs, sCOD removal was not impaired after backwashing and any reductions in removal were related to an unusually high, or low, influent concentration.

Nitrification was first observed on day 20 (Figure 10.4), and by day 37 the effluent ammonia concentration was 0.1 mg/l from the smaller medium reactor and 3.7 mg/l from the larger medium reactor. Ammonia removal did not stabilise, mainly because the influent concentration varied continuously between 13.3 and 46 mg/l. The low effluent ammonia concentrations on day 37 were only repeated on day 61, after influent concentrations dropped from above 40 mg/l to below 15 mg/l.

Performing paired t-tests on the effluent results during stable operation indicated that there were significant differences in the suspended solids, sCOD and the ammonia concentrations for the two reactors (p-values <0.001, 0.03 and <0.001 respectively).

The reactors had approximately 0.85 m clearance above the media beds to allow head loss development during the filter cycle. The maximum run time at each flow rate occurred when the maximum head loss was reached. At all flow rates, the reactor using the larger medium had longer filter run times. However, the difference between the maximum run times of the two reactors decreased as the flow rate was increased (Table 10.4). When a flow rate of 0.3 l/min was applied, there was a period when both reactors operated without much increase in head loss, followed by a logarithmic rise in pressure (Figure 10.5). Meanwhile, when a flow rate of 0.4 l/min was applied, the head loss above the reactor containing the larger medium increased more linearly with time (Figure 10.6). With a flow rate of 0.6 l/min the head loss above both reactors increased linearly for a period before a logarithmic rise in pressure.

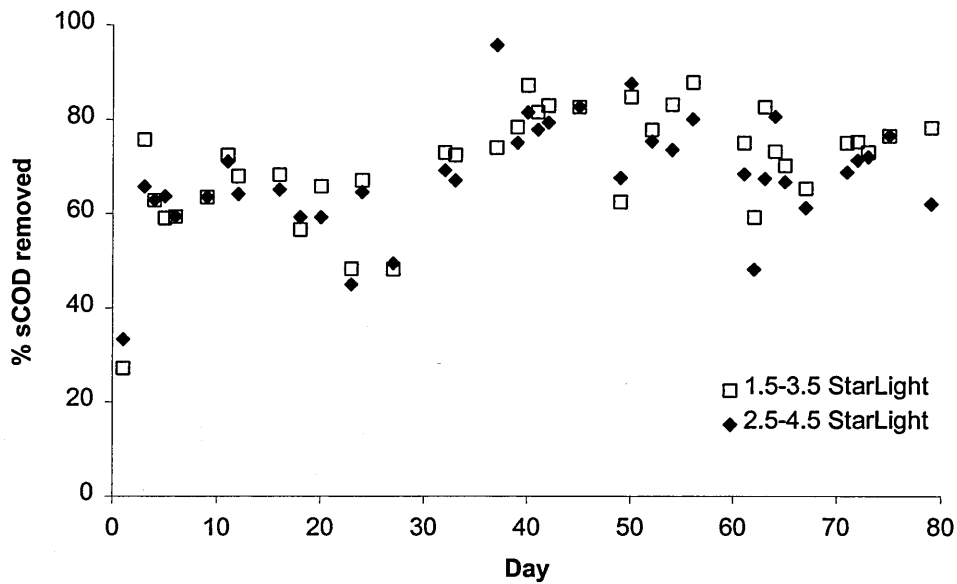


Figure 10.3: % removal of sCOD during operation

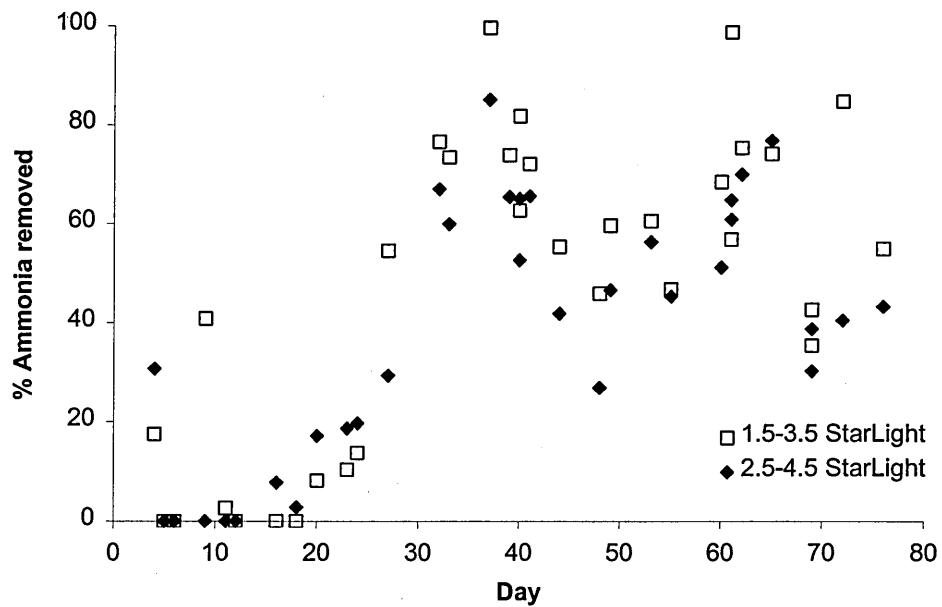


Figure 10.4: % removal of ammonia during operation

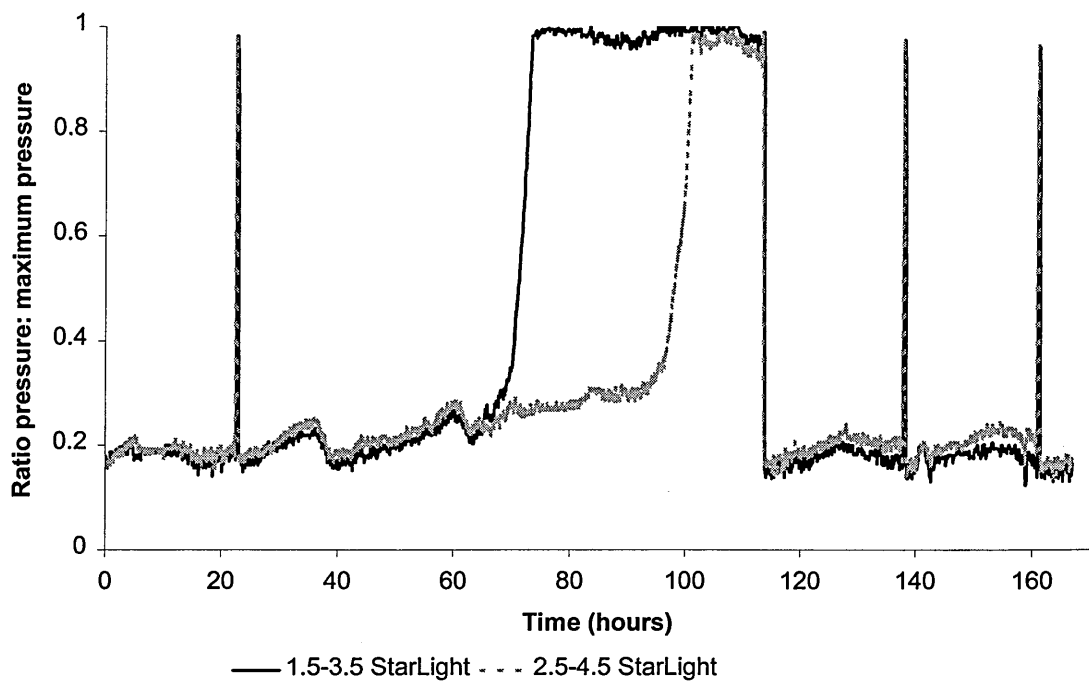


Figure 10.5: Pressure development with a flow rate of 0.3 l/min

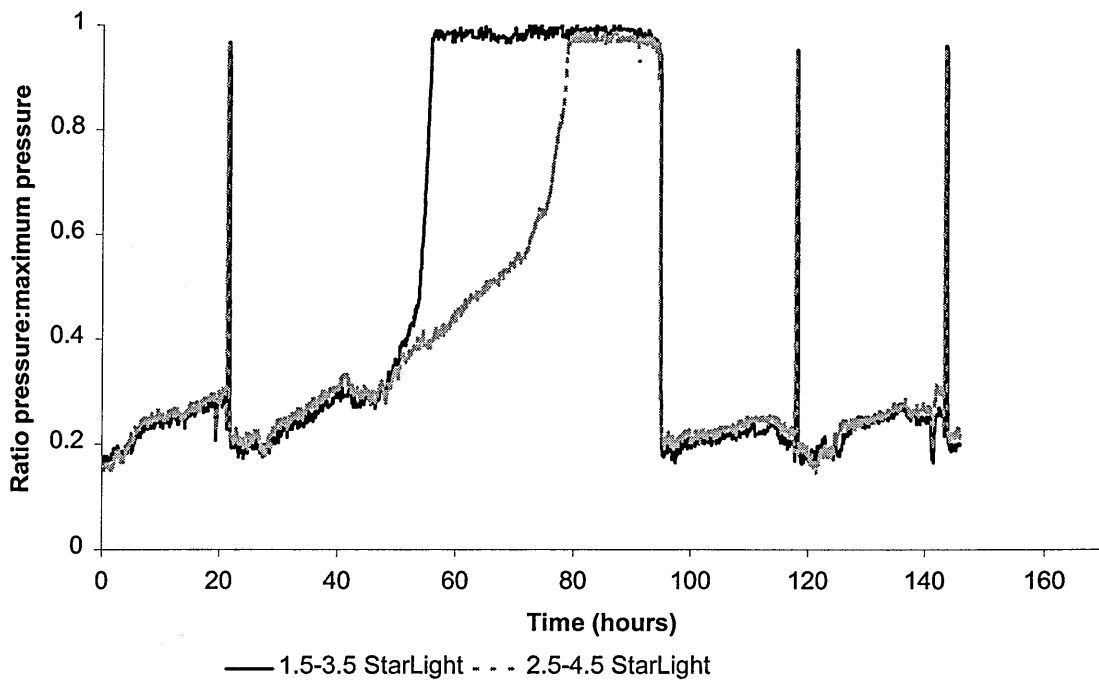


Figure 10.6: Pressure development with a flow rate of 0.4 l/min

Table 10.4: Mean maximum run times for reactors at each flow rate

Flow rate (l/min)	Mean of 2 maximum run times (hours)			
	Small medium	Larger medium	Difference	% difference
0.3	47	78	31	66
0.4	30	51	21	70
0.5	27	37	10	37
0.6	21	29	8	38

Plots of the SS concentrations (C) against reactor height (L) gave typical first-order curves which could be described by Equation 14.

-dC/dt = kC (or after integration: ln [C/C₀] = kt)

(14)

Where C₀ is the influent SS concentration and k is the rate constant. The time (t) corresponds to: LA/Q where, A is the cross-sectional area of the reactor and Q is the volumetric flow rate. For a first-order relationship, plotting ln(C/C₀) versus height results in a linear regression line with a gradient (m) equal to kA/Q (Figure 10.7). The first-order constant (k or mQ/A) for SS removal was higher for the BAF using small StarLight C than the BAF using larger StarLight C (Table 10.5). The R² values for these graphs were consistently over 0.9, which validates the assumption of first-order kinetics.

Table 10.5: Mean kinetic parameters of suspended solids removal

Medium	Q (l/min)	m	mQ/A	Depth (m)	R ²
	[no. profiles]	mean ± stdev	mean ± stdev	mean ± stdev	mean ± stdev
Smaller StarLight	0.3 [5]	1.8 ± 0.5	62 ± 18	1.7 ± 0.4	0.96 ± 0.02
	0.4 [6]	1.5 ± 0.2	70 ± 11	1.7 ± 0.2	0.91 ± 0.05
	Overall	1.7 ± 0.4	66 ± 14	1.7 ± 0.3	0.93 ± 0.04
Larger StarLight	0.3 [6]	1.4 ± 0.2	42 ± 6	2 ± 0.4	0.96 ± 0.02
	0.4 [3]	1.1 ± 0.1	45 ± 4	2.2 ± 0.2	0.95 ± 0.03
	Overall	1.3 ± 0.2	43 ± 6	2.1 ± 0.3	0.95 ± 0.03

When using the smaller medium, the majority of profiles indicated that only part of the filter was being utilised for SS removal. The mean active depth was 1.7 m and in the most extreme case, only the first 1.3 m of the filter bed removed SS before the concentrations stabilised (Figure 10.8). A greater proportion of the filter bed was used for SS removal in the BAF using the larger StarLight. In both reactors, there were no significant changes to the mean values of m , k or the depth of filter used for SS removal after the influent flow rate was increased.

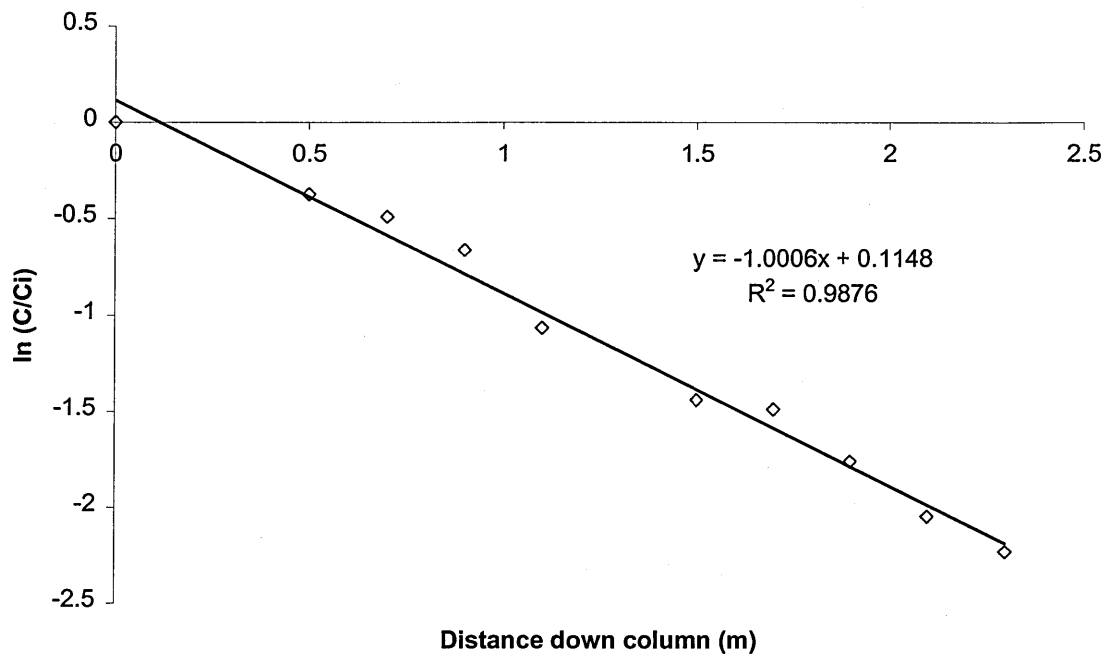


Figure 10.7: Logarithmic suspended solids removal profile using 2.5-4.5 mm StarLight with a flow rate of 0.4 l/min

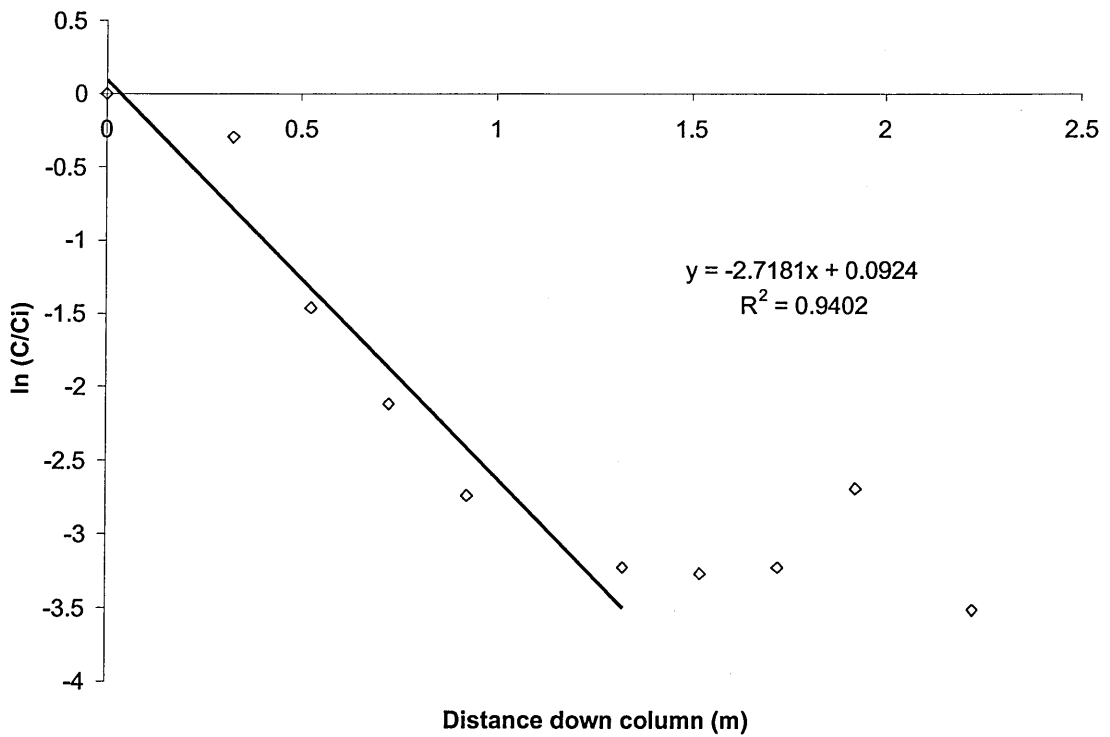


Figure 10.8: Logarithmic suspended solids removal profile using 1.5-3.5 mm StarLight with a flow rate of 0.3 l/min

10.4 DISCUSSION

The tracer studies for the BAF containing large StarLight were compared with those conducted previously under identical conditions using the smaller StarLight, expanded clay, sand and plastic media (Section 6.3.3). This indicated that packed beds of the larger StarLight and the expanded clay medium have similar hydraulic properties. Although these two media are different shapes, packed beds of both have voidage levels of approximately 45%. On empty bed trials using these two media, the peak tracer concentration appeared before the theoretical hydraulic residence time (tHRT), indicating the occurrence of channelling. However, when tracer studies were conducted on an operating bed of the expanded clay just before backwashing, channelling was no longer apparent, due to re-compaction and biofilm growth closing the preferential channels. In conclusion, the tracer study results for large StarLight highlight that poor performance, especially immediately after backwashing, may result from preferential channelling. Meanwhile, the reactor containing the smaller StarLight showed better plug flow conditions, which improves nitrification and solids removal (Mann *et al.*, 1995, Fdz-Polanco *et al.*, 1994b). Any further increases in the size of StarLight would probably increase the levels of mixing so that the flow through the reactor no longer approximated plug flow. Correspondingly nitrification would deteriorate as observed when using 3-6 mm, compared to 2-4 mm, expanded schist (Paffoni *et al.*, 1990).

The effluents contained consistently lower levels of organic matter than required by many consents. The medium with the smaller size range showed significantly better sCOD and ammonia removal. Ammonia removal is a good indicator of biological performance since nitrifying bacteria are easily out-competed by heterotrophic organisms. Therefore, nitrification only occurs once COD levels are low, at the base of a downflow filter. The results suggest that COD penetrated deeper into the reactor containing the larger medium. Thus, a BAF containing the smaller medium would treat a greater mass of substrate per volume of reactor. However, it may be possible to obtain similar performance and reactor footprint using the larger sized medium, by increasing the reactor height.

Both reactors showed improved COD removal when the flow rate was increased from 0.3 to 0.4 l/min. This could be accounted for by changes in the influent. Alternatively, the increased liquid velocity may have improved substrate transfer into the biofilm (Peladan *et al.*, 1996) and, by controlling the biofilm thickness, removed internal diffusion limitations (Lazarova & Manem, 1994). COD removal was not affected by further increases in the flow rate. However, the lowered ammonia removal observed at 0.6 l/min indicated that biological performance would eventually decrease with increasing flow rate.

Effluent SS consent levels are often set at or below 35 mg/l. The mean concentrations for both effluents at all flow rates were below this concentration. However, the standard could be exceeded for over twenty minutes after backwashing the larger medium reactor. Backwash recovery times of up to 45 minutes have been reported previously and, in a multi-cell arrangement, significant effects on effluent quality can be avoided (Budge & Gorrie, 1996).

A full-scale BAF may have improved operability and show lower media washout rates using the larger sized medium because it would be easier to separate media particles from the backwash water. Additionally, with its longer filter run times, the BAF using the larger medium would be the most efficient reactor in terms of operating costs. In BAF facilities with numerous cells, run-times below 24 hours are desirable to minimise the possibility that several cells require backwashing simultaneously (Stensel *et al.*, 1988). Therefore, the smaller medium may be limited to applications where the liquid velocity is below 2.4 m/h. As well as having different run times, the pattern of head loss development differed using the two sizes of media.

The head loss development and SS profile results can be interpreted by assuming that similar filtration mechanisms operate in depth filters and BAFs. Solids are transported to the surface of depth filter media grains by a combination of interception, diffusion and sedimentation (Bouwer, 1987). Deep bed filters are said to ripen because their performance improves initially during operation, as particle deposition increases the apparent filter grain size and reduces the filter porosity (Choo & Tien, 1993). In BAFs, this process will be combined with the metabolism of suspended matter and

biofilm growth (Takahashi *et al.*, 1969). Head loss development occurs in depth filters as deposited solids reduce the filter porosity.

Efficient depth filters exhibit linear head loss development, and block when the solids holding capacity is reached. Backwashing is required either at this point, or when the head loss profile causes air pockets to develop in the lower regions of the filter. Air pockets effectively block pores, causing an increase in local flow rates and subsequently a decrease in filtrate quality (Ives, 1969). Solids are removed throughout an efficient depth filter, and a first-order equation characterises the reaction.

Under-performing filters remove smaller proportions of solids in each successive layer of media. Prior to reaching the solids holding capacity, the porosity at the top of such filters has been reduced sufficiently to induce straining. This causes the rapid production of a surface mat of deposit, and subsequently there is a logarithmic rise in head loss with time (Ives, 1969). Under-performing filters are undesirable since they require more frequent backwashing and have higher operating costs without offering any advantages.

At flow rates of 0.3 and 0.4 l/min, the SS removal profiles indicated that the BAF using the smaller medium was under-performing as a filter. Solids were not removed throughout the filter depth, so it is unlikely that the filter's solids holding capacity was ever reached. Additionally, the reactor showed a tendency for logarithmic rather than linear head loss development. It is likely that the under-performance of the smaller StarLight as a depth filter medium was part of the reason that shorter run times were recorded.

There appears to be a complex relationship between the influent characteristics and flow rate, the reactor height and media size, and the performance in terms of both effluent standards and operating costs. The topic warrants further investigation to determine optimal media sizes for different conditions. Dual media BAFs also have potential since they could combine the high performance properties of smaller media with the lower operating costs associated with larger media. This may be particularly successful using foamed clay, since the particle density can be controlled so that a layer of larger medium could be maintained above a layer of smaller medium.

10.5 CONCLUSIONS

1. Using either of the size ranges of StarLight C, the BAFs produced effluents with mean concentrations of SS and COD within most consent levels, at loadings over 12 kg tCOD/m³/d and 4 kg SS/m³/d (based on working volumes).
2. The BAF utilising the smaller medium exhibited up to 70% shorter maximum run times than the BAF using the larger medium. The hydraulic loading of the small medium reactors may need to be restricted to below 2.4 m/h to prevent units requiring simultaneous backwashing. Additionally, the larger medium will be easier to separate from backwash water, which may improve process operability and reduce media washout.
3. The SS profiles and head loss development graphs indicated that the smaller medium was frequently exhibiting inefficient depth filtration. This partly explained the differences in filter run times and is highly undesirable.
4. The larger sized medium showed some inconsistency in SS removal immediately after backwashing at flow rates over 0.3 l/min, and inferior nitrification results. It is unclear whether this was due to the reactor having a decreased specific surface area or a higher degree of mixing.

CHAPTER 11

MODELLING THE SUBSTRATE REMOVAL PROPERTIES OF BAFS

ABSTRACT

Three empirical models were tested in an attempt to characterise the substrate removal properties of BAFs. Reaction constants were calculated for pilot-scale BAFs which utilised expanded clay, or either 1.5-3.5 mm or 2.5-4.5 mm diameter foamed clay media. These constants were then used to rank the three different media in order of their substrate removal performance.

The substrate profiles did not fit the model proposed by Mann & Stephenson (1997) because the first-order constant did not decrease with the influent substrate concentration. There appeared to be two distinct regions in the columns, each showing different substrate removal behaviour. Removal appeared to follow first-order kinetics in the first section and there were only small standard deviations for the first-order constants relating to a particular medium and flow rate. Applying a retardant equation allowed a greater proportion of the column to be characterised, but there were larger variations in the rate constant.

All three materials tested showed better substrate removal characteristics than the sand and plastic media used by previous workers. The expanded clay medium had the highest rate constant followed by the smaller and then the larger StarLight C.

11.1 INTRODUCTION

Process modelling is normally undertaken with one of two aims. One is to improve the understanding of the fundamental biological, chemical and physical reactions that drive the process. Mechanistic models are used for this purpose, and the overall equations tend to be complex expressions containing many dimensionless parameter groups (Jennings *et al.*, 1976). Alternatively, empirical process models are used to aid the selection and design of the process. This type of model uses relatively simple expressions to describe how key reactor parameters effect the performance characteristics of the process. Ideally, the mechanistic models for a particular type of wastewater treatment (WWT) reactor will provide a rational basis for selecting the correlation used in an empirical model.

In a BAF, the medium has a large influence on the reactor performance. Therefore, an empirical model for a BAF should contain a constant whose value depends on the medium. The calculation of such a constant would provide a rational and convenient method to compare the performance of different materials as BAF media.

11.2 THEORY

11.2.1 Background

If a biodegradation reaction is first-order with respect to a substrate, then the rate of substrate removal is proportional to its concentration (Chapter 10, Equation 14). However, if the reaction is zero-order, it will proceed at the same rate independent of the substrate concentration (Equation 15). Simplistically, observing first-order substrate removal in a biological WWT reactor suggests that there may be an excess of microbial activity compared to the substrate. Whilst, observing a zero-order reaction implies that there may be a substrate excess compared to microbial activity.

$$-dC / dt = k \quad (\text{which integrates to: } C_0 - C = kt) \quad (15)$$

11.2.2 Mechanistic Biofilm Reactor Models

Most mechanistic biofilm models couple microbial substrate removal kinetics, described in the Monod equation, with the fundamental principles of mass transport, described by Fick's Law (Arvin & Harremoes, 1989). The kinetics exhibited by a heterotrophic biofilm can be simplified by using first-order or zero-order linearisations of the Monod expression under different conditions (Mathusaran, 1978). A biofilm will tend to show first-order kinetics when the substrate concentrations are low and there is a large surface area for mass transport. A first-order reaction in the biofilm can be interpreted as a corresponding first-order bulk reaction (Henze *et al.*, 1995). As the substrate concentration increases, all biofilms will eventually switch to show zero-order kinetics. If this transition occurs at a low biomass concentration, the reactor will also switch directly to exhibit zero-order reaction kinetics. However, the increased biofilm depth and density usually associated with increased substrate concentrations, results in the internal mass transfer within the biofilm becoming limiting before the substrate is present in excess. The result is a partially penetrated biofilm and a reactor showing half-order kinetics (Equation 16).

$$-dC / dt = k (C)^{1/2} \quad (16)$$

Therefore, the observed reaction rate in a biofilm reactor can be either first, half or zero-order. When plotting the observed reaction rate against concentration, the region where half-order kinetic apply forms a soft transition between the regions of first and zero-order kinetics. This has often been confused with Monod kinetics, but is actually an expression of a diffusional limitation in the biomass (Henze *et al.*, 1995).

Half-order kinetics have been used to describe the performance of rotating biological contactors (RBCs) and it has been suggested that BOD removal in biofilms is predominantly a zero-order reaction, which becomes a half-order bulk reaction (Harremoes, 1977). However, the biofilm retention properties in a BAF differ in a number of ways to those in either trickling filters (TFs) or RBCs. The biofilm surface area is much greater in a BAF due to the high specific surface area of the medium. Additionally, biofilms in RBCs are 2-3 mm thick and so deeper than those reported in BAFs. Biofilm thickness in a BAF depends on substrate concentrations, fluid velocities and the backwash frequency. The depth is also quite variable across the medium surface but has been reported to be approximately 30-50 μm (Lee & Stensel, 1986). Thus, the biofilms in a BAF are more likely to be fully penetrated with substrate, and to show first-order kinetics at higher substrate concentrations, than the biofilms in TFs or RBCs.

In an attempt to model the performance of submerged biological filters, Jennings *et al.* (1976) combined a biofilm model (Williamson & McCarty, 1976) with a plug flow model for flow through a porous bed of spherical, biolayer coated particles. It was found that the reactor showed first-order kinetics even at relatively high values of influent substrate concentration. This justifies using a first-order correlation for empirical BAF models.

11.2.3 Empirical Biofilm Reactor Models

Most of the empirical biofilm reactor models have been designed for and tested on TFs. Velz (1948) was the first to propose that the removal of organic matter in a TF was a first-order reaction (Equation 17).

$$C / C_0 = e^{-kL} \quad (17)$$

As well as postulating that the rate of extraction per interval of depth was proportional to the remaining concentration of organic matter, Velz (1948) identified the importance of measuring organic matter in terms of its degradability. He proposed that not all organic matter is degradable, even with extended hydraulic retention times, and recognised that the dissolved fraction was the most difficult to remove. Despite his insight to its importance, Velz (1948) did not offer a method or procedure to quantify the degradability of organic matter and used BOD concentrations in his calculations.

Velz's equation is restrictive since it does not consider the influent flow rate and it was developed by Schulze (1960) to give Equation 18, where n is a constant.

$$C / C_0 = e^{-kt} \text{ where } t = LA / Q^n \quad (18)$$

Eckenfelder (1961) returned to the problem of accounting for differing BOD degradability. He stated that BOD removal probably followed a retardant form of the first-order reaction since, the more readily treatable components of BOD will be removed most rapidly. The basic equations for a retardant-order equation are given in Equation 19 and Eckenfelder's proposed relationship in Equation 20, where p , k' and K are constants.

$$-dC / dt = k' C (C / C_0) \quad (\text{which integrates to } C_0 / C = 1 + k't) \quad (19)$$

$$C / C_0 = 100 / (1 + [KAL^{(1-p)} / Q^n]) \quad (20)$$

The exponent (n) characterises the changes in hydraulic retention time with depth and was thought to be 1/3 for turbulent flow and 2/3 for laminar flow. The other exponent (p) allows for changes in the biofilm distribution with depth. When there is an even distribution of biofilm throughout the reactor, (p) will become zero.

Recently, these empirical trickling filter models have been applied successfully to other fixed film wastewater treatment reactors. Chung *et al.* (1996) found a good correlation between the BOD removal data from a fluidised bed and Equation 18, when using a bench scale reactor fed with artificial sewage. However, the model slightly underestimated the removal of dairy waste by the pilot plant.

Mann & Stephenson (1997) applied an adjusted version of Equation 18 to soluble COD removal data from BAFs. It was proposed that the first-order constant was dependent on the initial substrate concentration (Equation 21).

$$C / C_0 = e^{-Lm} \text{ where } m = k^* A / Q C_0^q \quad (21)$$

(which integrates to $-dC / dt = k^* C / C_0^q$)

Where k^* is the overall process constant and q is a constant depending on the medium. The results of Mendoza-Espinosa & Stephenson (1998) appeared to confirm the applicability of the model. However, Mendoza-Espinosa (1999) found that in upflow pilot-scale BAFs, Equation 21 tended to underestimate the removal performance at the bottom of the reactors and overestimate performance in the top section. These inaccuracies may be partly linked to the equation using soluble COD as a measure of biodegradable organic matter whilst making no allowances for changes in degradability with reactor depth.

11.2 4 Test Models

In this chapter, the substrate removal kinetics in BAFs will be compared to Mann & Stephenson's (1997) model (Equation 21) and to forms of both the simple first-order model (Equation 23) and the retardant model (Equation 24). The aim is to select the model that best describes the data and then link the rate constants to the BAF medium.

BAFs are submerged reactors in which the retention times can be simply calculated (Equation 22). Limited reports have also indicated that they have a uniform biomass distribution through their depth (Kent, 1998). Therefore, it has been assumed that neither the exponent (n) nor (p) from Equations 18 and 20 would be required, and as a result Equations 18 and 20 can be simplified to Equations 23 and 24 respectively.

$$t = LA / Q \quad (22)$$

$$C / C_0 = e^{-kLA / Q} \quad (23)$$

$$C_0/C = 1 + k' (LA / Q) \quad (24)$$

11.3 MATERIALS AND METHODS

Two identical, pilot-scale, down flow, BAFs were used to examine substrate removal with three different BAF media. A full description of the plant can be found in Section 5.2.1. During the first operational period, BAF 1 contained a standard expanded clay and BAF 2 contained 1.5–3.5 mm diameter StarLight C. During the second operational period, BAF 1 contained 1.5–3.5 mm diameter StarLight C, whilst BAF 2 contained 2.5–4.5 mm diameter StarLight C. A full description of the media can be found in Section 6.3.1 and Tables 5.1 and 10.1 summarise the operating parameters during the periods.

The concentrations of suspended solids, soluble COD (sCOD), total COD (tCOD), ammonia, nitrite and nitrate in the influent and effluents were determined using standard methods (APHA, 1992). Two tailed, paired t-tests were utilised to compare the performance of the two reactors during an operational period, and two tailed t-tests assuming unequal variance were used to compare the performance of the same medium in different operational periods. The observed significance levels (p-values) for these t-tests were calculated for a null hypothesis that there was no difference between the populations. When the p-values were lower than 0.05 the null hypothesis was rejected.

Sampling ports were located at 0.2 m intervals from the bottom of the reactors. When substrate removal had reached stable operation, effluent samples were taken at specific time intervals from the influent pipe, sampling ports and the final effluent pipe. The samples were analysed for dissolved oxygen (Jenway, UK), temperature, pH (Russel, UK) and sCOD. During the first operational period, six profiles were completed at a flow rate of 0.3 l/min, and five at both 0.4 and 0.5 l/min. In the second operational period, six profiles were completed at 0.3 l/min and three at 0.4 l/min.

For BAF 2 during the second operational period, soluble BOD (sBOD) concentrations were determined for the effluent profiles using standard methods (APHA, 1992). Additionally, attempts were made to quantify the biodegradability of the influent using respirometry. The oxygen uptake of 20 ml of activated sludge was compared

when the substrate was either, 30 ml of the filtered influent, or, 30 ml of artificial sewage with a sCOD concentration of 200 mg/l (Sterritt & Lester, 1981).

To test the validity of the three models, graphs were plotted and R^2 values were calculated to show the variation between the data points, and the lines of best fit for the models. As the R^2 value increases from 0 to 1, the correlation between the points and the line increases. For Mann and Stephenson's (1997) model, $\ln (C / C_0)$ was plotted against reactor height to obtain the gradient, $(-m)$. Then, $\ln (Q_m / A)$ was plotted against $\ln (C_0)$ to find (q) and (k^*) . To validate Equation 10, $\ln (C / C_0)$ was plotted against reactor height to obtain (k) and to test Equation 11, (C_0/C) was plotted against reactor height for (k') .

11.4 RESULTS

11.4.1 Overall Performance

A full analysis of the overall performance of the BAFs in the two operational periods can be found in Sections 5.3.2 and 10.3. The suspended solids and sCOD concentrations of the influent and effluents for the two runs are summarised in Table 11.1. T-tests indicated that the mean substrate concentrations in the influents were not significantly different for both experiments (p-value 0.35). A comparison of the results from the reactors utilising the smaller StarLight C, shows that significantly better suspended solids removal was achieved during the second experiment (p-value <0.01), but there was no significant difference between the sCOD removal results (p-value 0.12). The improved performance of the smaller StarLight in the second operational period was probably due to the introduction of a grid, which prevented washout of medium during backwashing. During the first experiment the media washout was measured and has been accounted for in the substrate profiles.

Table 11.1: Summary of performance by reactors 1 and 2 for the two experiments

Run (Reactor)	Suspended solids (mean value $\pm$ stdev)			sCOD (mean value $\pm$ stdev)		
	Influent (mg/l)	% removal @ 0.3 l/min	% removal @ 0.4 l/min	Influent (mg/l)	% removal @ 0.3 l/min	% removal @ 0.4 l/min
1(1)	120 $\pm$ 48	87 $\pm$ 12	85 $\pm$ 10	146 $\pm$ 35	76 $\pm$ 12	74 $\pm$ 12
1(2)	120 $\pm$ 48	85 $\pm$ 7	84 $\pm$ 7	146 $\pm$ 35	74 $\pm$ 6	72 $\pm$ 6
2 (1)	126 $\pm$ 10	91 $\pm$ 9	93 $\pm$ 5	142 $\pm$ 41	65 $\pm$ 13	78 $\pm$ 10
2 (2)	126 $\pm$ 10	82 $\pm$ 15	87 $\pm$ 10	142 $\pm$ 41	64 $\pm$ 13	74 $\pm$ 11

Note: 1 (1) = Expanded clay, 1 (2) & 2 (1) = Smaller StarLight C, 2 (2) = Larger StarLight C

The results of two tailed, paired t-tests showed that there was no consistent significant difference in the performance of the two BAFs during the first operational period (Section 5.3.2). Meanwhile, during the second operational period, the sCOD, suspended solids and ammonia removal were significantly better in the BAF using the smaller StarLight C (section 10.3).

Throughout both experiments, all samples had a dissolved oxygen concentration greater than 4 mg/l, a temperature above 11 °C and a pH above 6.7.

11.4.2 Mann & Stephenson's Model

The gradients (m) obtained after plotting $\ln(C/C_0)$ versus height did not appear to be dependant on the initial substrate concentration (C_0) as suggested by Mann & Stephenson (1997) for any of the BAFs (Figures 11.1 to 11.3). Previously, the values of $\ln(Qm/A)$, when $\ln C_0$ ranges between -1.6 and -2.3 , have been 1.6 to 2.3 for a sand medium (Mendoza-Espinosa & Stephenson, 1998), and 1.2 to 2.3 for a plastic medium (Mann & Stephenson, 1997). These values are significantly lower than the values obtained in this study over a similar influent range (Table 11.2).

Table 11.2: Mean and standard deviations of the expressions of m for the three media using all the data points

$f(m)$	Expanded clay	Smaller StarLight C	Larger StarLight C
$\ln(Qm/A)$	3.1 ± 0.3	3.3 ± 0.3	2.7 ± 0.4
Qm/A	21.9 ± 6.2	27.3 ± 7.5	16.4 ± 7.4

11.4.3 First-order Model

Essentially the first stage of the Mann & Stephenson model calculates the first-order constant (Qm/A) at a particular influent concentration. If a simple first-order model was appropriate, the values of Qm/A would remain constant for each medium at all flow rates and influent concentrations. In fact, as indicated by the standard deviations, the value of Qm/A was variable (Table 11.2). Additionally, the R^2 values for the data points and the linear regression line ranged between 0.25 and 0.92. The R^2 values reported for this relationship previously were between 0.978 and 0.993 (Mann & Stephenson, 1997), and 0.9 and 0.95 (Mendoza-Espinosa & Stephenson, 1998).

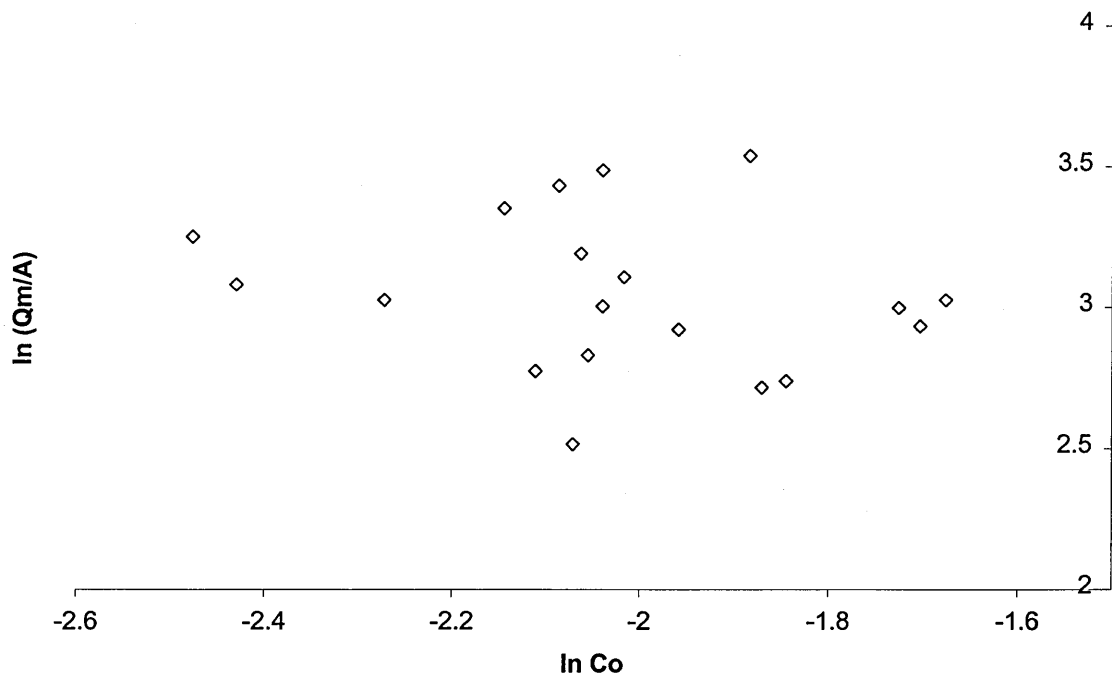


Figure 11.1: Relationship between $\ln(Qm/A)$ and $\ln C_0$ for the expanded clay BAF

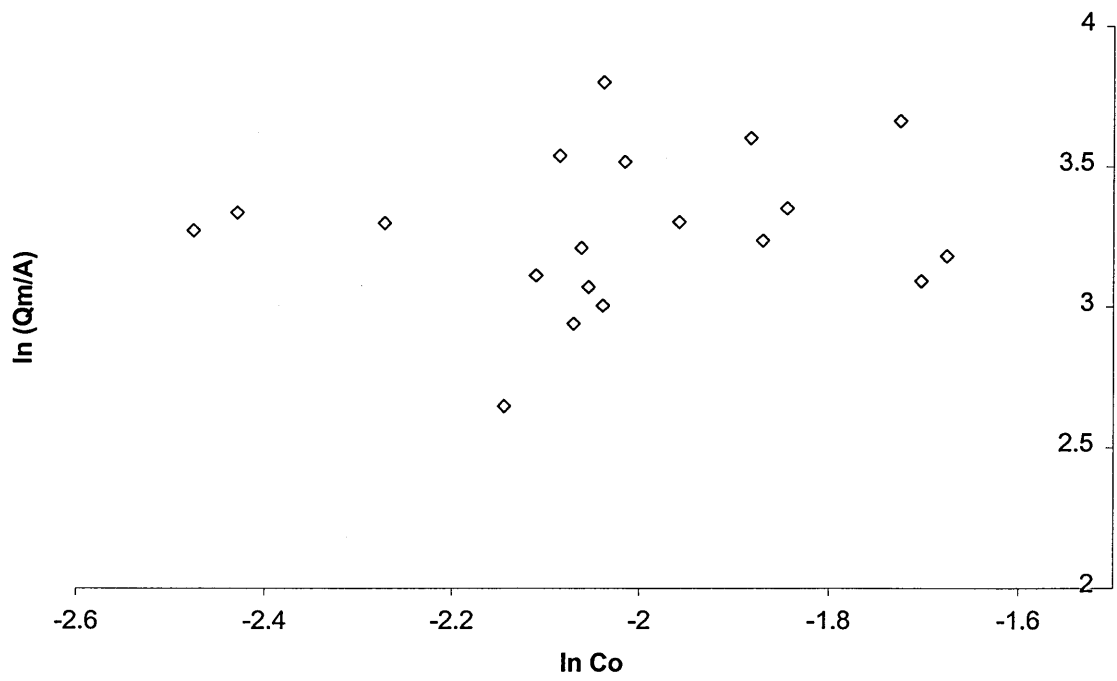


Figure 11.2: Relationship between $\ln(Qm/A)$ and $\ln C_0$ for the smaller StarLight C BAF

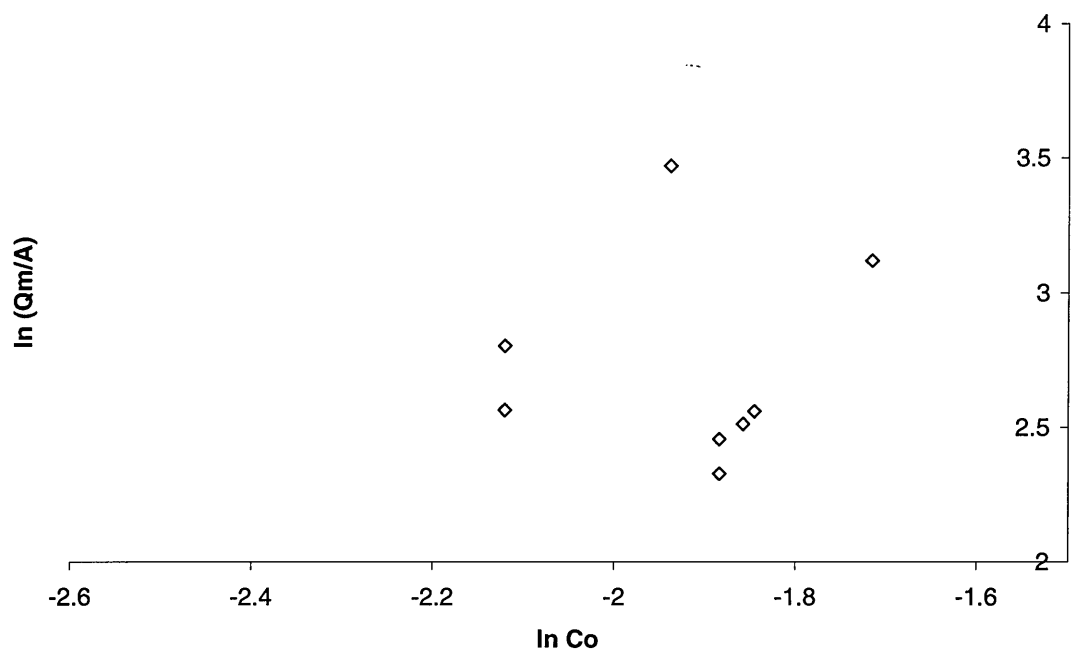


Figure 11.3: Relationship between $\ln(Qm/A)$ and $\ln C_0$ for the larger StarLight C BAF

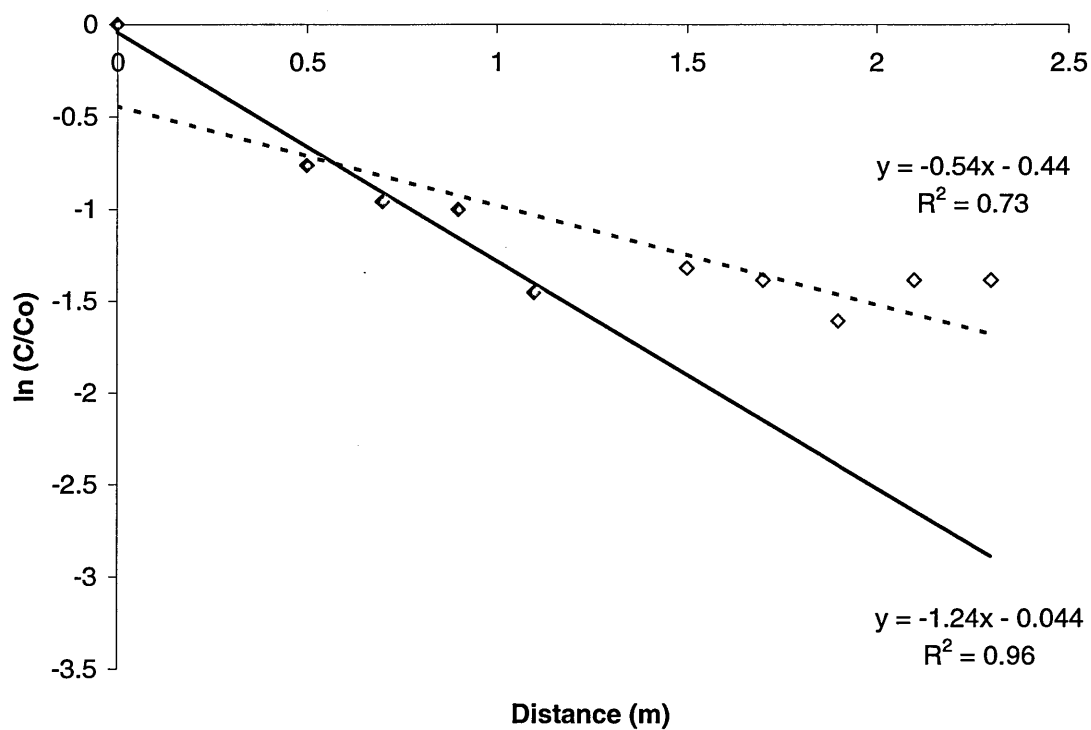


Figure 11.4: An example of the typical pattern of $\ln C/C_0$ against distance

Further examination of the graphs of $\ln C/C_0$ against reactor height, revealed that only the first part of the reactor appeared to have removed substrate. A typical example of the pattern of substrate removal recorded is shown in Figure 11.4. The first five points showed a close correlation to the first-order model ($R^2 = 0.964$) and there appeared to be no further removal in the second half of the reactor. There was less variation of the first-order constant when m was calculated for each run from just the first four or five data points (Figure 11.5 and Table 11.3). The standard deviations were even lower when each flow rate is considered separately.

Table 11.3: Expressions of m for the three media using the first 4 or 5 data points of the C/C_0 versus distance chart, when sCOD is the measured substrate

Medium	Q (m ³ /d)	mQ/A mean $\pm$ stdev	Ln (mQ/A) mean $\pm$ stdev	Mean no. of points	Mean R ²
Expanded Clay	0.432	61.1 $\pm$ 5.8	4.1 $\pm$ 0.09	4.2	0.85
	0.576	61.7 $\pm$ 5.8	4.1 $\pm$ 0.09	4	0.86
	0.72	94.3 $\pm$ 10.5	4.5 $\pm$ 0.1	4	0.81
	Overall	71.7 $\pm$ 17.3	4.3 $\pm$ 0.2	4.1	0.84
Small StarLight C	0.432	44.5 $\pm$ 6.3	3.8 $\pm$ 0.1	5.5	0.92
	0.576	49.5 $\pm$ 3.5	3.9 $\pm$ 0.07	4.2	0.97
	0.72	72.8 $\pm$ 3.7	4.3 $\pm$ 0.05	4.8	0.89
	Overall	54.9 $\pm$ 13.4	4.0 $\pm$ 0.2	4.9	0.93
Larger StarLight C	0.432	33.4 $\pm$ 2.8	3.5 $\pm$ 0.08	4	0.94
	0.576	47.4 $\pm$ 3.2	3.9 $\pm$ 0.07	4.6	0.97
	Overall	38.1 $\pm$ 7.6	3.6 $\pm$ 0.2	4.2	0.95

There was not a consistent trend in the substrate concentrations in the lower half of the reactors. Sometimes removal continued at a reduced rate, and on other occasions the substrate concentration was stable or even increased. The first-order removal of sCOD may have stopped half way through the columns because the remaining sCOD was not easily biodegradable, or because the concentration was insufficient to maintain a biofilm.

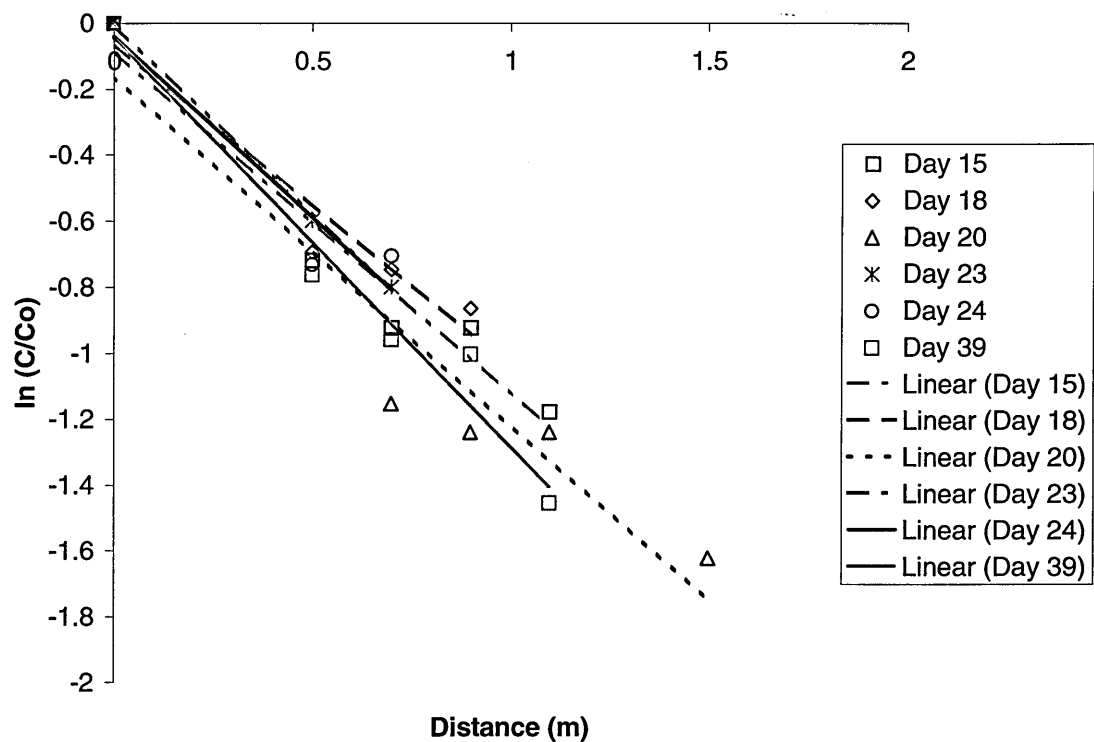


Figure 11.5: The parts of the $\ln C/C_0$ versus distance graphs showing first-order substrate removal for the larger StarLight C at a flow rate of 0.3 l/min

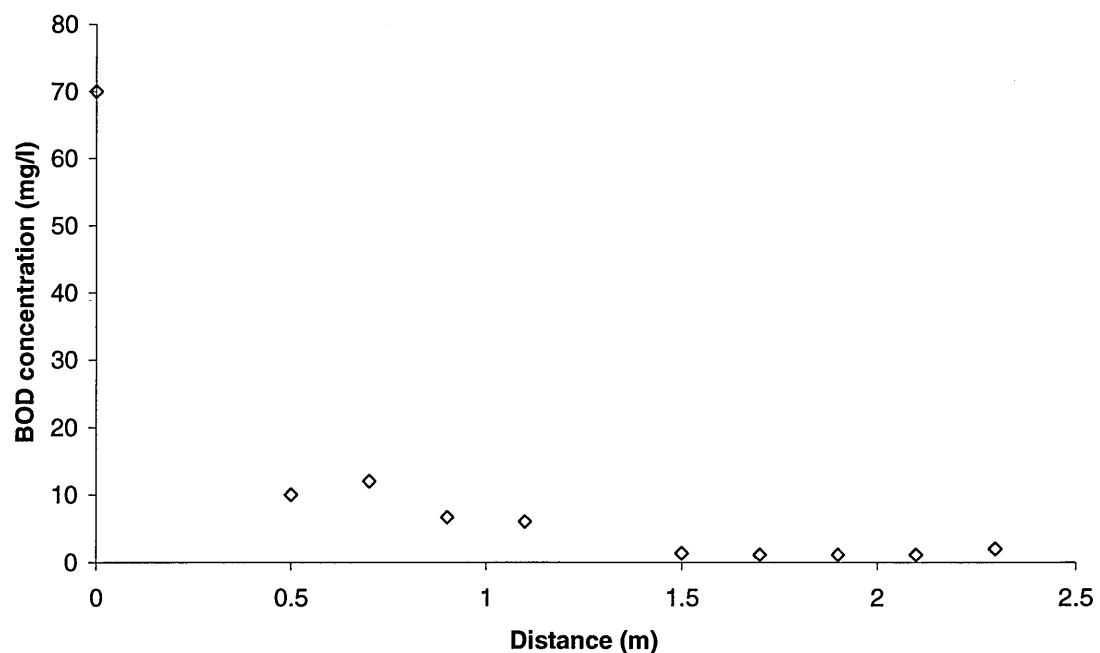


Figure 11.6: Example showing the concentration of BOD through the reactor

The sBOD removal profiles for the larger StarLight C BAF show that low concentrations of sBOD are present throughout most of the column (Figure 11.6). The mean sBOD concentration in the effluent was 2.4 ± 0.9 mg/l and this represented a mean of approximately 7 ± 5 % of the effluent sCOD. A summary of the preliminary results measuring sBOD degradation in the larger StarLight C BAF is given in Table 11.4. As expected the rate of BOD removal was higher than the rate of COD removal, but the standard deviations for (mQ/A) were also higher.

Table 11.4: Expressions of m for the larger StarLight C BAF using the first 4-7 data points of the C/C₀ versus distance chart, when sBOD is the measured substrate

Q (m³/d)	mQ/A mean ± stdev	Ln (mQ/A) mean ± stdev	Mean no. of points	Mean R²
0.432	91.2 ± 30	4.5 ± 0.4	5.2	0.94
0.576	99.0 ± 44	4.5 ± 0.4	5.7	0.97
Overall	93.8 ± 33	4.5 ± 0.4	5.3	0.95

On all the days tested, activated sludge showed a greater oxygen uptake rate with filtered sewage rather than synthetic sewage. This may have been because the micro-organisms had adapted to the sewage. For this preliminary set of results, no correlation was found between the difference in oxygen uptake rates of the filtered and synthetic sewage, and (mQ/A).

11.4.4 Retardant Model

The profile illustrated in Figure 11.4 has been compared to the retardant model in Figure 11.7. There was little change in the R² value when eight rather than five points were used in the comparison. For all three media, the retardant model appeared to fit better than the first-order model when an increased number of data points were included (Table 11.5). The mean R² values increased for the expanded clay BAF but decreased for the BAFs using either of the two types of StarLight C. All the standard deviation values for the expressions of m increased using the retardant model indicating that there was a larger variation in the gradients.

Table 11.5: Expressions of m for the three media using the retardant model

Medium	Q (m ³ /d)	mQ/A mean ± stdev	Ln (mQ/A) mean ± stdev	Mean no. of points	Mean R ²
Expanded Clay	0.432	74.3 ± 27	4.26 ± 0.35	7.3	0.95
	0.576	75.7 ± 12	4.31 ± 0.17	5.8	0.92
	0.72	143.1 ± 28	4.94 ± 0.23	5.8	0.96
	Overall	96.3 ± 39	4.49 ± 0.40	6.4	0.94
Small StarLight C	0.432	96.7 ± 26	4.54 ± 0.28	6.3	0.94
	0.576	77.1 ± 13	4.33 ± 0.16	6.4	0.88
	0.72	161.8 ± 34	5.07 ± 0.20	5.6	0.89
	Overall	110.9 ± 44	4.64 ± 0.37	6.1	0.91
Large StarLight C	0.432	44.4 ± 12	3.76 ± 0.27	7	0.86
	0.576	70.7 ± 27	4.21 ± 0.38	7.6	0.89
	Overall	53.2 ± 21	3.91 ± 0.36	7.2	0.87

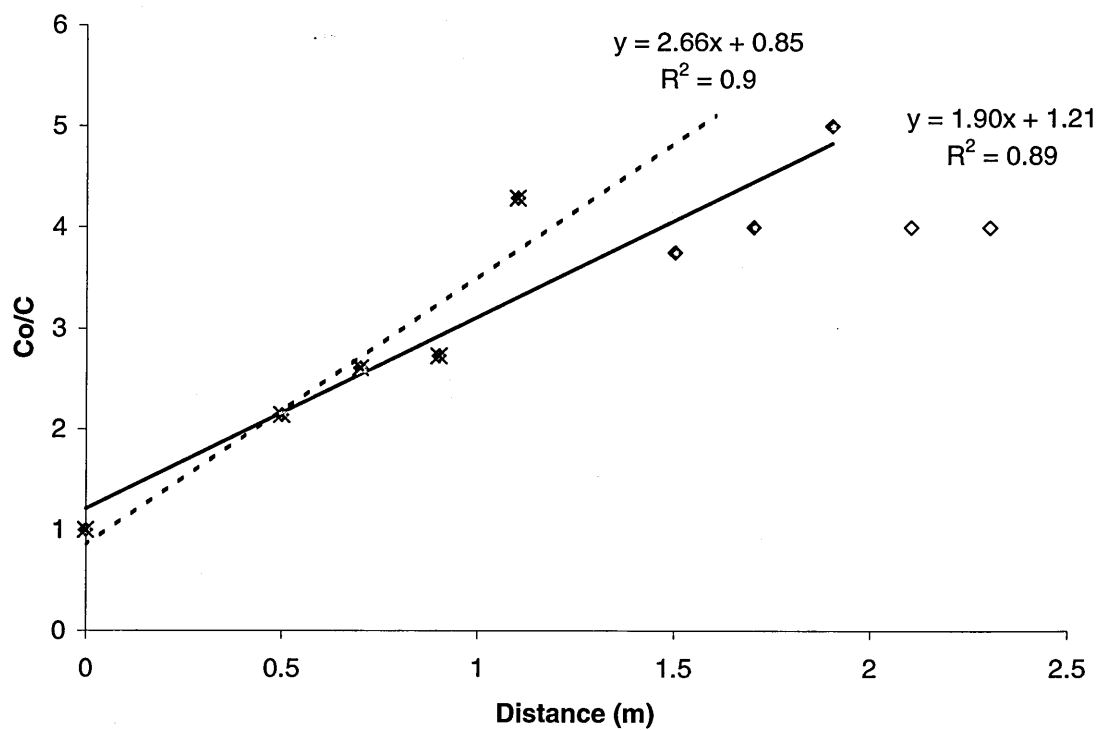


Figure 11.7: An example of the typical pattern of C_0/C against distance (same data as Figure 11.4)

11.5 DISCUSSION

Using the overall results and, assuming that any differences in the performance of the smaller StarLight C during the two operational periods were due to the prevention of media washout, it is possible to rank the performances of the three BAF media. In terms of substrate removal, the expanded clay and the smaller StarLight C are best followed by the larger StarLight C.

Mann & Stephenson's model states that $\ln (Qm/A)$ rises as the substrate influent concentration increases. The relationship did not hold true for the media used in this study and the values of $\ln (Qm/A)$ were generally higher than those previously reported. The function $\ln (Qm/A)$ increases with the value of the gradient (m) and the flow rate (Q). The flow rates used in this study ranged between 0.3 and 0.5 l/min, whilst in the previous studies the range was between 0.2 and 0.5 l/min in identical reactors (Mann & Stephenson, 1997). The higher values of m reported in this study indicate that the BAFs had a better substrate removal rate than those used previously. When all the data points were used to calculate m , the media could be ranked as; smaller StarLight C, expanded clay and then larger StarLight C, with all three being superior to the sand and plastic media used by previous researchers.

If the substrate removal in a BAF can be described by simple first-order kinetics, the expression $\ln (Qm/A)$ should be consistent. When only the first four or five points were utilised to calculate m , the R^2 values for the data and the first-order regression line increased. Additionally, the standard deviations between the values of $\ln (Qm/A)$ at a particular flow rate decreased. This indicated that first-order kinetics could be used to describe sCOD removal in these reactors, and using this methodology the mean values of m increased for all three media. However, the rise was slightly less for the BAFs in which substrate removal occurred deeper into the reactor. Thus, after this analysis the media would be ranked; Expanded clay, smaller StarLight C and then larger StarLight C. This illustrates that the inaccuracies from using all the data points may have led to the incorrect conclusion, that the rate of removal improved using smaller StarLight C rather than the expanded clay.

For all the media, the first-order constant increased as the influent flow rate increased. This is because increasing the flow rate will increase the liquid velocity, which improves the transport of substrate to the biofilm. However, if the velocity is too high, the increased shear stress will cause biofilm sloughing (Brading *et al.*, 1995). Previous work found that the flow rate did not influence the first-order constant in BAFs (Mann & Stephenson, 1997, Mendoza-Espinosa & Stephenson, 1998) and future research should examine this relationship.

The concept of a BAF being composed of two parts, where the first-order substrate removal region is followed by a volume where no further removal occurs, may not be relevant for BAFs which use sub-optimal media or low reactor heights. For instance, using a sub-optimal plastic medium Mann & Stephenson (1997) found a first-order rate of removal throughout the bed depth. However, it is possible that the reported decreases in the first-order constants as the influent substrate concentration was lowered, may have been due to a region of no, or lower, removal existing at the end of the reactor. Indeed, the graph of $\ln(C/C_0)$ versus height given in Mendoza-Espinosa & Stephenson (1998) appears to be composed of two separate regions, with the gradient decreasing significantly in the final part of the BAF. Further research is required to gain a better understanding of the region following first-order substrate removal and when the transition occurs for different media. An ideal medium would combine a high rate of removal with a high depth of first-order characteristics. The expanded clay and StarLight C medium will usually show similar removal performances because although the removal rate is slightly lowered using StarLight C, the removal continues deeper into the bed.

The small variations in the first-order removal constant for each medium may have been due to differing sewage degradabilities. To test this theory, it would be useful if a method for measuring biodegradability could be applied to sewage. The preliminary work in this study gave no conclusive results. The concentration of non-degradable sCOD in sewage appears to be more variable than the concentration of non-degradable sBOD. Therefore, it may be considered sensible to model the degradation of sBOD instead of sCOD in future work. However, the errors for sBOD data points were up to ± 20 mg/l and decreased down the column as different dilutions of the

effluents were used in the analysis. This may explain why the standard deviations for the expressions of m in the larger StarLight C BAF, were higher when the substrate measured was sBOD rather than sCOD.

Rather than trying to measure the changes in the degradability of sewage as it is treated, the effects can be modelled with a retardant model. Using a simple linear retardant model lead to the inclusion of a greater number of data points from the substrate removal profiles. However, the exact retardant expression may well depend on the specific components of the wastewater. It may be sufficient, for the purposes of comparing BAF media and for reactor design, to know the first-order constant obtained using the medium and the depth over which first-order kinetics dominate. It is note-worthy that the simplest empirical models have also been the most effective for the scale-up of substrate removal data in BAFs (Mendoza-Espinosa, 1999).

11.6 CONCLUSIONS

1. Data obtained for three media, at three influent flow rates, did not fit the model proposed by Mann & Stephenson (1997). The first-order constant did not decrease as the influent substrate concentration decreased.
2. Substrate removal along downflow BAFs appeared to follow first-order kinetics in the top section. The substrate concentrations in the lower section showed no obvious pattern and requires further investigation.
3. A simple, linear, retardant model allowed a greater proportion of the column to be modelled, but gave a less accurate portrayal of reactor performance, than the first-order model.
4. All three materials tested showed better substrate removal characteristics than the sand and plastic media used by previous workers.

CHAPTER 12

FINAL DISCUSSION, CONCLUSIONS & RECOMMENDATIONS FOR FUTURE WORK

12.1 FINAL DISCUSSION

Biological aerated filters (BAFs) have proven to be efficient, small footprint reactors capable of either complete secondary or tertiary wastewater treatment (Goncalves *et al.*, 1994). However, their application has been restricted by their comparatively high operating costs compared to traditional processes such as activated sludge reactors and trickling filters (Dillon & Thomas, 1989).

The BAF medium influences process performance since substrate removal is determined by biofilm retention, which may be affected by the surface's roughness (Baker, 1984), chemistry (Burchard *et al.*, 1990) and pore sizes (Messing *et al.*, 1979). Additionally, the BAF medium can influence the operating costs of the process in a number of ways. Using a medium with a low minimum fluidisation velocity could decrease the required volume of backwash water. Optimising particle shape, size and voidage could extend filter run times and reduce backwashing frequency. Additionally, increasing a medium's resistance to attrition would negate media replacement requirements (Chapter 2).

Previously the BEWA (1993) standards for filter media have been applied to assess the suitability of materials as BAF media (Kent *et al.*, 1996). Comparing the results from these tests to the performance of two potential BAF media in pilot-scale reactors, indicated that some of the BEWA standards should be modified when applied to BAF media. For instance, the foamed clay medium (StarLight C) exhibited 3% weight loss after attrition testing and so its suitability as a filter medium was doubtful. Meanwhile, the expanded clay (medium A) showed a weight loss of over 5% after the attrition test, which made it unsatisfactory as a filter medium (Ives, 1990). However, there were no increases in the mass of non-volatile solids in the

wastewater after passing through the media columns, and the hydraulic size of the particles was maintained. Therefore, it was suggested that the biofilm surrounding the medium may have protected it from attrition. This would allow the maximum weight loss acceptable after the attrition test to be slightly increased (Chapter 6).

The BEWA standards were considered appropriate for assessing media inertness, size, voidage and bulk density. However, to examine the potential of a material as a BAF medium prior to pilot-scale testing, a number of other parameters should also be measured. There is currently no standard method of measuring the surface roughness of BAF media. Comparing the performance of materials with a similar shape and size, but with a different degree of surface roughness under identical conditions at pilot-scale have indicated that surface roughness may influence reactor performance. Mercury porosimetry can be used to measure the internal porosity of a porous granular medium and this can be used to calculate the maximum soaked density of the particles. Additionally, the experimental minimum fluidisation velocity should be measured since many of the assumptions made in the Carmen-Kozeny (Coulson & Richardson, 1978) and Wen & Yu (1966) theoretical equations, do not hold true for BAF media (Chapter 6).

Previously, tracer studies have been conducted on empty BAF beds to examine the hydraulic characteristics of the reactors (Mann *et al.*, 1995). Comparing the results of tracer studies conducted prior to, and during operation, indicated that empty bed tracer studies provide a good indication of the flow regime in an operating BAF (Chapter 6). The flow characteristics of each media bed could be related, at least to some degree, to the reactor performance. For instance, tracer studies revealed that there was a degree of channelling through the bed of the larger variant of StarLight C. The operating results for that column indicated that the effluent from this BAF could deteriorate immediately after the backwashing procedure. However, generally the detrimental consequences of channelling through the reactor were obscured during operation, probably because the bed compacts and its porosity decreases as solids are captured and the biofilm grows (Chapter 10). Meanwhile, the ammonia removal results improved as the mixing decreased and the flow regime within the reactors came closer to plug flow. Plug flow conditions allow the concentrations of substrates to alter through the column. Nitrification is promoted because, in regions with lower

concentrations of organic material, there is reduced competition between the autotrophic nitrifying bacteria and the heterotrophic bacteria.

After qualifying as a potential BAF medium (Chapter 4), StarLight C was tested and compared at pilot-scale to a frequently used expanded clay BAF medium (Chapter 5). By applying the nominal flow rate immediately, start-up was rapid and stable sCOD and suspended solids removal was reached by day four. At loading rates over 10 kg tCOD/ m³ active volume/ d, both reactors achieved a mean sCOD removal of approximately 75% and a mean suspended solids removal of approximately 87%. These were superior to the results obtained using plastic and sand media under identical conditions (Mann *et al.*, 1999, Mendoza-Espinosa & Stephenson, 1998).

Both media were backwashed daily for ten minutes with a combined air scour and a liquid flow at 40% of the minimum fluidisation velocity of the medium. These conditions have been determined to provide optimal backwashing (Amirtharajah, 1993). There were no attempts to optimise the backwash procedure by reducing wash times to a minimum, and so it is probable that the required backwash water for both media could be reduced. However, StarLight C showed the potential of utilising half the volume of backwash water required by medium A and so increased productivity by 19, 14 and 11 % at influent flow rates of 0.3, 0.4 and 0.5 l/min respectively (Chapter 5).

Since StarLight C exhibited similar substrate removal characteristics to one of the best available types of BAF media, and potentially offered the benefit of lower backwash water requirements, it was considered worthy of further development. To ensure that development proceeded in appropriate areas, representatives of the wastewater treatment companies were asked to list their requirements in a novel BAF medium (Chapter 7). In the order of the parameters most frequently cited these were; better biomass retention; lower attrition rates; easier separation of particles from backwash effluent; reduced backwash velocity; lower cost; consistent substrate removal during the filter cycle; and improved solids removal. Comparing its performance to that of alternative BAF media, Starlight C has above average properties for all these parameters except for its attrition rate and ease of separation from the backwash liquid.

Increasing the concentration of the bentonite solution used in the feed clay from 35 g/l to 56 g/l produce a foamed clay variant which exhibited increased resistance to crushing and correspondingly showed lower levels of attrition. Altering the proportions of the other constituents of foamed clay increased the observed media friability. XRD analysis indicated that 10% of the crystalline structure of the medium was composed of cristobalite when the maximum firing temperature was 1300°C. This led to an undesirable reduction in crushing resistance compared to media samples fired to maximum temperatures of 1400 and 1500°C (Chapter 8). Therefore, the firing regime must ensure that all the media reaches a temperature of 1400°C.

For seven variants of foamed clay, higher crush resistance appeared to be correlated with a decrease in surface roughness. Under limited conditions designed to represent those in a BAF, there was no evidence that altering the surface roughness of foamed clay effected its biofilm retention properties. It was proposed that surface roughness provides anchoring sites for the biofilm, and that there was an excess of sites on all the variants of the foamed clay. It was therefore suggested that the high bentonite variant would exhibit the best overall performance as a BAF medium in terms of performance and operating costs (Chapter 8).

Increasing the size of BAF media particles would make them easier to separate from the backwash liquid, without risking grid blockages. Previous work also indicates that filter run times improve with increases in media size (Stensel *et al.*, 1988). However, for a given reactor to produce a particular effluent quality, media size is thought to dictate the maximum loading rate, since smaller particles provide a greater specific surface area (Smith *et al.*, 1999). In a second pilot-scale experiment the performance of the original StarLight C with particle diameters between 1.5 and 3.5 mm was compared to the performance a larger StarLight C with particle diameters between 2.5 and 4.5 mm (Chapter 10). The results concurred with those reported previously and indicated that the BAF containing the smaller medium was under-performing as a filter. Head loss during the filter cycle was more likely to be show logarithmic rather than linear development when using the smaller medium and suspended solids removal did not occur throughout the filter depth.

The environmental scanning electron microscope (ESEM) allows biofilms to be viewed in their fully hydrated state. The biofilm grown on foamed clay BAF media looked similar to those from previous ESEM studies, with cells completely covered in extracellular polymeric substances (Sutton *et al.*, 1994, Little *et al.*, 1991). The observed growth stages indicated that the biofilm does not form a uniform monolayer across the medium and this should be considered when developing biofilm mechanistic models in the future. The reduction in the viability of cells exposed to the conditions in the ESEM were found to depend on the accelerating voltage of the electron beam, the observation time and the age of the biofilm. Using a specially designed Peltier stand, exact areas of biofilm growing on foamed clay were relocated. When biofilms over five days old were examined for under 15 minutes, there were some signs of growth after 24 hours. If optimal conditions are found to reduce changes in viability even further, observing the growth and development of biofilms on BAF media may be feasible using the ESEM (Chapter 9).

Empirical models are useful aids in the selection and design of wastewater treatment processes. Recently Mann & Stephenson (1997) presented an empirical model designed specifically for BAFs, and one of their constants was proposed to be directly related to the BAF medium. The calculation of a media constant using either the Mann & Stephenson (1997) model or another empirical model would provide a rational and convenient method to compare the performance of different materials as BAF media. Substrate removal profiles were analysed for BAFs containing medium A and the two size ranges of StarLight C. The data did not fit the Mann & Stephenson (1997) model since the first-order constant did not decrease as the influent substrate concentration decreased.

The top section of the reactor appeared to follow first-order kinetics and there were only small deviations in the first-order constants relating to a particular medium and flow rate. The first-order constant and the length of the column that could be described by first-order kinetics depended on the medium (Chapter 11). By providing constant reactor conditions and sewage biodegradability, media could be characterised by the first-order constant for the upper section of a downflow BAF. The mean first-order constants were 72 for the expanded clay, 55 for the small StarLight C and 38 for the large StarLight C. All three materials tested had higher first-order constants than

the sand and plastic media used by previous workers, and showed better substrate removal characteristics.

In terms of the development of StarLight C as a BAF medium, the next stage may be to test the medium in a full-scale reactor. If possible this should be arranged with South West Water. The washout and attrition properties of the media may also be improved by further optimisation of the bentonite content. Additionally, it may be worthwhile developing a dual media BAF and this could be done in conjunction with one of the current BAF suppliers.

12.2 CONCLUSIONS

1. Many of the BEWA standards (1993) for water filter media are suitable for the assessment of BAFs. However, materials showing weight losses of up to 5% after attrition testing should not be disqualified as potential BAF media. Biofilm growth may reduce the exhibited attrition levels during operation.
2. Tracer studies on media beds prior to operation can provide useful insights into the future performance of BAFs. As channelling increases, it is more likely that substrate quality will deteriorate particularly immediately after backwashing. As conditions become closer to plug flow the nitrification characteristics should improve but head loss may develop more rapidly.
3. The sample of representatives from the wastewater treatment industry identified the BAF's operating costs as the biggest disadvantage of the process. At least half of the sample desired a BAF medium that exhibited superior biological performance, lower attrition, lower washout and reduced backwash velocity.
4. Rapid start-up of COD and suspended solids removal can be achieved with BAFs containing expanded or foamed clay media by using nominal flow rates, two days of re-circulation and reduced backwash frequency for a further four days.
5. A foamed clay (StarLight C) can show equivalent performance to a standard expanded clay (medium A) when used as a BAF medium. At pilot-scale, downflow BAFs containing these materials achieved a mean of 87% suspended solids removal and 75% soluble COD removal at loading rates $> 10 \text{ kg COD/ m}^3 \text{ working volume/ d}$.
6. Pilot-scale reactors containing StarLight C and medium A were backwashed daily for ten minutes at a liquid flow rate corresponding to 40% of the minimum fluidisation velocity (V_{mf}) of the media. The approximate experimental V_{mf} of StarLight C was 120 m/h compared to 215 m/h for medium A. Therefore, the BAF

using StarLight C exhibited between 11 and 19 % increased productivity at liquid velocities from 1.4 to 2.4 m/h.

7. Altering the degree of roughness exhibited by foamed clay did not effect its biological properties in small-scale reactors, under conditions representing those in a BAF. It was proposed that the foamed clay surface has an excess number of glycocalyx binding sites.
8. When StarLight C was produced using 56 g/l, rather than 35 g/l, bentonite solution the medium had lower surface roughness and higher crush resistance. These properties translated to a reduction from 3 % to 1 % weight loss on attrition testing, making the variant more suitable as a BAF medium.
9. Suspended solids removal in a BAF shows a good correlation to first-order degradation kinetics. The head loss above the filter bed can either rise linearly with time so the BAF acts as an efficient depth filter, or it can rise logarithmically indicating the reactor is an inefficient depth filter.
10. A more consistent effluent was obtained using the smaller StarLight C medium (1.5-3.5 mm diameter) compared to the larger medium (2.5-4.5 mm diameter). At flow rates > 0.4 l/min, the BAF using the larger medium produced an effluent containing > 20 mg/l of suspended solids for over 30 minutes after backwashing. However, up to 70% longer run times, as determined by reaching a set head loss, were recorded for the BAF containing the larger rather than the smaller medium. The development of pressure above the smaller medium filter bed tended to be logarithmic rather than linear, and a mean depth of 1.7 m was involved in suspended solids removal compared to 2.1 m by the larger medium BAF. This indicated that the smaller medium BAF was under-performing as a filter.
11. Substrate removal in downflow BAFs appeared to follow first-order kinetics in the upper section with little or no removal occurring in the lower section. The first-order constants were dependent on the liquid velocity and at 1.4 m/h were 61 d^{-1} for medium A, 45 d^{-1} for small StarLight C and 33 d^{-1} for large StarLight C.

12.3 RECOMMENDATIONS FOR FUTURE WORK

1. In addition to the BEWA standard tests, the surface roughness, porosity, sunken density and experimental minimum fluidisation velocity, of potential BAF media should be measured. A standard method to assess surface roughness needs to be developed.
2. In this study, attempts have been made to reduce the operating costs of the backwash procedure by lengthening the run times and decreasing the minimum fluidisation velocity of the media. In future work, the backwash procedure itself could be optimised by examining the performance of different combinations of liquid and air scours. Additionally, attempts could be made to reduce the aeration costs of the process by, for example, recycling part of the treated, oxygen-rich effluent.
3. It may be possible to develop a BAF medium with a surface chemistry that would particularly encourage the adhesion of nitrifying bacteria. This could decrease start-up times for nitrification in BAFs and increase the process stability. The silane coated StarLight C used in Chapter 8 could be adapted so that various functional groups were exposed and the medium exhibited differing hydrophobicities and surface charges. An apparatus based on that used in Chapter 8 could be inoculated with *Nitrosomonas* and *Nitrobacter* and supplied with an ammonia based synthetic feed (Hanaki *et al.*, 1990).
4. The ESEM has benefits over other electron microscopes since fully hydrated biofilms can be viewed. Therefore, the equipment has the potential to elicit new information about biofilm structure and development. However, further work is needed to optimise the operating conditions required to view biofilm growth on BAF media with minimal disturbance.
5. To verify the conclusions made after examining various empirical models for BAFs, substrate removal profiles from reactors containing different media need to be obtained. After confirming that the columns can be divided into two regions

with different substrate removal characteristics, it would be useful to have an increased understanding of the lower section of a downflow BAF. Changing the biodegradability or concentration of the wastewater should alter the depth of the region exhibiting first-order substrate removal. The lower section may act as a buffer zone that ensures low effluent substrate concentrations as the influent concentration varies.

6. For all the media, the first-order constant increased as the influent flow rate increased. This is because increasing the flow rate increased the liquid velocity, which would have improved the transport of substrate to the biofilm (Henze *et al.*, 1995). However, if the velocity is too high, the increased shear stress will cause biofilm sloughing (Brading *et al.*, 1995). Previous work found that the flow rate did not influence the first-order constant in BAFs (Mann & Stephenson, 1997, Mendoza-Espinosa & Stephenson, 1998) and future research should examine this relationship.
7. Further research is also required to optimise the heights of BAFs using different sized media. The height of the BAF containing the larger StarLight C should be increased until the reactor's performance is equivalent to the smaller medium. The difference in the maximum run times of the two reactors would then indicate how inefficient filtration effects reactor productivity.
8. A dual media BAF could be designed to combine the advantages of two sizes of StarLight C. In a downflow BAF, the larger StarLight C could form a top layer above a lower section containing the smaller sized variant. The top layer would reduce the suspended solids load on the lower section, which should extend the maximum run time of the filter. The lower section should ensure effluent consistency even immediately after backwashing. To maintain the division between the sections, the larger StarLight C could be produced with a lower density than the smaller version.

CHAPTER 13

PROJECT MANAGEMENT

13.1 PROJECT MANAGEMENT COMMITTEE

To ensure the project proceeded to meet the required cost, quality and time objectives the project management committee held regular steering meetings (Table 13.1). Nick Wilshaw represented English China Clays International (ECCI) throughout the four years. He established the requirements of ECCI and then ensured the project received company support. After the first year, academic supervision was passed from Prof. Tom Stephenson to Dr. Jo Quarmby. However, Prof. Tom Stephenson remained the chairman of the project management committee and continued to be involved with the project. Management supervision was passed from Dr. John Fielding to Prof. Paul Millier in August 1998.

Table 13.1: Records of the project's progress meetings

Meeting	Date	Present
1	20/10/95	R. Moore, T. Stephenson, N. Wilshaw
2	28/02/96	R. Moore, T. Stephenson, N. Wilshaw, S. Judd, D. Skuse, T. Jones
3	30/09/96	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby
4	21/02/97	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby, J. Fielding
5	24/04/97	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby
6	05/06/97	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby, J. Fielding
7	05/09/97	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby
8	12/02/98	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby, J. Fielding
9	15/05/98	R. Moore, N. Wilshaw J. Quarmby
10	19/10/98	R. Moore, N. Wilshaw, J. Quarmby, P. Millier
11	04/02/99	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby, P. Millier
12	20/07/99	R. Moore, T. Stephenson, N. Wilshaw, J. Quarmby, P. Millier

13.2 TIME AND QUALITY MANAGEMENT

At each committee meeting a full progress report was presented and the future direction of the project was discussed. The Gantt chart in Figure 13.1 summarises the project's activities over four years. The goals for the first year were met, but the project may have gained from better project planning by the management committee at the beginning of the second year. During December 1996 small-scale reactors were designed for a planned six months placement at ECCI's base in St Austell. However, early in 1997 it was decided that this phase of experimental work should be suspended for a year in favour of pilot-scale tests at Cranfield.

The delay was partly recovered by using the experience within the Water Sciences department to produce detailed plans of the requirements, costs and time-scales involved in building the pilot plant. This allowed the plant to become operational within budget just four weeks after the costs had been approved.

The experimental work carried out at St. Austell in 1998 was difficult to plan since there was little preceding work available in the literature and methods needed to be developed. Therefore, a detailed initial three-week schedule and several alternative plans were developed. This proved worthwhile when the inoculated nitrifying bacteria grew so slowly that significant amounts of ammonia were not being removed from the small-scale reactors after twenty days of operation. Immediately plans were switched to the alternative using heterotrophic bacteria and filling the bottom part of the reactor with media. There were therefore no unexpected delays and the planned work was completed within six months.

After nine months operational experience, the second experimental period using the pilot plant was relatively easy to plan. The weekly analysis was planned so that a placement student was responsible for maintaining the rig for two days a week. This allowed progress on the thesis write-up to be continued. However, the experiment was delayed for two weeks, due to problems with the supply of the media from ECCI. The project management improved throughout the project as more realistic time and quality objectives were set with increased levels of planning.

13.3 PROJECT COSTS

The budget was calculated after deducting the known fixed costs from the project input. Of these known costs the intra-school levies were rounded up to approximate figures. As shown in Table 13.2 the outgoings for the project were within the approximated budget by £382. However, the intra-school levies were £807.25 more than predicted and so the real final balance was -£425.52. This is reasonable considering the circumstances and it is recommended that more communication about the likely fixed costs are required to allow projects to remain within budget.

Table 13.2: Project costs

PROJECT INPUT (£):			
Pilot plant	6,500		
Small scale reactors	1,180		
Working budget	26,000		
FIXED COSTS (£):			
Grant	-8,800		
Intra-school levies (approx.)	-6,000	Intra-school levies (actual)	-6,807.25
Residual budget (approx.):	11,200	Residual budget (actual):	10,392.75
OUT GOINGS (£):			
Technical supplies	-9,528.72		
Internal supplies	-657.00		
Expenses (travel, entertainment)	-632.55		
TOTAL (£):	-10,818.27		
BALANCE (£):			
Based on approximate budget	382	Based on actual budget	-425.52

ID	Name	4Q95		1Q96		2Q96		3Q96		4Q96		1Q97		2Q97		3Q97		4Q97		1Q98		2Q98		3Q98		4Q98		1Q99		2Q99		3Q99		4Q99	
		O	N	D	J	F	M	A	M	J	A	S	O	N	D	J	F	M	A	M	J	A	S	O	N	D	J	F	M	A	M	J	A	S	O
1	MBA courses																																		
2	Literature review																																		
3	Management component																																		
4	Eng.D. core course 1 & 2																																		
5	Eng.D. core course 3																																		
6	Eng.D. core course 4																																		
7	ECCI																																		
8	MSC 1 & 2																																		
9	MSC 3																																		
10	Experimental program																																		
11	Physical tests 1																																		
12	Small scale run 1																																		
13	Small scale run 2																																		
14	Pilot run 1																																		
15	ECCI																																		
16	Physical tests 2																																		
17	Pilot run 2																																		
18	Write up																																		
19	JIWEM paper																																		
20	ICHEME paper																																		
21	IAWQ paper																																		
22	BAF3 paper																																		
23	Bio.Lett. paper																																		
24	Wat. Res. paper																																		
25	Modelling paper																																		
26	Finish thesis																																		
27	Hand in																																		

CHAPTER 14

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APPENDIX 4A:

DEFINITIONS OF DIFFERENT EXPRESSIONS OF DENSITY

Bulk (or packing) density (kgm^{-3}) = weight of medium (kg) / total volume of medium inclusive of voids (m^3)

Material density (kgm^{-3}) = weight of medium (kg) / volume of soaked particles (m^3)

Particle density (kgm^{-3}) = weight of medium (kg) / volume of particles (m^3)

Soaked density (kgm^{-3}) = weight of soaked medium (kg) / volume of particles (m^3)

APPENDIX 6A:

EXAMPLE CALCULATION OF N (NUMBER OF TANKS IN SERIES)

	A	B	C	D	E
1	Example calculation (Levenspiel, 1962) for medium A				
2	Conditions: Before operation with liquid flow @ 0.4 l/min and air flow @ 2 l/min				
3					
4	Time since input (T)	Conductance (mS)	Corrected conductance (C)	T*C	(T*T)*C
5	(minutes)		(conductance-0.55)		
6	1	0.55	0	0	0
7	2	0.55	0	0	0
8	Readings every minute until conductance returned to original level				
9					
10	305	0.55	0	0	0
11					
12		Column totals:	100.01	11914.29	1761853.13
13					
14					
15	Mean HRT =	Sum(T*C* time interval) / Sum(C*time interval)			
16	Time interval = 1, so calculate D12/C12 =	119.13			
17					
18	Variance =	(Sum((T*T)*C*time interval) / Sum(C*time interval)) - (Mean HRT*Mean HRT)			
19	so calculate (E12 / C12) - (C16*C16) =	3424.58			
20					
21	Dimensionless variance = Variance / (mean HRT*mean HRT)				
22	so calculate C19 / (C16*C16) =	0.24			
23					
24	N = 1 / dimensionless variance				
25	so calculate 1 / C22=	4.14			

APPENDIX 7A:

INTERVIEW WITH REPRESENTATIVES FROM THE WWT COMPANIES

Background

I am doing research in the School of Water Sciences at Cranfield University, and have produced this questionnaire as the management component of my research. My practical work is with a secondary wastewater treatment process and through this questionnaire I aim to:-

- Explore attitudes within the water industry towards the various secondary treatment processes.
- Determine who is involved in choosing the process type.
- Find out from the influential people what characteristics of secondary treatment processes are important, so that future research will be targeted to the industry's needs.

Thankyou very much for your participation in this interview.

If you have any queries please contact Rebecca Moore at:

School Of Water Sciences,
Cranfield University,
Cranfield,
Bedfordshire,
MK43 0AL

Name:

Company:

Job Title:

Background:

1. Please rate your knowledge of the following secondary treatment processes. (Tick appropriate box with; 1 = no knowledge and 5 =high knowledge). Feel free to add any other secondary treatment processes you think are appropriate.

PROCESS	1	2	3	4	5
Activated sludge					
Anaerobic processes					
Biological aerated filters (BAFs)					
Extensive processes (e.g. Reed beds)					
Fluidised beds					
Rotating contact reactors (RBCs)					
Trickling filters					

Other comments:

Name:
Company:

2. Please rate the importance of the listed characteristics (under general circumstances) between 1 and 5. (1= very unimportant & 5= very important & DN if you don't know). Spaces are available for adding any other characteristics of secondary wastewater treatment you feel are particularly important.

FEATURE	RATING
Suspended solids removal	
BOD removal	
Ammonia removal	
Process stability (e.g. to toxicity)	
Process flexibility (to input changes)	
Land usage (for a given organic loading)	
Aesthetics	
Capital costs	
Operating costs	
Operability	

Other comments:

Name:
Company:

3. Seven of the most important features of secondary treatment processes are listed in **row** 1 of the following table.

Rate the secondary treatment processes listed in **column** 1 of the table between 1 and 5 in respect to each of these characteristics. (1 = very poor & 5 =very good & DN if you don't know).

Consider the usual performance of the standard process configuration designed for a given population. Again spaces are available if you think any other features or processes are particularly relevant.

<div> <div>FEATURE</div> <div>PROCESS</div> </div>	S.S. rem.	BOD rem.	NH ₃ rem.	Process stability (e.g. to toxicity)	Process flexibility (to input changes)	Land usage (for a given load)	Aesthetics	Capital costs	Op. costs	Operability	
Activated Sludge											
Anaerobic processes											
BAFs											
Extensive processes											
Fluidised beds											
RBCs											
Trickling filters											

Other Comments:

Name:

Company:

4. Do different sewage treatment sites have different requirements/ constraints on their secondary treatment processes?

5. What are the main types of site and what requirements/ constraints make them unique?

6. Can you give examples of these types from your experience?

7. How big a proportion does each type represent?

Name:

Company:

8. Are there different requirements/ constraints for secondary wastewater treatment processes in...

- i) Urban and rural areas? What?
- ii) Do land restrictions tend to be in urban sites?
- iii) Do toxicity problems tend to occur in urban sites?
- iv) Are the requirements of secondary treatment proceeding, tertiary treatment different to other secondary treatment? How?
- v) Is different process selection shown when upgrading existing plants rather than putting in new plants?

9. Do these requirements/ constraints appear to influence process selection?
Which process with which conditions?

Name:

Company:

10. What are the possible challenges that secondary wastewater treatment processes are likely to face in the near future (within the next 5 years)?
11. Which secondary treatment processes have the necessary qualities to face these challenges?
12. How frequently are new secondary treatment processes introduced in your company?
(Every 6 months, between 1-2 years, 2-3 years, 3-4 years, 4-5 years, 5+years, don't know)
13. For what reasons are new secondary treatment processes introduced?
(What are the benefits most frequently sought e.g. better output standards, reduced land usage, to combat specific problems etc.)
14. Which secondary treatment processes are commonly used within your company?

Name:

Company:

15. For what reasons would a different **type** of secondary treatment processes be introduced? (Either for expansion or replacement)

16. Do you prefer to use traditional processes and if so what are the major risks associated with changing the type of secondary treatment process?

17. When are they considered inhibitory?

18. Who makes the decision that a new secondary treatment process is required?

19. Do they also set guidelines for the costs and the requirements of the new process? (or are there company wide standards?)

20. At what point is the decision of **process type** made? (Before or after quotes are obtained?)

21. Who is involved in determining the type of process to be used?

- Who influences the decision?

- Who are the decision-makers?

Name:

Company:

22. How long does the decision process take from the decision that a new secondary treatment process is required to the selection of the process and plant?

(Less than 6 months, 6-12 months, 1-2 years, over 2 years, don't know)

23. Is company image an influencing factor in process selection- are there show-pieces?

24. What other factors can influence process selection?

25. Which other wastewater treatment companies are considered to be the technology leaders?

26. Who are your key partners and suppliers of wastewater treatment systems?

27. How much influence do consulting companies have in your **selection** of secondary wastewater treatment processes? Delete as necessary.

HIGH / MEDIUM / LOW

28. How much influence do consulting companies have in the **design** of wastewater treatment processes? Delete as necessary.

HIGH / MEDIUM / LOW

Name:

Company:

29. Estimate the % of secondary wastewater treatment plants that will need to expand to deal with the EC urban wastewater treatment directive.

OR Estimate the amount of expansion in terms of;

HIGH / MEDIUM / LOW / DON'T KNOW

OR Whom could I contact to obtain figures?

30. Of sites requiring expansion estimate the % where land usage would have a big influence on process choice.(i.e. land restrictions).

OR Rate the influence of land usage on process choice;

HIGH / MEDIUM / LOW / DON'T KNOW

OR Whom could I contact to obtain figures?

31. What functions do your tertiary treatment processes fulfil?

32. What processes do you use for tertiary treatment?

33. What are the strengths and weaknesses of these tertiary treatment processes?

Name:
Company:

34. How many BAF plants are used within your company, what are their size and how have they performed overall (rate between 1 and 5 (1=very poor & 5=very good))?

Capacity	Number of BAFs	Performance

35. What are the greatest benefits of BAF systems to you? Rank (1=greatest benefit)

36. What are the weaker characteristics of BAF systems to you? Rank (1=weakest characteristic)

37. How much will BAFs be used in the future?
 (20+% less than now / 10-20% less than now / 0-10% less than now / the same as now / 0-10% more than now / 10-20% more than now / 20+% more than now)

Name:

Company:

38. When will BAFs be used? (Under what circumstances)

39. List the names of the companies involved in producing BAF plants, and their advantages and disadvantages.

Company	Advantages	Disadvantages

40. What advantages would you require in a new BAF media to induce switching of media type?

APPENDIX 7B:

**INTERVIEW WITH REPRESENTATIVE FROM THE
ENVIRONMENT AGENCY**

Job Title:

Background:

1. Do you have standard effluent consent levels for sewage treatment works or is each consent set individually?

What are the standards...?

2. How do you decide on and set the consent levels?
3. How do consent levels relate to the EC Urban Wastewater Treatment directives minimum standards of treatment (HNDA, normal, sensitive)?
4. How many sites require improvements to their treatment to meet consents?

5. What are the time-scales involved?

6. What are the consequences of failure to meet consents?

7. Do different sewage treatment sites have different requirements/ constraints on their secondary treatment processes?

Does the Environment Agency have categories?

8. What are the main types of site and what requirements/ constraints make them unique?

9. Can you give examples of these types from your experience?

10. Are there different requirements/ constraints for secondary wastewater treatment processes in.....

- i) Urban and rural areas? What?
- ii) Do land restrictions tend to be in urban sites?
- iii) Do toxicity problems tend to occur in urban sites?
- iv) Are the requirements of secondary treatment, proceeding tertiary treatment different to other secondary treatment? How?
- v) Is different process selection behaviour shown when upgrading existing plants rather than putting in new plants?

11. Do these requirements/ constraints appear to influence process selection? Which processes with which conditions?

12. What other factors influence wastewater treatment companies in their process selection?

13. Can the Environment Agency influence process selection? What is the nature of the interaction?

14. Do wastewater treatment companies prefer to use traditional processes?

15. Who do you consider to be the technology leaders?

16. How much influence do contractors have on process selection and design?

17. Is company image an influencing factor in process selection- are there show-pieces?

18. What do you consider to be the future influences (maybe linked to future regulations)?

19. How does the state of the industry in the UK compare to that in Europe?

Which are the leading countries?

20. When are BAFs the preferred treatment choice?

APPENDIX 7C:

INTERVIEW WITH REPRESENTATIVE FROM ECCI

Please rate ECCI's company strengths between 0 and 4 for the following questions where:

- | | | |
|--------------|----|-----------------|
| 0 represents | -- | (very bad) |
| 1 represents | — | (below mean) |
| 2 represents | = | (average) |
| 3 represents | + | (above average) |
| 4 represents | ++ | (very good) |

Q1. Experience and adaptability of the sales force

Q2. The company adaptability

Q3. Wealth of the project

Q4. Degree of synergy with other projects

Q5. Business activities integrated downstream

Q6. Service

Q7. Company image

Q8. Conformity with laws or regulations

Q9. Coherence between your strategy & the target market

Q10. The company's proficiency levels when using the technology

Q11. Coherence between project and companies technological strategy

Q12. Suppliers capacity to keep up with your growth

Q13. Risk of take-over

APPENDIX 7D:

SEGMENTATION OF THE DIFFERENT TYPES OF WWT WORKS

I have now interviewed representatives from all of the wastewater treatment companies and I am analysing the results. I would be very grateful if you could look at this proposed segmentation of the different types of sewage treatment works with secondary treatment. I have asked questions in CAPITALS.....

Proposed segmentation:

Urban				Rural			
Special area		Normal urban area		Special area		Normal rural area	
Site restricted	Site not restricted	Site restricted	Site not restricted	Site restricted	Site not restricted	Site restricted	Site not restricted
A	B	C	D	E	F	G	H

Description of the different segments:

1. Urban or rural.....

Urban sewage treatment works tend to be larger than rural sewage treatment works. This effects process choice since certain processes are more suited to being used on a large scale than others and smaller sewage treatment works need to be simple and easy to operate without frequent supervision.

Q1: DO YOU AGREE WITH THIS?

OTHER COMMENTS.....

2. Special or normal....

At some sewage treatment sites the processes need to be particularly robust and have low sensitivity to toxicity or changing volumes. These sites may have a significant proportion of industrial effluent or suffer from salinity infiltration. They may have seasonal population changes or they may feed a sensitive tertiary process (e.g. UV).

Q2: DO YOU AGREE WITH THIS?

OTHER COMMENTS....

3. Site restricted or not.....

Some sites are so compact that only certain processes will “fit in”. Sites that are located in densely populated areas tend to have huge aesthetic constraints, and a totally covered process is preferable. This means a low footprint process is required.

Q3: DO YOU AGREE WITH THIS?

OTHER COMMENTS....

The various combinations of these three differences produce eight segments A to H.

A: An urban site that requires a low footprint process which has minimal sensitivity to varying volumes/ toxicity.

B: An urban site that requires a process which has minimal sensitivity to varying volumes/ toxicity.

C: An urban site that requires a low footprint process.

D: An urban site (without particular process requirement).

E: A rural site that requires a low footprint process which has minimal sensitivity to varying volumes/ toxicity.

F: A rural site that requires a process which has minimal sensitivity to varying volumes/ toxicity.

G: A rural site that requires a low footprint process.

H: A rural site (without particular process requirement).

Q4: DO YOU AGREE THAT THE PREFERED SECONDARY PROCESSES WILL BE DIFFERENT FOR THESE SEGMENTS?

COMMENTS....

Q5: DO YOU AGREE THAT URBAN SITES TEND TO TREAT > 5000 p.e. WHILST RURAL SITES TEND TO TREAT < 5000 p.e.?
WHAT CUT OFF POINT IS MORE REPRESENTATIVE?

For questions 6-8 please answer in the table. Approximate percentages will be fine.

Q6: WHICH PROCESSES TEND TO BE USED FOR EACH SEGMENT?

Q7: WHAT % OF ALL SITES ARE LIKE SEGMENTS A TO H?

Q8: WHAT % OF THE TOTAL AMMOUNT OF WASTEWATER IS TREATED ON A SITE LIKE EACH SEGMENT?

	Q6	Q7	Q8
Segment	Processes used	% of sites	% of wastewater
A: urban, special & site restrictions			
B: urban & special			
C: urban & site restrictions			
D: urban			
E: rural, special & site restrictions			
F: rural & special			
G: rural & site restrictions			

APPENDIX 7E:

**DIAGNOSIS TOOL COMPLETED FOR DIFFERENT
CIRCUMSTANCES**

ECCI alone under present circumstances

Segments studied	S1	S2	S3	S4	S5	S6
Technology related risk						
1. Performance (Veto)	4	4	3	4	4	3
2. Enhancement potential	3	3	2	3	3	2
3. Technology's image	4	0	0	4	1	2
4. Company's proficiency	1	1	1	1	1	1
5. Project & company coherence	3	3	3	3	3	3
Risk	0.250	0.450	0.550	0.250	0.400	0.450

Segments studied	S1	S2	S3	S4	S5	S6
Risks related to the technological environment						
1. Suppliers capacity	4	4	4	4	4	4
2. Risk of takeover	4	4	4	4	4	4
3. Enhancement pot. of comp. technol.	3	3	3	3	3	3
4. General demand for technology	3	4	1	1	3	0
5. Customer implementation	3	3	0	3	3	0
Risk	0.150	0.100	0.400	0.250	0.150	0.450

Segments studied	S1	S2	S3	S4	S5	S6
Company properties						
1. Exp. & adapt. of sales function	3	3	3	3	3	3
2. Company adaptability.	1	1	1	1	1	1
3. Wealth	1	1	1	1	1	1
4. Synergy with other projects	3	3	3	3	3	3
5. Risk of being integrated	4	4	4	4	4	4
6. Service	4	4	4	4	4	4
7. Company image	2	2	2	2	2	2
8. Conformity with laws & regulations	4	4	4	4	4	4
Risk	0.313	0.313	0.313	0.313	0.313	0.313

Segments studied	S1	S2	S3	S4	S5	S6
Market appeal & constraints						
1. Market dynamics	3	3	1	3	3	1
2. Growth rate	4	4	1	4	4	1
3. Politico-economic risk	2	2	2	2	2	2
4. Intensity of competition	0	0	1	0	0	1
5. Clients seek leading-edge technol.	4	4	1	4	3	1
6. Price levels and ROI	3	3	2	3	3	2
7. Size and volume of market	2	3	1	0	2	0
8. Company & market coherence	2	2	2	2	2	2
Risk	0.375	0.344	0.656	0.438	0.406	0.688

Segments studied	S1	S2	S3	S4	S5	S6
Technology risk	0.250	0.450	0.550	0.250	0.400	0.450
Risk related to technological environment	0.150	0.100	0.400	0.250	0.150	0.450
Technology index	0.206	0.326	0.481	0.250	0.302	0.450
Company advantages & shortcomings	0.313	0.313	0.313	0.313	0.313	0.313
Market appeal & constraints	0.375	0.344	0.656	0.438	0.406	0.688
Commercial index	0.345	0.328	0.514	0.380	0.362	0.534
Strategic index	0.284	0.327	0.498	0.322	0.334	0.494

Partners now

Segments studied	S1	S2	S3	S4	S5	S6
Technology related risk						
1. Performance (Veto)	4	4	3	4	4	3
2. Enhancement potential	3	3	2	3	3	2
3. Technology's image	4	0	0	4	1	2
4. Company's proficiency	4	4	4	4	4	4
5. Project & company coherence	4	4	4	4	4	4
Risk	0.050	0.250	0.350	0.050	0.200	0.250

Segments studied	S1	S2	S3	S4	S5	S6
Risks related to the technological environment						
1. Suppliers capacity	4	4	4	4	4	4
2. Risk of takeover	4	4	4	4	4	4
3. Enhancement pot.of comp. technol.	3	3	3	3	3	3
4. General demand for technology	3	4	1	1	3	0
5. Customer implementation	3	3	0	3	3	0
Risk	0.150	0.100	0.400	0.250	0.150	0.450

Segments studied	S1	S2	S3	S4	S5	S6
Company properties						
1. Exp. & adapt. of sales function	3	3	3	3	3	3
2. Company adaptability.	2	2	2	2	2	2
3. Wealth	3	3	3	3	3	3
4. Synergy with other projects	4	4	4	4	4	4
5. Risk of being integrated	4	4	4	4	4	4
6. Service	4	4	4	4	4	4
7. Company image	3	3	3	3	3	3
8. Conformity with laws & regulations	4	4	4	4	4	4
Risk	0.156	0.156	0.156	0.156	0.156	0.156

Segments studied	S1	S2	S3	S4	S5	S6
Market appeal & constraints						
1. Market dynamics	3	3	1	3	3	1
2. Growth rate	4	4	1	4	4	1
3. Politico-economic risk	2	2	2	2	2	2
4. Intensity of competition	0	0	1	0	0	1
5. Clients seek leading-edge technol.	4	4	1	4	3	1
6. Price levels and ROI	3	3	2	3	3	2
7. Size and volume of market	2	3	1	0	2	0
8. Company & market coherence	2	2	2	2	2	2
Risk	0.375	0.344	0.656	0.438	0.406	0.688

Segments studied	S1	S2	S3	S4	S5	S6
Technolgy risk	0.050	0.250	0.350	0.050	0.200	0.250
Risk related to technological environment	0.150	0.100	0.400	0.250	0.150	0.450
Technology index	0.112	0.190	0.376	0.180	0.177	0.364
Company advantages & shortcomings	0.156	0.156	0.156	0.156	0.156	0.156
Market appeal & constraints	0.375	0.344	0.656	0.438	0.406	0.688
Commercial index	0.287	0.267	0.477	0.328	0.308	0.499
Strategic index	0.218	0.232	0.429	0.265	0.251	0.436

ECCI convince group 3

Segments studied	S1	S2	S3	S4	S5	S6
Technology related risk						
1. Performance (Veto)	4	4	3	4	4	3
2. Enhancement potential	3	3	2	3	3	2
3. Technology's image	4	4	4	4	3	2
4. Company's proficiency	1	1	1	1	1	1
5. Project & company coherence	3	3	3	3	3	3
Risk	0.250	0.250	0.350	0.250	0.300	0.450

Segments studied	S1	S2	S3	S4	S5	S6
Risks related to the technological environment						
1. Suppliers capacity	4	4	4	4	4	4
2. Risk of takeover	4	4	4	4	4	4
3. Enhancement pot.of comp. technol.	3	3	3	3	3	3
4. General demand for technology	3	4	1	1	3	0
5. Customer implementation	3	3	0	3	3	0
Risk	0.150	0.100	0.400	0.250	0.150	0.450

Segments studied	S1	S2	S3	S4	S5	S6
Company properties						
1. Exp. & adapt. of sales function	3	3	3	3	3	3
2. Company adaptability.	1	1	1	1	1	1
3. Wealth	1	1	1	1	1	1
4. Synergy with other projects	3	3	3	3	3	3
5. Risk of being integrated	4	4	4	4	4	4
6. Service	4	4	4	4	4	4
7. Company image	2	2	2	2	2	2
8. Conformity with laws & regulations	4	4	4	4	4	4
Risk	0.313	0.313	0.313	0.313	0.313	0.313

Segments studied	S1	S2	S3	S4	S5	S6
Market appeal & constraints						
1. Market dynamics	3	3	1	3	3	1
2. Growth rate	4	4	1	4	4	1
3. Politico-economic risk	2	2	2	2	2	2
4. Intensity of competition	0	0	1	0	0	1
5. Clients seek leading-edge technol.	4	4	1	4	3	1
6. Price levels and ROI	3	3	2	3	3	2
7. Size and volume of market	2	3	1	0	2	0
8. Company & market coherence	2	2	2	2	2	2
Risk	0.375	0.344	0.656	0.438	0.406	0.688

Segments studied	S1	S2	S3	S4	S5	S6
Technology risk	0.250	0.250	0.350	0.250	0.300	0.450
Risk related to technological environment	0.150	0.100	0.400	0.250	0.150	0.450
Technology index	0.206	0.190	0.376	0.250	0.237	0.450
Company advantages & shortcomings	0.313	0.313	0.313	0.313	0.313	0.313
Market appeal & constraints	0.375	0.344	0.656	0.438	0.406	0.688
Commercial index	0.345	0.328	0.514	0.380	0.362	0.534
Strategic index	0.284	0.268	0.450	0.322	0.306	0.494

Partners & group 3

Segments studied	S1	S2	S3	S4	S5	S6
Technology related risk						
1. Performance (Veto)	4	4	3	4	4	3
2. Enhancement potential	3	3	2	3	3	2
3. Technology's image	4	4	4	4	3	2
4. Company's proficiency	4	4	4	4	4	4
5. Project & company coherence	4	4	4	4	4	4
Risk	0.050	0.050	0.150	0.050	0.100	0.250

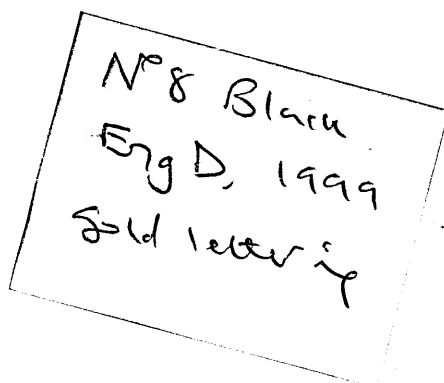
Segments studied	S1	S2	S3	S4	S5	S6
Risks related to the technological environment						
1. Suppliers capacity	4	4	4	4	4	4
2. Risk of takeover	4	4	4	4	4	4
3. Enhancement pot.of comp. technol.	3	3	3	3	3	3
4. General demand for technology	3	4	1	1	3	0
5. Customer implementation	3	3	0	3	3	0
Risk	0.150	0.100	0.400	0.250	0.150	0.450

Segments studied	S1	S2	S3	S4	S5	S6
Company properties						
1. Exp. & adapt. of sales function	3	3	3	3	3	3
2. Company adaptability.	2	2	2	2	2	2
3. Wealth	3	3	3	3	3	3
4. Synergy with other projects	4	4	4	4	4	4
5. Risk of being integrated	4	4	4	4	4	4
6. Service	4	4	4	4	4	4
7. Company image	3	3	3	3	3	3
8. Conformity with laws & regulations	4	4	4	4	4	4
Risk	0.156	0.156	0.156	0.156	0.156	0.156

Segments studied	S1	S2	S3	S4	S5	S6
Market appeal & constraints						
1. Market dynamics	3	3	1	3	3	1
2. Growth rate	4	4	1	4	4	1
3. Politico-economic risk	2	2	2	2	2	2
4. Intensity of competition	0	0	1	0	0	1
5. Clients seek leading-edge technol.	4	4	1	4	3	1
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7. Size and volume of market	2	3	1	0	2	0
8. Company & market coherence	2	2	2	2	2	2
Risk	0.375	0.344	0.656	0.438	0.406	0.688

Segments studied	S1	S2	S3	S4	S5	S6
Technolgy risk	0.050	0.050	0.150	0.050	0.100	0.250
Risk related to technological environment	0.150	0.100	0.400	0.250	0.150	0.450
Technology index	0.112	0.079	0.302	0.180	0.127	0.364
Company advantages & shortcomings	0.156	0.156	0.156	0.156	0.156	0.156
Market appeal & constraints	0.375	0.344	0.656	0.438	0.406	0.688
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Development of a Novel Medium to Improve the Performance
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