CRANFIELD UNIVERSITY

ANU TALVIKKI LAINE

TECHNOLOGIES FOR GREYWATER RECYCLING IN BUILDINGS

SCHOOL OF WATER SCIENCES

PhD THESIS

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Technologies for greywater recycling in buildings

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Abstract

The potential of four pilot-scale biological wastewater treatment processes for in-building greywater recycling has been assessed. The evaluation of three membrane bioreactors (MBRs) and a biological aerated filter (BAF) was principally with reference to the non-potable water reuse quality standards.

In the steady-state trials the feedwater quality was changed from synthetic greywater to blackwater (primary sewage influent) simulating the variability of domestic wastewater. The submerged MBR met both the chemical and microbiological water reuse quality standards independent of the feedwater strength. The BAF mostly achieved a significant reduction in organic pollutants but failed to meet the microbiological water quality criteria. Greywater treatment by the side-stream MBR resulted in a very good effluent quality whilst the membrane aeration bioreactor (MABR) had the most limited capacity to remove the pollutants measured.

The unsteady-state trials were carried out on the submerged MBR and the BAF. The performance of the former remained relatively unchanged in the feedwater transitions and the intermittent operation of feed and/or air with synthetic greywater. These tests on the BAF resulted in a loss of performance such that in some cases the recovery took several days.

In the supplementary experiments nutrient deficiency, variability in strength and degradation during storage were found typical of both real and synthetic greywater, hence potentially affecting the treatability of greywater.

A survey on behavioural patterns in households indicated that perceived harmfulness of a substance did not necessarily affect the frequency with which it was discharged. Laboratory experiments suggested that some substances discharged with the greywater may increase the risk of failure of the biological treatment process.

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Notation

Al	aluminium
BAF	biological aerated filter
BOD	biochemical oxygen demand (mg per litre)
BOD_5	5-day biochemical oxygen demand (mg per litre)
BOD_7	7-day biochemical oxygen demand (mg per litre)
sBOD	soluble biochemical oxygen demand (mg per litre)
tBOD	total biochemical oxygen demand (mg per litre)
cfu	colony forming units
Cl^-	chloride
Cl_2	chlorine
Co	cobolt
COD	chemical oxygen demand (mg per litre)
COD_{Mn}	chemical oxygen demand with permangate (mg per litre)
sCOD	soluble chemical oxygen demand (mg per litre)
susCOD	suspended chemical oxygen demand (mg per litre)
tCOD	total chemical oxygen demand (mg per litre)
Cu	copper
DO	dissolved oxygen (mg per litre)
DWF	dry weather flow
Fe	iron
GAC	granular activated carbon
HRT	hydraulic retention time
tHRT	theoretical hydraulic retention time
ICP-AES	inductively coupled plasma atomic emission spectrophotometer
MABR	membrane aeration bioreactor
MBAS	methylene blue active substance
MBR	membrane bioreactor
MF	microfiltration
MLSS	mixed liquor suspended solids (mg per litre)
MLVSS	mixed liquor volatile suspended solids (mg per litre)
Mo	molybdenium
N	nitrogen
NF	nanofiltration
NH_3-N (NH_3)	nitrogen as ammonia(mg per litre)
NO_3-N (NO_3)	nitrogen as nitrate (mg per litre)
NO_2-N (NO_2)	nitrogen as nitrite (mg per litre)
NTU	nephelometric turbidity unit
O_2	oxygen
P	phosphorus
pcc	per capita consumption
pe	population equivalent
pH	potential hydrogen

PVC	polyvinyl chloride
RBC	rotating biological contactor
RO	reverse osmosis
RTD	residence time distribution
SBR	sequencing batch reactor
SOUR	specific oxygen uptake rate
SRT	solids retention time
SS	suspended solids (mg per litre)
STW	sewage treatment works
TKN	total Kjeldahl nitrogen (mg per litre)
TMP	transmembrane pressure (bar)
TN	total nitrogen (mg per litre)
TOC	total organic carbon (mg per litre)
TP	total phosphorus (mg per litre)
UF	ultrafiltration
UV	ultra violet
Zn	zinc

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Chapter 1 *Introduction*

1.1 Background to water recycling and reuse

A 70% increase in fresh water demand has been experienced in the last 30 years (DETR, 1999) due to a rapid population growth, socio-economic factors and industrial activity. The situation has been highlighted by periodic regional droughts, such that the public have become increasingly aware of the insufficient water resources (Sayers, 1998; Surendran and Wheatley, 1998). Such events have emphasised the importance of sustainability, which can be achieved with water reuse in combination with other conservation measures. Water reuse is most readily employed for non-potable applications where the relatively high quality water is not a necessity. The estimated proportion of the water required for drinking and cooking is 3-5% (Okun, 1982; Sayers, 1998). Hence a large proportion of water can be reused for non-potable purposes, and these uses are divided into four categories:

- irrigation,
- industrial,
- groundwater recharge, and
- domestic.

Irrigation is the most widely employed water recycling and reuse application world-wide. The level of technology depends on the general level of both the development and the climatic conditions of the country concerned. For example in the Mediterranean countries the demand is over 70% of the total water use such that water withdrawal as a percentage of the total actual renewable water resources often exceeds 100% (FAO, 1997).

About 25% of global water demand is related to industrial applications (Asano and Levine, 1995) where a high level of technology is used to refine often onerous

effluents for reuse purposes. Increased pressure on industry to limit both the use of fresh water and the discharge of wastewaters will almost certainly lead to a higher number of industrial reuse applications world-wide.

Reclaimed water can also be recharged to aquifers either by infiltration through the soil or by direct injection into the aquifer, such that declining levels resulting from excessive extraction can be reduced, stopped or reversed. Coastal aquifers can be protected from intrusion of salt water and additional treatment for future reuse and storage of reclaimed water can be provided (Judd *et al.*, 1997).

This work focuses on domestic water reuse and particularly in the recycling of domestic wastewater within buildings. This is practised both in the industrial and developing countries driven by reasons such as dry climate, environmental protection and an aim to reduce overall costs. In addition to domestic wastewater, rainwater is commonly used as a supplementary source.

1.2 Research objectives

The research aims of this work are:

- to characterise greywater quality and dynamic behaviour,
- to investigate the feasibility of advanced biological wastewater treatment processes for in-building domestic wastewater recycling with reference to the existing non-potable water reuse quality standards,
- to assess the process recovery after intermittent operation of feed and/or air resulting from a system malfunction or a temporary power failure, and
- to identify components either occasionally or commonly present in greywater that could lead to an impaired performance of a biological treatment process.

Chapter 2 *Literature review*

2.1 Background to domestic water reuse

Domestic water reuse involves the purification and reuse or recycling of greywater, rainwater or blackwater, or a combination of these wastewaters, within or outside a building (Figure 2.1). Greywater originates from all the in the household other than the toilet, urinal and bidet, and hence does not usually contain faecal matter (Section 2.2). Blackwater, or foul water, includes all household wastewater that enters a sewer system (Section 2.2). Water reclamation in buildings can include the recycling of greywater and the use of reclaimed water supplied through a dual or separate distribution system. Another common application is the use of rainwater for either toilet flushing or garden irrigation during dry periods.

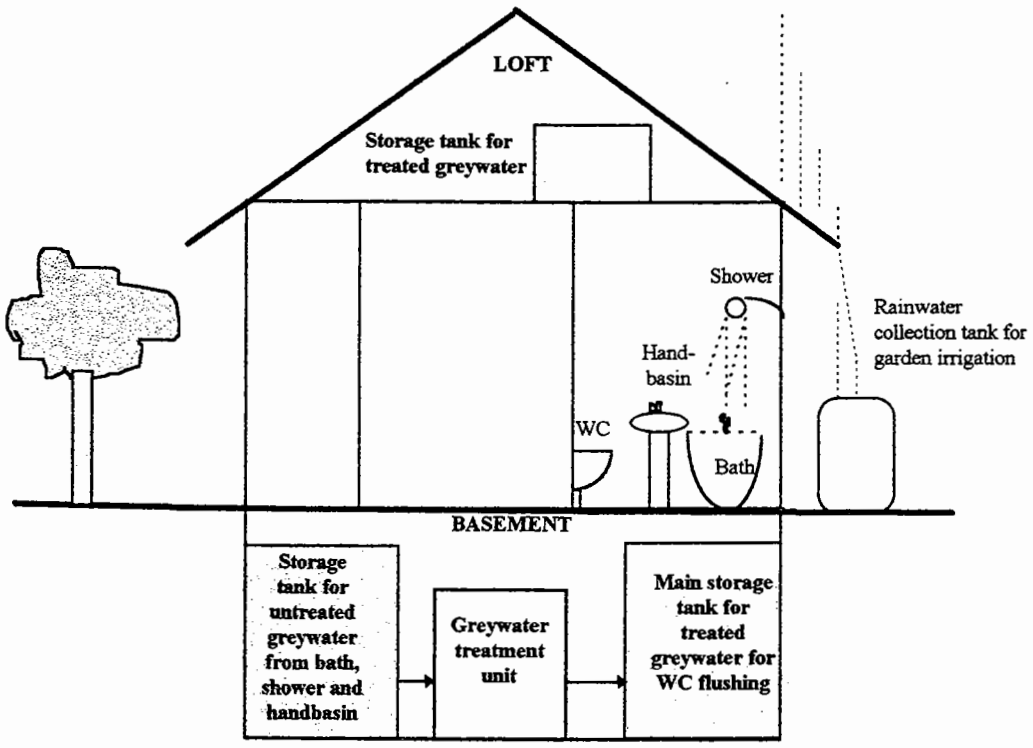


Figure 2.1. Schematic of domestic wastewater recycling showing common applications of internal use of greywater and external use of rainwater.

Domestic water recycling is an attractive option in the UK due to a relatively high domestic water consumption coupled with an intensive population: water use for domestic purposes constitutes 40% of the total demand, and over half of the population lives in towns of 100 000 or greater (Boschet *et al.*, 1998). In the near future there will be more pressure on the natural water resources due to the water use increase by 1.5-2% per annum (Butler, 1998b; Smith *et al.*, 1999) and the current plan to build about 4.4 million new households in 1991-2016 (DoE, 1996). However, there are several barriers to the uptake of greywater recycling systems (Mustow, 1998):

- environmental issues (Section 2.1),
- public health concerns (Section 2.3),
- lack of design structure (Section 2.3),
- technological limitations (Section 2.4),
- public perception (Section 2.6), and
- cost (Section 2.5.4).

Variation in climate provides an environmental barrier to greywater reuse, since supply and demand vary from one region to another. This may be the case, for example, in some areas in Europe where the abundant natural water resources exceeds the water demand. High annual rainfall does not necessarily preclude the implementation of water reuse. An example of this is Japan where the rainfall is highly seasonal and there is limited scope to store surface water. The majority of the population live in urban areas, which correspond to 3% of the geographical area (Takahashi, 1991). In many of these districts the groundwater resources have been overexploited, which has lead to the development of domestic and industrial water reuse schemes.

Another environmental barrier is the possibility of contamination of the natural water resources due to inadequately performing recycling systems or by-products in greywater (Section 2.4.3). This highlights the importance of selecting the most appropriate treatment process (Section 2.4) as well as the use of appropriate backup

and alarm systems. Environmental concerns may, however, promote the implementation of small-scale processes. This is the case in Japan, where greywater is the major pollutant in small rivers in cities, closed water basins and areas without public sewerage systems (Imura *et al.*, 1995). A number of conservation measures have resulted, including a rapid promotion of household wastewater treatment plants to reduce the organic and nitrogen load entering natural waters.

Political and social issues play an important role in water reuse, and ultimately may impede the process of introducing conservation methods: In the Middle Eastern countries water recycling is unacceptable on fundamental religious grounds. In contrast, the extensive water recycling is in place in the arid and semi-arid areas of the United States and Australia, where the climate has been the driving force towards extensive water recycling. In these areas the public generally view water recycling as an essential environmental protection measure.

A reduction in water use is an important element in progress towards sustainability and may require statutory means. The Netherlands have agreed a national target to cut potable water consumption by 10% to maintain a sustainable water supply, whereas no such targets exist in the UK (Environment Agency, 1998). One method employed to reduce demand is metering, which has been estimated to result in around 10-15% reduction in potable water consumption in households (Butler, 1998b; Edwards and Martin, 1995). This, however, may not be sufficient, and further action has been taken in many arid regions of the world. In Australia, for example, where many communities have achieved a 10-30% reduction on water use: the potential is estimated to be in the region of 25-30% if water efficient fittings and appliances are used (Williams, 1998), and as high as 30-50% if all greywater is reused (Jeppesen, 1996a). Also in Copenhagen, Denmark, the water use has dropped, as a result of the promotion of water saving appliances, by as much as 36% since the mid-1990's (Mikkelsen *et al.*, 1999; Stanner and Bordeau, 1995).

2.2 Water flows and strengths

2.2.1 Domestic water demand

Water consumption depends on regional, demographic and cultural conditions and is measured as per capita consumption (pcc in l person^{-1} d^{-1}). It is also recognised that water use has a tendency to increase with increasing income and decreasing household occupancy (Edwards and Martin, 1995). In the UK a range from 101 to 212 l person^{-1} d^{-1} has been reported (Butler, 1991, 1993 and 1998a; Edwards and Martin, 1995; Hall *et al.*, 1988; Surendran and Wheatley, 1998). This compares well with 115-260 l person^{-1} d^{-1} (Griggs *et al.*, 1997) presented for the rest of Europe and is low in comparison with 450 l person^{-1} d^{-1} in Zurich, Switzerland (Stanner and Bordeau, 1995). Water consumption at 133-223 l person^{-1} d^{-1} (Laak, 1974; Ligman *et al.*, 1974; Rose *et al.*, 1991; Siegrist, 1976) in some areas of the United States appears to be similar to that in the UK. Recently a figure as high as 1136 l person^{-1} d^{-1} has been reported (York and Burg, 1998), which is likely to include garden irrigation, in arid areas in the US.

Breakdown of typical domestic water usage in various countries (Table 2.1) demonstrates that the daily water volumes used for different purposes in households are similar. Toilet water usage is typically a third of the total domestic water consumption. It is higher in cases where high flush toilets are used, as could be assumed from two of the US examples (references 6 and 7, Table 2.1). Bath, shower and washbasin water comprises 25-30% of the total wastewater volume. Some 7-14% of the water is used in the kitchen and a further 11-31% for laundering operations. Most variation (7-41%) is found in the other uses that may or may not include water use in kitchen depending on the literature source. It can be seen from Table 2.1 that greywater from the bath, shower and washbasin roughly meets the toilet water demand. However, as greywater is produced at a time slightly offset from toilet flushing and generated over short time periods, whilst toilet flushing takes place more

consistently throughout the day, storage of both untreated and treated greywater is necessary.

Table 2.1. Breakdown of typical domestic water usage (l person^{-1} d^{-1}).

Reference	1	2	3	4	5	6	7	8
Country	UK	UK	UK	Denmark	The Netherlands	USA	USA	USA
Toilet	31	37	61.2	40	30.5	75	76	36
Kitchen sink	13	-	29.7	20	10.5	14	13	18
Wash basin	13	-	25.5	-	5.4	8	-	-
Bath and shower	28	19	34.4	45[b]	59.7	32	47	38
Washing machine	17	13	25.6	10	23.1	28	38	41
Other	-	48	35.9[a]	45[c]	15.4[d]	-	6	-
Total	101	117	212.3	160	144.6	156	180	133

1 Butler (1991, 1993) 2 Hall *et al.* (1988) 3 Surendran and Wheatley (1998)

4 Mikkelsen *et al.* (1999) 5 Van der Hoek *et al.* (1999) 6 Laak (1974)

7 Ligman *et al.* (1974) 8 Siegrist *et al.* (1976)

[a] dishwasher 6.8 l, outside tap 29.1 l

[b] includes handbasin

[c] dish washing and cleaning 25 l, other including leakage 20 l

[d] dishwasher 1 l, handwashing clothes 4.8 l, other 9.6 l

Information on the discharge volumes and patterns from households can be used to understand the rate and distribution of wastewater discharges into drains and sewers where characteristic patterns on annual, seasonal, daily, hourly and sub-hourly time scales can be observed. This variability is a useful tool in the design of both centralised sewage treatment works (STW) and localised greywater recycling systems. The water usage pattern from the different appliances is distinct, particularly in individual households, but it can also be observed in larger multi-occupancy buildings, though the variation in flows tend to be smaller (Webster, 1972).

Research on domestic wastewater inflows (Butler, 1991; Butler *et al.*, 1995; Edwards and Martin, 1995; Surendran and Wheatley, 1998) has shown that a morning peak discharge is followed by two major discharge peaks, one in the afternoon and the

other in the late evening. A minimum flow period of 4 h occurs late at night, corresponding to the occupants' sleeping hours (Butler *et al.*, 1995). The distribution of the appliance discharges shows that the smallest contribution comes from the washbasin (Butler *et al.*, 1995). A small-scale diary on domestic sanitary appliance usage in the south east of England (Butler, 1993) revealed that the appliance usage pattern is influenced both by the dwelling occupancy and the day of the week. It is more frequent in households with more occupants than in those with a smaller number of residents. At weekends the morning usage peak is more extended and smaller than during the week, appearing after a delay of 1-2 h.

According to Santala *et al.* (1998) water consumption peaks of 5-15 min long can contribute as much as 30-60% of the total measured daily consumption for washing purposes, depending on the demographic structure and life-style of the occupants. Butler *et al.* (1995) noted that the discharge from the bath and shower constitutes up to 66% of the total instantaneous discharge in the early morning (4-8 am) and evening periods (6-10 pm). Although the timescale in the latter study was longer than in the former, the findings regarding the ratio of shower and bath usage to total water use and discharge are in good agreement.

The most significant single wastewater-generating appliance, irrespective of the day of the week, is the toilet (Butler *et al.*, 1995). It contributes about 40% of the total instantaneous flow during the day and up to 90% at nights.

The discharges from greywater-generating appliances and the toilet have an impact on the overall wastewater quality. Biochemical oxygen demand (BOD) is largely variable throughout the day in comparison to orthophosphates, which provide a smaller concentration range, and to nitrate, which remains fairly constant (Butler *et al.*, 1995). Ammonia load in domestic wastewater is most changeable as it is affected by the toilet discharges. A one week survey on toilet usage (Friedler *et al.*, 1996) in southern England showed that the quantities of solid matter generated by the toilet were significant. All types of solid generation (faecal related flushes, toilet paper, and

sanitary refuse) showed distinctive morning peaks, and differences were noted between weekday and weekend usage, giving variable solids quality and loading which then had an impact on overall domestic wastewater quality.

2.2.2 Domestic wastewater strengths

Greywater

Greywater quality varies from one appliance to another, since wastewater is generated from washing machines, dishwashers and kitchen sinks. It is often alkaline and contains detergents, oil, grease and other organic matter (Table 2.2). These materials significantly impair conventional biological wastewater treatment, such that in most greywater reuse schemes only discharges from less polluted sources, i.e. baths, showers and handbasins, is recycled. Though this may exclude the treatment of some of the problematic substances, the heterogeneity of greywater composition nevertheless complicates both the treatment and risk assessment of reuse (Rose *et al.*, 1991). Pollutants such as grease may be substantially removed by appropriate pretreatment. An example of this is the use of grease traps for kitchen discharges, consequently improving BOD removal by a biological process (Adachi and Fuchu, 1991; Nakajima *et al.*, 1999).

Due to its composition greywater is prone to offensive odours and growth of micro-organisms during storage unless adequately treated (Jeppesen, 1996a; Section 4.2.3). Microbiological quality has been shown to deteriorate in distribution networks even within drinking water systems such that bacterial densities may increase 2-3-fold due to residence time, materials, flow conditions and temperature (Prévost *et al.*, 1997).

More than 95% of the pollution within greywater may consist of detergents (Santala *et al.*, 1998). Soaps contain surface active agents, i.e. surfactants, that have different properties of solubility, biodegradability and foam formation. Anionic detergents are known to produce an odour in water at concentrations of 0.4-3 mg l^{-1} (Chapman, 1992) and some combinations of surfactants may nullify disinfectants (Christova-Boal

et al., 1996), complicating the selection of the disinfectant. Soaps with reduced foaming properties can be used in greywater recycling schemes to prevent excessive foaming that can lead to operational problems (Section 5.4). It is important to limit the concentration of detergents to prevent foaming of the treated greywater in the toilet during flushing (Naisby, 1997).

Table 2.2. Characteristics of greywater (adapted from Hodges, 1998).

Greywater source	Characteristics
Washing machine	Bleach, foam, high pH, high temperature, nitrates, oil and grease, oxygen demand, phosphates, salinity, soaps, sodium, suspended solids and turbidity.
Dishwasher	Bacteria, foam, food particles, high pH, high temperature, odour, oil and grease, organic matter, oxygen demand, soaps, sodium, suspended solids and turbidity.
Bath, washbasin and shower	Bacteria, hair, high temperature, odour, oil and grease, oxygen demand, soaps, suspended solids and turbidity.
Sinks including kitchen	Bacteria, food particles, high temperature, odour, oil and grease, organic matter, oxygen demand, soaps, suspended solids and turbidity.

Greywater has a similar organic strength to aggregated domestic sewage but has a different chemical nature (Table 2.3). Low turbidity and suspended solids indicate that a great proportion of the contaminants in greywater are dissolved. An example of this is the share of the soluble BOD_5 ($sBOD_5$) at 60% (Laak, 1986). A further complication arises from the solids that are less settleable than in toilet wastewater (Laak, 1986) such that significant removal of solids by conventional sedimentation may be difficult. The ratio of chemical to biochemical oxygen demand (COD:BOD) is variable, and a value as high as 4:1 has been reported (Holden *et al.*, 1998). This is nearly double in comparison to the COD:BOD ratio of 2.2-2.5:1 typical of sewage (Metcalf and Eddy, 1991). Although greywater contains relatively little biodegradable matter, its BOD degrades faster than that of toilet wastewater (Laak *et al.*, 1981; Olsson *et al.*, 1968). Compared to sewage, greywater is often deficient in phosphorus (P) and nitrogen (N).

Table 2.3. Typical greywater strengths. Mean (standard deviation).

Parameter	Reference 1 UK	2 UK	3 UK	4 USA	5 Sweden	6 Sweden	6 Australia	7 Australia	8 Japan
COD ($mg\ l^{-1}$)	146	40	168 (91)	366	n/a	n/a	n/a	n/a	89[h]
BOD_5 ($mg\ l^{-1}$)	80	33	96 (103)	162	196	47[b] (25)	159 (69)	76-200[c]	349
COD:BOD (-)	1.8	1.2	1.8	2.3	n/a	n/a	n/a	n/a	n/a
TOC ($mg\ l^{-1}$)	n/a	n/a	49 (53)	125	n/a	n/a	n/a	n/a	n/a
Turbidity (NTU)	59	20	57 (138)	n/a	n/a.	n/a	113 (55)	60-240[c]	n/a
SS ($mg\ l^{-1}$)	n/a	n/a	45 (66)	162	141	n/a	113 (91)	48-120[c]	97
TN ($mg\ l^{-1}$)	n/a	n/a	n/a	n/a	6.5	3.7 (2.2)	11.6 (10.2)	4.6-20[c, d]	n/a
PO_4^- ($mg\ l^{-1}$)	n/a	0.4	2.4 (0.7)	n/a	7.8	3.7 (2.7)	n/a	0.11-1.8[c, e]	n/a
NH_3 ($mg\ l^{-1}$)	10	1.1	0.8 (0.7)	n/a	n/a	n/a	n/a	<0.1-15[c, f]	n/a
NO_3 ($mg\ l^{-1}$)	n/a	n/a	1.3 (0.7)	n/a	n/a	n/a	n/a	<0.05-0.20[c]	n/a
pH (-)	7.6	n/a	7.7 (0.4)	6.8	n/a	n/a	7.3 (0.6)	6.4-8.1[c]	6.6
Total coliforms ($cfu\ 100^{-1}\ ml^{-1}$)	n/a	n/a	5200000 (3600000)	24000000	3600000	n/a	n/a	24000000-5000000000[c]	n/a
Faecal streptococci ($cfu\ 100^{-1}\ ml^{-1}$)	n/a	n/a	479[a] (859)	1400000	880000[a]	35848 (34732)	n/a	2400-79000[c] 3300-170000[c, g]	n/a
Colon bacilli ($pp\ 100^{-1}\ ml^{-1}$)	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	185000

1 Ward *et al.* (2000) (Linacre college, Oxford) 2 Ward *et al.* (2000) (Fedden House, Cranfield) 3 Surendran and Wheatley (1999) 4 Brandes (1978)
5 Olsson *et al.* (1968) 6 Günther (2000) 7 Jeppesen (1996b) 8 Huitorel (2000)
n/a not available, [a] faecal coliforms, [b] BOD_7, [c] range, [d] total Kjeldahl nitrogen, [e] TP, [f] nitrate and nitrite, [g] faecal coliforms, [h] COD_{Mn}

A high count of coliforms in greywater (Brandes, 1978; Olsson *et al.*, 1968; Surendran and Wheatley, 1998) relative to sewage arises from the fact that organic matter from kitchen wastewater undergoes little breakdown, whereas all material in toilet wastewater undergoes considerable microbial and enzymatic breakdown during its passage through the human digestive tract (Brandes, 1978). Generally more faecal coliforms are present in shower and bath water than in laundry water (Christova-Boal *et al.*, 1996; Rose *et al.*, 1991). Total and faecal coliform count in greywater from families with young children can be up to 5 orders of magnitude higher than that from families without children (Rose *et al.*, 1991). Viruses may be present if the individuals are infected (Rose *et al.*, 1991; Yates and Gerba, 1998). For example, the concentration of rotaviruses can be as high as 10^{12} particles per gram faeces (Flewett, 1982). Due to the existing pathogens garden irrigation with untreated greywater is often not recommended particularly if the crops are to be eaten uncooked (EPA, 1992; Mustow *et al.*, 1997).

Personal hygiene products and pipework materials contribute sodium, sulphate, phosphorus, zinc, aluminium, copper and carbonate to greywater, and may sometimes arise in high concentrations in bathroom and laundry waters (Christova-Boal *et al.*, 1996; Hypes, 1974; Jeppesen, 1996b). Some of the nutrients and metals may be detrimental to soil condition if greywater is continuously used for irrigation.

Blackwater

Decentralised systems in rural areas often separate greywater and blackwater, or liquid and solid waste (Fittschen and Niemczynowicz, 1997; Skjelhaugen, 1999; Zeeman and Lettinga, 1999; Otterpohl *et al.*, 1999). There are several reasons for this: by separating the wastewater streams they can be treated accordingly for different reuse purposes; diluting blackwater with greywater increases the cost of treatment and reduces its agronomic value (Skjelhaugen, 1999).

Variations in blackwater quality (Table 2.4) depend upon the same factors that affect greywater quality. Blackwater contributes 80-91% of the total nitrogen (TN) and

phosphorus (Jenssen and Skjelhaugen, 1994; Laak *et al.*, 1981; Olsson *et al.*, 1968) as well as 50-75% of the organic matter (Jenssen and Skjelhaugen, 1994) to domestic wastewater.

Table 2.4. Examples of blackwater quality. Mean (range)

	Reference		
Parameter	1	2	3
	The Netherlands	USA	USA
COD	1720, 1200*	258 (175-490)	n/a
BOD	n/a	90 (38-160)	280
SS	n/a	77 (37-261)	450
Total solids	n/a	621 (521-745)	n/a
N	n/a	153 (140-170)	145
P	n/a	18.6 (16-22)	20

1 Zeeman and Lettinga (1999) 2 Brandes (1978) 3 Laak (1981)

* suspended n/a not available

Importance of nutrients

One of the key issues in biological wastewater treatment is the type and amount of nutrients. These have an important role in biological systems as cells require macronutrients for metabolic processes, the primary nutrients being nitrogen, phosphorus and carbon (C) (Beardsley and Coffey, 1985). The trace element, or micronutrient, requirements of bacterial cells can be determined from the composition of the cells (Metcalf and Eddy, 1991). However, as excess micronutrients can be adsorbed onto cell walls, the concentrations in biomass ash may exceed the actual amount required (Nicholas, 1963; Wood and Tchobanoglous, 1975). Actual requirements are influenced strongly by the organic and hydraulic loading rates, the cell growth rate (Speitel and Segar, 1995), the nature of the waste, and the mean cell residence time (Wood and Tchobanoglous, 1975). The retention time necessary for wastewater treatment decreases as environmental conditions and nutrient supply approach optimum levels (Benefield *et al.*, 1979; Speitel and Digiano, 1988). Several COD:N:P ratios at which nutrients should be supplied have been

presented: 100:10:1 and trace sulphur (Beardsley and Coffey, 1985), 250:7:1 (Franta *et al.*, 1994), 100:20:1 (Metcalf and Eddy, 1991) and 100:5:1 (Droste, 1997).

Nutrient addition is a useful tool in treatment of wastewaters which are deficient in one or several components required for effective performance by a biological process. This is often the case in industrial wastewaters. Nitrogen and/or phosphorus addition is often necessary for effluents from pulp and paper industry (Saunamäki, 1994; Section 5.3), food industry (Prendl and Nikolavcic, 2000) and chemical industry (Van Kempen *et al.*, 1997). Supplementary carbon can be added to wastewaters for enhanced denitrification and/or nitrification (Tam *et al.*, 1992; Watanabe *et al.*, 1995) or phosphorus removal (Jeon and Park, 2000; Tam *et al.*, 1992). Ghyoot *et al.* (1999) investigated the removal of a high nitrogen load in sludge reject water by a membrane-assisted bioreactor. In the first test the influent had a nitrogen concentration of 500-1000 mg l^{-1}. Methanol addition to the sludge at a COD:N ratio of 2.3 g g^{-1} increased the denitrification efficiency from 10% to 86% over a 7-day period. The mean COD removal was 87%. In the second test, where acetic acid was used as an additional carbon source, the influent nitrogen level fluctuated between 300 and 1200 mg l^{-1}. Acetic acid dose at a COD:N ratio of 4.0 g g^{-1} immediately increased denitrification to over 90%. During the 30 days of the trial a mean COD removal of 94% was measured. The difference in denitrification efficiencies relate to acetic acid being a readily available carbon source by many denitrifying bacteria whereas methanol needs a special bacterial population (Hallin *et al.*, 1996).

2.2.3 Impact of greywater recycling

The impact of greywater recycling are not restricted to local and immediate issues such as onsite storage and the selection of a technology. Domestic water reuse can result in a 35% reduction in discharged wastewater, leading to major impacts on small STWs (Butler, 1998b; Mustow *et al.*, 1997; Sayers, 1998). Though greywater reuse would decrease the overall pollutant load in the wastewater, it would increase pollutant concentration by up to 50% due to the decreased volume, potentially

encouraging septicity and consequently making the water less treatable (Butler, 1998b). On the other hand, reduced flow could have a positive impact on the downstream systems in terms of fewer capacity problems, leading to a design of a system with reduced capacity and lower pumping costs. Positive effects leading to improved quality discharges are likely, since storm events often lead to pollution peaks due to the flushing out of drains and sewers (Mustow *et al.*, 1997).

2.3 Legislation and water quality criteria

Legislation for water reuse is based on the necessity to protect human health and the environment. In the case of non-potable urban water reuse the public is exposed to reclaimed water via inhalation, direct skin contact or accidental ingestion. In addition to water quality standards, several of which are in place or under development abroad, adequate safety measures for non-potable reuse applications should include (Asano, 1994):

- separate storage and distribution systems,
- colour-coded labels to distinguish potable and non-potable systems,
- back-flow prevention devices,
- periodic tracer studies to detect cross-connections between potable and non-potable systems,
- off-hours usage to further minimise potential human contact, and
- information signs at sites using reclaimed water.

A lack of design guidance can cause confusion and impose a significant barrier to the use of greywater recycling systems.

Legislation and standards relating to domestic water reuse vary from country to the country. These are briefly appraised in the following section.

USA

The US federal laws on water reuse are broad and general in nature allowing the individual states to decide upon implementation. In general, federal laws apply to surface water only, whereas in the individual states the range is expanded to include groundwater. The Bureau of Reclamation conducts appraisal and feasibility studies supported by a government grant program and can participate in the design and construction of water reclamation projects (Cologne and MacLaggan, 1998). Loan programs are carried out by the individual states under the general US Environmental Protection Agency (US EPA) guidance and regulations. The US EPA guidelines (EPA, 1992; Table 2.5), which have been widely adopted within and outside of the US, list detailed information not only on water quality criteria but also the minimum level of treatment and the frequency of monitoring. They are thus one of the most stringent in the world. In comparison to the federal laws, the state laws are detailed and restrictive in prohibiting unreasonable use and control of water quality. For example, a state ordinance in California requires all large buildings to include dual plumbing (Crook, 1998). In contrast to California state, many less arid US states still have no legislation at all.

The World Health Organisation (WHO) has published guidelines (WHO, 1989) for wastewater use in agriculture and aquaculture, and some of these guidelines have been suggested as being suitable for domestic water recycling (Table 2.5).

Australia

In Australia public health regulations prohibit the use of greywater without specific prior approval (Stone, 1996). In general no regulations on acceptable reuse practices exist leaving the authorities to consider proposal water reuse schemes on a case-by-case basis. In 1993 the New South Wales (NSW) government published guidelines for urban and residential use of reclaimed water (NSW Recycled Water Coordination Committee, 1993; Table 2.5). These guidelines are applied in particular to new housing developments such as Rouse Hill, where a program was initiated in the early 1990's (Neal, 1996). The dual water supply on this site has been estimated to reduce

sewage flow by up to 40% (Williams, 1998). Since 1993 the regulations in Perth, Western Australia, have required the use of a dual flush toilet in all new installations (Stone, 1996). In such toilets a low (3 l) or a high flush (6 l) can be chosen. Promotion of retrofitting of existing toilets, shower heads and other water saving appliances and efficient irrigation are a part of the water efficiency program, which has potential to reduce the need for additional water supplies by about 23% (Stone, 1996).

General guidelines for domestic greywater reuse for Australia (Jeppesen, 1996b), which include requirements for design and installation of greywater systems, were published by the Urban Water Research Association of Australia (UWRAA) in 1996 (Mustow *et al.*, 1997). The performance and health and safety issues are also listed in detail. The functional requirements are described as follows:

- safe design and practice such that there is no human contact with greywater,
- the use of pressurised systems to avoid blockages in pipework,
- greywater reuse for lawn and ornamental garden irrigation only, and
- disposal of by-products in a hygienic manner.

The UWRAA guidelines define two treatment technologies suitable for domestic greywater treatment. The primary greywater reuse system comprises a coarse screen. Other treatment or storage of greywater is not permitted. This system may be used by permit for direct reuse of untreated bathroom and laundry greywater from a single-family home for sub-surface lawn and ornamental garden irrigation. The secondary greywater reuse system consists of an automatic filtration system followed by a membrane or sand filter. The system can be used by licence only in multi-occupancy buildings. Short-term storage is allowed if the greywater has BOD_5 <20 mg l^{-1}, SS < 30 mg l^{-1} and emits no odours. The treated greywater may be used for the same purposes as above.

Table 2.5. Summary of water quality standards and criteria suitable for domestic water recycling (adapted from Surendran and Wheatley, 1998).

Standard	Total coliforms (cfu 100^{-1} ml^{-1})	Faecal coliforms (cfu 100^{-1} ml^{-1})	BOD_5 (mg l^{-1})	Turbidity (NTU)	Cl_2 residual (mg l^{-1})	pH (-)
UK bathing water standard[a]	10000 (m) 500 (g)	2000 (m) 100 (g)	-	2 m (g) 1 m (m)	-	6-9
EC[b] bathing water standard	10000 (m) 500 (g)	2000 (m) 100 (g)	-	2 m (g) 1 m (m)	-	6-9
US NSF[c]	-	240	45	90	-	-
US EPA (g)	Non-detectable	14 for any sample 0 for 90% samples	10	≤ 2	1	6-9
Australia NSW[d]	1	4	20	2	-	-
Japan (m)	10	10 for any sample	10	5	-	6-9
Germany (g)	100 (g)	500 (g)	20 (g)	1-2 (m) 20	-	6-9
WHO[e] lawn irrigation	-	1000 (m) 200 (g)	-	-	-	-
UK (BSRIA)[f] (g)	-	14 for any sample 0 for 90% samples	-	-	-	-

[a] Bathing water standards suggested as appropriate for domestic water recycling

[b] EC = European Community

[c] NSF = National Sanitation Foundation

[d] NSW = New South Wales; also maximum of 2 viruses in 50 litres

[e] WHO = World Health Organisation

[f] toilet flushing

(g) = guideline

(m) = mandatory

Japan

The Japanese local and national government agencies have initiated numerous municipal and industrial wastewater reclamation and reuse and industrial water recycling facilities since 1970. Estimates of the share of urban water reused for toilet flushing range from 33% (Renaud *et al.*, 1997) to 37% (Murakami, 1989). Until recently laws and regulations governing reclaimed water were complex due to the established laws in potable water supply, sewerage and industrial water that were applied to water reuse (Asano *et al.*, 1981). An increasing need to incorporate water reuse into the traditional water supply lead to the revision of some of the procedures in the 1990's. For example, in Tokyo greywater recycling is mandatory for buildings

with a floor area >30 000 m^2 or potential water reuse of >100 m^3 d^{-1} (Rowden, 1996). Other Japanese cities also require smaller buildings to recycle greywater (EPA, 1992; Rowden, 1996). The Japanese domestic water reuse quality criteria are similar to many other standards in terms of BOD and physical parameters (Table 2.5), but do not require non-detectable bacteria in the reuse water.

European Union

The European Union (EU) is currently drafting a directive to assimilate all existing European regulations on water (Bontoux, 1998). The major problem is to find a uniform solution for all the member countries, which differ significantly geographically and climatically as well as in availability of water sources. Although greywater recycling for toilet flushing and fire fighting are emerging applications in France and Spain (Renaud *et al.*, 1997), these countries have worked towards a regulatory framework mostly for agricultural reuse which remains the major reuse application. Recently the Spanish government issued a draft of guidelines (Brissaud, 2000) which included non-potable urban reuse such as toilet flushing, with the water quality criteria being: <1 nematode egg l^{-1}, 0 *E.coli* 100^{-1} ml^{-1}, SS <10 mg l^{-1} and turbidity <2 NTU.

In Germany greywater reuse is not widely applied due to risk to hygiene, although research towards greywater treatment for in-building reuse has gained interest and resulted in operating systems on various sites (Section 2.4.2). Guidelines for treated greywater, or 'service water', were introduced on a local level in Berlin (Nolde, 1999), with the key water quality parameters being: 7-day BOD (BOD_7) < 5 mg l^{-1}, total coliforms < 100 cfu ml^{-1}, faecal coliforms < 10 cfu ml^{-1} and *Pseudomonas aeruginosa* < 1 cfu ml^{-1} (Nolde, 1999). Rainwater harvesting is recommended for toilet flushing and garden irrigation (Mustow *et al.*, 1997), encouraged to some extent by the high water charges (Grant, 1998). By 1998 there were around 600 000 rainwater systems in Germany (Grant, 1998). The criteria shown in Table 2.5 limit rainwater recycling so as not to apply to households with occupants with a limited immunity, such as elderly

and children. General conditions of the water supply legislation apply to rainwater systems (Fitsch and Koenig, 2000).

UK

In the UK water reuse is not directly restricted by the legislation though the government has three principle ways of intervening in the process (Dent, 1999). Firstly, water reuse can be encouraged by financial incentives such as funding research programs and favouring the use of water meters and monitoring water costs (e.g. Water Charging Bill 1999). Secondly, the government can act as a building regulator to protect public health. The third way is to raise public awareness by introducing educational campaigns (e.g. 'Are You Doing Your Bit').

Recently the government has taken steps towards safe water reuse. In 1999 the water byelaws were replaced by the new Water Supply (Water Fittings) Regulations (WRAS, 1999a and b) which made the identification of pipework compulsory and cross-connection with potable water illegal. Also greywater and rainwater are now recognised in the new regulation where five fluid categories of risk are identified (Hodges, 1998):

- Category 1 - Water of a quality that is not impaired by any change in taste, colour or odour,
- Category 2 - Water not representing a health hazard but whose aesthetic quality is impaired owing to a) change of temperature and/or b) presence of substance(s) causing a change in taste, odour or appearance,
- Category 3 - Fluid representing a slight health hazard due to presence of substances of low toxicity,
- Category 4 - Fluid representing a significant health hazard due to presence of toxic substances, and
- Category 5 - Fluid representing a serious health hazard due to presence of pathogenic organisms, radioactive or very toxic substances. E.g. greywater and rainwater.

At present water reuse quality standards do not exist in the UK. To meet the needs of the increasing number of recycling systems available in the country, the Building Services Research and Information Association (BSRIA) has proposed guidelines (Mustow *et al.*, 1997) for greywater, stored rainwater, and combined greywater and rainwater reuse systems. Based on the relevant guidelines, the proposed criteria are divided into three categories depending on the degree of human exposure to untreated reused water:

- Category A (high exposure) - drinking, cooking, bathing, irrigation of crops to be eaten raw: UK Water Supply (Water Quality) Regulations apply,
- Category B (medium exposure) - toilet flushing, vehicle and clothes washing, surface landscape irrigation, irrigation of crops to be eaten cooked, impoundments, use in fire protection systems and commercial air conditioners: non-detectable faecal coliforms in 100 ml, and
- Category C (low exposure) - sub-surface landscape irrigation, hand-basin toilets: faecal coliform limit not applicable.

The BSRIA guidelines are currently being revised partly due to the experience gained in water reuse systems and the need for amendments for the safety of plumbing systems and environment.

In 1997-2000 a group of experts representing industry and academia in the UK worked on risk and risk assessment issues related to greywater and rainwater recycling technologies. The Water Recycling Opportunities for City Sustainability (WROCS) project, sponsored by the Engineering and Physical Sciences Research Council (EPSRC), focused on risk identification, process and risk modelling and experimental testing. The team set out to investigate concerns with greywater and rainwater recycling and thereby support the development of water reuse in the UK. Using *Salmonella* as the indicative species, the probability of risk was defined in greywater reuse for toilet flushing or garden irrigation (WROCS, 2000). The following stages of a greywater system were identified in the models: initial

contamination, treatment, storage and regrowth, dose, infection and risk. It was concluded that there was no significant risk with reuse of treated greywater but that the risk is slightly higher if a person (user) is infected. The risk is greater with the use of untreated greywater. These findings highlighted that design, installation and maintenance of a treatment and storage system are critical factors in determining the overall acceptability of risk.

Quality concerns

In the existing guidelines and standards bacteriological quality, biodegrability, clarity and acidity are defined around the world but the actual permitted levels vary considerably (Table 2.5), leading to two ideologies. The first of these is based on the quality of greywater being comparable with its application. In such cases the standards are similar to those for bathing water as the level of risk to user is about the same. The alternative way is to consider greywater treatment in a similar manner to that of municipal effluent. The difference in these ideologies lies in the standard for coliform levels, which for the more pragmatic approach is in the range of few thousand colony forming units per 100 ml whereas the more conservative approach requires a very low or non-detectable level. In the UK concerns on the formation of droplets on adjacent toilet bowl surfaces following flushing have been expressed, such that a more suitable criteria for toilet flushing has been suggested as non-detectable *E.coli* in 1 ml (Dixon *et al.*, 1999a).

The focus on bacteriological quality reflects the potential for human exposure to recovered greywater resulting in the public health protection being the major criterion. Enteric viruses are known to be the most critical group of pathogens as they can cause illness at low doses, cannot be detected by routine microbial analysis and survive during storage. Therefore enteric viruses represent the component that is most difficult to remove. Experiments on poliovirus type 1 seeded into greywater showed that during a 6-day storage at 25°C and 17°C the counts declined by 99% and 90%, respectively (Rose *et al.*, 1991). The survival of enteric bacteria has also been studied. Rose *et al.* (1991) incubated greywater for 2 days at 25°C and observed a 1-2 log

increase in total bacteria standard plate count and faecal coliforms. Similar regrowth of *Salmonella typhimurium* and *Shigella dysenteriae* was not observed, but these species persisted for several days after having been seeded into greywater. Nolde (1999) reported that *Salmonella* seeded in greywater for 7 days and stored in dark at room temperature declined from the initial 10^7 cfu in 100 ml to below non-detectable level in 3 weeks. After the seeding was ceased, faecal coliform counts persisted at around 10^6 cfu in ml for a day before dropping by 4 log over the following 5 days. The differences in survival and regrowth in greywater between pathogens depend on the greywater quality and environmental conditions, as the examples above show. It is usual, however, that a process effective in removing bacteria will be similarly effective for other pathogens (Cooper and Olivieri, 1998).

It is normal to base standards on the more readily quantifiable indicator organisms of faecal or total coliforms. These species demonstrate a potential for disease transmission, rather than an actual risk of illness, but are more familiar bacteriological quality determinants than viruses and are more easily measured. On the other hand, no proven correlation exists between concentrations of indicator species and actual pathogen levels (Cooper and Olivieri, 1998), and some pathogens are known to be more resistant to treatment than the indicator species (Yates and Gerba, 1998). This has resulted in the more conservative approach being adopted in the USA, Japan and Australia where greywater recycling is an established operation. In the USA, specifically in the US EPA guideline for water recycling (EPA, 1992) the condition stipulated is non-detectable faecal coliforms for urban reuse combined with a specification for minimum level of treatment required. The guideline has improved the applicability of surrogate measures such as indicator organisms and is currently being adopted in California and Florida. Currently Arizona is the only US state that has numeric pathogen standards for reclaimed water and Florida has implemented regular monitoring for *Giardia lamblia* and *Cryptosporidium* (York and Walker-Coleman, 2000). The Australian guideline also includes a maximum limit for viruses (Table 2.5).

In many guidelines further requirements for the level of treatment are set by BOD and turbidity concentrations. Failure to reduce the amount of these pollutants in greywater can result in problems associated with regrowth of organic matter and micro-organisms downstream such that subsequent treatment of the product water by, for example, disinfection (Section 2.4.3) may be affected. Oxygen demand may cause odour problems and also has an impact on greywater biodegradability (Section 4.2.3). Turbidity is a largely aesthetic parameter. This is recognised in several water quality standards as recommendations for turbidity in treated water are similar in most standards shown in Table 2.5, with the exception of the US National Sanitation Foundation where a significantly higher turbidity is accepted.

2.4 Process technologies

2.4.1 Introduction

The process technologies for greywater reuse presented in Sections 2.4.2-2.4.6 can be divided into five categories depending on the level and type of treatment (Table 2.6). The differences in operational principles, cost and scale of use may in some cases limit the implementation of such systems. Concerns of the technology used often result from failed attempts to adequately treat greywater.

Table 2.6. A summary of greywater treatment processes.

Treatment process	Principle	Advantages	Disadvantages
Natural systems (Section 2.4.2)	Reed beds: plants grown in bed through which wastewater flows	Inexpensive Energy-efficient Chemicals not required for treatment	Located outdoors Large footprint Climate-dependent Product water quality variable
Basic two-stage systems (Section 2.4.3)	Coarse filtration and disinfection	Located indoors Suitable for small scale applications	Use of chemicals Low product water quality Long payback time
Chemical processes (Section 2.4.4)	Photocatalytic oxidation	Located indoors Small footprint Removal of turbidity, organic matter and efficient disinfection Potentially suitable for small scale applications	No removal of solids High capital cost
Physical and physiochemical processes (Section 2.4.5)	Filtration (membrane, sand filter etc.), chemical coagulation, disinfection	Located indoors Suitable for large scale applications Very high quality product water Not susceptible to chemical shocks Easy to scale up	High capital cost Cost of membranes and chemicals Chemicals required for membrane cleaning in membrane process May be susceptible to hydraulic shocks
Biological processes (Section 2.4.6)	Biological treatment of wastewater (suspended or attached growth)	Small footprint Located indoors or outdoors Suitable for large scale applications Chemicals not required for treatment Very high quality product water Easy to scale up	High capital cost, especially membrane bioreactors (MBRs) Chemicals required for membrane cleaning in MBRs May be susceptible to shock loading of organic matter and bactericidal chemicals

2.4.2 Natural systems

Artificial or constructed wetlands (reed beds, lagoons, ponds) are increasingly popular for treatment of secondary and tertiary domestic wastewater and stormwater. Wastewater is treated in horizontal or vertical flow reed beds, where the water is below the surface of a gravel bed to minimise undesired insect breeding and odour formation. Some flora have bactericidal properties (Batchelor *et al.*, 1990) and have a capacity to treat chemical pollutants. Common monoculture plants include *Phragmites*, *Baumea*, water hyacinth (*Eichhornia crassipes*), *Typha* and *Schoenoplectus*. A wider variety of organisms that recycle the nutrients and have a high growth as well as a rapid response to shock loading can also be grown in ponds (Mars *et al.*, 1999). The natural water levels, which have an effect on flowering and seed production, can be optimised to enhance treatment performance. Wastewater nutrient levels, seasonal temperature variation and flora characteristics determine the size of the pond and infiltration areas, with values of 0.7 m^2 (Green and Upton, 1995), 1 m^2 (Mars *et al.*, 1999), 5 m^2 (Cooper, 1990) and 5-8 m^2 (Bucksteeg, 1990) per person all quoted in literature for the latter depending on the wastewater characteristics and the target effluent quality. Odour formation can result from poor oxygenation, rather than organic overload, which then has an impact on ammonia concentration (Bonvillain *et al.*, 1998). Odour problems can generally be ameliorated through improved aeration, light and temperature.

Greywater is commonly treated by natural systems in areas without a public sewer system. Fittschen and Niemczynowicz (1997) reported a 100 population equivalent (pe) greywater treatment scheme in Sweden, which included a sedimentation tank, a reed bed and a sand filter followed by an artificial pond. *Phragmites communis* were planted on an area of 600 m^2 with a depth of 0.6 m to give a residence time of 4 days. This increased to 14 days in practice as only a quarter (10.7 ± 1.2 m^3 d^{-1}) of the design volume of the reed bed was used. BOD_7, COD, TN and total phosphorus (TP) removals of 97%, 87%, 59% and 64%, respectively, were achieved by the reed bed treatment, as well as a 2-4 log reduction in total coliforms. Further polishing by a sand

filter decreased both TN and TP to below 0.5 mg l^{-1}, and coliforms to a maximum of 2 cfu in 100 ml.

Lightly polluted handbasin and kitchen wastewater from a 500 pe college is currently being treated in a 1200 m^2 'wetpark' in southern Sweden (Günther, 2000). The a turnover time for the water is one year due to low winter temperatures. A triplicate shore-pond system comprises alternating shore purification zones and buffer ponds where a number of flora species were planted. The insect larvae population and leaf litter in the ponds is controlled by fish. After the third pond the water is filtered through a sand filter prior to reuse for handwashing and toilet flushing. The annual water use in the building is about 400 m^3. Early results show that a 3-4 log removal of faecal streptococci and thermostable coliforms is achieved, and the average BOD_7 of 47 mg l^{-1} is reduced to around 1 mg l^{-1}. A 99.5% P removal is also achieved by this pond treatment. Nitrogen removal (50%) is lower than the designed level possibly due to a leakage from the plant soil on the shore zones.

2.4.3 Basic two-stage process

The most common technology used for small-scale domestic reuse in the UK is a two-stage process based on coarse filtration followed by disinfection. This system forms the basis of a number of commercially-available products for installation in new buildings or retrofitting into existing ones. The residence time of the process is kept short to avoid alteration in the chemical nature of greywater and so reduce the complexity of treatment. The coarse filter usually comprises a foam blocker and a metal strainer. Christova-Boal *et al.* (1996) noted that filters require regular cleaning at least once a week independent of the source of greywater. Solid matter such as hair, lint and dirt captured by the filters may pose a risk to the person cleaning the filters despite similar protective measures such as wearing gloves, suggesting that disposable filters may be a preferable choice. Disinfection is achieved using chlorine (Cl_2), bromine (Br_2) or iodine (I_2) dispensed in slow release blocks or dosed as a liquid solution. A case study of a basic two-stage system is presented in Section 2.5.2.

The basic two-stage system is designed to meet the less stringent reuse standards analogous to those for bathing water. As the treated water remains high in organic load and turbidity, the effectiveness of the chemical disinfection stage is limited. This is because organic matter in the water imparts a disinfection demand, raising the amount of disinfectant required to give a free residue, which is required by some of the water quality standards (Table 2.5). Chlorine by-products such as chloramines and trihalomethanes are then generated which adversely affect human health (Olivieri *et al.*, 1998). They act as disinfectants themselves but are less active than chlorine.

Other options to disinfect treated domestic wastewater are UV light and ozone (O_3). The former has proven to be effective in destroying bacteria and viruses in wastewater, whilst forming no toxic by-products (Loge *et al.*, 1998). Its efficiency is influenced by turbidity and suspended solids in the water as well as the intensity and the age of the UV lamp. A rapid kill of bacteria and improved biodegradability of the water can be achieved by O_3 treatment, which also effectively reduces odour and colour in the water. This technology is often used for potable water treatment though examples also exist in large-scale domestic recycling schemes (Murakami, 1989). Like UV, ozone treatment is affected by turbidity, though not to the same extent, and presents problem in open systems due to its substantial toxicity (Loge *et al.*, 1998). Both disinfection methods have the disadvantage of leaving no residual in the treated water.

2.4.4 Chemical processes

Recalcitrant compounds can be oxidised by hydrogen peroxide in the presence of heterogeneous catalyst. Transition metal-based material, oxygen, air or ozone may act as a catalyst in the oxidation reaction (Pak and Chang, 1998).

Recent research (Parsons *et al.*, 2000) on photocatalytic oxidation using titaniumoxide (TiO_2) activated with UV light has shown potential of the process for greywater treatment. In the study reported adsorption of organic carbon on to a TiO_2

(10 mg l^{-1}) catalyst resulted in over 80% total organic carbon (TOC) removal and a further 10% was removed by the combined TiO_2 and UV process. The TOC removal and oxidation rate both appeared to improve at increased loading rates, suggesting an improved capability for shock loads of organics compared to a biological process. Though solids in the greywater were not removed, a 6 log total coliform reduction was achieved by TiO_2/UV process at a retention time below 30 minutes. The results suggest that chemical processes may be more suited to small systems with a variable organic loading than many biological processes.

2.4.5 Physical and physiochemical process

The treated water from physical and physiochemical processes is of a higher quality (Table 2.7) than that from the previously described basic two-stage system (Section 2.4.3). These advanced processes comprise a sand filter and/or a membrane filtration, usually combined with an appropriate pre-treatment.

Table 2.7. Performance of sand filter and membrane filtration process (tubular MF/UF* operated at up to 2.0 bar) for greywater treatment (Holden *et al.*, 1998).

	BOD_5 (mg l^{-1})	COD (mg l^{-1})	Turbidity (NTU)	Total coliform (cfu 100^{-1} ml^{-1})
Influent	33.3	143	44.5	-
Post sand filter	12.3	35.7	32.3	-
Post membrane	4.7	22.2	0.34	(0 *E.coli*)

* UF= ultrafiltration

Physical processes achieve a reasonable decrease in organic pollutant load and turbidity of greywater. Thus, compared to the two-stage systems, the aesthetic quality of the treated water is improved and problems associated with subsequent downstream disinfection, as encountered in the coarse filtration systems, substantially reduced. However, simple filtration based on fibrous (cloth) or granular depth filters presents no absolute barrier to suspended matter. This results in very substantial

coliform breakthrough and a propensity for solids unloading whenever hydraulic shocks occur.

Membrane systems offer a permanent barrier to suspended solids particles greater than the size of the membrane material, which can range from 0.5 μm for microfiltration membranes down to molecular dimensions for reverse osmosis (RO). The treated water is thus generally very low in turbidity and below the limit of detection for coliforms.

The key technical limitation of membrane systems is that imposed by fouling of the membrane surface by pollutants. This increases the hydraulic resistance of the membrane, commensurately increasing the energy required for membrane permeation and/or decreasing the permeate flux. Membrane fouling can occur through several physiochemical and biological mechanisms and is intensified in low-turbulence systems by concentration polarisation, which increases the concentration of foulants on the membrane surface (Wiesner and Aptel, 1996). Fouling by individual components tends to be specific to the membrane material and application, but organic fouling, unrelated to biological growth, can be attributed to proteins and colloidal and particulate matter present in wastewater. Proteins occur both in the dissolved and colloidal form and foul via several complex mechanisms. The main foulants in membrane bioreactors (MBRs) are the extracellular polymer substances (EPS) excreted from cells (Stephenson *et al.*, 2000). At high concentrations EPS substantially increase the hydraulic resistance of the fouling layer on the membrane surface (Nagaoka *et al.*, 1999), demanding a rigorous cleaning cycle.

Membrane fouling can be suppressed by operation at a lower membrane flux, but this increases the membrane area requirement and adds substantially to capital cost as a result. The foulant layer can be removed by vigorous cleaning, which then increases the operational cost as well as imparting an undesirable chemical load on the waste stream. Membrane fouling may also be suppressed by promoting turbulence to limit

of thickness of the hydrodynamic boundary layer, but this exerts an energy demand and so increases operational costs.

Problems of both inadequate permeate water quality and ineffective membrane cleaning have been reported (Holden *et al.*, 1998; Le Clech *et al.*, 2000) with membrane systems treating greywater. The greywater hold-up time of the system has been identified as the principal cause of these problems. Over extended time periods the greywater can become anaerobic, resulting in the generation of organic components which are less readily rejected by the membrane (Holden *et al.*, 1998). Membrane trials (Holden *et al.*, 1998; Le Clech *et al.*, 2000) have demonstrated that purely physical systems fail to reject all coliforms from the waste stream. This has been explained in terms of protein migration through the membrane pores which appear to aid the transport of coliform species (Judd and Till, 2000). A case study of a physical process is presented in Section 2.5.2.

Coagulation, an example of a physiochemical process, is widely used in water and wastewater treatment wherein aluminium and iron salts are used to improve the removal of colloidal solids and organic carbon by promoting the aggregation of suspended solids. It has recently been applied to greywater treatment. Parsons *et al.* (2000) achieved 97% and 88% removal of turbidity by aluminium sulphate and ferric chloride coagulation at the respective optimum doses of 60 mg l^{-1} and 100 mg l^{-1} at pH 7-10. TOC was significantly reduced by 70-85% over a coagulant dose range of up to 100 mg l^{-1}. At these high concentrations, the coagulant precipitates to form a voluminous solid which entraps particulate/colloidal solids and adsorbs dissolved matter. Significant sludge production results, which limits the efficacy of this process.

2.4.6 Biological process

2.4.6.1 Introduction

Filtration itself is not sufficient to guarantee an adequate reduction in organic contamination so as to prevent biological regrowth in distribution systems (Section

2.3). Biological treatment is required to remove biodegradable material especially for systems that include large distribution networks such as hotels or community-based centralised recycling schemes. The benefits of biological and physical treatment are combined in advanced processes such as membrane bioreactors and biological aerated filters (BAFs), which are small footprint processes capable of producing high-quality effluents. These processes are of a modular design such that scale-up of a treatment plant is relatively easy. The suitability of a biological process for a greywater reuse system is strongly influenced by its capability to remove both microbiological and biological contaminants.

2.4.6.2 Sequencing batch reactors

Sequencing batch reactors (SBRs) can be used in series with reed beds or on their own. An SBR is a suspended growth process with an anaerobic primary chamber followed by an aerobic chamber. These systems are used in areas without sewer network for primary or secondary treatment as small wastewater treatment plants, that are generally less than 2000 pe though plants ten times the size exist (Helmreich *et al.*, 1998). In a recent work (Shin *et al.*, 1998) a pilot scale SBR was set up for the removal of organic and nitrogen compounds from greywater collected from an office building. An equalisation tank of 2.5 m^3 buffered the incoming greywater such that the influent flow to the SBR was 2 m^3 d^{-1}. Effluent with 20 mg l^{-1} sCOD, 5 mg l^{-1} sBOD and 0.5 mg l^{-1} ammonia was produced during the cyclic mode operation. Prior to reuse turbidity and coliforms were removed by microfiltration (0.2 μm).

2.4.6.3 Rotating biological contactors

Rotating biological contactors (RBCs) are also commonly used as package plants in rural areas (Griffin and Findlay, 1998). The fixed-film process comprises an aerated tank in which a partly immersed disk rotates, allowing biofilm formation on the media. Although usually employed for sewage treatment, RBCs have also shown to be effective for greywater purification. In Germany a greywater treatment plant of 70 pe has been in operation for 10 years and has provided water for toilet flushing in a multi-storey building (Nolde, 1999). The process, situated in a 15 m^2 basement,

comprises a sedimentation tank followed by a four-stage RBC and final UV disinfection. The greywater BOD_7 of 50-250 mg l^{-1} is reduced to below 5 mg l^{-1} and the bacteriological effluent quality mostly meets the 'service water' quality guidelines (Section 2.3). A number of similar processes have been in operation for several years in housing estates and hotels in Germany. One of them is a 400-bed hotel where a six-stage RBC process has been operational since 1996 (Clarke, 1998). Located in the basement, it comprises four parallel anaerobic primary tanks followed by two RBCs and has a maximum output of 24 m^3 d^{-1}. The first primary tank, which has a 7-hour residence time, is desludged daily since most of the solids settle out in this stage. The other anaerobic tanks are desludged 2-3 times a week. The greywater is passed through lamella plate settling tanks and UV disinfection prior to entering service water tank.

2.4.6.4 Membrane bioreactors

The first combination of membranes with biological wastewater treatment 30 years ago (Smith *et al.*, 1969) led to the development of three generic membrane processes for biological treatment. The solid-liquid membrane separation bioreactors employ either submerged (Figure 2.2 a) or side-stream (Figure 2.2 b) bioreactors with UF or MF modules for the retention of biomass for recycle to the bioreactor. Gas permeable membranes are used to provide bubbleless oxygen mass transfer to degradative bacteria present in the bioreactor (Figure 2.2 c). An extractive membrane process has been designed for the transfer of degradable organic pollutants from hostile industrial wastewaters (Livingston, 1994). Here the first two MBR types are briefly discussed.

Solid-liquid membrane separation

In the solid-liquid membrane separation bioreactors a high biomass concentration and retention of high molecular weight compounds is maintained resulting in complete mineralisation of influent organic matter (Chiemchaisri and Yamamoto, 1993 and 1994). As a result of the membrane separation process solids retention time (SRT) is independent of hydraulic retention time (HRT). MBRs of this type, by far the most common, are most attractive for situations where long solids retention times are

necessary to achieve the removal of pollutants (Knoblock *et al.*, 1994). Due to high biomass concentrations, a high minimum maintenance energy is needed, in addition to energy for biosynthesis and cell growth. High oxygen concentrations are required in aerobic membrane systems to ensure continuous biosynthesis and cell growth. Maintaining a low food to micro-organism (F:M) ratio in the reactor results in minimum sludge wastage, reduced plant size, and the development and retention of waste specific micro-organisms (Aya, 1994; Chiemchaisri *et al.*, 1992; Smith and Scott, 1995). At steady state these systems can remove organic pollutants over a wide range of conditions, producing a high-quality permeate at high organic loading rates.

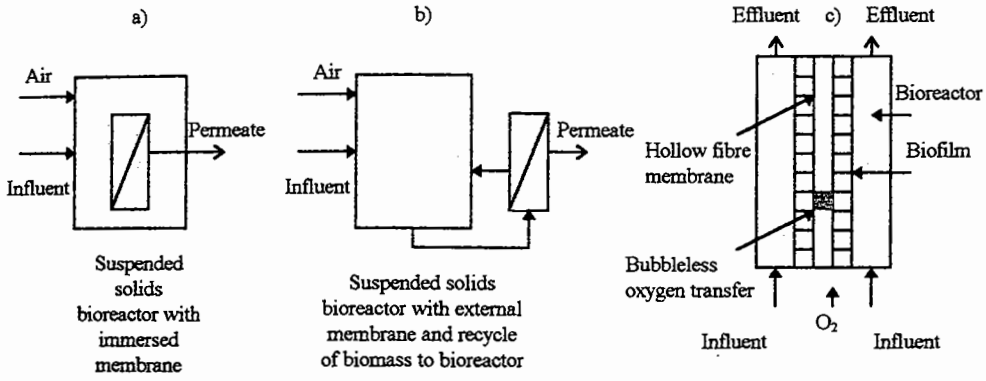


Figure 2.2. Membrane bioreactors: a) submerged MBR, b) side-stream MBR and c) membrane aeration bioreactor.

Membrane flux is influenced by a number of factors: suspended solids, temperature, cross-flow velocity, transmembrane pressure (TMP) drop, surface fouling and the extent of concentration polarisation (Cadi *et al.*, 1994; Ishiguro *et al.*, 1994; Sutton *et al.*, 1994). MBRs have been operated at a range of flux rates from around 10 to 250 l m^{-2} h^{-1} (Table 2.8), though both lower (5 l m^{-2} h^{-1}; Gander *et al.*, 2000) and higher (300 l m^{-2} h^{-1}; Krauth and Staab, 1993) values have been reported. Side-stream MBRs are generally operated at higher flux rates than submerged systems due to operation at higher pressures (Table 2.8). Due to this the specific flux rate, i.e. the flux per unit pressure, is often lower in side-stream systems than in submerged configurations. Examples of solid-liquid separation MBRs are listed in Table 2.8.

Table 2.8. Examples of aerobic MBR performance in domestic/municipal wastewater treatment.

Type	Volume (m^3)	Pore size (μm)	Flux ($l\ m^{-2}\ h^{-1}$)	HRT (h)	MLSS ($kg\ m^{-3}$)	OLR ($kg\ m^{-3}\ d^{-1}$)	Influent concentration ($mg\ l^{-1}$)	Removal (%)	Reference
HF/S	2.1	0.4	16.6	n/a	27.9	n/a	98[a]	>99[a]	Murakami *et al.* (1999)
PF/S	n/a	0.4	20.8-69.4	7.6-11.4	12-18	0.32-0.63[a]	176[a]	99[a], 96[b]	Ishida *et al.* (1993)
HF/S	1.5	n/a	n/a	6-15	<23.6	1.96[b]	490[b]	>96[b]	Chiemchaisri *et al.* (1992)
HF/S	0.04	0.1	10-250	10-15	6-14	n/a	127[a], 343[b]	99[a], >96[b]	Nah *et al.* (2000)
HF/S	3.12	0.4	16.6	13	12.9	0.245[a]	133[a]	99[a]	Ueda and Hata (1999)
S	n/a	0.2	40-200	4.98	16	0.36[a], 1.76[b]	115[a], 365[b]	>91[a], >99[b]	Peters *et al.* (1999)
T/SS	0.61	0.2	34-170	7.4-14.8	40-50	0.9-2.0[b]	130[b]	>99[b]	Müller *et al.* (1995)
HF/S	21.4	0.1	12.1-80.5	13	12	0.315[a]	133[a]	>99[a]	Ueda *et al.* (1996; 1997)
PF/S	n/a	0.4	n/a	n/a	18	n/a	216[a], 538[b]	>97[a], >95[b]	Churchouse (1997); Judd (1997)
HF/S	n/a	0.02	35-70	2	5-15	2.24[a], 4.27[b]	187[a], 356[b]	>97[a], >95[b]	Côté *et al.* (1997)
HF/S	1	200 000[c]	n/a	4.8	16	2.3[b]	457[b]	>96[b]	Côté *et al.* (1998)
	1	200 000[c]	n/a	6.5	21	1.7[b]	457[b]	>97[b]	
	1	200 000[c]	n/a	9.2	26	1.2[b]	457[b]	>97[b]	
PF/SS	0.0045	n/a	n/a	1-8	12.8	0.45-1.5[b]	550[b]	>94[b]	Chaize and Huyard (1990)
T/SS/C	1	0.1	60-80	24	2.5[d]	0.18[a], 0.49[b]	182[a], 488[b]	>94[a], >95[b]	Trouve *et al.* (1994)
T/SS/C	0.13	0.1	73-104	n/a	3.7	0.67[b]	410[b]	87[b]	Ghyoot *et al.* (1999)
	0.13	0.1	77-154	n/a	20.5	3[b]	7025[b]	>97[b]	

n/a not available [a] BOD [b] COD [c] Daltons [d] volatile

HF= hollow fibre, PF= plate and frame, T= tubular, S= submerged, SS= side-stream, C= ceramic, WC= woven cloth

By the mid-1990's there were 51 plants treating up to 160 m^3 d^{-1} of domestic wastewater from large buildings in Japan (Aya, 1994). MBRs have been successfully employed in Japan for greywater recycling in office blocks and residential buildings (Ishida *et al.*, 1993). A case study of an MBR treating greywater is presented in Section 2.5.2.

An example of an MBR process is the Kubota submerged membrane bioreactor, which has been successfully used for several years at many sites in Japan to treat domestic and industrial biodegradable wastewater (Ishida *et al.*, 1993). The first pilot plant became operational in 1989, and the first commercial operational plant was installed in 1991 (Churchouse, 1997). By the late 90's there were over 40 Kubota processes in Japan and two in the UK (Churchouse, 1997). The installations vary widely in capacity from 16 to 110 m^3 d^{-1}. At the Higashi plant, Hiroshima, the volumetric loading rate is 0.32-0.63 kgBOD m^{-3} d^{-1}.

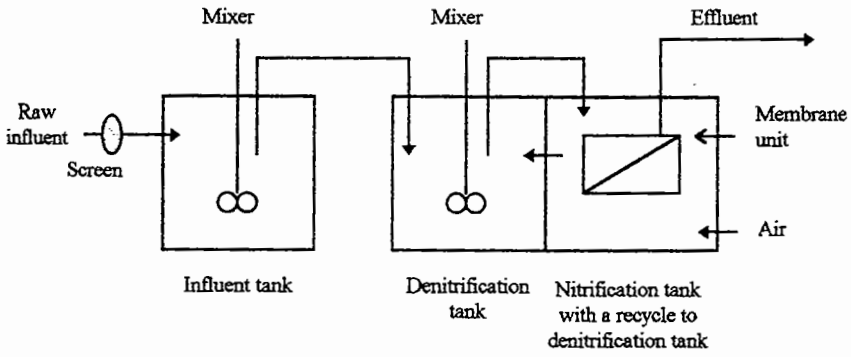


Figure 2.3. Operating principles of Kubota submerged membrane bioreactor.

Screened, degritted wastewater is passed to an aerated equalisation tank followed by an optional denitrification tank and a fine screen (Figure 2.3). The membranes are submerged in the nitrification tank and a part of the mixed liquor is returned from the nitrification tank to the denitrification tank. The final effluent is collected by a permeate suction pump. Excess sludge is removed through a valve, which is located between denitrification and nitrification tanks. The polyolefine-based membranes have an average pore size of 0.4 μm and are of flat plate geometry. These are submerged in the bioreactor and the permeate is withdrawn under suction (-0.3 bar).

The flat-plate membrane cartridges are stacked in a box-like module. The number of membranes in one module varies from 24 to 140. Solids and liquids are completely separated by the submerged membrane, such that high mixed liquor strengths are possible. Eutrophic salts such as nitrogen and phosphorus, as well as organic pollutants, can be simultaneously effectively removed (Table 2.9).

Table 2.9. Influent and effluent quality and operational parameters of the Kubota submerged membrane process at Higashi plant (Ishida *et al.*, 1993).

Water quality parameter	Influent	Effluent	Operational parameters
SS (mg l^{-1})	153	<1	Flux 25-30 l m^{-2} h^{-1}
BOD (mg l^{-1})	176	1.5	MLSS 12 000-18 000 mg l^{-1}
COD_{Mn} (mg l^{-1})	79	6	Volumetric BOD loading 0.32-0.63 kg m^{-3} d^{-1}
TN (mg l^{-1})	29.6	5.4	TN-MLSS loading 0.004-0.007 kg kg^{-1} d^{-1}
NH_4^+-N mg l^{-1}	22.4	0.1	HRT 7.6-11.4 h
NO_2^--N (mg l^{-1})	0.2	4.6	SRT 25-40 d
NO_3^--N (mg l^{-1})	0.1	0.1	Temperature 18.5-22.0°C
TP (mg l^{-1})	3.7	1.2	
Cl^- (mg l^{-1})	51.5	53.6	
pH (-)	7.3	7.1	
E.coli and heterotrophes	-	n.d.*	

* n.d. = not detected

The Kubota system is possibly the most extensively tested in-building water recycling system in operation: some plants have now been running for ten years or more without significant problems. Operational data from Japanese plants indicate that average membrane lifetime exceeds 5 years (Churchouse, 1997), reducing the frequency of membrane replacement. The plants produce 30% less sludge than a conventional activated sludge plant and, because no sludge concentration tanks or pre-sedimentation is needed, the plant size is considerably less than that of a conventional scheme. Running costs are also reasonable, but the capital cost is comparatively high because of the requirement for a relatively large membrane area to allow low-flux operation (about 25 l m^{-2} h^{-1}) so as to avoid permanent membrane fouling. MF membranes usually operate at fluxes in excess of 80 l m^{-2} h^{-1}, about three times that of the Kubota system (Table 2.9). On the other hand, the transmembrane

pressure is consequently very low, and almost all of the running cost is associated with the aeration system.

Oxygen mass transfer membranes

Oxygen mass transfer using a synthetic membrane to provide bubbleless aeration for the biological treatment of primary sewage, synthetic sewage and brewery effluent has been demonstrated at a laboratory scale (Hirasa *et al.*, 1991; Kniebusch *et al.*, 1990). Bubbleless oxygen mass transfer can be accomplished using gas permeable dense membranes or hydrophobic microporous membranes (Côté *et al.*, 1988). COD removal of 63-91% at 0.06-8.94 kgCOD m^{-3} d^{-1} has been reported (Debus and Wanner, 1992; Pankania *et al.*, 1994; Timberlake *et al.*, 1988; Yeh and Jenkins, 1978). Since no oxygen bubbles are formed, gas stripping of volatile organic compounds and foaming due to the presence of surfactants can be prevented (Rothemund *et al.*, 1994; Semmens, 1991; Wilderer *et al.*, 1985). Both plate and frame and hollow fibre membrane configurations have been used though research has focused on hollow fibres, with the gas phase on the lumen side and the wastewater on the shell side of the fibres. These provide a high surface area for oxygen transfer and biofilm formation while occupying a relatively small volume within the bioreactor. Since lumen gas partial pressure is independent of tank depth, large transfer driving forces in shallow tanks can be achieved (Côté *et al.*, 1988). Examples of membrane aeration bioreactors (MABRs) performance are shown in Table 2.10.

Table 2.10. Examples of MABR performance.

Influent	Loading rate (kg m^{-2} d^{-1})	Pollutant concentration (mg l^{-1})	% removal	Reference
Primary sewage	TOC 0.003-0.011 organic N 0.011 NH_4-N 0.001-0.002	70-92 17-27 14-30	33-50 55-75 n.g.	Timberlake *et al.* (1998)
Synthetic	BOD 0.011 TOC 0.007	200 114	n.g. 95	Yamagiwa and Ohkawa (1994)
Brewery effluent	tCOD 0.068 susCOD* 0.013	1782 ± 40 343 ± 49	83 84	Brindle *et al.* (1999)

n.g. not given, * suspended COD

Research needs

Application of MBR technology makes it possible to recover valuable components from effluent streams, reuse contaminated process water, and provide the means for the development of pollutant-specific microbial populations within the bioreactor (Diels *et al.*, 1993; Livingston, 1994). Though commercial size membrane separation bioreactors exist, there remains a need to investigate methods to maintain biomass viability, reduce salt accumulation in the bioreactor and develop cheaper and/or more fouling-resistant membranes in order to make the technology more attractive.

2.4.6.5 Biological aerated filters

The first types of contact aerators for sewage treatment, comprising aerated tanks and layers of slate for biofilm attachment, were introduced in 1913 (Clark, 1930). Problems with clogging media resulted in research in different medium materials such as cotton, veneer and copper gauze (Buswell and Pearson, 1929). In the late 1970's and early 1980's increasing interest in novel fixed-film reactors lead to the development of granular and structured media biological aerated filters where solids removal is combined with a fixed-film biological reactor (Pujol *et al.*, 1994; Stensel and Reiber, 1983). Thus, unlike MBRs, they present no absolute barrier to suspended material and so do not substantially disinfect water. BAFs have shown to be comparable in performance to other secondary and tertiary treatment methods and in many cases superseded established processes (Stensel and Reiber, 1983). In comparison to conventional treatment processes, BAF operation can be automated and is unaffected by sludge settleability. High effluent quality is achievable even at high organic or hydraulic loadings.

In BAFs the pollutants are removed by different mechanisms depending on the pollutant type. Solids removal is achieved mainly through filtration (Stensel *et al.*, 1988; Ryhiner *et al.*, 1994) and is dependent on the properties of the captured solids, the media support used, the biofilm structure and the hydraulic characteristics of the reactors used (Arvin and Harremoes, 1989). Carbonaceous matter is removed by solids filtration, adsorption and oxidation (Stensel *et al.*, 1988).

Media selection affects BAF reactor performance through variation in size, shape, density and roughness (Smith *et al.*, 1999a). Media materials range from various types of plastic to stone and clay. It has been suggested that large media (>6 mm) should be used for roughing, intermediate size media (3-6 mm) for general treatment and fine media (<3 mm) for effluent polishing and/or for tertiary treatment (Quickenden *et al.*, 1992). For some BAF types a media size of 3-6 mm is recommended when the BAF is used exclusively for BOD removal and a slightly smaller media (2.5-5 mm) when the process is used primarily for nitrification (Smith and Brignal, 1996). Backwashing is required to remove excess biological growth and accumulated solids. The reactor can be of either upflow or downflow configuration, depending on the direction of the influent flow, and can be used with or without aeration (Grasmick *et al.*, 1984) (Figure 2.4).

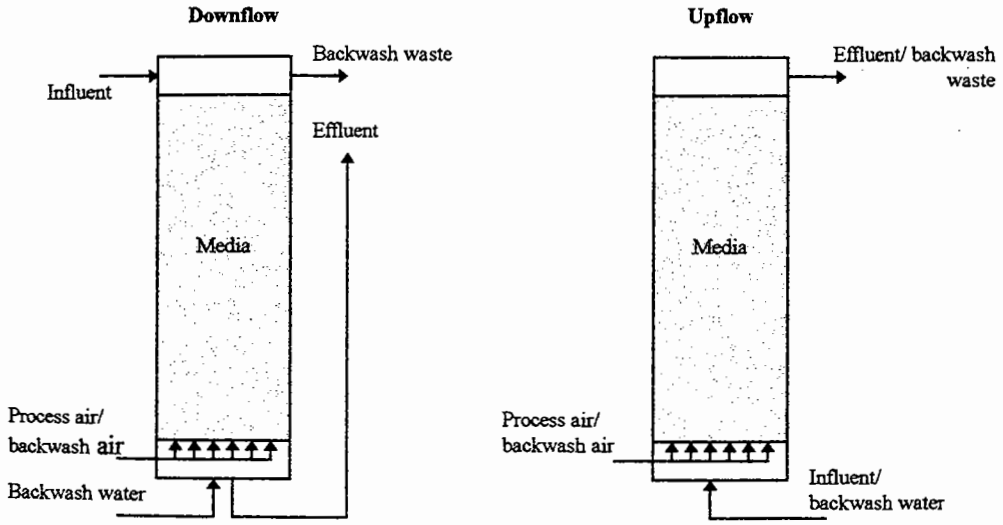


Figure 2.4. Down- and upflow configurations of a granular biological aerated filter.

Most commonly the BAFs are used for removal of carbonaceous material but they can also be used exclusively for nitrification. Hybrid BAFs combine both treatments, i.e. BOD and ammonia removal. Some applications offer denitrification, and the most recent advances in BAF design have been in phosphorus removal (Goncalves *et al.*, 1994; Sagberg *et al.*, 1992). BAF process employed for this duty demand lower coagulant doses for precipitation of phosphorus than reactors which are not designed for phosphorus removal (Sammut *et al.*, 1992).

Table 2.11. Examples of BAF performance in domestic wastewater treatment

Reactor configuration	Plant size (m^3)	HRT (h)	Organic loading rate ($kg\ m^{-3}\ d^{-1}$)	Influent concentration ($mg\ l^{-1}$)	Removal (%)	Reference
Upflow	Full scale	n/a	3.7[b]	n/a	86[b]	Carrand *et al.* (1990)
Downflow	0.141	0.4-0.76	2.5-4.6[a]	424[a]	>90[a]	Dillon and Thomas (1990)
Downflow	0.14	0.5	10.5[b]	<200[a]	~55[b]	Rogalla *et al.* (1990a)
Downflow	Full scale	n/a	7.5[b]	354[a]	75[b]	Rogalla et al. (1990b)
	6 cells, 54 m^3 each					
Downflow	0.2-0.3	0.4-0.6	<15[b]	324[b]	81-85[b]	Rogalla and Bourbigot (1990)
Downflow	0.019	n/a	8-10[b]	350[b]	90[b]	Bacquet *et al.* (1991)
Upflow and downflow	Full scale	n/a	0.5-8.0[b]	35-607[b]	55-85[b]	Pujol *et al.* (1992)
Downflow	Full scale	n/a	0.83-2.65[a]	n/a	93[a]	Smith and Hardy (1992)
Downflow	Full scale	n/a	n/a	6-35[a]	69[a]	Wheale and Cooper-Smith (1995)
	4 cells	1.3	1.5[a]	131[a]	91-94[a]	
Upflow	0.091	n/a	<16.6[b]	300[b]	84-87[b]	Visvanathan and Nhien (1995)
Upflow	Full scale	n/a	4[a]	109-250[a]	>93[a]	Brewer (1996)
	8 cells, 151.2 m^3 each					
Upflow	Full scale, two cells	n/a	1.3[a]	88[a]	85[a]	Rundle (1996)
	Full scale, four cells	n/a	3.7[a]	284[a, c]	91[a]	
Upflow	Full scale	n/a	5.4[a]	n/a	80[a]	Bédard (1999)

n/a not available [a] BOD [b] COD [c] municipal/industrial wastewater

BAFs are primarily employed for sewage treatment (Table 2.11), though a case study relating to greywater treatment is presented in Section 2.5.3.2. They can substitute conventional secondary and tertiary treatment processes or else be retrofitted in upgrading existing sewage treatment works (Budge and Gorrie, 1996; Sagberg *et al.*, 1992). Treatment of high-strength industrial wastewater by BAFs is less common than that of domestic wastewater (Kleiber *et al.*, 1994), though petroleum-contaminated effluents (Bouwer *et al*, 1992; Hamoda *et al.*, 1987), food industry wastewater (Rundle, 1996; Rusten and Ødegaard, 1986) and paper mill effluents (Kantardjieff and Jones, 1997; Rovel *et al.*, 1994) have all been successfully treated.

2.5 Water reuse schemes and economic factors

2.5.1 Introduction

The following case studies provide examples of the application of the various technologies applied to wastewater reuse schemes both abroad and in the UK. Cost is discussed with respect to the treatment technologies in Sections 2.5.2-2.5.3 and more generally in Section 2.5.4. A list of domestic water reuse schemes is presented in Table A.1 in Appendix A.

2.5.2 Abroad

A number of domestic water reuse schemes exist abroad and, as already stated, in arid climate regions direct reuse for non-potable purposes is now common. By the mid-1980's there were already around 380 reuse schemes in California alone (Okun, 1984). Around 840 in-building recycling units based on membrane or biomembrane processes were operating in Japan by the mid-1990's (Aya, 1994). The Japanese district and municipal systems, however, are based on conventional sewage treatment with disinfection. On a new housing development in a district scheme in Sydney, Australia, tertiary treated and disinfected sewage effluent is recycled to houses (Law,

1996). Another example of a large-scale water reuse for toilet flushing is a 12-storey apartment building of 25 000 pe at the Jurong Industrial Estate in Singapore (WERF, 1989).

Case study 1: Irvine Ranch Water District, California, USA

The Irvine Ranch Water District (IRWD) is a full service water and sewer agency serving approximately 120 square miles (Holliman, 1992; Parsons, 1990) and an existing population of 138 000 (Young *et al.*, 1998) in California. In 1987, with the planned intense development of high-rise offices in the area, IRWD began to investigate the feasibility of using reclaimed water in commercial buildings for non-potable uses (Lewinger and Young, 1988).

The 66 000 m^3 d^{-1} reclamation plant provides effluent for all the uses in the district (Young *et al.*, 1998). Wastewater is treated by biological oxidation, in-line chemical coagulation, and dual media filtration followed by disinfection, with all of the processes meeting the requirements of the State of California Department of Health Services' "Wastewater Reclamation Criteria" (Holliman, 1992; Lewinger and Young, 1988).

Since the mid-1960's IRWD has maintained a separate irrigation/reclamation dual system which provides reclaimed and untreated water for irrigation uses (Lewinger and Young, 1988; Young *et al.*, 1994). The reclaimed water contains less than 2.2 coliforms in 100 ml and is thus classified as Type 1 or Class A of Title 22 of the California Administrative Code (Holliman, 1992). In 1991 the district was the first one in the US to obtain health department permits for the use of reclaimed water in interior spaces such as for toilet flushing (Young *et al.*, 1998). Initially reclaimed water was used in two high-rise buildings. By the late 1990's the scheme was extended by connecting two 20-storey high-rise and two low-rise buildings to the dual water supply with five additional high-rise towers awaiting for service (Young *et al.*, 1998).

Water quality was an important factor in pipework and equipment selection. Discolouration to vitreous china fixtures, odour, and corrosivity due to the quality of the reclaimed water were identified as potential problems (Lewinger and Young, 1988). Analyses showed no noticeable difference in colour, corrosivity, and odour between the reclaimed water and fresh water. COD was used as an indicator of odour generation propensity, the maximum measured value being 50 mg l^{-1}. A 'food grade' dye was chosen to differentiate between fresh water and reclaimed water.

At the beginning of the project it was estimated that 70-90% of the water used in the commercial building was used for toilet and urinal flushing (Holliman, 1992; Young *et al.*, 1994). It was further estimated that 80% of the total water used could be reclaimed water if employed for toilet and urinal flushing and landscape irrigation duties. A significant amount of the remaining 20% of fresh water was directed to cooling tower operations, suggesting that a further 10% saving in water use could be made if the cooling tower supply was switched to reclaimed water. Early results of the operation showed that these goals were met as fresh water demand in the high-rise development dropped by 75% (Holliman, 1992) as a result of the recycling system.

The life-cycle cost of supplying reclaimed water to at least half of the high-rise towers in the district was less than purchasing and distributing domestic water over a 50-year period (Lewinger and Young, 1988). It was estimated that a deficiency in water supply would be experienced some time after the year 2000 but that this could be alleviated by expanding the dual distribution system (Lewinger and Young, 1988).

Case study 3: Mori building, Tokyo, Japan

An example of an in-building greywater treatment and reuse scheme is the 500 m^3 d^{-1} Ubis plant in the Mori building in Tokyo, Japan. This plant has been operational since 1986. The greywater is screened prior to the sedimentation tank from where it is passed through another screen to a side-stream MBR. The product water is chemically disinfected prior to reuse for toilet flushing. The 1 h residence time of the greywater in the system is possible due to the high MLSS (20 000 mg l^{-1}) and strong mixing

compared with a conventional sewage treatment plant (Huitorel, 2000). The plant has a capacity of 100 m^3 d^{-1}, and has a 45 m^3 footprint (Huitorel, 2000). The membrane module has a membrane surface area of 34 m^3 within a 6 m^3 bioreactor. The design of an UF module using turbulent generators allows very high permeate fluxes to be maintained over long periods. Over a 45-day period the flux decreases from 120 to 100 l m^{-2} h^{-1}, after which the membranes are regenerated by chemical cleaning. Electrical consumption is about 3 kWh m^{-3} of treated water. The performance of the Mori plant is summarised in Table 2.12. By early 2000 there were more than 45 Ubis systems in buildings and large hotels in Japan (Huitorel, 2000).

Table 2.12. Performance of the Ubis plant in the Mori building (Huitorel, 2000). Mean (range).

Parameter	Influent	Treated water for reuse
COD_{Mn} (mg l^{-1})	89.1 (12-140)	12 (<5-21)
BOD (mg l^{-1})	349 (120-577)	3.7 (1.8-5.5)
SS (mg l^{-1})	96.5 (33-160)	b.d. (b.d.)
n-hexane extract (mg l^{-1})	11.7 (11-12.4)	<1 (<1)
MBAS (mg l^{-1})	6.5 (3.1-9.9)	0.3 (0.2-0.4)
pH (-)	6.6 (5.7-8.5)	6.8 (6.1-7.9)
Temperature (°C)	19.5 (15.5-22.5)	26.3 (21.0-30.5)
Colon bacilli (cfu 100^{-1} ml^{-1})	19 x 10^4 (15 x 10^4-22 x 10^4)	b.d (b.d.)

b.d. = below limit of detection, MBAS = methylene blue active substance

2.5.3 UK

2.5.3.1 Introduction

Water recycling is a fairly new phenomenon in the UK, although the number of water reuse schemes has rapidly risen over the last few years. Several domestic water recycling systems have been installed in new dwellings and the number of such schemes either under construction or in the planning stage is rising. The schemes represent a variety of scales from single house systems to large-scale units for housing estates, office blocks or schools. On the basis of the water matrix treated, the cases can be divided into four categories :

- rainwater,
- rainwater/greywater,
- greywater, and
- blackwater.

Rainwater reuse represents the most significant application. The three other matrix categories are discussed with reference to specific case studies pertinent to the current work, which focuses on greywater and blackwater treatment.

2.5.3.2 Rainwater/greywater reuse

Combined rainwater/greywater recycling systems offer the advantage of water supply even during dry periods, maximising the water supply. Most of the existing schemes shown in Figure 2.5 represent grouped houses or multiple occupancy buildings. Treatment processes range from natural systems to basic two-stage systems and more advanced technologies.

Case study 4: Loughborough University

In 1994 research on the feasibility of recycling greywater and rainwater drained from the roof of a building for toilet flushing was initiated at Loughborough University. Water usage patterns at the halls of residence revealed that water demand was similar to some previous studies and that the quantity of greywater was sufficient for toilet flushing. A survey (Surendran and Wheatley, 1999) conducted prior to the demonstration stage at four universities, three hotels and three recreational centres showed that as many as 96% of customers would accept greywater use for toilet flushing and 70% would invest additional 9.8% of their water bill-equivalent for long-term benefit. Most of the dissenting 4% expressed concerns about the purity and safety of recycled water. During the demonstration stage of the full-scale plant cost was found to be the main concern of the occupants at the halls of residence.

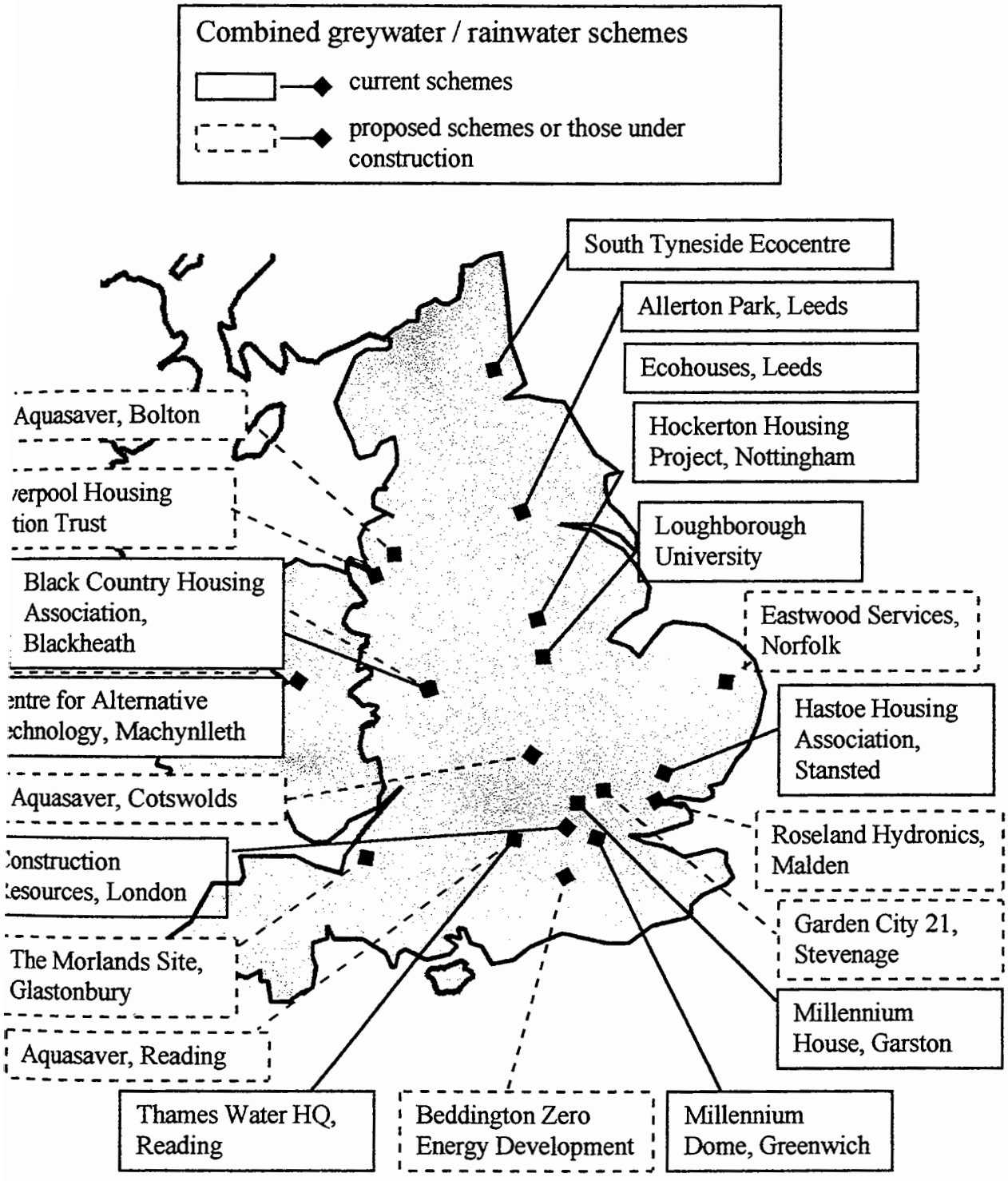


Figure 2.5. A map of the current combined rainwater and greywater reuse schemes in England and Wales (Jefferson *et al.*, 2000a).

A demonstration full-scale plant of 40 pe was built in a halls of residence (Surendran and Wheatley, 1998). The treatment process illustrated in Figure 2.6 comprises four or five stages, the fifth being optional:

1. 1400-litre greywater balancing tank with a filter,
2. anaerobic solids treatment tank with large pore size foam,
3. aerated bioreactor with large pore size foam and beads,
4. active slow filter with small pore size reticulated foam, and
5. activated carbon stage (for potable water quality).

Treated water is collected into two storage tanks: a low-level tank (700 l) attached to the treatment plant and a high-level tank (500 l) connected to the toilets. The low-level tank is equipped with a timer to initiate pumping of treated water to the high-level tank. Excess water is returned to the low-level tank via a return pipe. A standby mains water supply is connected to the high-level tank to ensure adequate water supply for when insufficient water is treated for reuse. Water usage and some water quality determinants are regularly monitored by means of flow meters and on-line monitors.

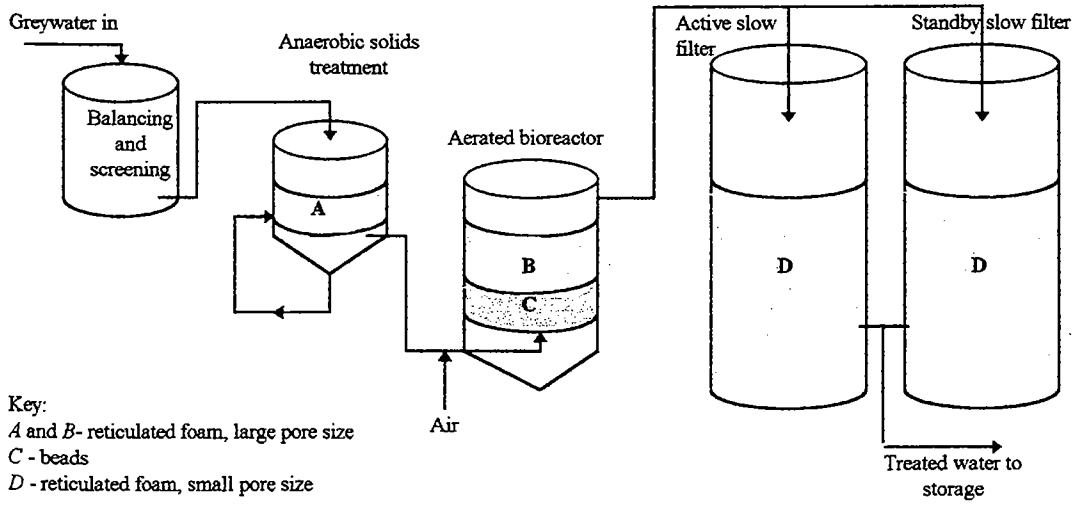


Figure 2.6. Greywater treatment unit at Loughborough University (adapted from Surendran and Wheatley, 1998).

For the long-term trials greywater from 27 occupants was collected and treated. Twelve months of operation demonstrated that the treated water (Table 2.13) met the mandatory limits of both the EC and UK bathing water quality criteria (Table 2.5) in terms of turbidity, BOD_5, faecal coliforms and total coliforms (Table 2.13). Odour problems or sludge blockages were not experienced (Surendran and Wheatley, 1999). Water usage and some water quality determinants are regularly monitored by means of flow meters and online monitors. The unit has been evaluated to have a pay-back time of 8-9 years and a life-span of 20 years.

Table 2.13. Performance data of the demonstration plant at Loughborough University (Surendran and Wheatley, 1999). Mean.

Parameter	Influent	Effluent	Removal (%)
BOD (mg l^{-1})	83.1	4.0	95.2
TOC (mg l^{-1})	34.9	8.3	76.3
Total solids (mg l^{-1})	426.2	361.4	15.2
Total dissolved solids (mg l^{-1})	379.7	356.6	6.1
Total suspended solids (mg l^{-1})	39.4	2.7	93.0
Total volatile solids (mg l^{-1})	189.4	97.4	48.6
Turbidity (NTU)	34.8	1.6	95.4
pH (-)	7.5	7.6	-
Temperature (°C)	22.8	18.7	-
Faecal coliforms (cfu in 100 ml)	942.6	4.8	2 log

Case study 5: Millenium Dome, London

In the largest combined rainwater/greywater scheme in the UK reuse water is collected from three sources (Smith *et al.*, 1999b):

- greywater from the handbasins (120 m^3 d^{-1}) in the toilet blocks,
- rainwater (maximum of 100 m^3 d^{-1}) from the roof (100 000 m^2), and
- groundwater (600 m^3 d^{-1}) from the chalk aquifer below the site.

500 m^3 d^{-1} of reclaimed water is supplied to flush some 400 WCs and 130 urinals in the building (Smith *et al.*, 1999b). The treatment process comprises a BAF process for greywater, hydrogen peroxide (H_2O_2) and granular activated carbon (GAC) for groundwater, and reed bed/lagoon for rainwater. After these treatment stages the waters are combined and filtered through ultrafiltration/reverse osmosis membranes. The product water is disinfected with chlorine and its pH is adjusted prior to reuse.

Greywater quality was assumed to be based on that recorded in previous work (Christova-Boal *et al.*, 1996; Murrer and Bateman, 1998) and trials conducted using the handbasin waters in the Thames Water R&D office complex. Handwash soaps for large-scale public applications were analysed, and most of them were found to be significantly biodegradable. Such soaps were used in the greywater recipe during the evaluation trials.

Pilot trials were carried out using two downflow BAF columns (diameter 150 mm, height 2 m and total bed volume 0.036 m^3) with pulverised fuel ash media. Total BOD (tBOD) of 60 mg l^{-1} in synthetic greywater was reduced to 20-25 mg l^{-1} by the BAF treatment. Post-treatment of this effluent by ultrafiltration, nanofiltration and reverse osmosis membranes resulted in a BOD of 2.3-10.6 mg l^{-1}. This was regarded as a sufficient product water quality for toilet flushing.

As rainwater quality assessment at the Dome was not possible during the plant design stage, roof runoff water quality was established using a report by Gromaire-Mertz *et al.* (1999), which highlighted the significant amount of hydrocarbon and heavy metals. The variation of the runoff water quality was accounted for by the intensity, duration and period between rainfall events. It was recommended that the use of heavily contaminated "first flush" from the roof should be avoided (Smith *et al.*, 1999a). At the Dome the rainwater runoff is treated by two 0.6 m deep reed beds (250 m^2 each, design maximum flow of 100 m^3 d^{-1}) with a 0.5% gradient and an intermediate storage lagoon (400 m^2). The reed beds consist of washed river gravel planted with salt tolerant *Phragmites australis*.

High concentrations of NaCl, hardness, hydrocarbons and hydrogen sulphide gas were found in groundwater. Chemical oxidation by H_2O_2, which does not form by-products, was selected for converting sulphide to sulphate. After a 15-minute contact time with H_2O_2 groundwater is passed through two GAC absorbers (designed hydraulic flow rate of 10 m^3 m^{-2} h^{-1}) containing 5 m^3 of carbon.

The two-stage membrane plant consists of a hollow fibre UF membrane (nominal pore size of 0.01 μm) made of hydrophilic polyether sulphone/polyvinyl pyrrolidone and an RO membrane made of cellulose acetate with a 90% salt rejection. The UF plant (designed feed flow 700 m^3 d^{-1} and a 85% recovery) is backwashed at the end of each filtration cycle and periodically dosed with 100 mg l^{-1} of chlorine to inhibit bacterial growth in the system. Mineral scale and organic membrane fouling are controlled by dosing with acid and caustic. Prior to feeding the UF filtrate to the RO plant it is dosed with sodium bisulphite to remove free chlorine, acidified and dosed with antiscalant. The filtrate from the RO plant (designed feed flow 600 m^3 d^{-1} and a 85% recovery) is disinfected with chlorine followed by pH adjustment to reduce its corrosion potential on plumbing fittings prior to storage.

2.5.3.3 Greywater reuse

Greywater-only schemes in the UK (Figure 2.7) represent a range of treatment technologies from basic two-stage systems to advanced physiochemical and biological processes. The commercially-available basic two-stage systems generally retail at £500-£1000 installed. The relatively low cost of potable water implies that, even at these modest system costs, domestic greywater reuse system incur extended pay-back periods. Retrofitting a greywater recycling system in an existing building for external uses may be in the same price range as for a new housing development, but if such a retrofitted system is employed both for external and internal uses it becomes more expensive.

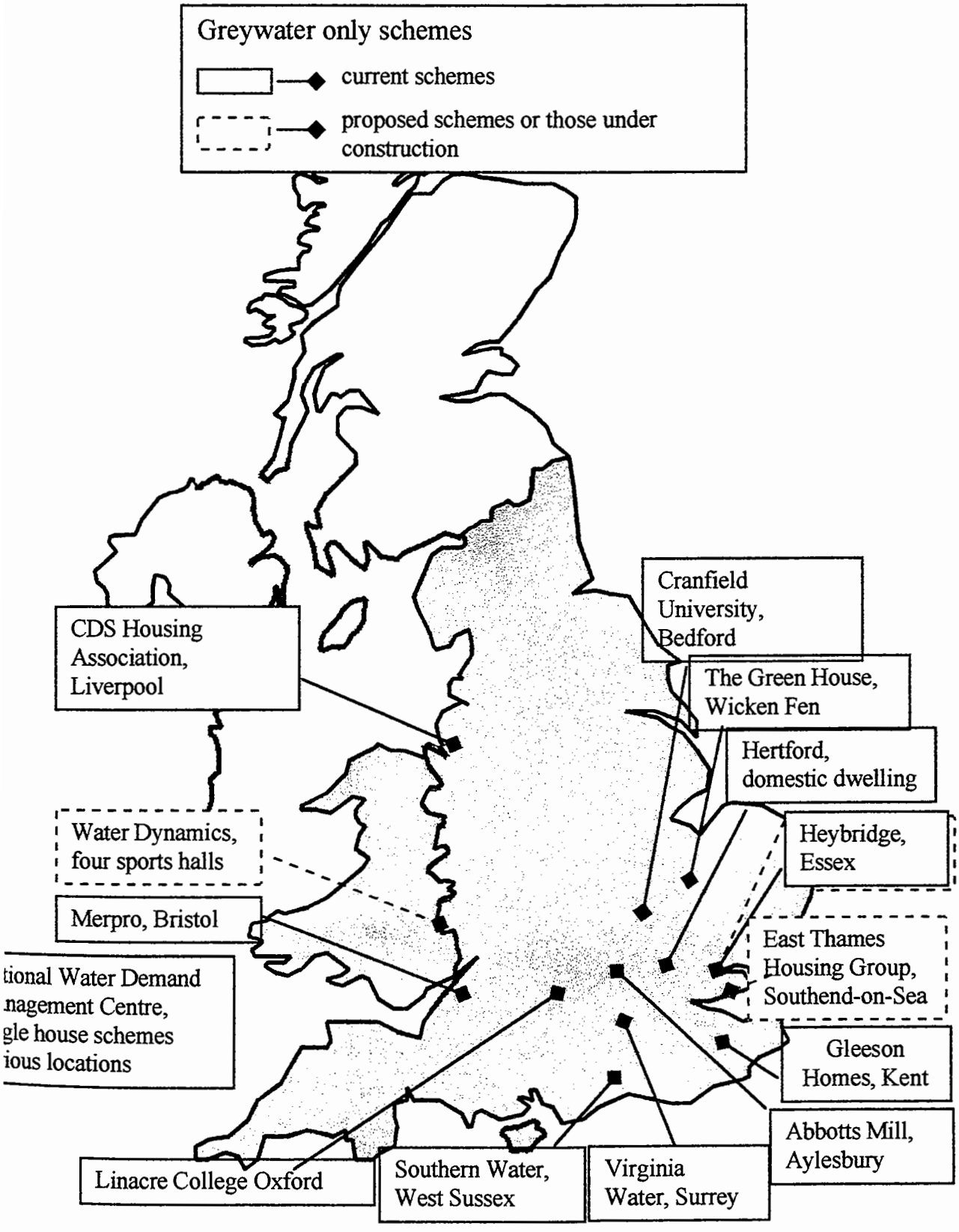


Figure 2.7. A map of the current greywater-only schemes in England and Wales (Jefferson *et al.*, 2000a).

Case study 6: Water Dynamics systems, various locations

A two-year project has been carried out by the Environment Agency (EA) to assess the feasibility of single household greywater systems. The practical aspects of water consumption and cost savings, water quality and user acceptability were evaluated. Water Dynamics recycling systems, which permit recycling of water from handbasins, baths and showers for toilet flushing, were retrofitted to ten houses. Water meter readings, along with water samples for analyses from the storage unit, the header tank and the toilet cistern were taken by the volunteer customers on a monthly basis. Additionally, they completed a report every month to identify operational problems and user perception issues.

Water consumption data were collected over a 4-year period prior to the installation of the recycling systems. After the first year of the trial savings from 5.2% to 30.6% were measured, followed by 5.3% to 35.9% in the second year when the number of trial houses dropped to eight (Sayers, 1998). The range of savings reflected the household occupancy, the number of toilets in each house and the number of toilets connected to the recycling system. The most consequential factor, however, was found in the customers' approaches to maintenance such as cleaning filters. Those prepared to carry out the tasks if necessary achieved the greatest savings whereas those who did not inspect the systems on a regular basis benefited the least.

Acceptable water quality in terms of pH (6-8), phosphorus (around 1 mg l^{-1}) and ammonia (<1 mg l^{-1}) were measured in the toilet cisterns during the two years. High chlorine levels, typically <8 mg l^{-1} but rising to 40 mg l^{-1} on occasions resulted in odour problems, indicating poorly adjusted disinfectant dosing (Sayers, 1998). The frequency of the incidences led to the introduction of a bromine-based disinfectant by the manufacturer. Anionic detergent levels of up to 60 mg l^{-1} were measured in treated greywater with typical values of around 10 mg l^{-1}, leading to increased odour problems when combined with chlorine. Despite the SS of up to 80 mg l^{-1}, the water appearance was defined by the customers as "always clear" and "more clear than turbid" in 26.5% and 40.9% of the cases, respectively, and "more turbid than clear" in

27.3% of the cases. Although additional aeration of greywater could have reduced both the organic pollution and odour formation, the effluent BOD level at 50-300 mg l^{-1} was not regarded as a problem, provided reuse occurred within a few hours (Mustow *et al.*, 1997). Over 90% of the time either non-detectable faecal coliforms or counts close to the detection limit were measured in the toilet cisterns. On a few occasions levels of over 10^4 cfu 100^{-1} ml^{-1} were recorded, exceeding the BSRIA guideline and EU bathing water regulations limits (Sayers, 1998). Investigation revealed that increased bacteria counts in the greywater were caused by the malfunction of the disinfection and filtration units, some customers having washed their pets in the bath.

The following operational concerns were raised from the study:

- blocking of filtration system (more frequent cleaning of filter),
- pump failures (replacement or maintenance; mains water used for toilet flushing in the meantime),
- excessive chlorine dosing leading to odour (use of bromine-based disinfectant),
- staining of the toilet bowl (more frequent use of cleaning products),
- foaming, and
- build-up of sediment in the cistern (longer time required to refill the cistern).

Some of these long-term effects suggest that further investigation into plumbing materials used for greywater recycling is necessary. Improvements to system design, such as the location of the disinfection top-up and alarms in case of blockages or low levels of disinfectant, were suggested by the customers. The customers generally found the recycling unit and appearance of treated greywater aesthetically acceptable though the retrofitted infrastructure was visually unattractive. The "fit and forget" attitude by some users reflected that of the greater public, indicating that education on greywater recycling should not be overlooked.

Pay-back periods were calculated based on a range of water and sewerage charges and household occupancy excluding running and replacement costs. The most economic pay-back period was 13 years in the case of a 4-person household (high water charges) and the most uneconomic at 138 years in the case of a single person household (low water charges). Though water savings depend on many factors, these figures show that recycling systems for individual households are currently not financially attractive. Even at 25% of its market price only 22% of the users were prepared to purchase a Water Dynamics system.

Case study 7: Linacre College, Oxford

Linacre College is the first domestic water recycling scheme in the UK. A student residence for 23 occupants was built in 1995 using "environmentally friendly" or recycled materials in order to cut down energy and water demand. One of the conservation aspects was reuse of greywater for toilet flushing. A survey (Fuentes, 1997) conducted prior to the project showed that 40% of the occupants were concerned about the odour and smell of the treated water but would consent to the plan if these problems could be eliminated.

The first scheme comprising a bag filter and a depth filter was built and operated by a contractor (Read, 1997). Due to severe problems the plant was on-line only for two days. Consequently, the local water company was contacted in order to have the plant fully operational. Anglian Water Services Ltd, Huntingdon, undertook a series of process selection trials (Murrer and Ward, 1997) to identify a suitable system for the scheme, and a number of sand filters and membranes were tested. A trial house with a selected process was evaluated as a small-scale experiment to investigate the cost of such an application, which was found to be substantially reduced by operating at night when the electricity rate was at its lowest (Murrer and Ward, 1997). The trials lead to the second stage of the Linacre scheme where the greywater was treated by a depth filter and a membrane.

The physical process used at Linacre (Figure 2.8) was situated in an underground chamber. Greywater from baths, showers and handbasins was collected in a storage tank and filtered through a 4 inch diameter sand filter (Murrer and Ward, 1997). This was followed by a hollow fibre UF membrane with a pore size of 0.01 μm. The effluent was collected into a header tank in the loft and topped up with mains water when necessary to supply enough water for toilet flushing. The effluent was disinfected with chlorine prior to use. Some of the effluent from the UF membrane was used for backwashing the sand filter. A 5 log reduction in bacteria was attained by this treatment. Viruses were not detected in the effluent.

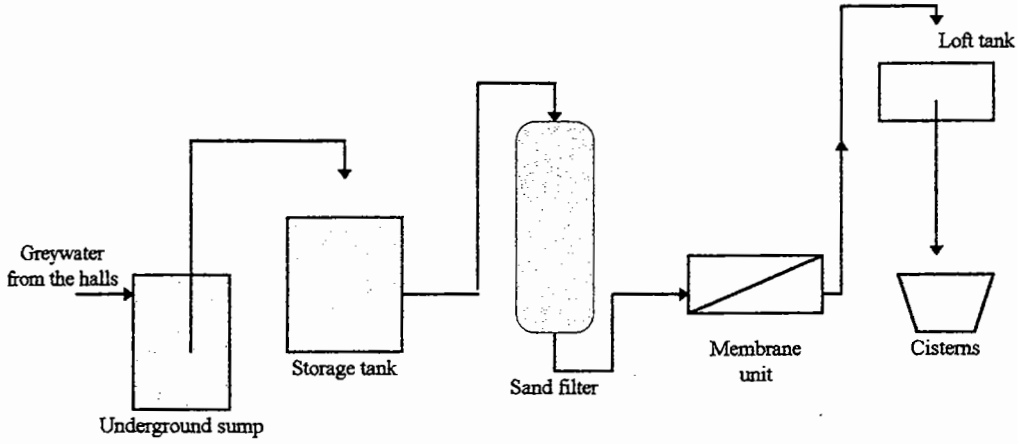


Figure 2.8. The greywater treatment process at Linacre College (Ward, 2000).

The scheme suffered some operational difficulties. The operating and maintenance costs were found to be high due to excessive membrane fouling resulting in a low flux (Ward, 2000). Raw greywater had partially digested under anaerobic conditions in the lengthy collection network resulting in poor permeate quality. Further development of the membrane cleaning procedure was needed to reduce membrane fouling from fats and other organic material in the greywater.

The problems related to the physical process and the poor initial design of collection and storage systems. A further process modification, this time incorporating a biological system (Ward, 2000) has since been made, and the process scheme now

comprises a bioreactor is followed by a sand filter, an activated carbon column and chemical disinfection.

2.5.3.4 Blackwater reuse

Few examples of blackwater recycling exist in the UK (Figure 2.9). The schemes in Doncaster and Blackburn treat blackwater locally to the source in a similar way to greywater. The level of technology depends on the reuse application: natural systems such as reed beds are used in cases where the water is reused for irrigation. Such a treatment is used at a theme park in Doncaster, where other water conservation measures include porous pavements for water collection as well as waterless urinals and combined air/water toilet flushing. At the Hockerton scheme greywater and blackwater from five households is collected for reed bed treatment and subsequently used for irrigation. More advanced technologies are employed for recovered water reused for toilet flushing. This is the case in Blackburn where Anglian Water, the local water company, in collaboration with the estate builder is introducing a blackwater recycling collection and treatment system for toilet flushing in a new housing development consisting of 130 houses. In Cheshire the water for golf course irrigation is treated in a similar way to the scheme in Essex described below, with the exception that no UV light is used for disinfecting the treated water for the irrigation application.

Case study 8: Essex and Suffolk Water

As a result of a severe drought in 1995-97 the levels in the two rivers in Essex normally used for filling a reservoir were too low for extraction, such that an alternative water supply was required (Wishart *et al.*, 2000). A pilot project, approved by the government in 1997, consisted of two stages. In the first one UV pre-treated effluent was pumped into Hanningfield reservoir where the residence time was 250 days at a full reservoir capacity (Simmons and Walker, 1998). Due to health concerns possible environmental impacts were studied based on levels of bacteria, viruses and oestrogen and eutrophic effects. Studies showed that a 3 log reduction in faecal coliforms in the effluent would be required for the river discharge to meet the

Environment Agency condition of 'no deterioration' in water quality (Wishart *et al.*, 2000). UV disinfection was installed to provide a safeguard. Algal growth in the reservoir did not increase due to the recycling as phosphorus was removed to 1 mg l^{-1} at the Chelmsford STW. The oestrogenic studies revealed that a 3:1 dilution of treated water with river water removed any oestrogenic activity (Walker, 2000). Additionally, it was observed that the steroid removal improved to 94% by dosing the effluent with powdered activated carbon (50 mg l^{-1}). This equals to a 20 time dilution of treated water with river water, thereby substantially removing oestrogenic activity even at low river flows.

The second stage of the project was initiated in 1999, with the aim of indirectly recycling 30 Ml d^{-1} of treated effluent (Walker, 2000). The sewage is treated at an existing STW where the tertiary effluent is then chemically disinfected prior to storage in a drinking water reservoir and reuse. In 2000 Essex and Suffolk Water got the planning permission to upgrade the existing STW (Essex and Suffolk Water, 2001). This will enable a long-term reuse of effluent at 40 Ml d^{-1}. The construction work is scheduled to be completed by summer 2002.

In the early stages of the project Essex and Suffolk Water undertook a consultation exercise as part of the EA's consent to discharge (Wishart *et al.*, 2000). Adequate response was received neither from the statutory consultees nor the general public. Subsequently the scheme was shadowed by negative media coverage and this lead to a strong opposition from the public. In this case the social barriers were found more difficult to overcome than the technological ones, however, these problems have since been alleviated. Upgrading the existing STW at Langford has involved "extensive public programme and co-operation with local businesses" (Essex and Suffolk Water, 2001).

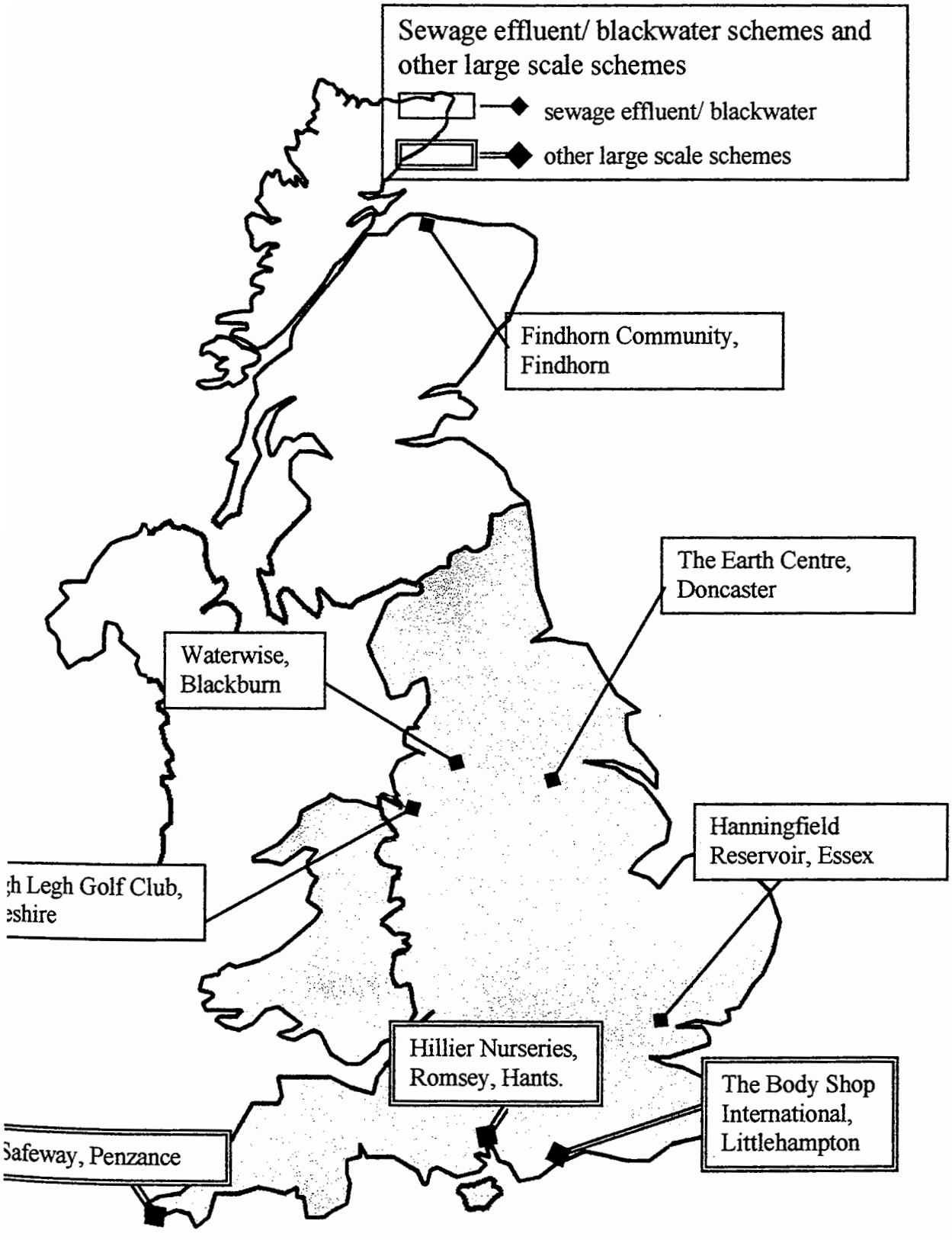


Figure 2.9. A map of the current blackwater-only recycling schemes in England and Wales (Jefferson *et al.*, 2000a).

2.5.4 Economic factors

Cost is one of the biggest barriers to a wider uptake of water recycling systems (Mustow *et al.*, 1997). Long pay-back periods tend to infer non-profitability, reducing interest. Since financial issues are the most sensitive, to many companies, difficulties are encountered in obtaining operating cost data for existing in-building wastewater recycling schemes.

BSRIA have evaluated costs for existing and new buildings based on rainwater, greywater and combined rainwater/greywater reuse (Mustow *et al.*, 1997). Commercially available basic systems treating and recycling water for external, internal and combined uses were considered for scheme sizes ranging from houses to hotels. The latter appeared to be the most cost-effective building type: in all cases the payback time was less than 10 years with the majority being less than 5 years. The most economic application for every scheme size considered was the external use of rainwater, with the pay-back times ranging from less than one year in the case of a hotel to 6-9 years in the other building types. The systems utilising greywater for both external and internal uses had a pay-back time of over 44 years in all building types but hotels. Large housing developments thus provide more tangible economic benefits than smaller ones, though worthwhile savings may still be provided in all cases (Dixon *et al.*, 1999b).

Models of suitable systems for greywater recycling have been presented by Jeffrey *et al.* (1998) and Dixon *et al.* (1999b and 2000). In the former an infrastructure model involving the use of MBRs showed that best savings in potable water and transfer costs are a function of household occupancy, the number of plants and the number of connections linking these processes. Dixon *et al.* (1999b) also linked water saving efficiency with occupancy and evaluated systems utilising both greywater and rainwater, finding that small-scale systems would offer little advantage if both sources of water were used. Water savings and disinfection efficiency could be compromised if larger storage sizes were used at domestic scale (Dixon *et al.*, 2000). For multi-

occupancy there is an ideal storage size of 1000 l beyond which additional savings are not achieved.

Recently an assessment of capital and operating costs for a submerged MBR and a BAF was conducted (Stephenson and Judd, 2000) for a range of population sizes and two influent sources (greywater and blackwater). Results are expressed both by calculating the cumulative present worth as a function of time (Figure 2.10) and by varying either income or costs to identify the break-even point for each technology over the range of conditions outlined (Figure 2.11). Required cost of water per unit volume to attain parity for a 20-year project life at an interest rate of 5% (Figure 2.11) clearly demonstrates that the economic viability of all processes is sensitive to population, water quality and, most importantly, water cost. Even under the most favoured cost-benefit conditions of high water costs, the pay-back periods exceeds 6 years for the MBR technology.

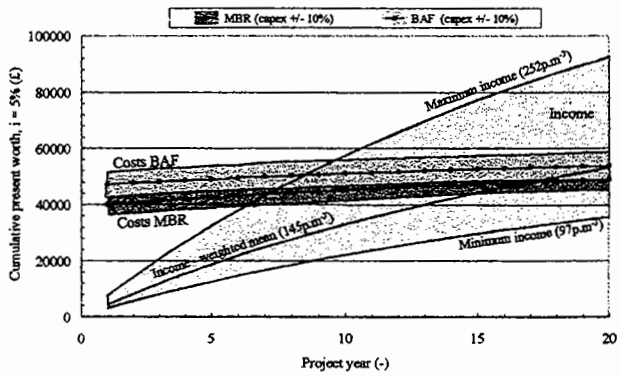


Figure 2.10. Cumulative present worth for a 200 pe greywater recycling scheme (adapted from Stephenson and Judd, 2000).

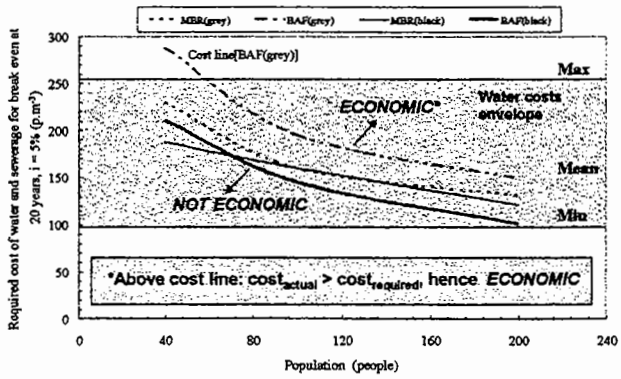


Figure 2.11. Required water costs for economic break-even point for greywater and blackwater recovery at a 20-year project life (adapted from Stephenson and Judd, 2000).

Sometimes cost may be the only barrier to the implementation of greywater recycling. This was the case in a new housing estate with 18 000 houses (45 000 pe) in Amsterdam, the Netherlands (Van der Hoek *et al.*, 1999). In the planning stage alternative water sources for toilet flushing and laundry, i.e. 'household water', were evaluated to reduce demand on the freshwater supply. The 'household water' was to be supplied from greywater, rainwater, lake water, pre-treated surface water, sewage effluent, brackish groundwater or infiltrated rainwater. For each option the following were assessed: public health, technology, legislation, acceptance, cost, and environmental impact. Interestingly, it was not the public health or acceptance but the financial issues that lead to the rejection of both greywater and rainwater options. The respective costs of 14.67 Dutch guilder (NLG) m^{-3} and 15.60 NLG m^{-3} did not favourably compare with the chosen dual water supply of lake water at 2.57 NLG m^{-3}. The assessment emphasised that only some 20% of the total water demand could be met by either greywater or rainwater whilst, via the other alternatives, this figure would be nearly 40%. A survey of 400 residents revealed that 97% accepted 'household water' for toilet flushing and 80% also for clothes washing, provided that the results were as good as for a drinking water supply (Van der Hoek *et al.*, 1999). As many as 70% of the surveyed residents were willing to pay a higher price for 'household water'.

2.6 Perception issues

Public acceptance is generally the most crucial element in domestic water reuse, reflecting cultural and personal values as well as economic interests. Several studies (Bruvold, 1972; Flack and Greenberg, 1997; Neal, 1996; Olson and Pratte, 1978; Water Reuse, 1992) have demonstrated that the public is generally interested in and willing to contribute towards water conservation, especially during a severe shortage of water. It has also been shown that metered customers, who are believed to use less water than flat rate customers, are more willing to recycle water than non-metered customers (Flack and Greenberg, 1997; WROCS, 2000). Opposition towards reuse

increases with the degree of human exposure to reclaimed water, such that the loss of identity of reclaimed water plays an important role in its acceptability.

Surveys have shown different issues to be important for different people depending on educational background, life-style, previous knowledge of water industry and perception of environmental issues. In a survey of Californian customers by Bruvold and Ward (1972) the most frequently mentioned concerns were psychological repugnance and concern over the purity whilst the cost of treatment appeared to be a less important factor in the use of reclaimed water. Interviews conducted in England and Wales revealed the majority of respondents were not concerned with the possibility of water recycling, particularly from communal sources, as long as there would be no increased risk to health (WROCS, 2000). Interestingly, poor aesthetic quality of recycled water appeared to have only a minor impact on the positive perception of reuse. Reliability of the process equipment was identified as a key parameter, since safe operation of an in-building treatment process must be assured.

Choosing the terminology to describe water reuse is critical in order for it to be positively perceived. A telephone survey in the US noted that the terms reclaimed water, recycled water and repurified water had approval rates of 20%, 34% and 70%, respectively, when people were asked whether they would drink water with such descriptions (Katz and Tennyson, 1997). This related to a proposed scheme in San Diego where management of the public perception was begun prior to implementation of a water reuse scheme.

In most reuse schemes in the US today, a detailed public awareness and education programme is initiated before the technical part of the programme begins and continues afterwards (Wegner-Gwindt, 1998). The major program elements include communication materials, general and targeted outreach efforts, legislative and media relations as well as agency partnerships (Wegner-Gwindt, 1998). Reclaimed water is advertised as not being the best compared to potable water, but the second best.

2.7 Summary

Greywater recycling in combination with other water conservation methods can reduce potable water consumption and improve sustainability of communities. Valuable experience has been gained in schemes ranging from single homes to large buildings. However, a number of factors have been recognised as barriers to a wider uptake of such schemes. Appropriate legislation and water quality standards exist in some countries abroad, and they are under development in the UK.

Various technologies have been employed for greywater treatment. These range from a combination of simple filtration and disinfection, where little reduction in pollutant levels is achieved, to advanced physical and biological processes, which produce high-quality product water.

Studies have shown greywater quality to be location-specific and variable. Due to the biochemical and microbiological pollutants present biological treatment of greywater is required such that the effluent meets the reuse water quality standards and regrowth can be prevented. Although operational data on biological processes exist, further investigation is required so as to assess the suitability of such systems for greywater treatment under steady- and unsteady-state conditions.

Chapter 3 Materials and methods

3.1 Introduction

Four biological wastewater treatment processes were set up to treat a variety of simulated household wastewaters ranging from greywater to blackwater. The processes were operated under both steady-state and shock load conditions to test their robustness and reliability. Evaluation was principally through reference to specific water quality determinants pertinent to existing water reuse quality criteria. In parallel with these trials several experiments associated with the feasibility of in-building wastewater recycling were carried out.

3.2 Materials

3.2.1 Cranfield University's sewage treatment works

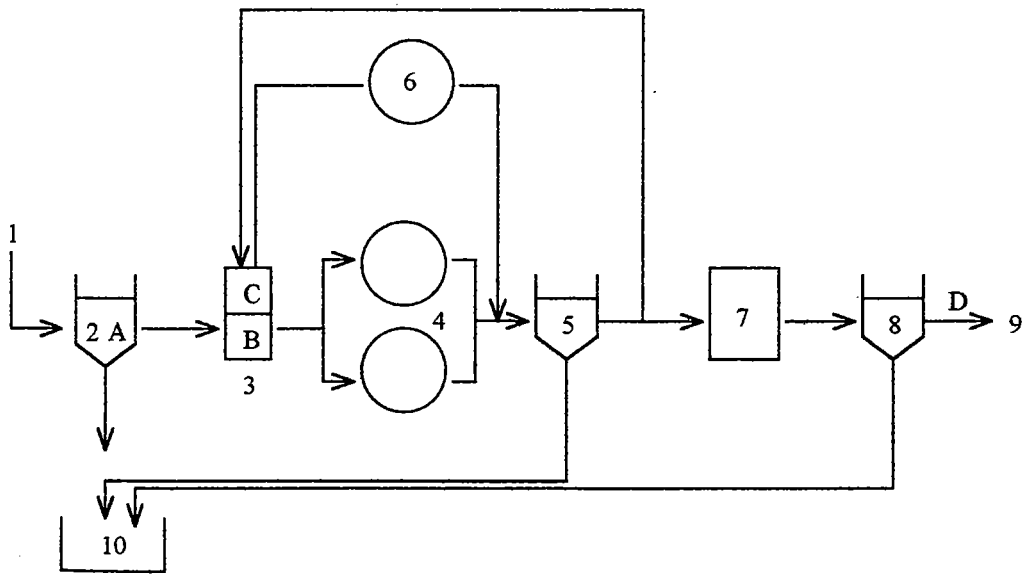
The consented dry weather flow (DWF) for the Cranfield University's sewage treatment works (STW) is 675 m^3 d^{-1} with a typical DWF of 325 m^3 d^{-1} and a rainfall flow of approximately 500 m^3 d^{-1}. The STW (Figure 3.1) has primary sedimentation in a pyramidal tank-square plan area (8.6 m x 8.6 m), straight walls down for 1 m, then angled in at 60° to a total depth of 7.8 m. The sedimentation tank is fitted with V-notch weirs, scum boards and has gravity desludging.

Secondary biological treatment is achieved with circular trickling filters (12.3 m diameter, 1.95 m depth), with randomly packed plastic media and radial distributors. There are three trickling filters: two treating primary settled effluent from the sedimentation tank and one treating secondary effluent from the first humus tank (with dimensions identical to the primary sedimentation tank).

Tertiary treatment is achieved using post-nitrifying filter (5 m x 10 m x 2.8 m average depth) with stone media. The distribution of the media is:

- top 0.25 m contains 63 mm nominal diameter media,
- centre 2 m contains 28 mm nominal diameter media, and
- bottom 0.55 m contains 63 mm nominal diameter media.

Following the post-nitrifying treatment final settlement is achieved using a circular humus tank (7 m diameter, average depth 2.25 m) before discharge. Sludge is tankered off site.



Key:

1. Raw sewage inlet
2. Primary sedimentation tank (A)
3. Distribution channels (B, C)
4. Trickling filters
5. Humus tank
6. Secondary trickling filter
7. Post-nitrifying filter
8. Humus tank
9. Final effluent (D)
10. Sludge tank

Extraction points:

A - primary sewage influent
B - primary settled sewage effluent
C - secondary settled sewage effluent
D - tertiary sewage effluent

Figure 3.1. Schematic presentation of Cranfield University's STW showing treatment processes, direction of effluent flow and extraction points for sewage samples.

3.2.2 Feed

3.2.2.1 System design

The feed system (Figure 3.2) comprised four $1m^3$ covered polyethylene (PE) tanks (*Mailbox International Ltd*, Stalybridge) enabling the feedwaters of different compositions to be generated by mixing individual components of concentrated greywater, blackwater or mains water. Filling of the tank was automated using an actuated valve board linked to the feed pump (*Mono Pumps Ltd*, Manchester) and a Siemens program controller (*RS Components*, Corby; type Logo24 R). The feed pump was calibrated with water (Table B.1 in Appendix B) prior to the experiments.

Submersible pumps (*BSS Group plc*, Leicester. Type Boss 30; *Acal Auriema*, Slough. Type DA12) were installed in all the tanks except the water tank to keep solids suspended. Polyvinylchloride (PVC) tank connectors with valves were fitted 15 cm from the bottom of the tanks, and reinforced PVC tubing was used in connections between the tanks, the actuated valve board and the feed pump.

Concentrated greywater (Section 3.2.2.2) was prepared in a separate holding tank. Depending on the feed ratio, between 250 and 750-litre batches were made up regularly, daily or every other day, to avoid problems of compositional changes during prolonged storage (Section 4.2.3). The blackwater tank was positioned level with the actuated valve board and the feed pump to prevent solids build-up in the tubing. The tank was continuously fed with primary influent sewage (Figure 3.1, i.d. 2) via a DTR18 submersible maceratoring pump (*Acal Auriema*, Slough). The excess blackwater was returned via an overflow to the primary sewage influent basin of the treatment works. The mains water tank was equipped with a ball float switch to maintain a full tank with the overflow directed to the main drain. The feed tank was kept at least half full at all times, with the overflow directed to the sludge holding basin (Figure 3.1, i.d.10). A PVC pipe with an outlet to each rig and a separate valve for emptying the feed tank were both fitted 15 cm from the bottom of this tank.

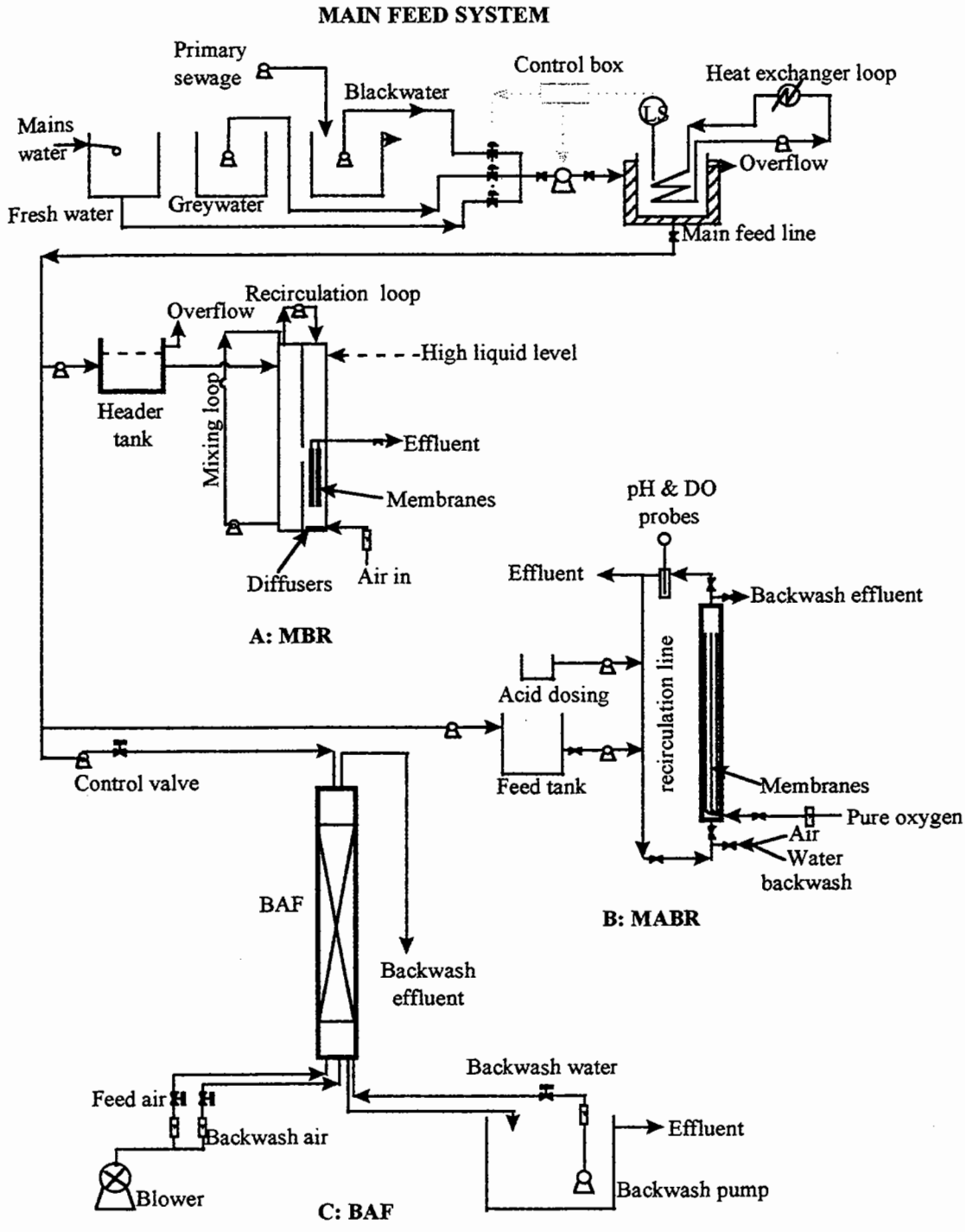


Figure 3.2. Process flow diagram of the pilot plant for the submerged MBR, the MABR and the BAF.

The entire feed system was placed outdoors with all electrical equipment protected by retangular PVC containers. The 1m^3 storage tanks were covered with black PE sacks to minimise algae formation during the summer, and insulation jackets to prevent the contents from freezing in winter. Additionally, frost protection (*RS Components*, Corby) was installed in all electrical parts and pipework.

3.2.2.2 Feed components

Greywater

Synthetic greywater was used throughout the pilot scale trials. Two formulae for synthetic greywater, which are be referred to as matrices no. 1 and no.2, were provided by Anglian Water Services, Huntingdon. Synthetic greywater stock solution used in the current study comprised double the concentration of each constituent of Matrix no.2 (Table 3.1). The mixture contained tertiary sewage effluent to provide bacteria to the greywater and its expected COD values were higher than those generated from Matrix no.1.

The synthetic greywater stock solution was prepared by adding vegetable oil to homogenised commercial soap bar and homogenising by vigorous mixing before adding this mixture, along with the other materials listed in Table 3.1, to a small amount of water. This concentrate was diluted to the desired volume with mains water mixed continuously by a submersible pump. On several occasions during the start-up and greywater trial hair blocked the influent tubing to the treatment processes leading to reduced or no flow. Consequently, hair was no longer used as a part of the greywater stock solution after day 5 of the steady state trial. In order to increase COD, the greywater stock solution was doubled again in concentration on day 35 of the steady state trial. Thus the resulting concentration was four times the original matrix.

Table 3.1. Synthetic greywater stock solution matrices no.1 and no. 2.

	Matrix no.1 [a]		Matrix no.2 [b]	
Material	Amount (in $1m^3$)	Brand, supplier	Amount (in $1m^3$)	Brand, supplier
Soap	62.5 g	Radox Wheatgerm Shower Gel (*Tesco*, Milton Keynes)	16 g	Palmolive Original bar soap (*Tesco*, Milton Keynes)
Shampoo	160 ml	Tesco's Head and Shoulders (*Tesco*, Milton Keynes)	200 ml	Almond shampoo (*Gentlemen's Hairdresser*, Cranfield)
Vegetable oil	8 ml	Tesco's vegetable oil (*Tesco*, Milton Keynes)	8 ml	Tesco's pure sunflower oil (*Tesco*, Milton Keynes)
Hair	3.2 g	- (*Gentlemen's Hairdresser*, Cranfield)	a pinch	- (*Gentlemen's Hairdresser*, Cranfield)
Tertiary sewage effluent	-	-	600 ml	- (*Cranfield University's STW*)

[a] Murrer and Ward (1997) [b] Holden *et al.* (1998)

Blackwater

As a blackwater source was not available for this work, raw sewage was used to represent this component. Primary sewage influent from the works (Figure 3.1, i.d. 2) was monitored for two weeks prior to the start of pilot plant work to determine total chemical oxygen demand (tCOD), ammonia (NH_3) and suspended solids (SS) concentrations. The mean values were 491 mg l^{-1} (182-760 mg l^{-1}), 36 mg l^{-1} (18-55 mg l^{-1}), and 185 mg l^{-1} (82-303 mg l^{-1}), respectively. In addition the Cranfield sewage works monthly report (Branston and Taylor, 1996) was referred to as a guide to the concentration ranges. Apart from the low ammonia, these values were comparable with those reported for blackwater (Brandes, 1978; Laak, 1981; Zeeman and Lettinga, 1999).

Feed ratios

Three different feed ratios were used to produce a range of domestic wastewaters for the work:

- greywater (1:3 solution of synthetic greywater stock solution with mains water),
- greywater/blackwater (50% greywater, 50% blackwater), and
- blackwater.

Activated sludge

Activated sludge was used for the inoculation of the membrane bioreactors. It was provided from the activated sludge plant at Cotton Valley STW (*Anglian Water*) in Milton Keynes, sampled and stored with aeration for no longer than 6 h before use.

3.2.3 Technologies

3.2.3.1 Submerged MBR

Process configuration

The submerged MBR (*Model Products*, Wootton) consisted of two perspex chambers with a working volume of 0.035 m^3 each (Figure 3.2). The chambers were divided into denitrification and nitrification units by a partition wall. The biomass in the denitrification unit was recycled by a centrifugal pump (*Brook-Crompton*, Doncaster) and the also via the recirculation window (50 mm x 50 mm) on the partition wall, allowing biomass transfer between the units. Two 0.24 m^2 plate and frame polysulphone microfiltration membranes *(Kubota Corporation*, Japan) (210 mm x 295 mm), with a mean pore size of 0.4 µm, were immersed into the nitrification units of each bioreactor.

Flow through the MBR was hydraulically driven via a 100-litre PVC constant head device which maintained the immersion depth, i.e. the distance between the water level and the top of the membrane modules, at 0.6 m. The constant head device was continuously fed with a 505S/RL Watson-Marlow pump (*Watson-Marlow Ltd*, Falmouth) and the overflow returned to the feed tank. The effluent outlets from the two MBR chambers were connected via a Y-piece for effluent collection. A commercial compressor supplied air at a 4 bar pressure to an air main from which it

was supplied to the bottom of the MBR chamber. The air flow rate was monitored by means of a flow meter with a 5-25 l min^{-1} range (*Platon Flowbits Ltd*, Basingstoke).

Modifications to the submerged MBR configuration were made at a later stage (Sections 6.3-6.4) mainly to minimise bulking of the sludge in the denitrification tank. Work by Gander *et al.* (1999) demonstrated operation of this type of MBR without the denitrification unit to be successful. The first alteration was to seal the partition wall between the units in each MBR chamber to operate solely the nitrification unit to which the feed was directed. The second modification was to operate the two MBR chambers separately in order to increase the number of trials possible.

Start-up procedure

The submerged MBR was filled with freshly collected activated sludge and aerated for 24 h without feeding. After this period feed was pumped into the constant head device, and so the MBR unit, and effluent outlet ports were opened to allow flux through the membrane.

Membrane cleaning

The membrane cleaning procedure was carried out during configuration modifications and also at the end of the project according to the supplier's recommendations. The membranes were first mechanically cleaned with mains water and a sponge to remove any cake from the membrane surface and then immersed in a 0.6% sodium hypochlorite solution (*Hays Chemical Distribution Ltd*, Leeds. Sodium hypochlorite 10/11%) for an hour. The membranes were kept wetted until the next use to maintain their hydrophilicity. The same set of membrane plates were used throughout the work.

3.2.3.2 Side-stream MBR

Process configuration

The side-stream MBR (*Thames Water Utilities*, Reading) included a FP100 tubular ultrafiltration membrane (*PCI Membrane Systems Ltd*, Whitchurch) of 12.5 mm diameter and 1.2 m length. A centrifugal feed pump (*Mono Pumps Ltd*, Manchester.

Type CB012AC1R4/G) supplied the membrane unit with feed from a 380-litre bioreactor. Feed from the feed tank to the bioreactor was continuously supplied by a 505S/RL Watson-Marlow pump (*Watson-Marlow Ltd*, Falmouth). Overflow from the bioreactor was directed to main drain.

Start-up procedure

The membrane was assembled into the unit and flushed with mains water to removed the preservative bacteriocide solution. Feed was pumped into the bioreactor which had been inocculated with activated sludge. The side-stream MBR was fed with the bioreactor liquor.

Membrane cleaning

The membrane was cleaned prior to and after the greywater trial with a 0.25% Ultrasil P10/11 (*Henkel-Ecolab Ltd*, Swindon) solution prepared in water at 40-60°C. The membrane unit was first flushed with mains water for 5 min prior to recirculating the Ultrasil solution in the system for a further 30 min. This alkali solution was flushed from the system with mains water until pH in permeate reached the pH of mains water.

When off-line for longer than 2 h, the membrane was preserved to prevent it from membrane fouling. For periods up to 3 days a 0.1% (for periods longer than 3 days 0.25%) solution of sodium metabisulphite (*Merck Ltd*, Lutterworth. AnalR grade) was prepared with cold mains water and recirculated in the system for 30 min. After this the solution was left in the system and flushed with mains water when required for next use.

3.2.3.3 MABR

Process configuration

The MABR (Figure 3.1) consisted of a urethane/polyethylene composite hollow fibre bundle (*Membran*, Minnesota, USA) held within a 0.077 m perspex column with a total working volume of 0.0073 m^3. The reactor was fed with pure oxygen (*BOC Gases*, Manchester) at a rate of 9.73×10^{-3} l min^{-1} through a flowmeter (*Platon Flowbits*

Ltd, Basingstoke) and a pressure of 2 bar. The membrane fibres were of a 0.04 μm pore diameter and packed with a parallel orientation to the flow with a specific surface area of 444 m^2 m^{-3} and a voidage of 97%, both with reference to the total working volume. The feed to the MABR was supplied via a 302S Watson-Marlow pump (*Watson-Marlow Ltd*, Falmouth).

Start-up procedure

Two approaches for the start-up procedure of the MABR were used. In the first one a 10% activated sludge solution was recirculated in the MABR for 24 h prior to pumping feed water through the system. In the second approach, instead of feeding by a pump after 24 h, 200 ml doses of diluted activated sludge were injected into the MABR loop daily for five days to advance the biofilm growth on the membrane fibres. After this period feed was pumped into the system as in normal operation.

Membrane cleaning

Previous work on the MABR used for treating tannery wastewater (Brindle, 1997) demonstrated the necessity of cleaning the membrane regularly by backwashing using the MABR effluent. In the pilot scale trials the greywater used had lower COD and SS concentrations (Tables 5.5-5.6) than those of the tannery wastewater. No backwashing was required in this work due to operational problems (Section 5.2.2). When not in use, the membrane was washed with mains water and air dried.

3.2.3.4 BAF

Process configuration

The BAF unit was supplied by Thames Water Utilities, Reading. The unit was housed in a perspex-covered metal frame and comprised an air compressor, an effluent collection tank, and a feed tank (not used in this work). The rig was located outdoors and a temperature actuated heater was used as a frost protection device.

The BAF column (Figure 3.1) comprised a 0.165 m diameter vertical perspex container with plastic 3-4 mm media (*Thames Water Utilities*, Reading. Lytag) of 1.64 m depth

onto which a biofilm was developed. Feed was pumped using a 504U Watson-Marlow pump (*Watson-Marlow Ltd*, Falmouth) counter current to the compressed air, which was supplied at 15 l min^{-1} and monitored by means of a Platon flowmeter (*Platon Flowbits Ltd*, Basingstoke). Due to occasional loss of media, the column was refilled as required.

Start-up procedure

The BAF column was filled with feed to the top level on the column. Feed was continuously pumped into it to allow biofilm growth on the media.

Media backwashing

BAF effluent for backwashing purposes was collected to a 1.5 m^3 PVC tank from which an overflow was directed to drain. The head loss of the media in the column initiated the automatic backwash program as follows:

1. process air off,
2. air scour at 15 l min^{-1} for 1 min,
3. air scour at 15 l min^{-1} and backwash water at 20 l min^{-1} for 6 min,
4. air scour and backwash water off, and
5. process air at 15 l min^{-1}.

A more thorough cleaning of the media was carried out after the steady-state trials (Section 5.2). The column was emptied of feed, filled up with mains water, and backwashed with mains water several times. The rig was run with mains water for two days to remove any residual feed and then with greywater for 1 month prior to the unsteady-state trials.

3.3 Methods

3.3.1 Sampling protocol and analyses

The weekly sampling protocol and the analyses carried out during the steady-state trials on the submerged MBR, the MABR and the BAF (Section 5.2) are listed in Table 3.2. After the greywater run tCOD, soluble COD (sCOD), ammonia (NH_3-N), nitrite (NO_2-N), nitrate (NO_3-N), total nitrogen (TN), phosphorus (P) as well as the bacteriological determinants were measured once a week. During the unsteady-state operation (Sections 6.3 and 6.5) the sampling frequency was higher.

Table 3.2. Weekly sampling protocol during the steady-state trials (Section 5.2).

Frequency (d)	Tasks
7	Overall check-up (e.g. pumps, tubing, tanks) on the plant, preparation of greywater stock solution
5	Measurements: Effluent flow rate, temperature, DO, pH, air flow, power
3	Measurements: tCOD, sCOD, BOD_5, NH_3-N, NO_2-N, NO_3-N, TN, P, SS, MLSS, MLVSS, turbidity, total coliforms, *E.coli*, faecal streptococci

The effluent samples were collected in 1000 ml Nalgene bottles (*Merck Ltd*, Leicester) at appropriate outlet ports of each rig, and the influent sample was taken in a 250 ml Nalgene bottle (*Merck Ltd*, Leicester) from the feed tank. Dissolved oxygen (DO) and pH levels of the effluents and the influent were measured at the pilot plant facility using portable meters. The remaining samples were analysed within 3 h in the laboratory or stored at 4°C for as short period of time as possible prior to analyses and in any case no longer than 24 h.

Samples of the side-stream MBR influent and effluent were taken daily for 32 days for BOD_5 (5-day biochemical oxygen demand), tCOD, SS, turbidity, flow rate, DO and total/*E.*coliforms as well as from the bioreactor for mixed liquor suspended solids (MLSS) and mixed liquor volatile suspended solids (MLVSS).

3.3.2 Analyses

3.3.2.1 Chemical analyses

All chemical analyses were carried out as described in the Standard Methods (APHA, 1992), using its adapted versions for the Hach methods or according to the supplier's instructions in the case of total organic carbon (TOC). For all the analyses Finnpipettes (*Fischer Scientific UK*, Loughborough) with ranges 200 -1000 µl and 2-10 ml and Finntip sizes 61 and 63 (*Fischer Scientific UK*, Loughborough) were used. Deionised water was used for all the solutions and dilutions. The materials used for the chemical analyses are summarised in Table 3.3.

Tracer studies

Tracer studies with sodium chloride (*Merck Ltd*, Poole. GPR grade) were carried out as described in Mann *et al.* (1995). Previous work concluded that low molecular weight substances as salts can be used as representative tracer for aqueous solutions (Fdz-Polanco *et al.*, 1996; Mann *et al.*, 1995). The 1 M NaCl tracer was injected as a pulse input such that the volume applied was smaller than 1% of the reactor volume and the time of injection was less than 5% of the theoretical hydraulic residence time (tHRT). The tracer concentration was determined from the conductivity readings of the samples.

The corresponding lithium (*Fischer Scientific UK*, Loughborough. AAS grade) tracer studies were carried out on the BAF during greywater and blackwater treatment to determine if the presence of biomass had an effect on the hydraulic pattern. The procedure was adapted from Moore *et al.* (1999) using 1000 mg l^{-1} Li^{+} and the injection method as above. Anderson *et al.* (1991) concluded that at this concentration lithium has no effect on the methanogenic activity of bacteria. The concentration in the samples was measured using an inductively coupled plasma with an atom emission spectrophotometer (ICP-AES). In all experiments the residence time distribution (RTD) curves were monitored over a period of twice the actual time or theoretical

hydraulic retention time (tHRT) or until >90% of the tracer injected had been recovered. The tracer study data are presented in Section 5.2.3.1.

Table 3.3. Materials used for the chemical analyses.

Constituent	Materials (Supplier, location)
DO, temperature	Jenway DO2- 9071 meter (*Jenway Ltd*, Dunmow)
pH	Hanna H18424 pH meter (*Merck Ltd*, Leicester)
Conductivity	DPC2 conductivity meter (*LTH Electronics*, Luton)
tCOD, sCOD, NH_3-N, NO_3-N, TN, P (Section 5.2 and 6.3-6.5)	Hach test-n-tube vials and Hach DR/2010 portable spectrophotometer (*Camlab Ltd*, Cambridge)
P (Sections 4.2-4.3), S, Ca, K, Fe, Mg, Mn, Cu, Al, Zn, Mo, Co, Li	Thermo Jarrell Ash ICP-AES (*Sci-tek Instruments*, Olney. Type Atom Scan 16)
BOD_5 with allylthiourea (ATU)	AnalR grade chemicals (*Merck Ltd*, Poole) for the dilution water and ATU, GEC Avery balance (*Avery Berkel UK*, Walsall), Philip Harris mechanical stirrer (*Merck Ltd*, Leicester)
TOC	TOC analyser Shimadzu Model TOC-5000A (*Shimadzu*, Milton Keynes)
NO_2N	AnalR grade chemicals for the reagent solution (*Merck Ltd*, Poole), Precisa 125 balance (*Precisa Balances Ltd*, Milton Keynes), PU 8720 UV/VIS Philips spectrophotometer (*Science Exchange*, Walton on Thames)
SS, MLSS, MLVSS	Whatman GC/F 70 mm diameter filter papers (*Fischer Scientific UK*, Loughborough), Speedivac 2 rotary vacuum pump (*D.Benway Ltd*, Boreham Wood), GEC Avery balance (*Avery Berkel UK*, Walsall), Gallenkamp Hotbox oven with fan, size one (*OVB 350*, Walton on Thames), Camlab super range oven (*Camlab Ltd*, Cambridge), Townson+Mercer furnace model M4-1200 (N/A)
Turbidity	Hach 2100N turbidimeter (*Camlab Ltd*, Cambridge), Ultrawave ultrasonic bath (*Scientific Laboratory Supplies*, Hull)

Storage trials

The greywater storage trials described in Section 4.2.3 consisted of both small- and large-scale experiments. In the former 10 l batches of synthetic greywater prepared according to the matrices shown in Table 3.1 were stored in 30 l containers for 5-7 days and mixed with air. In the case of the latter the batches were stored in the 1 m^3 tanks covered with PE as described in Section 3.2.2.1. For the 18-day trial period 500 l of synthetic greywater (Matrix no.2, Table 3.1) was mixed with a submersible pump (Section 3.2.2.1). Shower and bath waters collected from the staff and students in the department were combined for the real greywater storage trials where 270 l and 230 l were stored for 18 days with and without mixing (i.e. quiescent), respectively. The samples from the quiescent greywater were withdrawn 15 cm from the top and bottom of the tanks.

3.3.2.2 Biological analyses

The biological analyses were divided into two groups:

- bacteria counts to monitor the dynamic behaviour of greywater (Section 4.2) and the process performance (Sections 5.2-5.3, 6.3 and 6.5), and
- respirometry to investigate the growth rate of the biomass of the submerged MBR and separate experiments (Sections 4.3 and 6.4).

Bacteria counts

Samples for the bacteria counts were asceptically collected in autoclaved 100 ml Nalgene bottles (*Merck Ltd*, Leicester) at the respective sampling points of the treatment processes and the storage tanks. The analysis procedure was carried out asceptically using sterilised deionised water, and Finnpipettes (Section 3.3.2.1) with appropriate sterilised pipette tips. Colilert18 and Enterolert reagents (*Idexx Ltd*, Chalfont St.Peter) were used for determining total/E. coliforms and faecal streptococci, respectively. For both sets QuantiTray2000 and QuantiTray sealer (*Idexx Ltd*) were used as proscribed by the supplier.

Respirometer

Activated sludge samples for respirometry in both test series were collected in 250 ml Nalgene bottles (*Merck Ltd*, Leicester). Two separate series of tests were performed using a closed cell 10-channel aerobic respirometer (*CES Ltd*, Sittingbourne), which operates in a similar way to the Warburg respirometer (Heddle, 1980). The continually stirred glass cells are maintained at 20°C, with the readings taken every 5 min. The respiration rate is the rate at which oxygen is used as the terminal electron acceptor, and therefore can be used to indicate the condition of the sludge in terms of metabolic activity.

Respirometry work in Section 4.3

The sludge was taken from two MBRs: one treating real greywater (COD:TN:P of 13:4:1) and the other treating synthetic greywater (Matrix no.2, Table 3.1; COD:TN:P of 894:106:1). The sludge was maintained in porous pots (Section 3.3.2.2) at a pH of 7.5 ± 0.2 and fed with real greywater collected daily and synthetic greywater prepared daily (Table 3.4). The total sample volume of 55 ml consisted of 50 ml aliquots of activated sludge and predetermined quantities of nutrient supplements (Table 4.5, Section 4.3.1), as well as deionised water. Control samples received N/P supplements, deionised water in the tests with and without N/P balancing, respectively, in addition to activated sludge. With reference to the microbial nutrient requirements for sludge (Table C.7 in Appendix C), aluminium (Al), cobalt (Co), copper (Cu), iron (Fe), molybdenum (Mo) and zinc (Zn) were identified as missing elements using inductively coupled plasma atomic emission spectrophotometer (ICP-AES; Table 3.3). These nutrients were tested on the respirometer in single supplements to the following greywaters to observe possible effects on the biomass (Section 4.3):

- N/P-limited real greywater,
- N/P-balanced real greywater,
- N/P-limited synthetic greywater , and
- N/P-balanced synthetic greywater.

A sample of the original activated sludge was filtered for influent sCOD and MLSS, and each sample was analysed for effluent sCOD and MLSS after a 15-hour respirometer run. The tests were duplicated to assess repeatability (Section 4.3.3). Respiration and COD removal rates per unit MLSS were calculated from the MLSS, oxygen utilisation and influent and effluent sCOD.

Respirometry work in Section 6.4

Activated sludge samples were collected from a submerged MBR unit treating synthetic greywater (Matrix no. 2, Table 3.1) with a mean flow of 0.091 ± 0.006 m^3 d^{-1}. The mean MLSS and pH of the sludge were 9788 ± 1588 mg l^{-1} and 7.4 ± 0.2, respectively. Each respirometer cell received 30 ml of activated sludge, a predetermined quantity of a test substance (Table B.2 in Appendix B) and deionised water, with the total volume being 60 ml. The control cells received only sludge and deionised water. The pH of the mechanically stirred samples was measured after the substance addition. The MLSS and sCOD were measured before and after the 12-hour respirometry test for calculation of the respiration and COD removal rates. The concentration of the test substances were increased until a reduced performance was observed or to the point where operational difficulties prevented or hindered the measurements.

Porous pots

Work described in Section 4.3 was based on the WRc Porous-pot method (*Water Research Centre*, Marlow). The activated sludge inoculae were taken from membrane bioreactors treating real and synthetic greywater (Matrix no.2, Table 3.1) and maintained in porous pots (*Yorkshire Water plc*, Bradford). Real greywater was collected every two days from the greywater holding tank at the Fedden House student accommodation at Cranfield University. Synthetic greywater was prepared every two days by mixing the constituent materials (Table 3.4) with mains water.

Table 3.4. Synthetic greywater matrix used for porous pots.

Material	Brand, supplier	Amount (in 10 l)
Soap	Tesco's soapfree cream wash (*Tesco*, Milton Keynes)	0.64 g
Shampoo	Tesco's value shampoo (*Tesco*, Milton Keynes)	8.0 ml
Sunflower oil	Tesco's pure sunflower oil (*Tesco*, Milton Keynes)	0.1 ml
Tertiary sewage effluent	- (*Cranfield University's STW*)	24.0 ml

3.3.3 Product evaluation: RACOD

3.3.3.1 Introduction

A number of on-line analysers for the measurement of wastewater BOD are applied in the industry. RACOD (*USF Wallace & Tiernan*, Tonbridge), a Rapid Assimilable Chemical Oxygen Demand, is an instrument based on the respirometric method (Table 3.5). The meter measures dissolved oxygen in the bioreactor to give the easily biodegradable fraction of COD or BOD, from which it determines the readily assimilable fraction of the sample on a ten minute cycle. The culture of organisms is maintained in a substrate-limited condition so that addition of fresh nutrient results in a fast change in respiration rate as the organisms respond to the influx of nutrient.

Table 3.5. Components of the RACOD meter and their functions.

Component	Function
Bioreactor (500 ml) with an overflow	Contains culture of organisms (usually derived from wastewater source)
Stirrer blade	Mixes sample in bioreactor and cleans surface of bioreactor during cleaning cycle (programmed)
DO and temperature probes	Fitted on wall of bioreactor to measure DO and temperature of biomass (operating temperature 15 to 30°C)
Aeration pump	Aerates sample in bioreactor for 1 to 9 min
Peristaltic feed sample pump (*Watson-Marlow Ltd*, Falmouth. 300 series)	Provides continuous feed sample flow from 1 to 3 ml min^{-1}
Electronics module	Process control unit

3.3.2.2 Evaluation trials

The RACOD meter was set up in a laboratory for the 3-month evaluation trial period (Section 6.2). The feed sample was kept in a covered 5-litre polyethylene container at room temperature. The start-up procedure was carried out using primary sewage influent (Figure 3.1, i.d. 2) diluted 1:1 with mains water as a feed sample at a flow rate of 3 ml min^{-1} for two hours followed by a constant flow rate of 1 ml min^{-1}. After 5 days of operation a cooling fan was placed in front of the process control unit to prevent it from overheating. Five types of wastewater (Figure 3.1) were evaluated using the RACOD in comparison with BOD_5:

- primary sewage influent (i.d. 2),
- primary settled sewage effluent (i.d. B),
- secondary settled sewage effluent (i.d. C),
- tertiary sewage effluent (i.d. D), and
- synthetic greywater (Matrix no.1, Table 3.1).

Waste from the unit was collected via an overflow into a covered polyethylene container and disposed of daily. The bioreactor, the feed sample tubing and the overflow were drained and mechanically cleaned weekly and between the experiments with mains water and a brush. The feed pump was calibrated every two weeks.

3.3.3.3 Online monitoring

The RACOD was set up as an on-line BOD monitor for the submerged MBR and the BAF effluents (Section 6.3) in the pilot plant facility. Effluent from the respective outlet ports was directly pumped into the RACOD bioreactor at 3 ml l^{-1}. The BAF effluent was occasionally cooled down by recirculating mains water around the effluent outlet port to maintain the RACOD temperature within the operational range. Samples for BOD_5 were taken at predetermined times depending on the individual experiment, and the values were compared to the RACOD readings. The RACOD unit was cleaned as described in Section 3.3.3.2 weekly and when the effluent type was changed.

Chapter 4 Greywater characterisation

4.1 Existing information

One of the key elements of wastewater treatment is the nature of the wastewater matrix itself. Blackwater characteristics are well known since they are similar to strong sewage (Droste, 1997; Metcalf and Eddy, 1991; Section 2.2.2). Greywater, on the other hand, is a much less familiar matrix.

Information on greywater derives from several recycling projects, many of which have concluded greywater quality to depend largely on the source (Section 2.2). It resembles domestic wastewater in terms of the organic strength, but some differences arise from its often high COD:BOD ratio (Holden *et al.*, 1998) as well as nitrogen and phosphorus concentrations (Christova-Boal *et al.*, 1996), which may then demand nutrient addition for effective biological treatment (Section 4.3). The most variable quality parameter is possibly the pathogen count (Table 2.3, Section 2.2.2). Greywater quality is subject to dynamic changes when stored such that pathogen levels are maintained over storage periods of several days (Nolde, 1999; Rose *et al.*, 1991) whilst the other quality determinants begin to change in value within the first couple of days (Dixon *et al.*, 1999c; Jefferson *et al.*, 2000b; Neal, 1996). These changes may have an impact on the treatability of the greywater. Models interpreting degradation patterns under different storage conditions have been presented (Dixon *et al.*, 1999c) so as to understand the transient behaviour of greywater.

Given the limited information available on greywater characteristics a number of requiring investigation prior to the pilot-scale trials were identified:

- greywater quality variation between different sources in households,
- greywater quality variation within a single source,

- effect of specific hygiene products on the quality parameters,
- synthetic and real greywater quality comparability,
- effect of storage conditions on degradation, and
- potential enhancement of a biological process performance through nutrient addition.

4.2 Real greywater vs. synthetic greywater

4.2.1 Real greywater

4.2.1.1 Individual user

Staff and students within the department were asked to provide shower, bath and handbasin samples along with information on gender and personal hygiene products used (Table C.1 in Appendix C). The samples were analysed on the day of collection so as to minimise deterioration of water quality. Results (Table 4.1) refer to the combined data from shower, bath and handbasin samples.

In addition to the experiments described above, two sets of more controlled experiments were carried out to assess the repeatability of these measurements (Table 4.1). The first involved a series of showers by the same individual. The amount of water used for each shower was kept constant by controlling the time taken to shower. The amount of hygiene products used for each shower was also kept constant by visual estimation. A similar series of tests was carried out by another individual who had baths of constant volume. The shower and bath samples were analysed on the day of collection.

Organics and nutrients

The shower, bath and handbasin samples had a similar biodegradable content with the mean BOD_5 values ranging from 129 to 155 mg l^{-1}, whilst larger differences were

observed in terms of the non-biodegradable fraction at a mean COD of 367-587 mg l^{-1} (Table 4.1). The COD:BOD ratio was generally lower in the bath (2.9 ± 1.3) and shower samples (2.8 ± 1.0) than in the handbasin samples (3.6 ± 1.6). The mean TOC concentration in the individual samples was in the range of 60-99 mg l^{-1}, with the lowest being in the bath samples and the highest in the handbasin samples, giving an overall COD:BOD:TOC ratio of 7.8:2.8:1 due to the relatively high COD and low organic carbon.

Table 4.1. Comparison of real greywater characteristics. Mean ± standard deviation.

Parameter	Shower [a]	Bath [b]	Handbasin [c]	Combined	Shower repeat [d]	Bath repeat [e]
BOD_5 (mg l^{-1})	146 ± 55	129 ± 57	155 ± 49	146 ± 54	90 ± 26	185 ± 31
tCOD (mg l^{-1})	420 ± 245	367 ± 246	587 ± 379	451 ± 289	181 ± 63	651 ± 112
COD:BOD (-)	2.9 ± 1.3	2.8 ± 1.0	3.6 ± 1.6	3.1 ± 1.3	2.2 ± 0.6	3.5 ± 0.2
TOC (mg l^{-1})	65 ± 45	60 ± 43	99 ± 142	73 ± 79	-	-
COD:BOD:TOC (-)	7.4:2.8:1 [f]	6.7:2.8:1 [f]	9.1:2.8:1 [f]	7.8:2.8:1 [f]	-	-
SS (mg l^{-1})	89 ± 113	58 ± 46	153 ± 226	100 ± 145	26 ± 9	49 ± 11
Turbidity (NTU)	85 ± 71	60 ± 43	164 ± 171	101 ± 109	18 ± 4	52 ± 11
TN (mg l^{-1})	8.7 ± 4.8	6.6 ± 3.4	10.4 ± 4.8	8.7 ± 4.7	-	-
P (mg l^{-1})	0.3 ± 0.1	0.4 ± 0.4	0.4 ± 0.3	0.4 ± 0.2	-	-
C:N:P (-)	1687:38:1 [f]	2129:62:1 [f]	1749:31:1 [f]	1641:38:1 [f]	-	-
pH (-)	7.5 ± 0.3	7.6 ± 0.3	7.3 ± 0.3	7.5 ± 0.3	7.7 ± 0.3	7.7 ± 0.0
Total coliforms (cfu 100^{-1} ml^{-1})	6800 ± 9740	6350 ± 9710	9420 ± 10100	7387 ± 9759	17000 ± 11000	23900 ± 20000
E.coli (cfu 100^{-1} ml^{-1})	1490 ± 4940	83 ± 120	10 ± 8750	2022 ± 5956	590 ± 1630	21800 ± 22100
Faecal streptococci (cfu 100^{-1} ml^{-1})	2050 ± 4440	40 ± 49	1710 ± 5510	1740 ± 4488	32 ± 41	10 ± 1

[a] 65 samples [b] 25 samples [c] 13 samples [d] 10 samples [e] 6 samples [f] mean

The controlled experiments also exhibited some variability such that COD was 181 ± 63 and 651 ± 112 mg l^{-1} respectively for shower and bath replicate samples. In the

former the COD:BOD ratio was similar to sewage at 2.2 ± 0.6 whereas in the latter it was somewhat higher at 3.5 ± 0.2 due to the larger proportion of non-biodegradable matter.

A deficiency in phosphorus and nitrogen in greywater was observed (Table 4.1) and this corroborates well with values from literature (Table 2.3, Section 2.2.2). In the shower, handbasin and bath samples the C:N:P ratio was as high as 1687:38:1, 1749:31:1 and 2129:62:1, respectively, with the combined data giving an overall ratio of 1641:38:1. The nutrient balance of greywater and its impact on a biological treatment process is discussed in Sections 2.2.2 and 4.3.1.

Solids and turbidity

The handbasin samples had the highest mean and variability in suspended solids (153 ± 226 mg l^{-1}) and turbidity (164 ± 171 NTU) (Table 4.1), due to the variable and predominantly low dilution compared to the shower and bath samples. For shower greywater solids and turbidity of 89 mg l^{-1} and 85 NTU, respectively, were measured. The bath greywater was slightly more diluted with corresponding values of 58 mg l^{-1} and 60 NTU.

In the controlled experiments the mean values were lower than in the individual samples, but none-the-less subject to significant standard deviation. The replicated samples in the study appeared not to follow the previously mentioned dilution factor as the shower replicates gave SS and turbidity values of 26 ± 9 mg l^{-1} and 17.9 ± 3.8 NTU respectively whereas the corresponding values for the bath replicates were higher at 49 ± 11 mg l^{-1} and 51.7 ± 11.0 NTU. Given the overall natural variability of the samples this is within the boundaries of imprecision. The mean pH of all the greywater samples was in the neutral range (7.3-7.7).

Pathogens

The largest variability was found in the pathogen counts. The individual samples from all sources had a total coliform counts of 3 ± 3 log. The faecal contamination of the

handbasin samples was surprisingly high at 3 ± 3 log for faecal streptococci and a mean 1-3 log for *E.coli*, confirming previous findings (Surendran and Wheatley, 1999).

The controlled experiments had a generally higher total coliform count at 4 ± 4 log than in the individual samples. The faecal contamination in terms of faecal streptococci at 1 ± 1 log was lower than in the individual samples (Table 4.1). However, the bath replicate samples exhibited *E.coli* counts as high as 4 ± 4 log, being 3 log higher than the individual bath samples. In the shower replicate levels were similar to the individual shower samples at 2 ± 3 log.

Gender of greywater donor

The products used and the gender of the donors of the greywater samples are shown in Table C.1 in Appendix C. As the work set out to assess greywater quality rather than consumer behaviour, neither the amounts of products nor the water volume were recorded as these would have been difficult to verify accurately. However, the quantities of hygiene products and water in combination with particular ingredients naturally have a great impact on the pollutant concentrations.

The shower samples did not reveal major differences between males and females, and similar concentrations in all the individual samples were measured for phosphorus and pH. Possibly due to shaving products the handbasin samples given by males had higher pollutant concentrations than those by females, whilst the opposite was the case for the bath samples.

4.2.1.2 Product effect

Personal hygiene products contribute significantly to the pollutant load (Christova-Boal *et al.*, 1996) and to 10% of the COD in domestic wastewater (van der Wijst and Groot-Marcus, 1999). The changes in consumer behaviour and life-style, however, are more significant factors than the type or amount of products used. Van der Wijst and Groot-Marcus (1999) reported that in the Netherlands the frequency of showering

doubled between 1980 and 1992, whereas the COD originating from shampoo, soap and bathing products increased by 39%. High pollutant loads may be mitigated in large scale greywater systems whilst they have a significant impact on small treatment units.

4.2.2 Synthetic greywater

4.2.2.1 Variation in instantaneous quality

Examples of greywater analogues used in previous studies are shown in Table 4.2. By applying modification to basic matrices shown in Table 3.1 (Section 3.2.2.2), the impact of certain ingredients on the key water quality parameters, COD in particular, has been assessed. Microbiological quality was not determined since the recipes contained little or no tertiary sewage effluent.

Organics, solids and turbidity

Increasing the amount of all ingredients increased the COD accordingly (Table 4.2). This trend was less obvious in the case of the vegetable oil possibly due to its sparing solubility. The biodegradable matter in the greywater samples appeared not to change as a result of adding individual ingredients but more so after doubling all the components in Matrix no.1. The mean COD:BOD ratios of the samples derived from Matrix no.1 were 3.0-4.9:1, and were thus in reasonable agreement with that of the synthetic greywater of Anglian Water (3.0:1) and other reported ratios for real greywater (Holden *et al.*, 1998).

Two batches of Matrix no.2 were prepared using the specific commercial products reported by Gunstead (1998). The higher COD (62-164 mg l^{-1}) in Matrix no.2 was a likely result of the increased concentration of shampoo over that of Matrix no.1 (Table 3.1, Section 3.2.2.2). The correspondingly higher BOD (22-57 mg l^{-1}) was partially attributed to the tertiary sewage effluent component. As a result the COD:BOD ratio at 2.8-2.9:1 was lower than that of Matrix no.1. In general the water quality determinant values of Matrix no.1 in the current work were at the lower limit

of the reference range of the Anglian Water data (Table 4.2). At 2.3-3.2 NTU the turbidity was similar in the basic Matrix no.1 and changed little on dosing the levels of some of the individual components. On doubling all the ingredients a value of 4.0 NTU was measured. Generally the water quality values were low compared to those of the synthetic greywater used in the pilot scale trials.

Table 4.2. Comparison of synthetic greywaters. Mean ± standard deviation (range).

Synthetic greywater type (re: Table 3.1, Section 3.2.2.2)	Parameter COD ($mg\ l^{-1}$)	BOD_5 ($mg\ l^{-1}$)	COD:BOD (-)	SS ($mg\ l^{-1}$)	Turbidity (NTU)
Matrix no.1	43	12	3.6:1	1	2.8
Matrix no.1, oil 2x	59	12	4.9:1	3	3.2
Matrix no.1, soap 2x	52	14	3.7:1	3	2.8
Matrix no.1, shampoo and hair 2x	30	10	3.0:1	0	2.3
Matrix no.1, oil and soap 5x	387	n.m.	n.m.	n.m.	n.m.
Matrix no.1, oil and soap 10x	515	n.m.	n.m.	n.m.	n.m.
Matrix no.1, oil 20x	287	n.m.	n.m.	n.m.	n.m.
Matrix no.1, all 2x	78	17	4.6:1	4	4.0
Matrix no.1, all 3x	212	n.m.	n.m.	n.m.	n.m.
Matrix no.1, all 10x	812	n.m.	n.m.	n.m.	n.m.
Matrix no.1, all 20x	1527	n.m.	n.m.	n.m.	n.m.
Matrix no.2	113 ± 72 (62-164)	40 ± 25 (22-57)	2.85:1 (2.8-2.9:1)	10 ± 13.4 (1-20)	10.6 ± 12.9 (1.4-19.7)
Anglian Water synthetic [a]	117 ± 52 (38-237)	39 ± 39 (7-117)	3.0:1	35 ± 23 (8-102)	29.6 ± 21.7 (0.9-74.6)
Thames Water synthetic [b]	153, 109 [c]	59, 37 [c]	2.6:1	18	n.g.
Loughborough University synthetic [d]	n.g.	215 ± 120 (45-551)	n.g.	196 ± 91 (65-439)	72 ± 38 (9-155)

[a] Gunstead (1998), bar soap, shampoo, hair, cooking oil, tertiary sewage effluent
[b] Smith *et al.* (1999b), synthetic soap only
[c] soluble
[d] Surendran and Wheatley (1998), soap, detergent, starch, yeast extract, cooking oil, settled sewage
n.g. not given
n.m. not measured

Differences in reported synthetic greywater characteristics often relate to differences in specific target real greywaters which the analogues are designed to represent. The effect of different ingredients on the greywater is demonstrated in Table 4.2. The Anglian Water matrix was designed to represent a greywater from domestic baths, showers and handbasins. Thames Water matrix consisted of synthetic soap only, simulating handbasin discharge from a commercial building (Section 2.5.3.2). The organics are mostly in soluble form and the COD:BOD ratio at 2.6:1 is comparable to that of sewage (Section 2.2.2). The Loughborough University scheme values are significantly higher than those of the other two schemes (Table 4.2) because a range of ingredients were used so as to represent greywater from baths, showers, handbasins and washing machines in a household.

Reproducibility

The reproducibility of the greywater quality arising from Matrix no.2 (Table 3.1, Section 3.2.2.2) was assessed by preparing a greywater batch 24 times and measuring the basic parameters of BOD_5, COD, SS, turbidity, pH, phosphorus, ammonia and nitrate of the fresh mixtures within 10 min from preparation (Table 4.3). Different shampoo and soap than those of Gunstead (1998) were used for economic reasons.

Table 4.3. Repeatability of Matrix no.2 (24 samples).

Parameter	Mean ± standard deviation
COD (mg l^{-1})	181 ± 37
BOD_5 (mg l^{-1})	113 ± 42
COD:BOD (-)	1.9 ± 1.1
SS (mg l^{-1})	31 ± 23
Turbidity (NTU)	26 ± 5
NH_3-N (mg l^{-1})	0.9 ± 0.2
NO_3-N (mg l^{-1})	7.2 ± 0.8
P (mg l^{-1})	0.0 ± 0.0
pH (-)	7.6 ± 0.2

The mean and standard deviation for COD, BOD, SS and turbidity at 181 ± 37 mg l^{-1}, 113 ± 42 mg l^{-1}, 31 ± 23 mg l^{-1} 26 ± 5 NTU respectively are more consistent than

those of the Anglian Water synthetic greywater (Table 4.2). Some of these differences may relate to the use of the specific products used in the current work, or possibly to sampling. It is not known if the Anglian Water data represented fresh greywater or not.

4.2.2.2 Product effect

Personal hygiene products generally contain similar chemical ingredients. To identify any major quality differences, greywater batches were prepared using the same matrix (no.2, Table 3.1, Section 3.2.2.2) but different commercial products, selected on the bases of different cost. As these batches were prepared once, the results can be regarded as an indication only. The chemical compositions of the ingredients are shown in Table C.4 in Appendix C.

At BOD_5 and COD values of 69-78 mg l^{-1} and 144-178 mg l^{-1} respectively greywaters prepared with the more expensive products had an organic concentration nearly twice that of the one with the least expensive products at 30 mg l^{-1} and 88 mg l^{-1} (Figure 4.1). The COD:BOD ratio of the samples was 2.0-3.0:1, with the highest ratio being in the greywater with the inexpensive products and the lowest in those prepared with the mid-range (no.2) and expensive products. As the solids concentration at 11-17 mg l^{-1} was similar in all cases (Figure 4.1), differences in the COD and BOD appear to have arisen from the soluble fraction of the components.

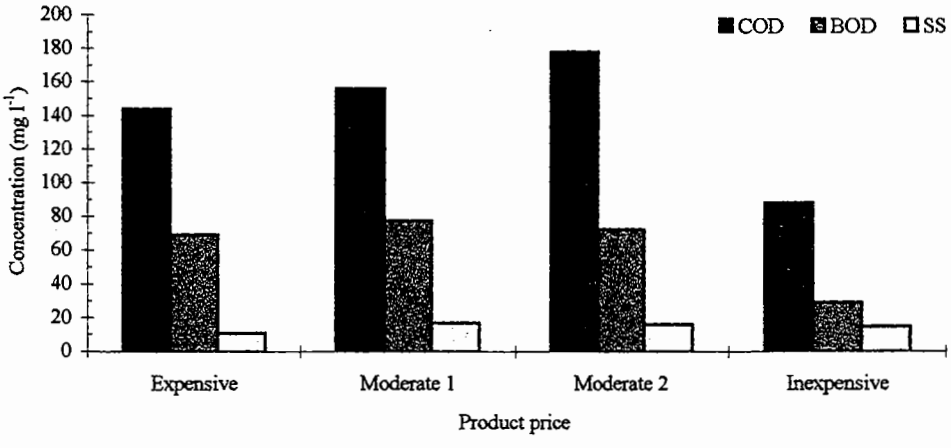


Figure 4.1. COD, BOD and solids of the fresh synthetic greywaters prepared with different products.

The mid-range product yielded water of a higher turbidity at 22-24 NTU than the other two samples at 14 NTU. Phosphorus at around 1 mg l^{-1} was found only in the mid-range products (Figure 4.2). Both these and the less expensive products had ammonia at around 2 mg l^{-1}, whilst the most expensive products contained least ammonia at 0.2 mg l^{-1} (Figure 4.2). Smaller differences in terms of nitrite (7.6-8.4 mg l^{-1}) and pH (7.5-7.8) levels were measured in the greywater samples.

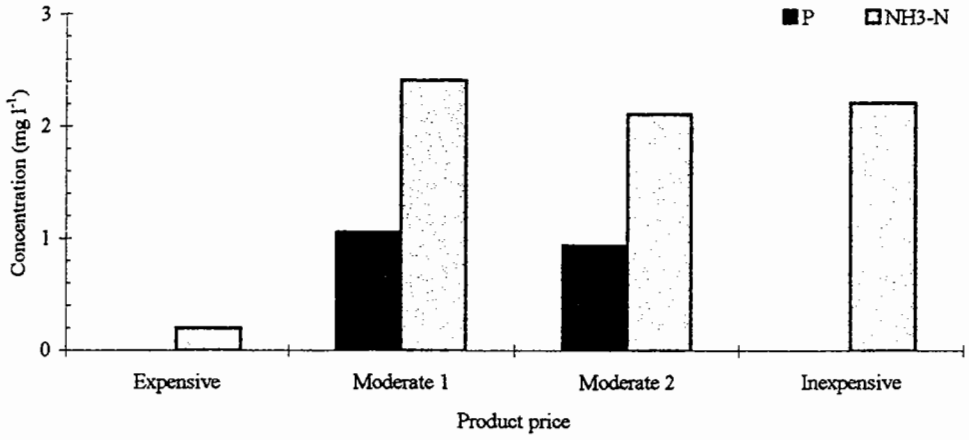


Figure 4.2. Phosphorus and ammonia of the fresh synthetic greywaters prepared with different products.

4.2.3 Storage

Small-scale, air-mixed synthetic greywater samples were stored at ambient temperature for 5 and 14 days (Section 3.3.2.1) so as to identify a basic pattern of degradation. Two types of greywater were prepared: the basic matrices (Table 3.1, Section 3.2.2.2) and the modified versions of Matrix no.1 (Table 4.2). This was a joint work to which G. Tricault, an overseas student, contributed by preparing and analysing some of the small-scale greywater samples. The results are analysed in the current work.

These experiments were followed by large-scale storage trials on synthetic and real greywaters where the tank contents were mixed with submersible pumps, the synthetic greywater being four times the concentration of that of Matrix no.2 (Section

3.2.2.2). By this stage the stock solution had been chosen for the pilot-scale trials (Sections 5.2 and 6.3), hence its long-term degradation was of interest. An experiment was also carried out on real greywater stored under quiescent conditions, i.e. without mixing. In the following section small- and large-scale experiments are discussed separately.

COD

The small-scale samples of the basic matrices no.1 and no.2 had initial COD concentrations in the region of 43-164 mg l^{-1}, respectively (Figure 4.3). In all cases the values declined over the first 2 days and then slowly increased.

The air-mixed modified greywater samples exhibited various COD dynamic trends during 5 days of storage (Figure 4.4). The sample with 2x oil was similar in behaviour to the basic matrices. When all the ingredients were doubled, a gradual decline occurred over a 4-day period after which the COD appeared to increase. Doubling only soap or shampoo and hair level resulted in a slower degradation, suggesting that at increased concentrations a change in COD is seen only after 2 days-or-so of storage. In general all the small-scale greywaters showed little decline in non-biodegradable matter.

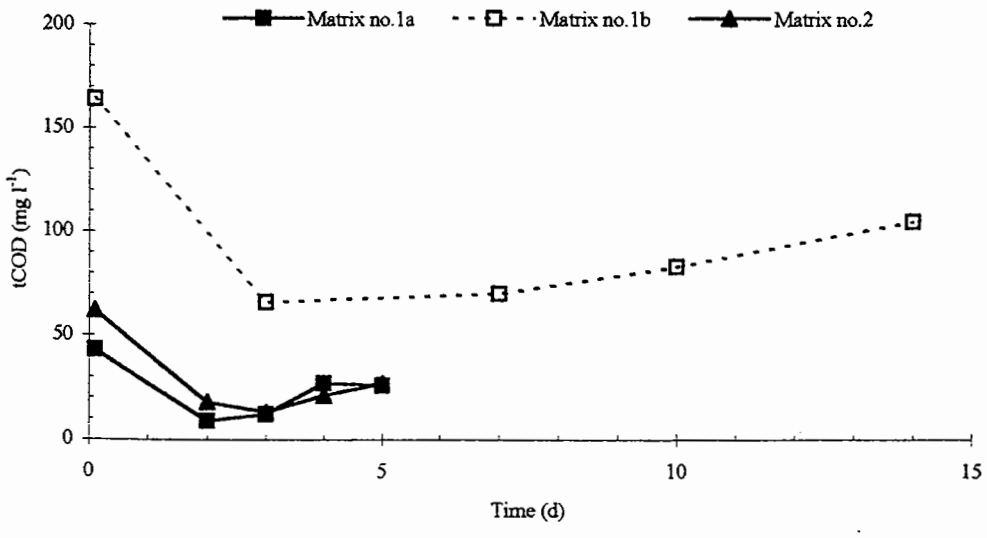


Figure 4.3. COD transient of air-mixed small-scale greywater batches prepared according to Matrices no.1 and no.2.

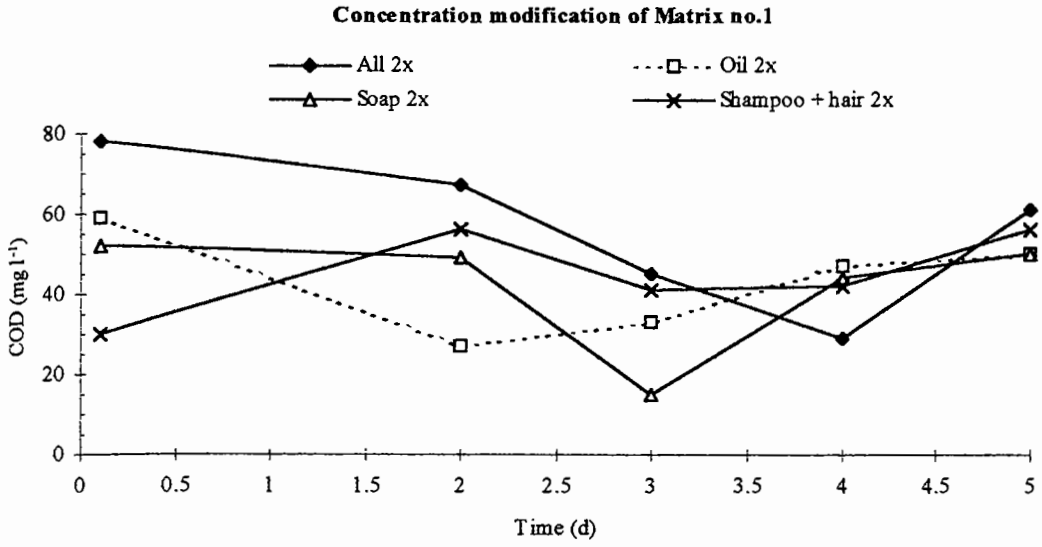


Figure 4.4. COD transient of air-mixed modified greywater samples (Matrix no.1).

The COD degradation patterns were distinct in the 18-day large-scale trials. The initial tCOD concentrations of 420 and 520 mg l^{-1} in the stirred synthetic and real greywaters respectively dropped to 87 and 80 mg l^{-1} (Figure 4.5); degradation was accelerated by vigorous stirring combined, possibly enhanced by, the corresponding temperatures of 30 ± 3°C and 26 ± 3°C (Figure C.11 in Appendix C). The tCOD of the stirred real greywater batch dropped by 36% over the first 24 h whereas that of the synthetic solution began to decline only after 3 days, taking much longer to stabilise. A possible reason for this is that the organic matter in the synthetic water, being freshly prepared, takes longer to degrade whereas the active ingredients in the real water are already partially degraded by the much higher microbial and enzyme content.

Samples from the quiescent real greywater were taken from the top 15 cm layer and the bottom 15 cm layer of the tank (Section 3.3.2.1) to evaluate degradation patterns in different parts of the storage tank. The initial concentration of 384 mg l^{-1} dropped by 14-15% over the first 24 h (Figure 4.5). After 17 days corresponding values of 156 and 217 mg l^{-1} were measured at the top and bottom of the tank (Figure 4.5). In these layers the soluble fraction of the COD declined slowly from 164 mg l^{-1} to 55 mg l^{-1} at the tank top and 70 mg l^{-1} at the base (Figure C.7 in Appendix C). The difference

between the two readings is probably due to the larger organic particulate material settling to the base of the tank. In the bottom layer the settled organic matter introduces more sCOD by dissolution (Dixon *et al.*, 1999c).

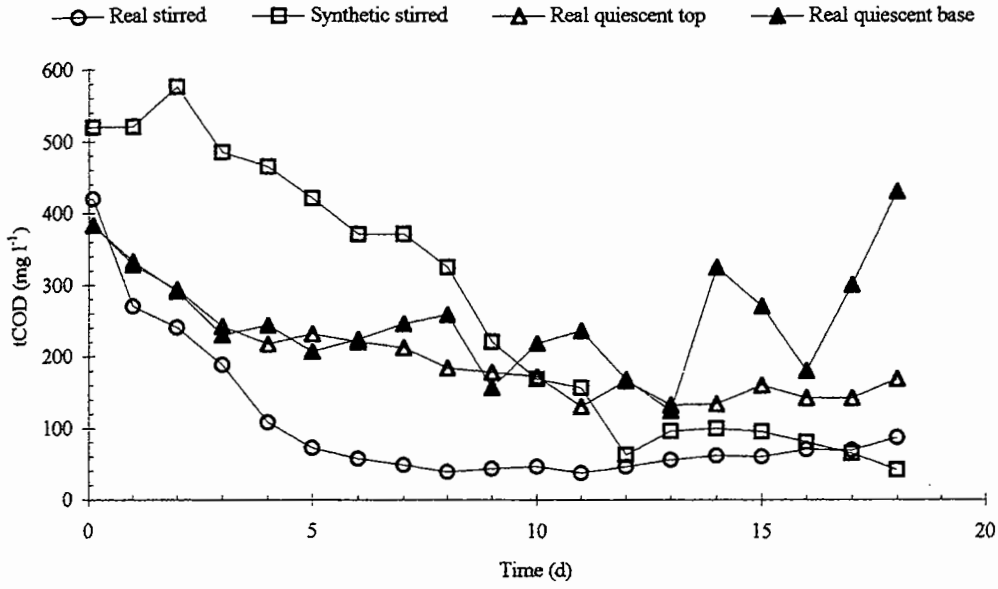


Figure 4.5. tCOD transient of large-scale real and synthetic greywaters stored under different conditions.

The temperature at 19 ± 3°C at the top and base was lower than in the stirred samples (Figure C.11 in Appendix C) where mixing elevated the temperature.

BOD

The BOD of the small-scale samples prepared using the basic matrices (Figure 4.6) decreased over a 3-day period after which the values stabilised. This was also generally the case for the modified greywaters. The biodegradable matter in the sample with 2x oil, however, appeared to be relatively stable for the first 2 days after which the BOD began to drop (Figure 4.7). It is possible that the oil hindered biodegradation in some way. The results from the small-scale samples show that the rates of BOD decline are different but in some cases the patterns are similar independent of the actual matrix and concentrations used. Odour formation was not observed in any of the small-scale samples.

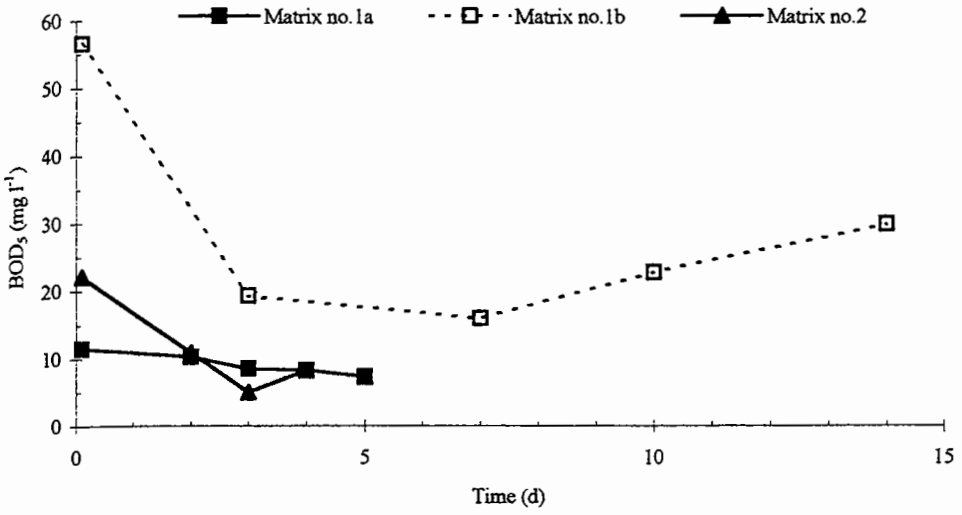


Figure 4.6. BOD_5 transient of air-mixed small-scale greywater samples prepared according to Matrices no.1 and 2.

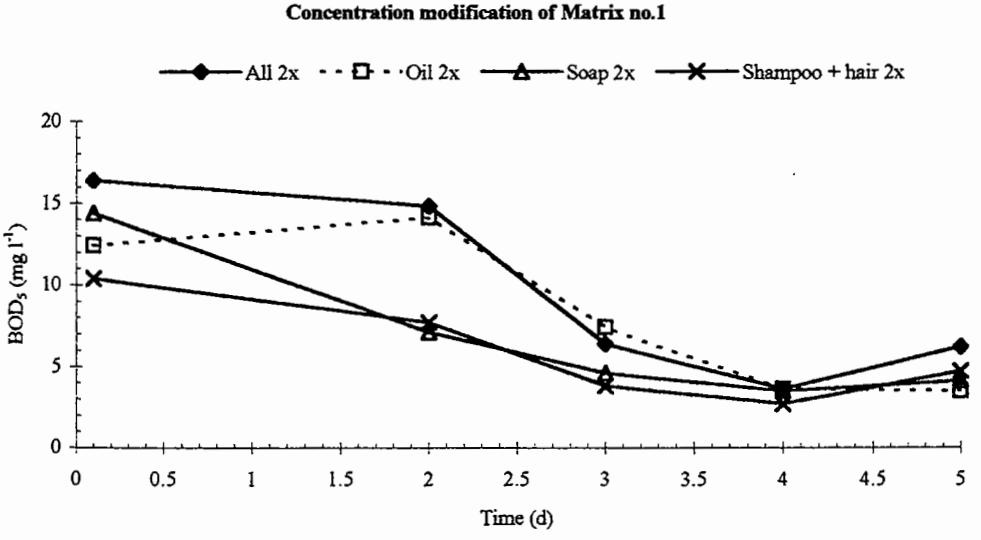


Figure 4.7. BOD_5 transient of air-mixed small-scale modified greywater (Matrix no.1) samples.

The transient BOD data of most of the large-scale studies were scattered (Figure 4.8), though generally decaying over an initial 3-day period in much the same way as the sCOD (Figure C.7 in Appendix C). The scatter related to variable SS levels (Figure C.9 in Appendix C), which also tended to decrease over the course of time.

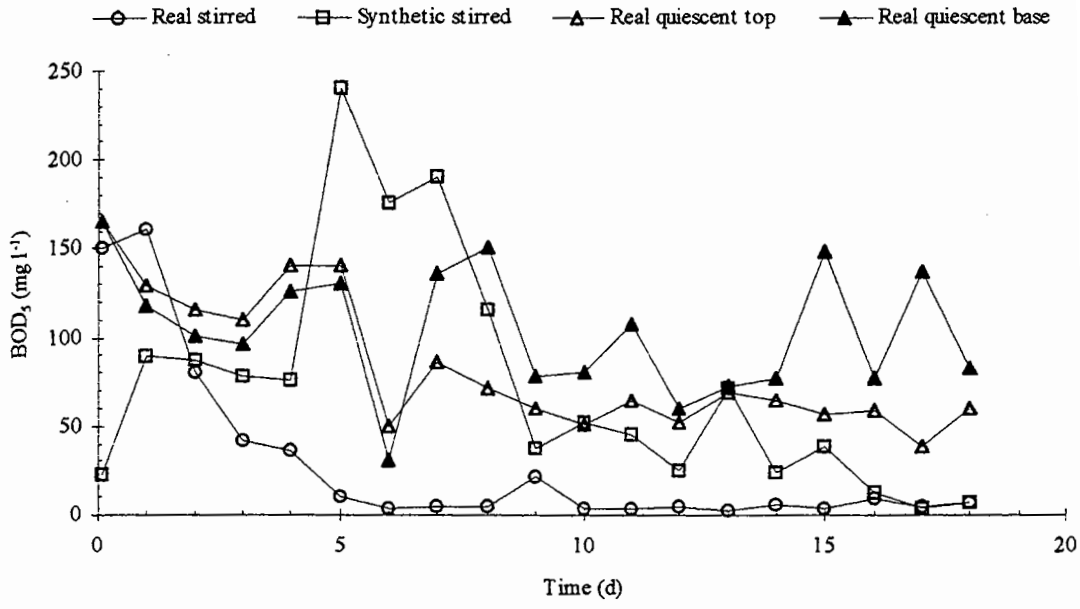


Figure 4.8. BOD_5 transient of large-scale real and synthetic greywaters stored under different conditions.

In the stirred real greywater the DO dropped from 7.8 to 4.5 mg l^{-1} over the first two days and afterwards fluctuated between 4.9 and 9.1 mg l^{-1} (Figure C.8 in Appendix C). This pattern has been reported in another study (Dixon *et al.*, 1999c). The DO of the quiescent greywater dropped from 7.6 mg l^{-1} to below 3 mg l^{-1} in a day after which the bottom layer had slightly lower DO values than the top (Figure C.8 in Appendix C). As these were grab samples taken once a day, it is possible that the DO in the tank occasionally dropped below 1.0 mg l^{-1}, hence the odour formation within a few days. Dixon *et al.* (1999c) observed similar fluctuation in the DO (0.2-3.0 mg l^{-1}) after the first drop in the initial value in quiescent real greywater. The author attributed this behaviour to the variation in ambient temperature, which seems the most probable explanation in the current work.

Solids and turbidity

In the current study trends in the suspended solids and turbidity data for all the small-scale trials were not readily discernible (Figures C.1-C.2 and C.4-C.5 in Appendix C). The solids increased in the basic matrix samples stored for 5 days (Figure C.1 in Appendix C). In contrast, turbidity in the samples stored for 5 days remained stable

(Matrix no.2) or decreased (Matrix no.1) until the second day after which the values began to rise (Figure C.2 in Appendix C). In the case of the 14-day storage trial a sharp decline from 19.7 NTU to 3.6 NTU over 3 days was followed by a steady increase (Figure C.2 in Appendix C). The solids concentration fluctuated over the entire period (Figure C.1 in Appendix C), suggesting an intermittent solids build-up on the tank walls.

In the large-scale trials the SS transient (Figure C.9 in Appendix C) was highly scattered due to the influence of two factors, the first one being the sediment layer formed at the base of the tank and the second was the scale formation on the tank walls. The soft scale layer appeared to increase in thickness over course of the 18-day storage and intermittently detach from the tank walls, to add to the suspended matter. This was also observed during the steady-state trials where spot samples of the greywater scale were taken (Section 5.4). Dixon *et al.* (1999c) reported floc formation in some of the bath samples after 4 days of storage under quiescent conditions and visible sediment more from both washing machine rinse water samples than the bath water samples after the 20-25-day experiment.

In both stirred, large-scale experiments the initial turbidity was 89-101 NTU whereas in the quiescent batch it was less than half this value (Figure 4.9). The turbidity declined rapidly over the first 5 days in the stirred real greywater whereas it declined very slowly in the quiescent tank. In sharp contrast, the turbidity of the synthetic matrix remained at a high level for several days, declining rapidly only after about 7 days of storage. After 18 days the samples from the stirred tanks and the top layer of the quiescent tank had a turbidity of 7.1-16.5 NTU whilst the bottom layer of the latter had increased to 87.4 NTU due to settlement.

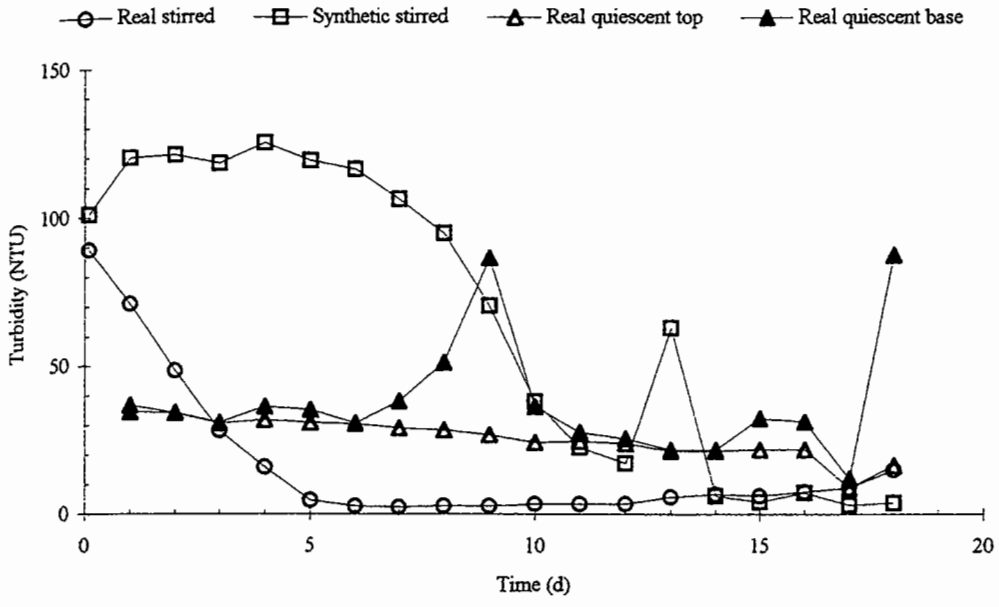


Figure 4.9. Turbidity transient of large-scale real and synthetic greywaters stored under different conditions.

pH

In Matrices no.1 and 2 the pH declined from 8.1-8.2 to 7.6 but in the modified samples it increased from 7.2-7.6 to 7.7-8.0 (Figures C.3 and C.6 in Appendix C). This parameter was most stable at 7.5-8.0 in quiescent real greywater, with little difference in readings between the two sampling points (Figure C.10 in Appendix C). In the stirred real and synthetic greywaters the pH levels had a noticeable upward trend, fluctuating between 8.1-9.6 and 7.1-7.7, respectively (Figure C.10 in Appendix C). Dixon *et al.* (1999c) also reported a small fluctuation in the pH of bath greywater samples during storage.

Pathogens

Pathogen levels were determined only for the quiescent real greywater due to the limited time available. The initial 4 log total coliforms reached a maximum of 9 log after 3-4 days of storage and remained at this level until the end of the experiment (Figure 4.8). *E.coli* counts increased over a couple of days from the 4 log to a relatively stable 6 log (Figure 4.10). The faecal streptococci at 2-7 log (Figure C.12 in Appendix C) was the most variable of the measured parameters, suggesting a pattern

of bacteria growth and decay. The high pathogen counts in all cases indicate sufficient organic matter and nutrient levels to support microbial populations during the 18-day storage period (Figures 4.10 and C.12 in Appendix C).

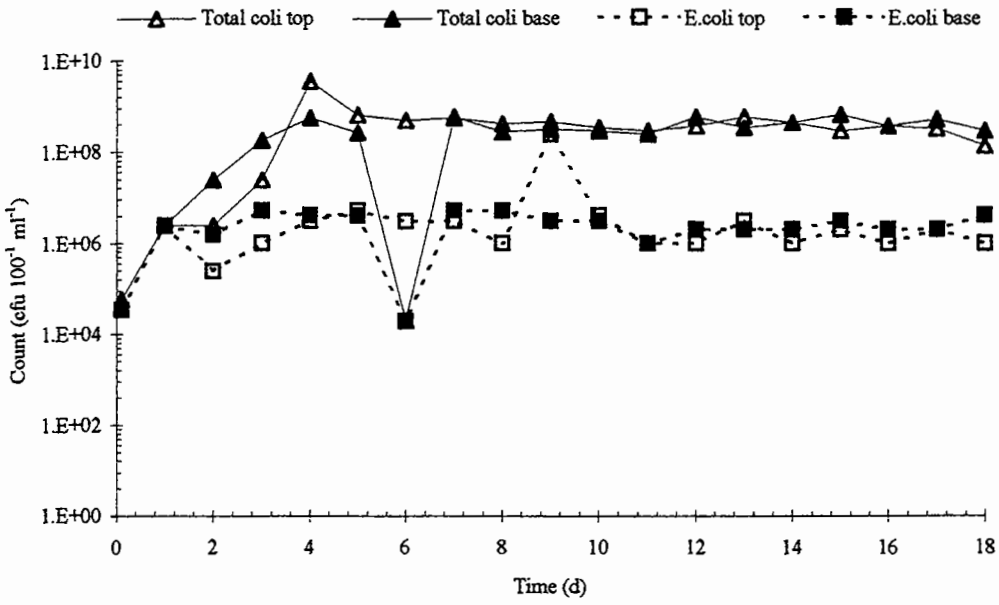


Figure 4.10. Total and *E.*coliform transients of real greywater stored under quiescent condition.

Types of storage vs. rate constants

The effect of mixing intensity on degradation is most easily quantified with reference to pseudo-first order rate constants, which in the current study were calculated in with reference to BOD and COD for both real and synthetic greywater (Table 4.4). When real greywater is mixed, the organic content degrades faster than under quiescent conditions (Figures 4.5 and 4.8). It also appears that under quiescent conditions the BOD and tCOD on the bottom of a greywater tank degrades faster than on the top of the tank (Table 4.4), possibly due to differences in the nature of the materials accumulating in these regions: the soluble fraction of COD appears to degrade at a similar rate independent of the sampling point. When greywater is mixed, >80% of the tCOD degrades over the first five days of storage after which the level remains relatively stable (Figure 4.5). If greywater is stored under quiescent conditions, the degradation is slower and less complete with a reduction of only about 65% of the tCOD.

Table 4.4. Summary of greywater degradation during storage.

Storage conditions	DO (mg l^{-1})	BOD rate constant (d^{-1})	COD rate constant (d^{-1})
Real stagnant, top of tank	1.3-7.6	0.1364 (days 0-3)	0.0906 total (days 0-12) 0.1079 soluble (days 0-10)
Real stagnant, bottom of tank	1.3-7.6	0.1825 (days 0-3)	0.1246 total (days 0-5) 0.1018 soluble (days 0-5)
Real stirred	4.5-9.1	0.6214 (days 0-6)	0.3352 total (days 0-6)
Synthetic stirred	-	0.2436 (days 5-8)	0.1631 total (days 1-12)
Synthetic air-mixed	-	0.2352*	-
Synthetic 2x strength air-mixed	-	0.3137 (days 0-3)	0.2473 total (days 0-4)
Synthetic 2x oil air-mixed		0.3092 (days 0-4)	-
Synthetic 2x soap air-mixed	-	0.3536 (days 0-4)	0.4144 total (days 0-3)
Synthetic 2x shampoo air-mixed	-	0.3371 (days 0-4)	-

* average of three trials

The results (Table 4.4) indicate that the method of mixing the samples, i.e. by pump or air, has little effect on the BOD degradation rate of synthetic greywater but that stirring motion and temperature significantly enhance the degradation process. Stirred synthetic greywater degraded slower than stirred real greywater but faster than quiescent real greywater (Table 4.4). 80% of the tCOD in the synthetic matrix declined over 14 days, after which the concentration remained relatively stable (Figure 4.5), hence there appears to be a delay in the degradation process compared to real greywater. The time dependency is an important finding, in particular for greywater storage modelling purposes if based on a simulant greywater.

In the air-mixed samples the sample with 2x shampoo had the highest BOD degradation rate, followed by 2x soap, the double strength matrix and 2x oil, with the lowest rate being for the basic matrix (Table 4.4). In some cases the initial concentration of these modified matrices was similar to that of the basic matrix (Figures 4.6-4.7). The higher degradation rates were possibly due to the type of compounds present.

4.3 Nutrient addition

4.3.1 Nutrients in biological processes

Supplementation with macronutrients can enhance degradation of organics in STWs treating nutrient-limited wastewater (Singleton, 1994; Valo *et al.*, 1985). Adequate trace nutrient concentrations, including metal ions and vitamins, are required to support all the genera present in the biological treatment system in order for a diverse community to survive. The lack of specific nutrients can result in an unbalanced mixed culture, leading to a poor quality effluent, reduced treatment efficiency and sludge handling problems (Blackall *et al.*, 1996; Pala and Sponza, 1996; Soddell and Seviour, 1996). A clear example of this can be seen with activated sludge, where pin point flocs, dispersed growth and poor flocculation have been attributed to deficiencies in trace elements and have been shown to be ameliorated by nutrient dosing (Gostick, 1991). Chemical effects of metal ions on activated sludge in other wastewater applications such as in treatment of phosphorus-limited industrial effluent (Burgess *et al.*, 1999a and b) are well known whilst knowledge on the mechanistic roles of micronutrients in aerobic bacterial cells is scarce.

Compared to municipal and industrial wastewater, evaluating the importance of nutrients on greywater treatment is in its early stages. A large range of trace metal concentrations are present in greywater from different sources due to the detergents, shampoos, household plumbing and collection tank materials used (Christova-Boal *et al.*, 1996; Hypes, 1974). The aim of this work was to investigate the feasibility of optimising the biological treatment of greywater through nutrient supplementation. This was a joint work, a substantial part of which was carried out by A. Pichon, a visiting student. As the work has a significant influence on the data regarding the general body of the work, it is discussed in the following.

4.3.2 Respirometry work

The missing or deficient trace elements as well as nitrogen, phosphorus and sulphur in synthetic and real greywaters as compared to theoretical requirements of micro-organisms were identified. Previous findings (Section 4.2) related to low nitrogen and phosphorus concentrations in greywater were confirmed with a new set of samples. The analysis (Section 3.3.2.2) carried out also showed that real and synthetic greywaters were deficient in aluminium (Al), copper (Cu), cobalt (Co), iron (Fe), molybdenum (Mo) and zinc (Zn) (Table C.3 in Appendix C). A list of supplementary nutrients required are presented in Table 4.5

Table 4.5. Nutrient additions to the greywater samples.

Nutrient	Dose added to real greywater (mg l^{-1})	Dose added to synthetic greywater (mg l^{-1})
N	10.00	10.00
P	1.63	2.95
S	none	none
Ca	none	none
K	none	none
Fe	0.39	0.39
Mg	none	none
Mn	none	none
Cu	0.04	0.05
Al	0.05	0.05
Zn	0.97	1.00
Mo	0.70	0.70
Co	5.00	5.00

The respirometry work was carried out on N/P-balanced and -limited real and synthetic greywaters as described in Section 3.3.2.2. The oxygen uptake and COD removal rates are shown in Table C.4 in Appendix C. The results shown in Figure 4.8 are presented as a percentage of the control value. It is assumed that the control value represents the state of the sample before nutrient addition. This percentage is therefore a measure of the effect of adding of a nutrient. This method allows direct comparison between results from different tests (Figure 4.8).

Four types of phenomena were observed in the samples:

- stimulation indicated by increased COD removal and increased or unchanged oxygen uptake,
- metabolic inhibition indicated by decreased COD removal and oxygen uptake,
- uncoupling of metabolism by decreased COD removal and increased oxygen uptake; anabolism is inhibited whereas catabolism is not, and
- chemical reactions indicated by increased COD removal and decreased oxygen uptake; increased adsorption of COD-imparting substrate components onto the bacterial cell walls.

4.3.3 Results

In the following the potential mechanism of each nutrient is discussed with each test.

Nutrient additions on real and synthetic greywaters: N and P

Addition of nitrogen or phosphorus to N/P-limited real greywater resulted in stimulation of the biomass (Table 4.6, Figure 4.11), as expected with reference to the measured C:TN:P ratio (Section 3.3.2.2). This beneficial effect was most clearly observed in the improvement by 160% and 213% respectively, in the COD removal for nitrogen and phosphorus supplements. The corresponding improvement in respiration rates were 120% and 109% in real greywater. Addition of nitrogen to synthetic greywater was also stimulatory, with the oxygen uptake being 106% and COD removal 150% of the concurrent control (Table 4.6, Figure 4.11). Adding phosphorus to synthetic greywater, however, led to effects indicative of adsorption/coagulation mechanisms at a COD removal 88% and respiration rate 125% of the control (Figure 4.11). This suggests that the sludge receiving synthetic greywater was under nitrogen-limited conditions such that the overall imbalance remained nitrogen controlled.

Table 4.6. Effects of nutrient addition on real and synthetic nutrient-balanced and -limited greywaters.

	Real greywater		Synthetic greywater	
Nutrient	N/P-balanced	N/P-limited	N/P-balanced	N/P-limited
Co	stimulatory	stimulatory	chemical	chemical
Cu	stimulatory	stimulatory	inhibitory	stimulatory
Fe	chemical	stimulatory	stimulatory	chemical
Mo	stimulatory	stimulatory	stimulatory	chemical
Zn	stimulatory	chemical	inhibitory	stimulatory
Al	inhibitory	chemical	inhibitory	stimulatory
P	-	stimulatory	-	chemical
N	-	stimulatory	-	stimulatory

Co

Addition of Co to all types of greywater increased the COD removal but did not appear to be beneficial to respiration rate in all cases. As shown in Table 4.6 and Figure 4.11, real greywater was stimulated by Co: the COD removal of nutrient-balanced real greywater increased to 130% of the control and that of its nutrient-limited counterpart was 131%. The oxygen uptake rate increased to 118% and 103% of the control in nutrient-balanced and -limited real greywaters, respectively. Chemical reactions resulting from the addition of Co were observed in both nutrient-limited and -balanced synthetic greywater. In the former the respiration rate decreased to 95% and the COD removal increased to 135% of the control whereas in the latter the corresponding values were 81% and 100% (Table 4.6, Figure 4.11).

Co has been shown to stimulate metabolism in other studies. As a metallic enzyme activator it is known to be required by some bacteria for synthesis of cyanocobalamin (vitamin B_{12}), and is also used in the production of carboxypeptidase (Wood and Tchobanoglous, 1975). Stimulatory action on metabolism is suggested by the fact that Co-chloride and Co-sulphate can stimulate the synthesis of cyanocobalamin and some other B_{12}-like factors in aerobically fermenting sludge (Sathyanarayana Rao and Srinath, 1961). The addition of <1.0 mg l^{-1} Co has shown potential in optimising the activated sludge process, although increases in metabolic activity may not be

associated with any improvement in BOD or COD removal, nor linked to any change in the nitrogen and phosphorus content of the sludge (Sathyanarayana Rao and Srinath, 1961). In the current study these observations on nitrogen and phosphorus interactions are confirmed for real greywater.

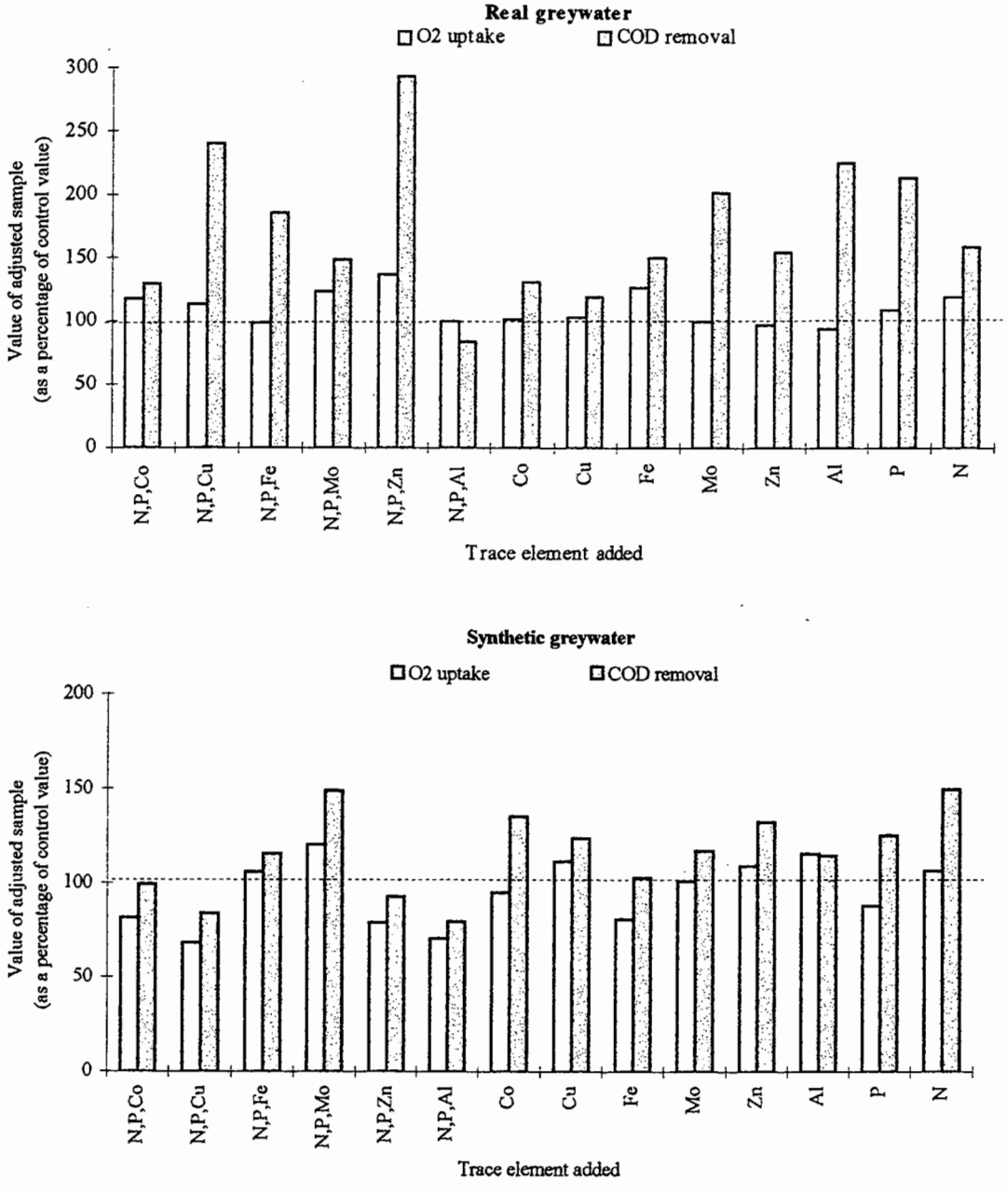


Figure 4.11. The mean oxygen uptake and COD removal of real and synthetic greywaters with and without nitrogen/phosphorus balancing, as a percentage of the control value. The line at 100% represents the control.

Cu

As expected in Cu-limited substrates, Cu addition resulted in metabolic stimulation in both real greywater types, with a COD removal increase by 240% in the case of the balanced real greywater and 119% in its nutrient-limited counterpart (Table 4.6, Figure 4.11). In both real greywater types the oxygen respiration rate increased, though not as much as the COD removal (Figure 4.11). Stimulatory effects were observed also in the case of nutrient-limited synthetic greywater, with the increase in COD removal and oxygen uptake rate being 124% and 111%, respectively (Table 4.6, Figure 4.11). In sharp contrast, toxic effects were observed in balanced synthetic greywater, indicating a different level of sensitivity even to the low concentrations of Cu in this work (Table C.3 in Appendix C).

Two mechanisms relating to Cu addition have been discussed in previous studies. Toxic effects on activated sludge are common when Cu is found in wastewater streams from industrial and domestic sources (Gracia *et al.*, 1994). This is because Cu forms complexes with enzymes and other metabolic agents connected with respiration such that they become inactive (Gray, 1989). The beneficial effects of low doses of Cu in wastewater have been demonstrated by Vandevivere *et al.* (1997) who added 0.6-3.0 mg l^{-1} Cu to industrial wastewater, showing that concentrations above the accepted toxicity threshold can chelate other toxins in the effluent and improve process performance. However, the possibility of the stimulatory effects being temporary cannot be ignored, as the effects of Cu addition have been shown to diminish as the metal becomes adsorbed onto other binding sites (Vandevivere *et al.*, 1997).

Fe

Fe dosing showed irregular effects on the biomass fed with the different types of greywaters, though in all cases the COD removal increased. In the nutrient-limited and -balanced real greywaters it was 150% and 185% of the control, respectively, whilst in the limited and balanced synthetic greywaters the corresponding COD removal changes were 103% and 115% (Table 4.6, Figure 4.11). As in the case of Co,

differences in effects were observed in the respiration rates. Stimulatory effects on addition of Fe to nutrient-limited real and nutrient-balanced synthetic greywaters were observed, the corresponding oxygen uptake rates being 126% and 106% (Table 4.6, Figure 4.11). Chemical reactions indicated in the case of balanced real and limited synthetic greywaters such that the respiration rates decreased to 98% and 81%, respectively (Table 4.6, Figure 4.11).

One possible explanation for these observations relates to the fact that small quantities of Fe^{3+}-reducing bacteria are found in most activated sludge mixed cultures as Fe^{3+} and sulphate reduction are important in floc formation. Fe-based co-precipitants are commonly used in aerobic biological wastewater treatment. Fe^{3+} forms strong complexes with the condensed phosphates, pyrophosphate and tripolyphosphate, which may then be removed by adsorption onto Fe^{3+} hydroxophosphate surfaces (Jenkins *et al.*, 1971). Previous work in which Fe-based co-precipitants were added to an activated sludge system revealed adverse effects on COD and SS removal, although the most significant changes were a floc size and structure rather than microbial population (Clark *et al.*, 1999). In the current study, however, the effects of Fe indicate chemical reactions rather than biological effects, possibly partly because of the low phosphorus concentration in greywater. This confirms the previous findings in literature and suggests that Fe may not be a limiting nutrient in terms of bacterial metabolism.

Mo

Mo, commonly found as a limiting nutrient in sewage (Grau, 1991), had large stimulatory effects on both nutrient-deficient and -balanced real greywater with respective COD removal improvements of up to 200% and 150% (Table 4.6, Figure 4.11). Also in the case of balanced synthetic greywater the COD removal increased to 150% whereas conclusive effects were not seen in nutrient-limited synthetic greywater though the COD removal (117%) appeared to improve (Table 4.6, Figure 4.11).

The mechanistic effects of Mo are poorly understood but it is believed to enhance wastewater treatment by the activated sludge process. Burgess *et al.* (1999b) found Mo to have beneficial effects on activated sludge treating recalcitrant industrial wastewater.

Zn

A range of effects of the addition of Zn to greywaters were observed. Nutrient-limited real greywater showed chemical effects at a normalised COD removal of 154% and respiration of 97% (Table 4.6, Figure 4.11). Biomass in nutrient-balanced real and nutrient-limited synthetic greywaters were stimulated by the addition of Zn, such that the respective COD removals were 290% and 132% of the control (Table 4.6, Figure 4.11). Zn was inhibitory in balanced synthetic greywater, resulting in a reduction of COD removal to 80% and oxygen uptake rate of 79% (Table 4.6, Figure 4.11).

Zn has been demonstrated to non-competitively inhibit bacterial metabolism (Gökçay and Yetis, 1996) and in these instances inhibited enzyme kinetics can be applied to the metabolism of the cells. Zn interacts with other metals, such as Cu, to exacerbate their effects and has been shown to reduce the rates of reactions so that the kinetics of biodegradation in the presence of zinc and copper resembled those of biodegradation without metal addition, but with a very short sludge age (Beyenal *et al.*, 1997). The action of Zn on biomass is strongly influenced by the chemical conditions. As Zn and Cu were detected in real but not in synthetic greywater, interactions were expected to occur in real greywater.

Al

Al addition demonstrated a significant impact on biological activity. Chemical effects were observed in nutrient-limited real greywater after addition of Al, resulting in the COD removal and respiration of 225% and 94% of the control, respectively (Table 4.6, Figure 4.11). Addition of Al to nutrient-limited synthetic greywater led to stimulation of biomass with normalised COD removal at 115%. Al in balanced real and synthetic greywaters, however, inhibited the metabolism of the biomass resulting

in a significant drop of COD removals to 85% and 80% respectively, whilst the oxygen uptake of the former remained the same and that of the latter decreased to 70% (Table 4.6, Figure 4.11).

The mechanism of the action of Al in aerobic bacteria is not known. The findings from the current study suggest that if Al is required, the toxicity threshold for greywater is lower than previously thought for sewage. Al has varying coagulation effects that are dependent on the prevailing chemical conditions.

Effects of greywater source and N/P balancing

Paired t-tests were applied to the respiration and COD removal rates calculated per unit MLSS, and also to the normalised data. Eight tests were carried out with confidence limits of one standard deviation (Table C.5 in Appendix C). Significant differences were found when (a) comparing respiration and COD removal rates of sludge with and without N/P balancing, and (b) comparing data obtained with real and synthetic greywater. Although there was no significant effect on the rates per unit MLSS per day, it was observed that in the normalised data the biomass receiving synthetic greywater showed a greater response to nutrient dosing than that supplied with real greywater. The results show that the presence or absence of nutrient balancing in the respirometer does not affect the normalised data, but has a significant effect on the COD removal and respiration rates per unit MLSS regardless of the source of greywater. This indicates that nitrogen and phosphorus play vital roles in the biodegradation of greywater pollutants.

Reproducibility of trends

The repeatability of the duplicates was quantified by dividing the difference between duplicates by twice the mean, a method which indicates the percentage repeatability of a measurement. Percentage values in the range -10.0% to +10.0% were taken as an indication of a good reproducibility.

The respiration and COD removal rates per unit MLSS per unit time showed that sludge supplied with synthetic greywater required more oxygen and degraded more COD than sludge receiving real greywater under every dosing regime (Table C.10 in Appendix C), although a t-test at a 95% confidence level showed this difference not to be significant (Table C.9 in Appendix C). It can also be seen that both COD removal and oxygen uptake rates increase on the addition of N/P to any dosing regime, indicating significantly increased metabolic activity of the sludge. The results obtained during tests using unbalanced greywater were more reproducible than tests using N/P-balanced greywater, and synthetic results were more reproducible than real results (Table C.10 in Appendix C). Results from tests using Co, Fe, Mo and Al were the most variable with both types of greywater. In all cases except Zn addition without N/P balancing, respiration was found to be less reproducible than COD removal rates. The most noticeable trend was the lack of reproducibility in data obtained from N/P-balanced tests compared to data from unbalanced greywater tests.

4.4 Summary

Analysis of greywater samples collected from showers, baths and handbasins revealed the handbasin samples to be most concentrated due to the small water volumes used whereas the bath samples were most diluted. Generally the organic and nutrient content was similar to those reported in previous studies (Butler *et al.*, 1995; Surendran and Wheatley, 1998). The largest variation in the measured parameters was found in the pathogen levels, as previously found by Surendran and Wheatley (1998) who reported total and faecal coliforms of 6 ± 6 log and 2 ± 2 log, respectively. Data from the current work (Table 2.3, Section 2.2.2) suggests that categorisation of greywater on the basis of strength may be of limited value.

Variation within a controlled sample shower and bath samples appeared to be significant for many water quality determinants (Table 4.1), and arises from such factors as the personal hygiene products used and the cleanliness of the person at the

time of washing. These factors may have some impact on greywater quality even if water volume and the amount of hygiene product is controlled. Overall the shower samples were lower in COD, BOD, SS and turbidity than the respective control samples but higher in terms of total coliforms (Table 4.1). The bath samples were the most contaminated of all greywater samples in terms of COD, BOD and total coliforms (Table 4.1).

Hygiene products influence the greywater quality (Christova-Boal *et al.*, 1996) though consumer behaviour and life-style have a greater influence (van der Wijst and Groot-Marcus, 1999). In the current study the exact amounts of personal care products used were not recorded but the types of products used were listed (Table C.1 in Appendix C). These suggested similarities in behaviour such as common use of shower gel and liquid soap instead of bar soap, which none-the-less lead to greywater samples of a variable quality (Table 4.1). Single controlled samples prepared with products representing different price ranges indicated that the chemical ingredients have some impact on the bulk water quality determinant levels (4.2.2.2).

Simulating real greywater is made difficult by its site specificity. Published information on synthetic greywaters (Gunstead, 1998; Smith *et al.*, 1999b; Surendran and Wheatley, 1998) demonstrates that a number of matrices exist which are specific to a particular site. However, small-scale synthetic greywaters (Table 4.2) did not appear to represent the real counterparts in the current work (Table 4.1) and gave a substantially different water quality analysis to many of those quoted in literature (Table 2.3, Section 2.2.2). The basic synthetic greywaters had relatively low bulk determinant levels (Table 4.2), concurring with findings from multiple-occupancy housing estates (Ward, 2000; Ward *et al.*, 2000) where COD, turbidity and phosphorus values of around 52-95 mg l^{-1}, 1.9-2.3 NTU and <0.5 mg l^{-1} were measured. The replicate greywater quality measurement (Table 4.3) correlated well with real greywater data from a multiple-occupancy property where COD, turbidity and ammonia of 50-298 mg l^{-1}, 11-29 NTU and ammonia of <0.5-4.3 mg l^{-1} were reported (Ward, 2000).

A four-fold increase in the standard synthetic greywater concentration (Table 4.3) lead to better correlation with the real greywater samples from this study (Table 4.1) and those reported in the literature (Table 2.3, Section 2.2.2). In the modified versions of the basic recipes either individual or all components were increased in concentration. The resulting COD values (Table 4.2) were similar to other real greywater data collated both in the current work (Table 4.1) and other studies (Brandes, 1978; Butler *et al.*, 1995; Christova-Boal *et al.*, 1996; Olsson *et al.*, 1968; Sayers *et al.*, 1999; Surendran and Wheatley, 1998). The dynamic behaviour of the synthetic greywater during storage, however, is somewhat different to the real greywater. The synthetic greywater showed an induction period of a few days prior to commencement of degradation whereas the pollutant concentrations in the real matrix began to decline immediately (Section 4.2.3). The likely reason for this is the presence of microbes and enzymes in the real greywater.

The sampling points of greywater stored under quiescent conditions vary from one study to another, though are often located at the top of the tank: Dixon *et al.* (1999c) sampled 5 cm from the top of the tank, Rose *et al.* (1991) from the top of the storage sump, Brandes (1978) below the liquid level from a greywater septic tank. In the current work the samples from the quiescent greywater were taken 15 cm from the top, being comparable to the reported data, and 15 cm from the bottom of the tank.

Dixon *et al.* (1999c) identified settlement of suspended particulate matter and depletion of DO as being the dominant processes in the first few hours of storage of untreated real greywater. The rapid increase in coliform counts suggest that depletion of DO is probably caused by growth of aerobic biomass. Two other possible processes suggested, though not directly observed by the author, were identified as reaeration of DO through water surface and release of soluble COD (sCOD). An example of the latter is a release of sCOD from fatty acids in settled degradable particulate matter. To represent aerobic growth, decay and hydrolysis, a model based on Monod-type equations, similar to those in Activated Sludge Model No.1 (Henze *et al.*, 1987), was developed and calibrated to the data from quiescent samples. Correlation with DO

was found to be better than that with COD. The former followed the same trend as the test samples: the initial DO of about 6-9 mg l^{-1} declined to around 1-2 mg l^{-1} over 24-48 h, stabilising at that level. In the case of the COD the model represented the characteristic decline and subsequent increase of greywater in many samples. However, it was generally not a good fit to the data, and the deviation varied from one sample to another. In some cases the model gave higher COD values than those measured in the samples, in others the model values were lower than those in the samples. Dixon *et al.* (1999c) emphasised that greywater quality and products used affect the predictability of changes in water quality determinants. Both the model and the measurements, which are discussed later in this section, supported the hypothesis of four major processes described earlier. In the current work the quiescent greywater samples from the top of the tank exhibited a steady decline in the tCOD rather than a decline followed by an increase as in the described model. Measurements of soluble COD from the top and base of the tank revealed that also the soluble component decreased over time. An exception to this was a high value at the base of the tank measured on day 9 (Figure C.7 in Appendix C). Therefore this pattern does not concur the model by Dixon *et al.* (1999c). There are several possible explanations to this: different types of products used, different initial COD levels and generally lower temperature in the current work (15.1-23.5°C in the top layer after the initial 28.2°C) than in that of Dixon *et al.* (1999c; 20.3-26.1°C throughout the study). On the other hand, the DO levels exhibit a similar trend though generally higher values (1.3-3.6 mg l^{-1}; Figure C.8 in Appendix C) than those in the model (0.3-2.6 mg l) after the drop in the initial DO.

The findings from the current work relating to the differences in storage conditions agree with those of Neal (1996) where real greywater was stored for 2 weeks. The initial values of around 180 mg l^{-1} and 85 mg l^{-1} of BOD and SS respectively, declined more slowly in samples stored under quiescent conditions than when air-mixed, regardless of the temperature (13°C and 23°C; Figures 4.13 and 4.15-4.17). The quiescent greywater quickly became anaerobic with the biological components deteriorating, but it took longer for the chemical quality to degrade such that this type

of storage was not recommended. Within a week the air-mixed greywaters achieved BOD and SS of 20 mg l^{-1} and 10 mg l^{-1}, respectively, declining to 10 mg l^{-1} and below 10 mg l^{-1} after another week of storage. In both quiescent and air-mixed samples the stable decline in BOD is related to the stable SS decline (Figures 4.16-4.17). A rapid degradation in COD and BOD_5 of fresh soap solution has been noted in another study (Jefferson *et al.*, 2000b) where depletion by 14-50% and 20-67%, respectively, was observed within a 24-hour storage. This was accelerated by a high temperature in the feedtank and accompanied with scaling on the feedtank walls, both of which were observed in the steady-state trials of the current work (Sections 5.2 and 5.4). The storage trials on quiescent bath and washing machine rinse waters by Dixon *et al.* (1999c) demonstrated that substantial reductions in COD and TSS associated with the settlement of solids occur over a 24-hour period. In this respect greywater can benefit from such a long storage though adverse aesthetic effects result from odour formation through the declining DO levels.

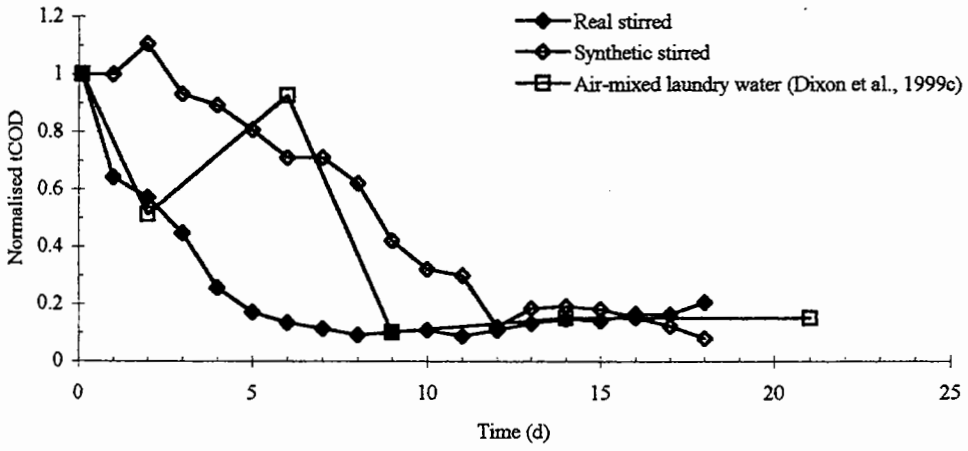


Figure 4.12. Comparison of tCOD transient in stirred/air-mixed greywater between different studies.

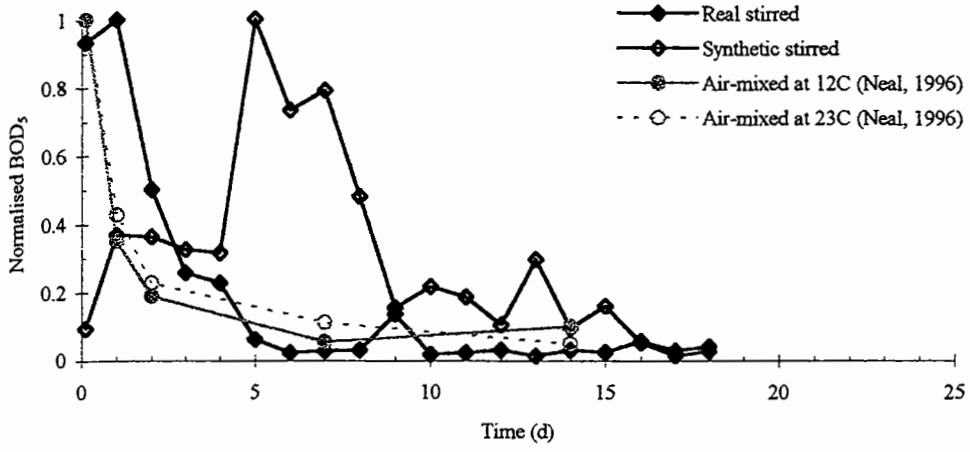


Figure 4.13. Comparison of BOD_5 transient in stirred/air-mixed greywater between different studies.

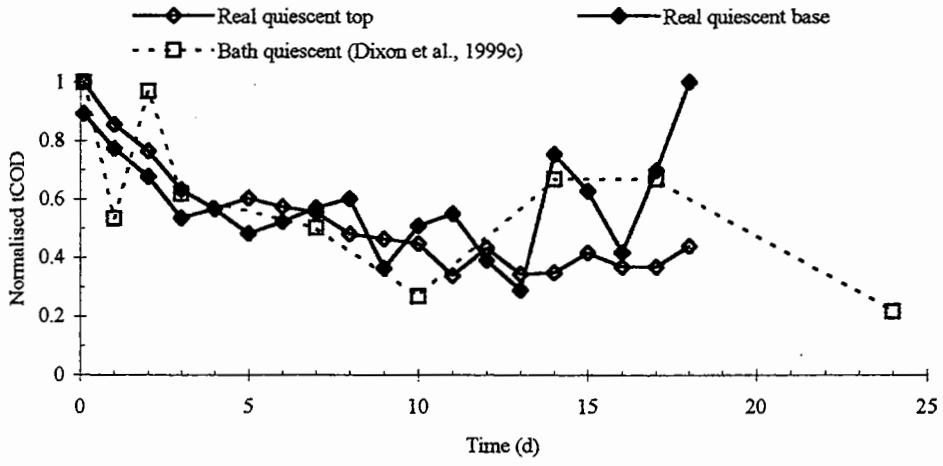


Figure 4.14. Comparison of tCOD transient in quiescent greywater between different studies.

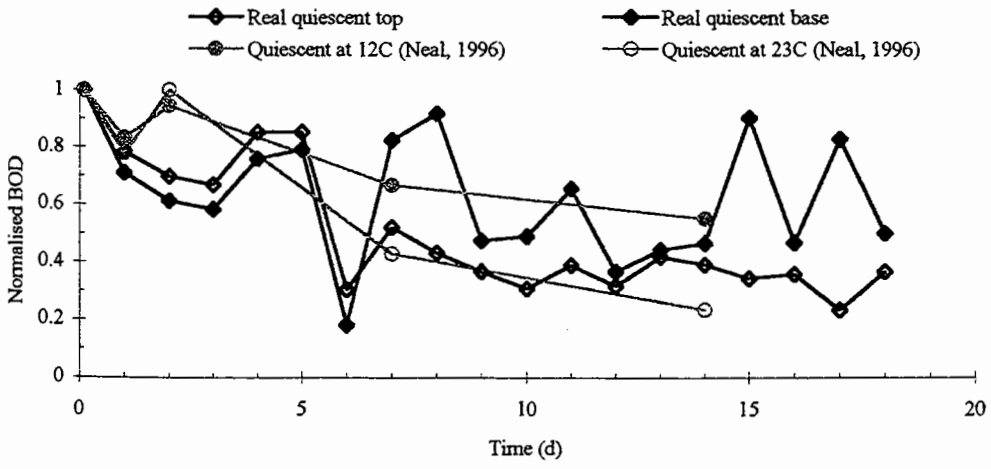


Figure 4.15. Comparison of BOD_5 transient in quiescent greywater between different studies.

n the current work solids attachment on the storage tank walls and sedimentation was)bserved both in stirred and quiescent greywater samples, resulting in variable solids evels throughout the storage (Figures 4.16-4.17 and C.9 in Appendix C). These seem ikely also in the study by Dixon *et al.* (1999c), where the SS levels in the air-mixed aundry remained relatively stable for a few days after the initial decline and in the quiescent bath water where the SS degradation was very slow (Figures 4.16-4.17). The stable SS decline in the work by Neal (1996) suggests that none of his greywater samples were subject to solids accumulation (Figures 4.16-4.17).

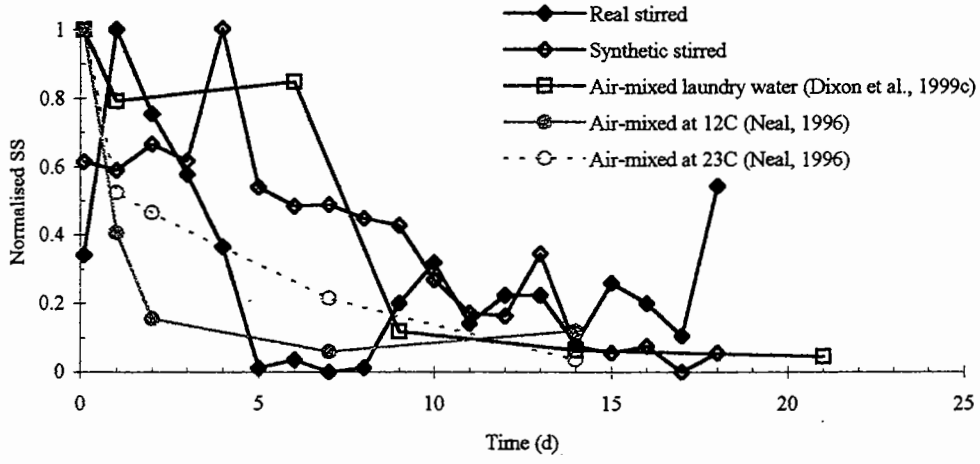


Figure 4.16. Comparison of SS transient in stirred/air-mixed greywater between different studies.

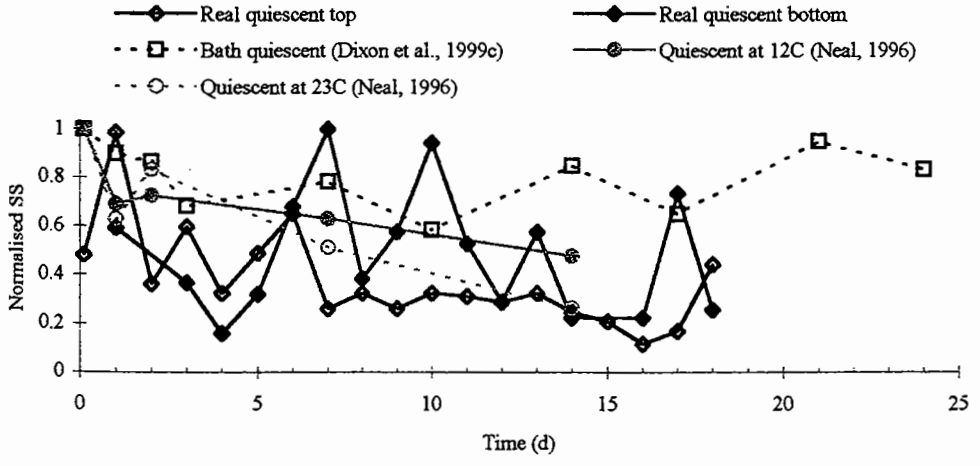


Figure 4.17. Comparison of the SS transient in quiescent greywater between different studies.

Research (Dixon *et al.*, 1999c) has shown that turbidity decline in air-mixed samples is most pronounced in cases where the initial values are high. This was also observed in the current work where reduction in turbidity of the stirred real greywater was distinct (Section 4.2.3). The stirred synthetic greywater however appeared to have an induction time of 7-8 days before the turbidity began to decline. Neal (1996) noted that within two weeks of storage the initial turbidity value in air-mixed real greywater dropped by about 95% independent of the temperature (Figure 4.18). In contrast, quiescent samples tend to exhibit slower and more variable degradation patterns than those of stirred greywater (Figure 4.19). Over the entire storage period Neal (1996) measured a decline of around 55% and 85% at 13°C and 23°C, respectively. In contrast Dixon *et al.* (1999c) observed only a 16% decline in greywater turbidity (Figure 4.19). In the current study the samples from the top of the tank showed a steady decline in contrast to the samples from the base which had generally lower turbidity but higher occasional peaks (Figure 4.19).

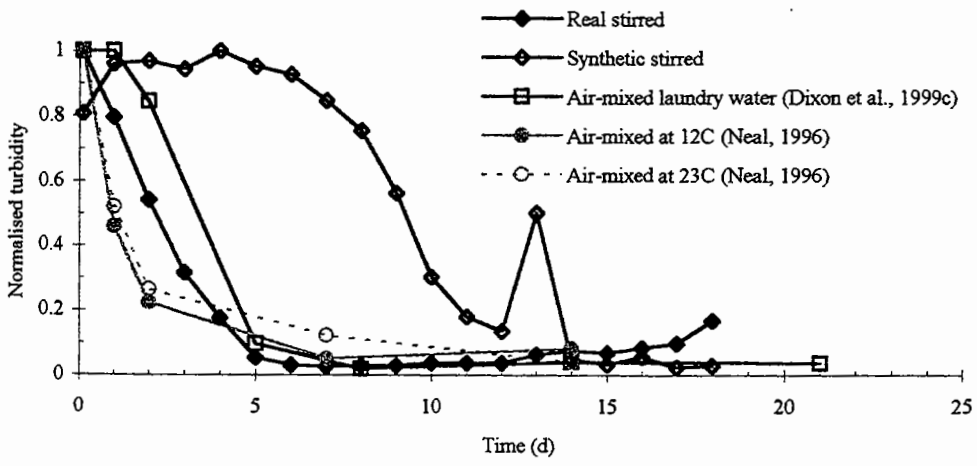


Figure 4.18. Comparison of turbidity transient in stirred/air-mixed greywater between different studies.

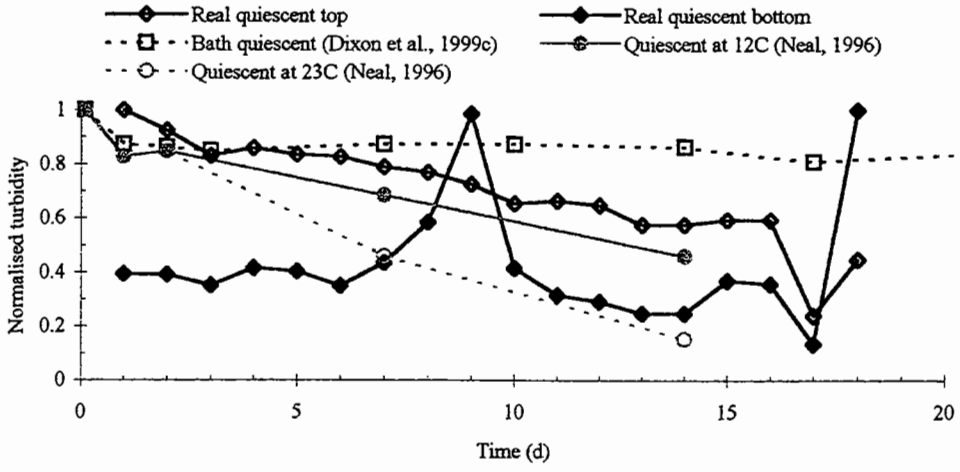


Figure 4.19. Comparison of turbidity transient in quiescent greywater between different studies.

The pathogen levels in terms of total/*E.*coliforms in quiescent real greywater increased by 2-5 log over a few days after which they remained at that level due to sufficient substrate in greywater (Figure 4.20). Increases in phosphates, ammonia and turbidity in greywater indicate that nutrients may be available for micro-organisms (Rose *et al.*, 1991). In the current study this was confirmed by the fluctuating turbidity and high bacteria levels. The findings agree with Rose *et al.* (1991) who reported a 1-2 log increase in total and faecal coliforms in greywater within the first 48 h of storage (Figure 4.20). Although this was followed by a slow decline, the numbers remained higher than those initially present even after 12 days. Brandes (1978) observed a slightly different pattern for the pathogen levels collected from a greywater septic tank. The initial total and faecal coliform levels increased to maximum values of 6 log within 24-96 h from sampling, persisting at that level several days longer than in the other studies (Figure 4.20). After this period the levels rapidly declined to below the initial values. The reason for the difference in results for pathogen survival are related to the raw wastewater composition and retention time which affect the bacteria growth (Brandes, 1978). In a study by Dixon *et al.* (1999c) the total coliform counts remained high at >5 log after the initial increase and decline in numbers (Figure 4.20). Although there appears not to be a uniform pattern for growth and survival of pathogens, the studies demonstrate that the numbers increase over the first few days.

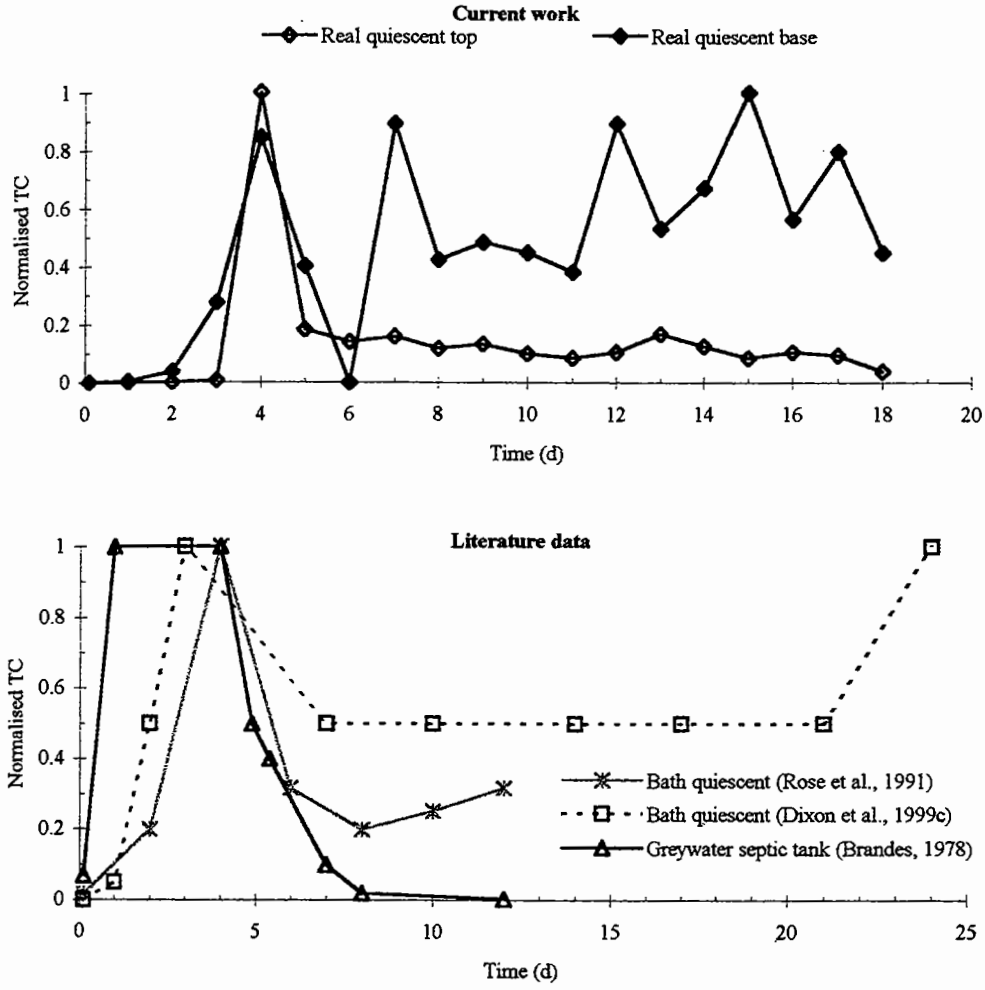


Figure 4.20. Comparison of total coliform transient in quiescent greywater between different studies.

The higher variability in tests with real greywater than in a synthetic matrix can be accounted for by the nature of the greywater. Although the real greywater was collected daily, its composition was affected by trends in water use in the source building, the wider range of possible constituents and the water level in the collection sump at the time of sampling. In contrast, the daily prepared synthetic greywater was freshly used and contained the same ingredients every time. Hence the strengths were different as well as the degradation rates (Section 4.2.3) though the latter was not investigated. Though synthetic greywater is a reasonable analogue of real greywater and more reproducible in quality, it brings the problem of significant difference in nutrient levels between these greywaters, making nutrient addition site-specific. These results imply that studies using synthetic greywater to simulate and predict

treatability in greywater recycling systems may give more consistent results than using nutrient addition to real greywater. In both real and synthetic greywaters more variability was observed in the measurement of respiration rate than in COD removal rate.

The reproducibility of the results appeared to depend on the nutrient added. Co, Fe, Mo and Al were less reproducible than the other micronutrients. This may be owing to their action on biomass as Fe and Al are known to have biochemical roles as well as varying coagulation effects that are dependent on the pH. All four metals exhibited effects ranging from stimulatory to inhibitory in the current study (Table 4.6), and have inconsistent effects on other biological wastewater treatment processes; Co-stimulatory effects being recorded by Wood and Tchobanoglous (1975) and Sathyanarayana Rao and Srinath (1961) and Co-inhibitory effects by Geradi (1986) and Hunter *et al.* (1966). A decrease in reproducibility was seen when greywaters were N/P-balanced and then dosed with micronutrients. This may be because the unbalanced greywaters were N/P-limited, such that N/P balancing caused another characteristic of the greywater to become the limiting factor.

Nutrient addition can be used as a process enhancement tool in cases where the wastewater itself is deficient in carbon or nitrogen (Sections 2.2.2 and 5.5). In the current work a range of nutrients were added to synthetic and real greywaters. The respirometry tests demonstrated increased COD removal rates in 24 of the 28 nutrient dosing regimes tested. This suggests that macro- and micronutrient balancing has a potential role in greywater treatment and recycling in the future. However, the effects of dosing with additional nutrients beyond simple nitrogen and phosphorus balancing are unclear. N/P balancing appeared to have a large impact on the relative normalised effects of each of the nutrients in the case of real greywater (Figure 4.8). The effects of the added trace elements on the nutrient-balanced and nutrient-limited synthetic greywaters were not so clear (Figure 4.8). Further, variations in results increased once nutrient balancing had occurred indicating that either:

- other substances than those tested were imposing process limitations after N/P balancing, or
- potentially toxic overdoses due to undetected variation in the existing background concentrations of micronutrients that were being dosed.

Chapter 5 Process evaluation - steady state

5.1 Introduction

Biological treatment is required to adequately remove organic pollutants from wastewater, so as to prevent regrowth in the distribution systems, that cannot be achieved solely by filtration (Section 2.4.3 and 2.4.6). This is of a particular importance in large distribution networks (Section 2.4.6).

The performance of four wastewater treatment processes was evaluated under steady-state operation during a 350-day test period. Influent quality was changed from greywater to blackwater simulating the variability of domestic wastewater. Start-up of the processes (Section 3.2.2) was followed by a four-stage steady-state trial period:

- days 1-169: greywater (the submerged membrane bioreactor (MBR), the membrane aeration bioreactor (MABR) and the biological aerated filter (BAF)),
- days 170-236: grey/blackwater (the submerged MBR and the BAF),
- days 237-300: blackwater (the submerged MBR and the BAF), and
- days 400-450: greywater (the side-stream MBR). This stage was carried out as a joint work with P. Le Clech, a visiting overseas student, who undertook the experimental work. The results are analysed in the current study.

5.2 Compliance and reliability

5.2.1 Assessment

The treatment processes were operated at feed and air flow rates recommended by the manufacturers or suppliers. Overall performance and robustness during a range of

volumetric loading rates (VLRs) were determined (Section 5.2.2). The compliance of the treatment processes was assessed in two principal ways from the grab samples. Firstly, the removal efficiencies were used as an indication of the performance. Secondly, the effluent quality was compared to the existing water quality standards (Table 5.1) such that the concentration distribution of the samples was examined.

Table 5.1. Water quality standards chosen as reference.

Parameter	Standard	Limit
BOD_5	US EPA	10 mg l^{-1}
Suspended solids	US EPA	5 mg l^{-1}
Turbidity	US EPA	2 NTU
Total coliforms	UK bathing water	500 cfu in 100 ml *
	US EPA	non-detectable
Faecal streptococci	UK bathing water (faecal coliforms)	100 cfu in 100 ml *
and *E.coli*	US EPA (faecal coliforms)	14 for any sample, 0 for 90% samples

* guideline

5.2.2 MABR

Operational difficulties experienced with the MABR reduced the length and type of trials to a relatively short period of exclusively greywater treatment. The start-up of the process was complicated by foaming. Additionally, surfactants in greywater prevented the biomass from attaching on to the fibres. An alteration in the start-up procedure (Section 3.2.3.3) lead to a slightly improved biofilm formation on the membrane.

The results from the MABR are insufficient for a complete evaluation for greywater treatment and hence are not discussed in detail. The performance data obtained in the early stages of the greywater trial (Section 5.2.4-5.2.6, Figures D.3-D.24 in Appendix D) indicated that the MABR was the least stable and inappropriate of the four processes tested, with a mean BOD_5, tCOD and SS removal of 72%, 31% and 66%, respectively (Tables 5.4-5.5). At a later stage a leak was found in the membrane

module. After the membrane was resealed, the oxygen regulator broke down and it was decided to curtail the trials due to the problematic start-up and the limited treatment capacity.

5.2.3 Flows

5.2.3.1 Tracer studies

The hydraulic characteristics of a system have a large impact on the process performance, and as such require optimisation. In this work the treatment processes were operated under predetermined hydraulic conditions as determined by tracer studies with sodium chloride (NaCl) and lithium (Table 5.2). The former was carried out using mains water (without biomass) whilst in the latter the processes were run with greywater and blackwater (with biomass). Due to the limited time available the hydraulic patterns for the side-stream MBR and the MABR were not determined.

Table 5.2. Summary of the key parameters from the tracer studies with sodium chloride and **lithium**. Feed and air flow, theoretical hydraulic retention time (tHRT), mean retention time of the tracer (t_m), number of tanks in series (N) and dispersion number (*D/uL*).

Parameter	Submerged MBR	BAF
Feed flow ($l\ h^{-1}$)	32.1	13.2
	-	**13.2**
Air flow ($l\ h^{-1}$)	20	15
	-	**15**
tHRT (min)	144	159.5
	-	**159.5**
t_m (min)	67	167
	-	**122-123**
t_p (min)	43	105
	-	**80-85**
N	2.3	7.3
	-	**5.9-7.9**
D/uL	0.22	0.07
	-	**0.06-0.09**
Reactor type [a]	CSTR [b]	Plug flow

[a] determined as by Levenspiel (1999) with reference to the tracer study data

[b] completely stirred tanks in series

In the submerged MBR the difference between the theoretical hydraulic retention time (tHRT) and the mean retention time of the tracer (t_m) implied the presence of dead zones (Table 5.2, Figure D.1 in Appendix D). The most likely places for the dead zones are in the denitrification chambers and possibly beneath the aeration devices in the nitrification units. The BAF represented a plug-flow reactor with a N of 5.9-7.9 and a low dispersion number in both tracer studies. Some channelling ($t_m \neq$ tHRT) and mixing ($t_m \neq$ peak of tracer, t_p) occurred when the system was run with greywater and blackwater (Figure D.2 in Appendix D), suggesting that there were areas in the media bed with higher concentrations of biomass than in some other areas of the bed. Mann *et al.* (1995) have highlighted that a certain degree of mixing is needed in BAFs to allow the biomass to remove the maximum amount of nutrients from the liquid and to distribute nutrients along the biofilm.

5.2.3.2 Flow rates

The mean flow rate of the submerged MBR was the lowest of the treatment processes, decreasing from a mean value of 0.071 to 0.047 m^3 d^{-1} over the 300-day period (Figure 5.1, Table 5.3). The side-stream MBR and the MABR treated greywater at the respective flow rates of 0.137 m^3 d^{-1} and 0.225 m^3 d^{-1}. For the first 201 days the BAF was operated at a mean flow rate of 0.345 m^3 d^{-1}. After this it was increased by 50% to 0.546 m^3 d^{-1} for 32 days so as to assess the effects of the increased loading rate. At the beginning of the blackwater trial the flow rate was returned to its original level.

Table 5.3. Flow rates (m^3 d^{-1}) of the treatment processes during the greywater, *grey/blackwater* and **blackwater** trials (does not include the start-up). Mean (range).

Submerged MBR	Side-stream MBR	MABR	BAF
0.071 (0.042-0.170)	0.137 (0.104-0.180)	0.225 (0.126-0.275)	0.328 (0.122-0.410)
0.056 (0.037-0.072)			*0.362 (0.310-0.439)*
			0.546 (0.446-0.594)**
0.047 (0.038-0.056)			**0.309 (0.043-0.396)**

**, 0* when flow to the BAF increased by 50%*

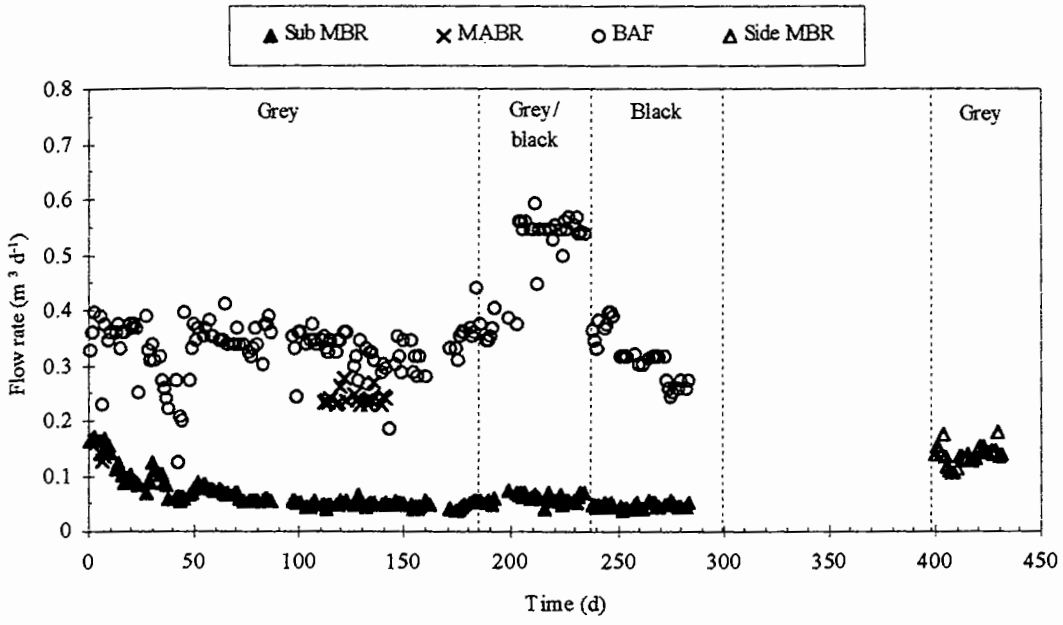


Figure 5.1. Flow rates of the treatment processes during the greywater, grey/blackwater and blackwater trials.

5.2.3.3 Flux

The operational difference between the submerged and side-stream membrane bioreactors lies in the former being driven by a hydrostatic head of pressure (Section 3.2.3.1), therefore having a limited potential for increasing through the flux, and the latter having a greater range in operating conditions due to adjustment of the transmembrane pressure (TMP) and the crossflow velocity. The submerged and side-stream MBRs were operated at respective TMPs of 0.06 bar and 2.5 bar, thus the significantly higher specific flux in the former than the latter.

The flux of the submerged MBR declined from the initial 60 l m^{-2} h^{-1} (0.300 m^3 d^{-1}, 1000 l m^{-2} h^{-1} bar^{-1}) to 38 l m^{-2} h^{-1} (0.187 m^3 d^{-1}, 633 l m^{-2} h^{-1} bar^{-1}) in two hours followed by a slower decline to 32 l m^{-2} h^{-1} (0.162 m^3 d^{-1}, 533 l m^{-2} h^{-1} bar^{-1}) over 14 days of the start-up period (Figure 5.2). This further reduced to a stable 12 l m^{-2} h^{-1} (0.046 m^3 d^{-1}, 200 l m^{-2} h^{-1} bar^{-1}) over 76 days in the greywater trial (Figure 5.2), with the total drop in the flux being some 80%. In the grey/blackwater and blackwater trials respective flux values of 12.8 l m^{-2} h^{-1} (0.056 m^3 d^{-1}, 213 l m^{-2} h^{-1} bar^{-1}) and 11.6

l m^{-2} h^{-1} (0.047 m^3 d^{-1}, 193 l m^{-2} h^{-1} bar^{-1}) were measured. The gradual decrease in the flux observed was possibly due to a slow build-up of a fouling layer on the membrane surface, such that insignificant further external fouling occurred after a stable flux was achieved. This assumption was supported by visual observation of the membrane after 300 days. The flux remained stable (Figure 5.2) independent of the influent quality without the need for cleaning for 300 days, suggesting operation under sub-critical conditions, the critical flux being the flux above which colloidal deposition on the membrane surface takes place, leading to irreversible fouling (Howell, 1995). Operation below this value reduces the requirement for cleaning, thus lowering operational costs (Stephenson *et al.*, 2000).

The side-stream MBR was operated without membrane cleaning for 32 days at a flux ranging from 38 to 64 l m^{-2} h^{-1} (from 0.104 to 0.180 m^3 d^{-1}, Figure 5.2) with the specific flux being 15-26 l m^{-2} h^{-1} bar^{-1}. This indicates that the process may be suitable for greywater treatment for long periods without membrane cleaning.

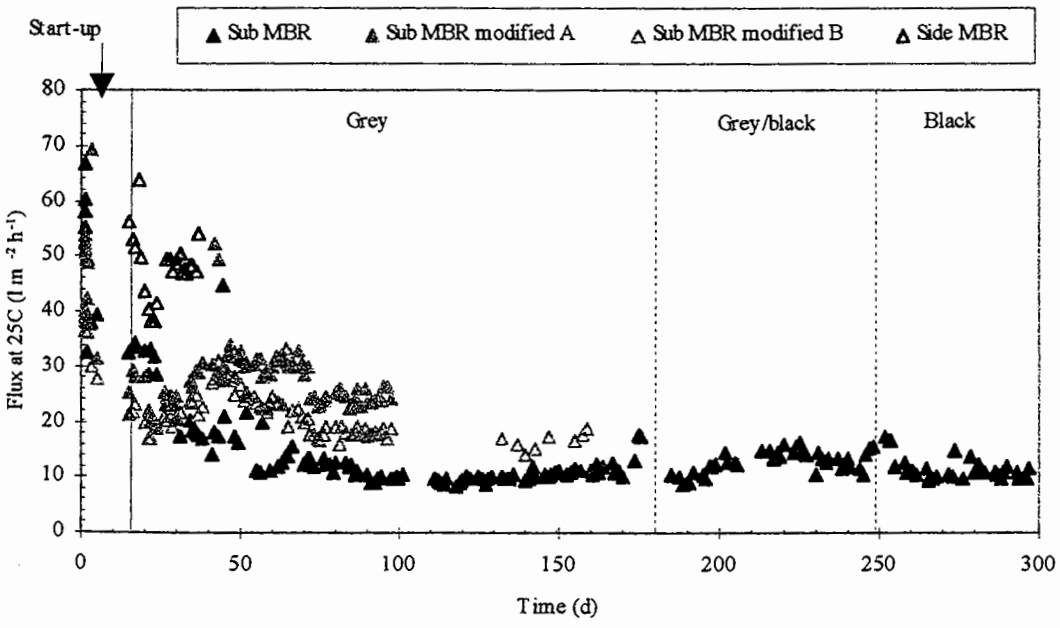


Figure 5.2. Flux of the submerged, modified and side-stream MBRs during the steady-state trials including the start-up period.

At a later stage the configuration of the submerged system was changed so as to operate the nitrification unit alone, i.e. without denitrification (Section 3.2.3.1). This provided two MBR units where a significantly slower flux decline was observed. In the unit A the initial value of 54 l m^{-2} h^{-1} (0.266 m^3 d^{-1}, 900 l m^{-2} h^{-1} bar^{-1}) declined to a stable flux of 24 l m^{-2} h^{-1} (0.121 m^3 d^{-1}, 400 l m^{-2} h^{-1} bar^{-1}) over a 94-day period (Figure 5.2), with the total decline being around 55%. In the unit B the initial flux of 39 l m^{-2} h^{-1} (0.193 m^3 d^{-1}, 647 l m^{-2} h^{-1} bar^{-1}) declined to a mean value of 22 l m^{-2} h^{-1} (0.109 m^3 d^{-1}, 358 l m^{-2} h^{-1} bar^{-1}) over 100 days of operation (Figure 5.2). These suggest that membrane fouling occurred faster in the original unit than in the modified one although the start-up procedure and the influent type were the same. Between days 132 and 160 the flux in the unit B was 17 l m^{-2} h^{-1} (0.082 m^3 d^{-1}, 277 l m^{-2} h^{-1} bar^{-1}), suggesting that further flux decline may be occur over a longer period of operation. In the unit B the total flux decline was 56%. The higher flux in the modified MBR allows higher organic loading rates to be employed without detriment to the performance (Section 5.3.2.1).

5.2.4 Organic matter and nutrients

5.2.4.1 BOD_5

The organic loading rates (OLRs) applied to the processes (Table 5.4) were low compared to literature values on similar systems (Tables 2.8 and 2.11, Sections 2.4.6.4-2.4.6.5). In the case of the MBRs this was due to the limitation imposed by the membrane hydraulic resistance, leading to long hydraulic residence times (Table 5.4). The organic loading rate for the non-barrier processes, which were run at recommended values, were limited by the influent strength (Table 5.4). Two of the lowest flow rates and hence the longest hydraulic retention times (HRTs; 9.7 and 27.6 h) for the BAF were measured when the flow to the column was exceptionally, low possibly due to a blockage in the pipe. Since these occurred on days on which the full range of analyses were not carried out, the effect of the longer HRT on BOD removal is not known.

Table 5.4. Hydraulic residence time and process performance in terms of BOD and COD during the greywater, *grey/blackwater* and **blackwater** trials. Mean ± standard deviation.

Parameter	Influent	Side-stream MBR influent	Submerged MBR		Side-stream MBR		MABR		BAF	
			Effluent	Removal (%)	Effluent	Removal (%)	Effluent	Removal (%)	Effluent	Removal (%)
HRT (h)			13.6 (4.9-20.1) *15.4 (11.7-22.4)* **18.0 (15.0-22.0)**		67.8 (50.7-88.0)		0.8 (0.6-1.4)		3.7 (2.9-9.7) *3.3 (2.7-3.8)* *2.2* (2.0-2.7)** **4.5 (3.0-27.6)**	
OLR (kgBOD m^{-3} d^{-1})			0.08 ± 0.06 *0.07 ± 0.04* **0.14 ± 0.07**		0.06 ± 0.02 - -		0.83 ± 0.47 - -		0.27 ± 0.20 *0.37 ± 0.18* *(0.44 ± 0.23)** **0.67 ± 0.27**	
BOD_5 (mg l^{-1})	41 ± 30 *41 ±21* **110 ± 50**	181 ± 58 - -	1 ± 1 *1 ±1* **1 ± 1**	96 ± 6 *97 ± 2* **98 ± 3**	1 ± 1 - -	100 ± 1 - -	9 ± 9 - -	72 ± 19 - -	4 ± 4 *5 ±3 (5 ±4)** **12 ± 10**	88 ± 10 *92 ± 10* *(84 ± 9)** **86 ± 13**
DO (mg l^{-1})	- - -	- - -	8.3 ± 0.8 *8.1 ± 1.3* **9.1 ± 1.2**		9.2 ± 0.6 - -		9.3 ± 1.0 - -		8.7 ± 1.1 *9.4 ± 0.7* *(8.1 ± 1.4)** **8.7 ± 1.2**	
COD loading rate (kgCOD m^{-3} d^{-1})			0.25 ± 0.15 *0.25 ± 0.21* **0.43 ± 0.14**		0.10 ± 0.02 - -		3.73 ± 2.30 - -		0.84 ± 0.59 *1.37 ± 1.27* *(1.49 ± 0.83)** **2.03 ± 0.74**	
tCOD (mg l^{-1})	128 ± 89 *159 ±129* **300 ± 91**	273 ± 57 - -	7 ± 6 *9 ±7* **16 ± 7**	92 ± 9 *92 ± 9* **95 ± 3**	2 ± 3 - -	99 ± 1 - -	17 ± 34 - -	31 ± 102 - -	13 ± 13 *19 ±12 (22 ±18)** **57 ± 29**	81 ± 23 *90 ± 5 (74 ± 20)** **84 ± 7**
sCOD (mg l^{-1})	46 ± 31 - **80 ± 36**	- - -	3 ± 3 - **11 ± 9**	88 ± 18 - **78 ± 38**	- - -	- - -	31 ± 22 - -	27 ± 24 - -	7 ± 7 - **24 ± 11**	78 ± 38 - **58 ± 40**

() when flow to the BAF increased by 50%*

The loading and removal rates for the treatment processes are presented in Figure 5.3 and Table 5.4. The submerged MBR achieved a 96% BOD removal from greywater at 0.08 ± 0.06 kgBOD m^{-3} d^{-1}. The slightly higher efficiency (99%) demonstrated by the side-stream system at 0.06 ± 0.47 kgBOD m^{-3} d^{-1} was related to the smaller pore size of the membrane. Another reason was the HRT which was longer in the side-stream MBR than in the submerged configuration such that the micro-organisms had a longer time to treat organic matter. This difference in the removal is relatively insignificant, demonstrating that the biological stage is fairly independent of the membrane configuration, i.e. submerged or side-stream, in these MBRs. The BAF removed 89% of the BOD in greywater at a loading rate of 0.27 ± 0.20 kgBOD m^{-3} d^{-1} (Table 5.4, Figure 5.3).

The observations of smaller increases in the loading rates than expected on changing to grey/blackwater may be attributed to the range of influent quality (Table 5.4) arising from the greywater characteristics (Section 4.2) and the fluctuations in the diurnal flow and strength of blackwater. Nevertheless in the grey/blackwater trial the BAF performance improved to a 94% BOD removal (Table 5.4), which declined by 10% on increasing the flow rate to 0.546 m^3 d^{-1}. This decline in performance was related to the increased filtration velocity.

The blackwater component in the influent increased the OLRs of the submerged MBR and the BAF by 75% and 148%, respectively. In the blackwater trial the corresponding loading rates were 0.14 ± 0.07 and 0.67 ± 0.27 kgBOD m^{-3} d^{-1}. The BOD removal of these processes remained relatively unchanged at 98% and 84%, respectively (Table 5.4, Figure 5.3).

The distribution of the effluent BOD_5 with respect to the US EPA standard of 10 mg l^{-1} during the steady-state trials is presented in Figures 5.4-5.6, and the BOD_5 transient in Figure D.6 in Appendix D. Throughout the trials the submerged and side-stream MBRs produced very good quality effluent such that the majority of the samples had a BOD_5 20% below the US EPA water quality standard. A low effluent BOD is a key

factor for in-building treatment processes as regrowth in the distribution system can effectively be avoided and a more consistent disinfection achieved.

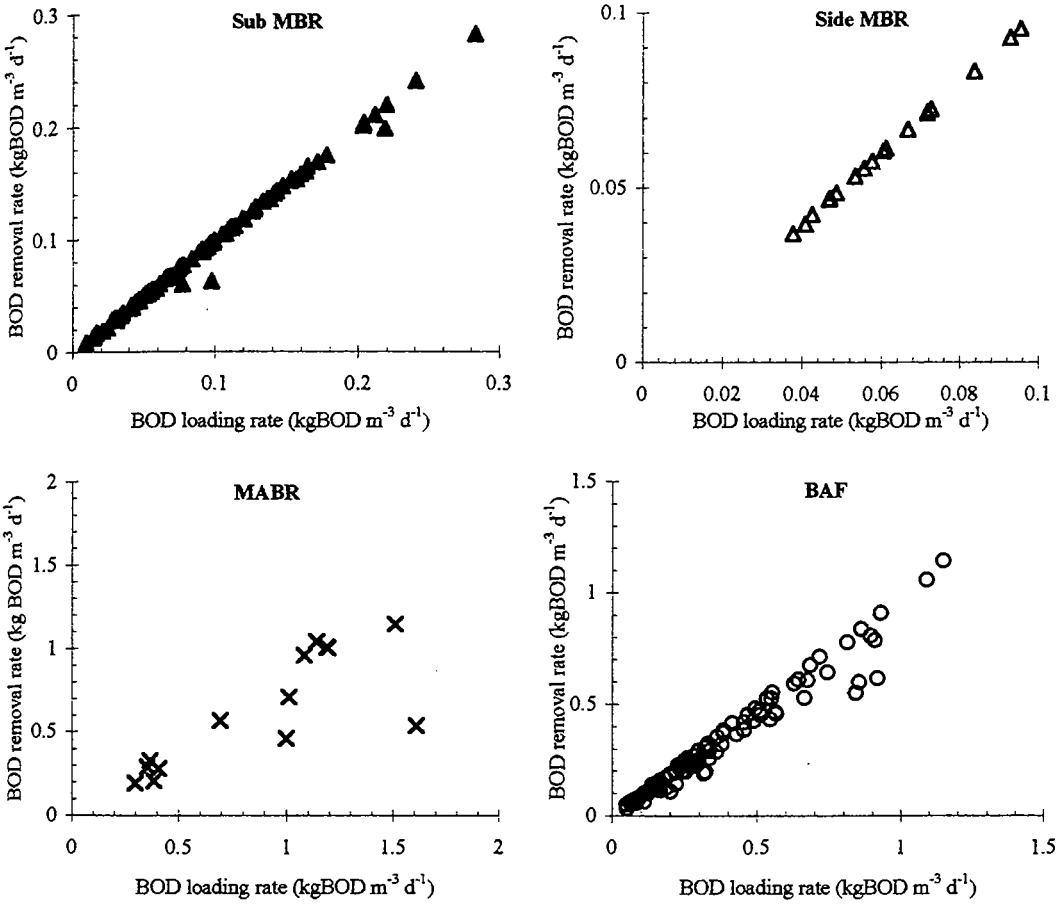


Figure 5.3. Loading rate versus removal rate in terms of the BOD.

A large proportion of the BAF treated water samples were within the water quality limit for a greywater feed (Figure 5.4, Table 5.4), with the highest values being up to 20 mg l^{-1}. When treating grey/blackwater at 0.362 m^3 d^{-1}, the BAF met the standard at all times (Table 5.4). The increased flow rate lead to 10% of the samples having a BOD_5 of up to twice the US EPA limit (Figure 5.4, Table 5.4). During the blackwater trial the increased failure rate of the BAF (Figure 5.6) reflected the increased influent strength (Table 5.4). The results show that although the majority of the samples were generally below 10 mg l^{-1}, a wider concentration range as a result of a higher influent BOD was evident.

Due to the high air flow to the processes the effluents had a relatively high mean dissolved oxygen at >8.1 mg l^{-1}. The values are shown in Table 5.4 and Figure D.7 in Appendix D.

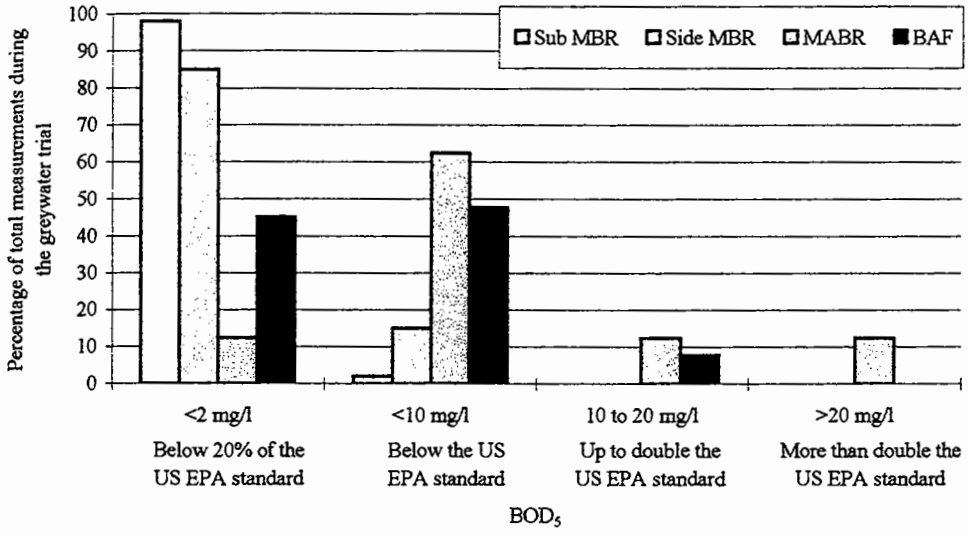


Figure 5.4. Distribution of the effluent BOD_5 during the greywater trial with respect to the US EPA water reuse guideline.

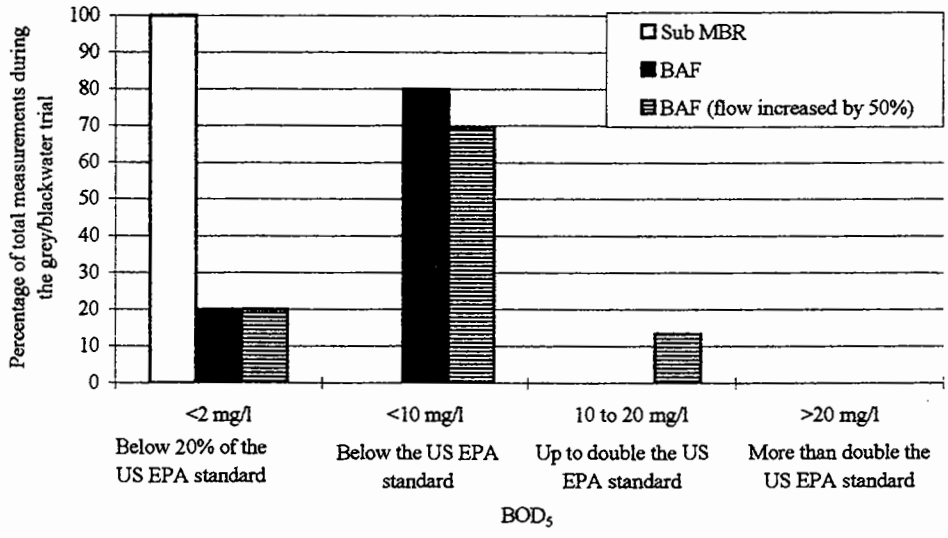


Figure 5.5. Distribution of the effluent BOD_5 during the grey/blackwater trial with respect to the US EPA water reuse guideline.

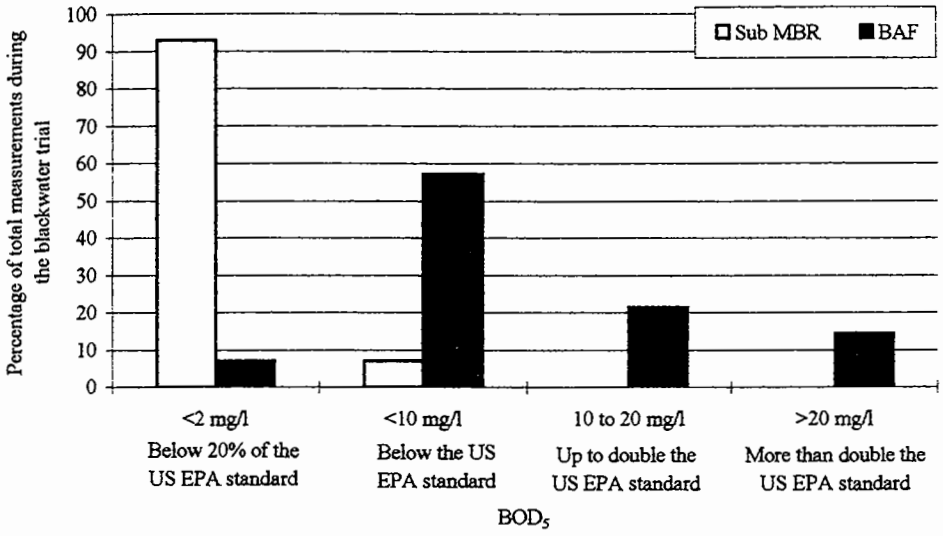


Figure 5.6. Distribution of the effluent BOD_5 during the blackwater trial with respect to the US EPA water reuse guideline.

5.2.4.2 COD

For the submerged MBR, the MABR and the BAF the mean influent COD:BOD ratio over the entire period of operation was 3.8 with a standard deviation of 84%. The variation was related to the factors regarding the variability of the greywater and blackwater influent compositions (Section 5.2.4.1), and accounted for some of the variation in the actual performance of the individual processes. The share of the biodegradable component in the side-stream MBR greywater was higher at a COD:BOD ratio of 1.61 ± 0.24.

The COD loading rates versus the removal rates for the treatment processes are presented in Figure D.3 in Appendix D. In the side-stream MBR the removal rate of 99% indicated removal of nearly all 'hard' chemical oxygen demand at a volumetric loading rate of 0.10 ± 0.02 kgCOD m^{-3} d^{-1} (Table 5.4). The long HRT aided the breakdown of the non-biodegradable components. In the greywater trial the load applied to the submerged configuration was higher at 0.25 ± 0.15 kgCOD m^{-3} d^{-1}, which in the subsequent stages increased by 72% to 0.43 ± 0.14 kgCOD m^{-3} d^{-1}. This had no significant impact on the relatively stable COD removal of 92-95% (Table 5.4). The performance of these processes compares favourably with membrane only-

systems. A 68-92% COD reduction was achieved from screened greywater (41-85 mg l^{-1}) by 15 kDa ultrafiltration (UF) membrane at crossflow velocities of 1-4 m s^{-1} and TMPs of 1.47-3.66 bar (Ahn *et al.*, 1998). Treating screened greywater (75 mg l^{-1}) by a 0.1 μm microfiltration membrane (4 m s^{-1}, 1.2 bar) resulted in a 89% COD removal. The small difference between the performance of these membranes were attributed to the majority of the particles in the greywater being larger than 0.1 μm and the influent having little organic soluble matter. In another study a 72% COD removal from greywater (111 mg l^{-1}) was achieved by a UF membrane (4000 Da, 2.6 m s^{-1}, 0.8 bar; Le Clech *et al.*, 2000). The superior treatment in the MBR systems is attributed to the membrane's ability to retain macro-organic materials that then can be slowly degraded in the bioreactor. Xing *et al.* (2000) estimated that of the total COD removal of 97% by an MBR treating urban wastewater 85% was removed by the bioreactor and 12% by the membrane.

In comparison to the MBRs, the COD removal of the BAF at a range of 74 to 90% was more dependent on the influent strength and flow rate. The lowest treatment efficiencies were observed at the lowest (0.84 ± 0.59 kgCOD m^{-3} d^{-1}) and the highest (2.03 ± 0.74 kgCOD m^{-3} d^{-1}) loading rates as well as at the higher flow rate (Table 5.4). The best performance occurred in the grey/blackwater trial at a flow of 0.362 m^3 d^{-1}, equating to a more optimal loading rate (1.37 ± 1.27 kgCOD m^{-3} d^{-1}). A maximum rate of 3.3 kgCOD m^{-3} d^{-1} for a 80% COD removal for a BAF treating soap solution has been reported (Jefferson *et al.*, 2000b). This indicates the possibility of increasing the load to the process without reducing the treatment performance, provided foaming resulting from the greywater components can be suppressed. In the current work the effect of nutrient balancing of greywater at higher OLRs to the BAF was investigated at a later stage (Section 5.3).

Shin *et al.* (1998) reported limited organic and nitrogen removal by a sequencing batch reactor due to the initial operation at low organic loading rates. The main reason for this was the fluctuating greywater discharge from the office building. Consequently an equalisation tank was installed so as to improve COD removal, such

that a reduction from the initial sCOD of 26-197 mg l^{-1} to 20 mg l^{-1} was achieved. The results from the current study confirm that the low organic loading did not affect the treatment by the MBRs or the BAF.

The soluble fraction of the COD in the greywater was reduced by the submerged MBR and the BAF by 90% and 80%, respectively (Table 5.4). On changing to blackwater these declined to 78% and 58% (Table 5.4, Figure D.8 in Appendix D), reflecting the increased sCOD in the influent.

Non-potable domestic water reuse quality standards for chemical oxygen demand do not exist but as this parameter indicates the level of non-biodegradable matter in water it is a significant water quality determinant. The most consistent effluent quality at a mean tCOD of 2 mg l^{-1} was produced by the side-stream MBR (Table 5.4, Figure D.9 in Appendix D). The mean values of 7 and 13 mg l^{-1} measured in the greywater trial for the submerged MBR and the BAF respectively increased to 16 and 57 mg l^{-1} as a result of the higher sCOD in the blackwater (Table 5.4, Figure D.9 in Appendix D).

5.2.4.2 Nitrogen, phosphorus and temperature

The optimum biological growth requires, among other factors, a number of nutrients in balanced proportions (Section 4.3.1). The greywater was generally low in both nitrogen and phosphorus (Table 5.5), and the nutrient ratio changed during degradation. According to the literature (Section 2.2.2), the greywater C:TN:P ratio of 228:13:1 in the current study was not ideal for biological systems. It changed to 183:27:1 on mixing greywater with blackwater. The blackwater with a C:TN:P ratio of 20:4:1, a high pathogen count and solids concentration had characteristics more typical of domestic wastewater. Tests carried out on the side-stream MBR excluded nitrogen and phosphorus analyses, such that the nutrient ratio in the influent was not determined and hence limitations imposed by these nutrients on this treatment process are not known. The transients for the nitrogen and phosphorus compounds are presented in Figures D.10-D.14 in Appendix D.

Table 5.5. Ammonia loading rate and process performance in terms of nitrogen, phosphorus and temperature during the greywater, *grey/blackwater* and **blackwater** trials. Mean ± standard deviation.

Parameter	Influent	Side-stream MBR influent	Submerged MBR		Side-stream MBR		MABR		BAF	
			Effluent	Removal (%)	Effluent	Removal (%)	Effluent	Removal (%)	Effluent	Removal (%)
Ammonia loading rate ($kgNH_3$-N m^{-3} d^{-1})			0.001 ± 0.001 *0.013 ± 0.006* **0.037 ± 0.006**		- - -		0.011 ± 0.016 - -		0.002 ± 0.003 *0.067 ± 0.044 (0.097 ± 0.020)** **0.170 ± 0.023**	
TN (mg l^{-1})	6 ± 3 *18 ± 11* **43 ± 10**	- - -	6 ± 4 *12 ± 5* **33 ± 4**	-3 ± 66 *22 ± 20* **18 ± 25**	- - -	- - -	6 ± 1 - -	-5 ± 32 - -	3 ± 2 *14 ± 6 (8 ± 4)** **32 ± 7**	48 ± 38 *41 ± 28 (22 ± 24)** **21 ± 26**
NH_3-N (mg l^{-1})	0.4 ± 0.4 *9.1 ± 4.5* **26.5 ± 5.2**	- - -	0.5 ± 0.8 *0.1 ± 0.3* **2.6 ± 5.4**	3 ± 146 *99 ± 3* **89 ± 23**	- - -	- - -	0.5 ± 0.8 - -	-247 ± 622 - -	0.7 ± 0.8 *4.1 ± 5.7 (1.4 ± 2.0)** **0.1 ± 0.1**	-148 ± 316 *72 ± 36 (81 ± 29)** **100 ± 1**
NO_3-N (mg l^{-1})	2.2 ± 1.9 *0.5 ± 0.6* **1.9 ± 3.2**	- - -	5.2 ± 2.9 *12.4 ± 4.5* **32.7 ± 2.2**		- - -		3.9 ± 1.5 - -		1.6 ± 1.6 *6.7 ± 4.7 (7.6 ± 4.9)** **24.5 ± 5.6**	
NO_2-N (mg l^{-1})	0.07 ± 0.27 *0.06 ± 0.05* **0.56 ± 0.91**	- - -	0.30 ± 0.10 *0.03 ± 0.06* **0.00 ± 0.00**		- - -		0.11 ± 0.07 - -		0.12 ± 0.33 *1.32 ± 2.19* *(0.01 ± 0.01)** **0.05 ± 0.03**	
P (mg l^{-1})	1.7 ± 1.4 *4.4 ± 2.4* **17.8 ± 12.5**	- - -	2.5 ± 1.5 *2.8 ± 1.3* **8.6 ± 1.0**	14 ± 24 *8 ± 39* **19 ± 18**	- - -	- - -	0.5 ± 0.3 - -	36 ± 30 - -	0.3 ± 0.2 *1.4 ± 1.4 (2.6 ± 0.4)** **9.2 ± 6.2**	50 ± 73 *18 ± 55 (29 ± 9)** **18 ± 13**
T (°C)	- - -	- - -	21 ± 4 *16 ± 3* **15 ± 2**		20 ± 2 - -		17 ± 3 - -		20 ± 6 *13 ± 3 (13 ± 3)** **13 ± 3**	

() when flow to the BAF increased by 50%*

The most significant characteristic of the greywater was the depleted level of ammonia (Table 5.5). During the first part of the steady-state trial a limited and variable ammonia removal by the submerged MBR and the BAF was observed at the very low respective loading rates of 0.001± 0.001 and 0.002 ± 0.003 $kgNH_3\text{-}N\ m^{-3}\ d^{-1}$. The addition of the blackwater component to the feedwater increased the activity of the autotrophic bacteria, and hence the ammonia removal (Table 5.5). Over the course of the steady-state operation the entire ammonia load increased by 3600% and 8400% in the submerged MBR and the BAF (Table 5.5, Figure D.5 in Appendix D).

Ammonia removal is widely reported for MBRs (Chiemchaisri *et al.*, 1992; Côté *et al.*, 1997) and BAFs (Rogalla *et al.*, 1990; Smith and Hardy, 1992) treating domestic wastewater similar to blackwater in the current study. The sampling frequency for ammonia was reduced from three times a week in the greywater trial to once a week after the changeover to grey/blackwater. It can thus only be estimated that it took up to 6 d and 11 d for the nitrifiers to build-up in the submerged MBR and the BAF, respectively. The corresponding temperatures of 11-17°C and 9-17°C measured in the effluents during these time periods suggest that the initial nitrification could have been adversely affected by the low temperature, documented both for the MBRs (Chiemchairisi and Yamamoto, 1993) and the BAFs (Paffoni *et al.*, 1990).

The effluent temperatures of the submerged MBR and the BAF (Table 5.5) resulted from the seasonal variation during the 300 day-trial, such that this may have influenced nitrification to some extent. The stable value of 20 ± 2°C for the side-stream MBR (Table 5.5, Figure D.19 in Appendix D) suggests that temperature did not have a large impact on the process operation. Aerobic biological processes are often operated at ambient temperatures, though for most micro-organisms the optimum temperature is in the mesophilic temperature range at 25-40°C (Metcalf and Eddy, 1991). Increased temperature enhances carbonaceous removal, but it can lead to problems of increased oxygen demand resulting in a system becoming oxygen-limited (Gray, 1989). Different micro-organisms may dominate as a result of

temperature fluctuation, altering the biological growth and substrate removal rates in both activated sludge systems (Gray, 1989) and BAFs (Visvanathan and Nhien, 1995).

The low and variable percentage phosphorus and total nitrogen removal by the treatment processes was related to the low influent concentration (Table 5.5), with removal improving for the higher strength blackwater (Figures D.12-D.13) though not stabilising in all cases.

5.2.5 Solids, turbidity and pH

5.2.5.1 Solids

The solids loading versus the removal rates are presented in Figure D.4 in Appendix D. During the greywater trial the side-stream MBR and the submerged MBR achieved a 93% and 89% suspended solids (SS) removal respectively at corresponding loading rates of 0.02 ± 0.01 kgSS m^{-3} d^{-1} and 0.10 ± 0.10 kgSS m^{-3} d^{-1} (Table 5.6). The difference in rejection was due to the side-stream system having a smaller pore size than the latter. The removal efficiency of the submerged MBR increased to 94% at the significantly higher solids loading from blackwater, which more than doubled the solids load to the system. At a mean loading rate of 0.33-1.01 kgSS m^{-3} d^{-1} the BAF achieved a solids removal of 81-94% throughout the trials.

The distribution of the effluent SS is presented in Figures 5.7-5.9 with respect to the US EPA standard of 5 mg l^{-1}. The transient values of the influent and effluent water quality are shown in Figure D.15 in Appendix D. The submerged and side-stream systems produced effluent mostly with a SS <5 mg l^{-1} (Figures 5.7-5.9). The unusually high solids readings at above 5 mg l^{-1} (Figures 5.7-5.9) were a result of regrowth in the effluent pipes rather than permeation through the membrane. When run with grey/blackwater at 0.362 m^3 d^{-1} the BAF produced a relatively consistent product SS of <10 mg l^{-1} (Figure 5.8). The significantly larger proportion (around 40%) of the product sample above 10 mg l^{-1} at the increased flow rate in the grey/blackwater trial and later in the blackwater stage (Figures 5.8-5.9) reflected the higher loading rate.

Table 5.6. Solids loading rate and process performance in terms of solids, turbidity and pH during the greywater, *grey/blackwater* and **blackwater** trials. Mean ± standard deviation.

Parameter	Influent	Side-stream MBR influent	Submerged MBR		Side-stream MBR		MABR		BAF	
			Effluent	Removal (%)	Effluent	Removal (%)	Effluent	Removal (%)	Effluent	Removal (%)
Solids loading rate (kgSS m^{-3} d^{-1})			0.10 ± 0.10 *0.09 ± 0.65* **0.21 ± 0.11**		0.02 ± 0.01 - -		1.39 ± 1.55 - -		0.33 ± 0.39 *0.49 ± 0.26* *(0.60 ± 0.60)** **1.01 ± 0.60**	
SS (mg l^{-1})	52 ± 58 *59 ± 39* **158 ± 97**	58 ± 16 - -	4 ± 4 *4 ± 4* **3 ± 4**	89 ± 26 *90 ± 14* **94 ± 4**	4 ± 4 - -	93 ± 6 - -	13 ± 13 - -	66 ± 21 - -	6 ± 6 *4 ± 4* *(6 ± 7)** **17 ± 19**	81 ± 22 *94 ± 7* *(83 ± 25)** **91 ± 10**
Turbidity (NTU)	- - -		0.3 ± 0.3 *0.4 ± 0.3* **0.4 ± 0.2**		0.9 ± 1.4 - -		6.6 ± 6.5 - -		3.2 ± 9.0 *2.0 ± 2.2* *(2.0 ± 1.9)** **12.7 ± 27.3**	
pH (-)	7.0 ± 0.2 *7.3 ± 0.2* **7.7 ± 0.2**	7.1 ± 0.4 - -	7.4 ± 0.4 *7.5 ± 0.4* **7.1 ± 0.3**		7.2 ± 0.4 - -		6.9 ± 0.2 - -		7.6 ± 0.3 *7.7 ± 0.2* *(7.6 ± 0.2)** **7.2 ± 0.4**	

() when flow to the BAF increased by 50%*

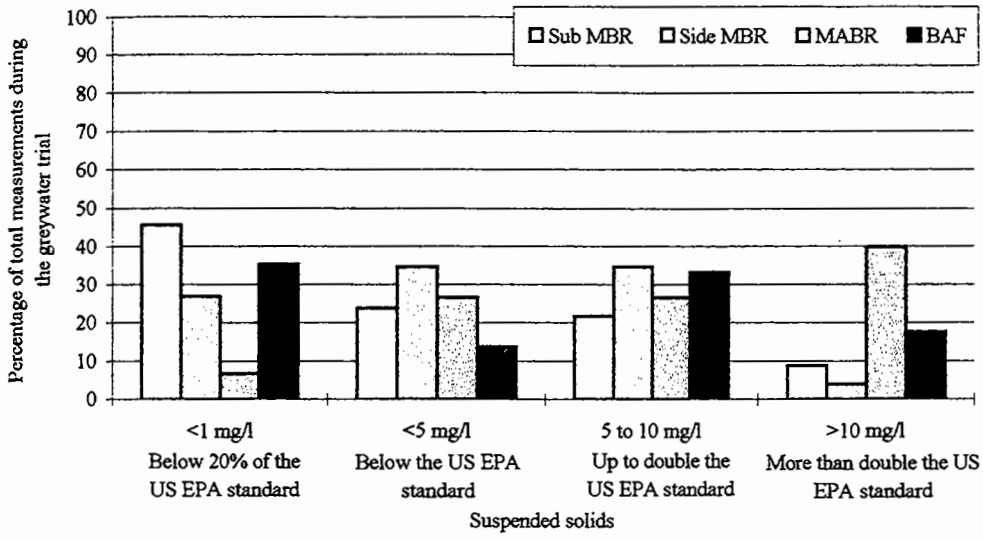


Figure 5.7. Distribution of the effluent suspended solids during the greywater trial with respect to the US EPA water reuse guideline.

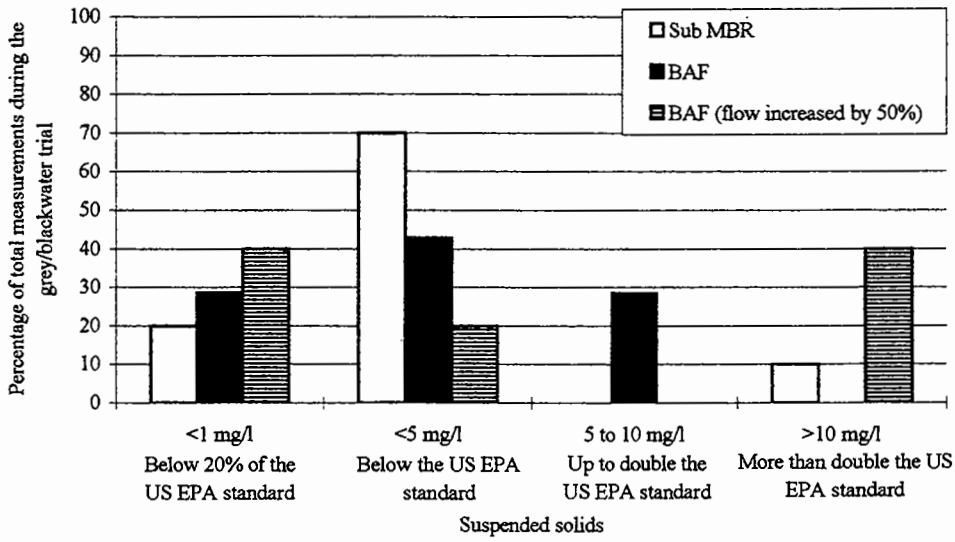


Figure 5.8. Distribution of the effluent suspended solids during the grey/blackwater trial with respect to the US EPA water reuse guideline.

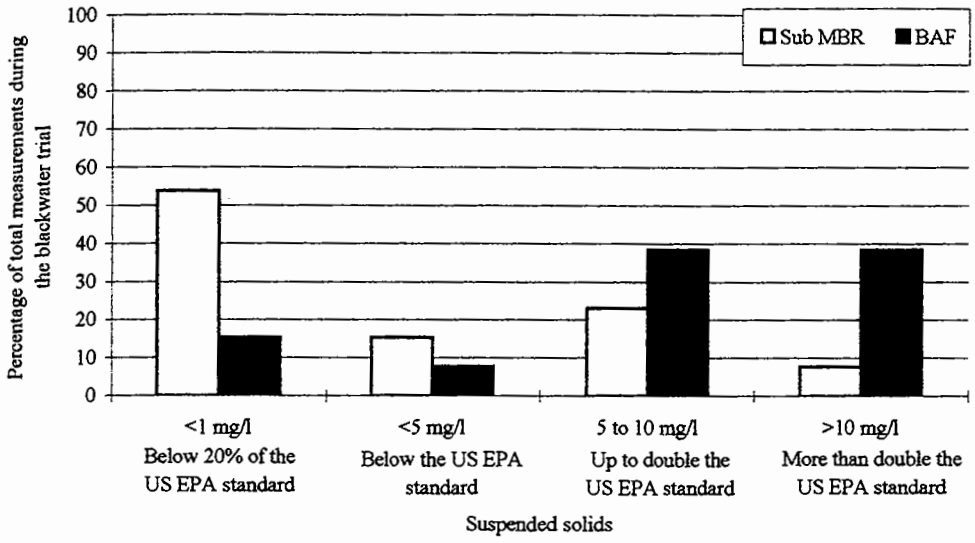


Figure 5.9. Distribution of the effluent suspended solids during the blackwater trial with respect to the US EPA water reuse guideline.

5.2.5.2 Turbidity

The turbidity of the influent was above the upper determination limit of the equipment used. At a later stage in the programme when a turbidimeter with a higher measuring range was available typical values for greywater and blackwater were measured. For greywater the value (19.7 NTU) from the large-scale synthetic greywater experiment (Table 4.2, Section 4.2) was used. The blackwater value of 300 NTU was determined from a grab sample of the primary sewage influent. The turbidity of the side-stream MBR influent (33.3 NTU) was evaluated from greywater batches prior to the steady-state experiment on the process.

Most of the samples from the submerged and side-stream MBRs were below 20% of the standard, i.e. <0.4 NTU, with the rest of the samples being <2 NTU (Table 5.6, Figures 5.10-5.12 and D.16 in Appendix D). These values are typical for such barrier processes, and presents a major advantage offered by the MBRs in domestic wastewater recycling for obvious aesthetic reasons.

The BAF, as a non-barrier process, had a limited particle resolution such that smaller particle sizes were not retained by the media. Manual backwashing of the media bed at the end of the greywater run resulted in a number of samples with a turbidity >4 NTU (Figures 5.10 and D.16 in Appendix D). During the grey/blackwater trial at the flow rate of 0.362 m^3 d^{-1} nearly 90% of the BAF samples were within the water quality limit (Figure 5.11). This good performance was reduced at the 0.546 m^3 d^{-1} flow rate in the grey/blackwater trial (Figure 5.11) due to higher filtration velocity. The significant proportion of the high values in the blackwater trial (Figure D.16 in Appendix D) resulted from the high solids load in the influent (Table 5.6). The results demonstrate that an additional polishing stage for the BAF effluent would be necessary to achieve similar effluent quality to the MBRs.

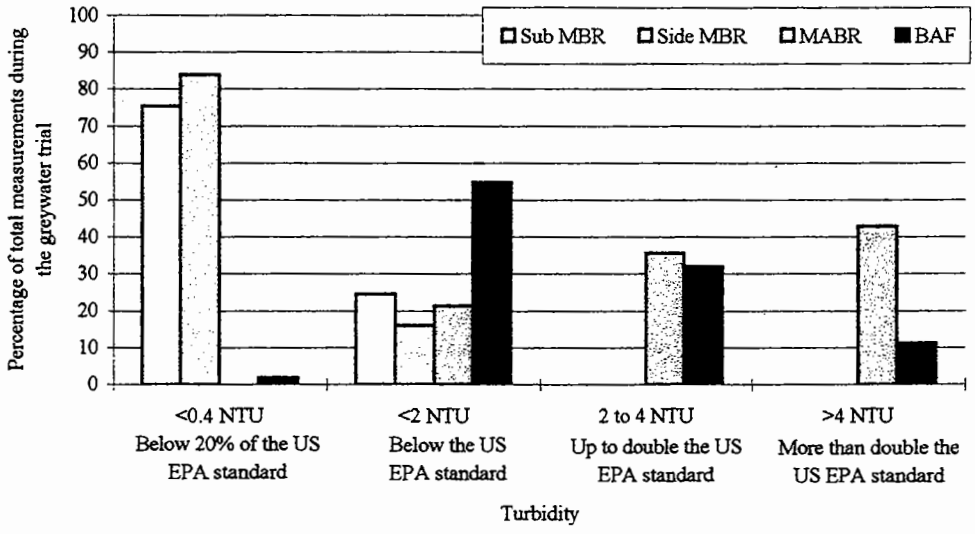


Figure 5.10. Distribution of the effluent turbidity during the greywater trial with respect to the US EPA water reuse guideline.

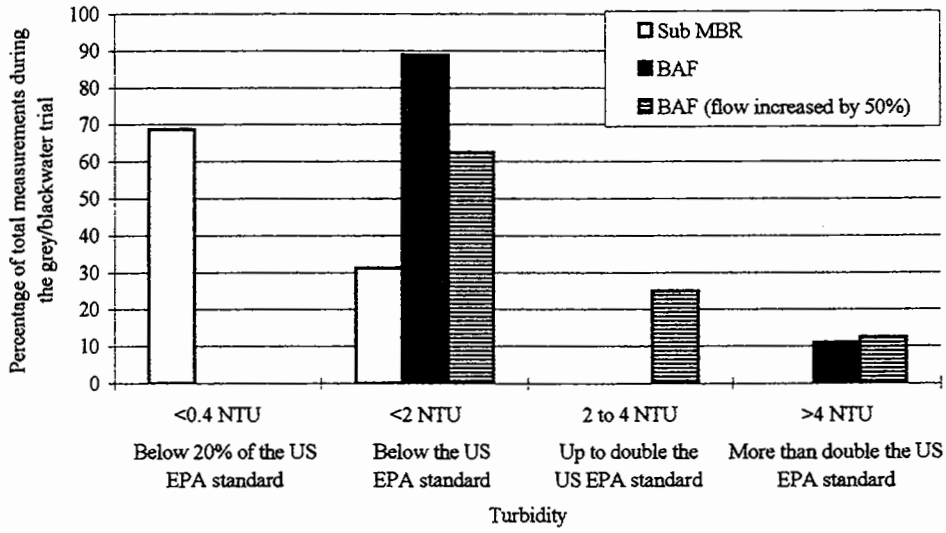


Figure 5.11. Distribution of the effluent turbidity during the grey/blackwater trial with respect to the US EPA water reuse guideline.

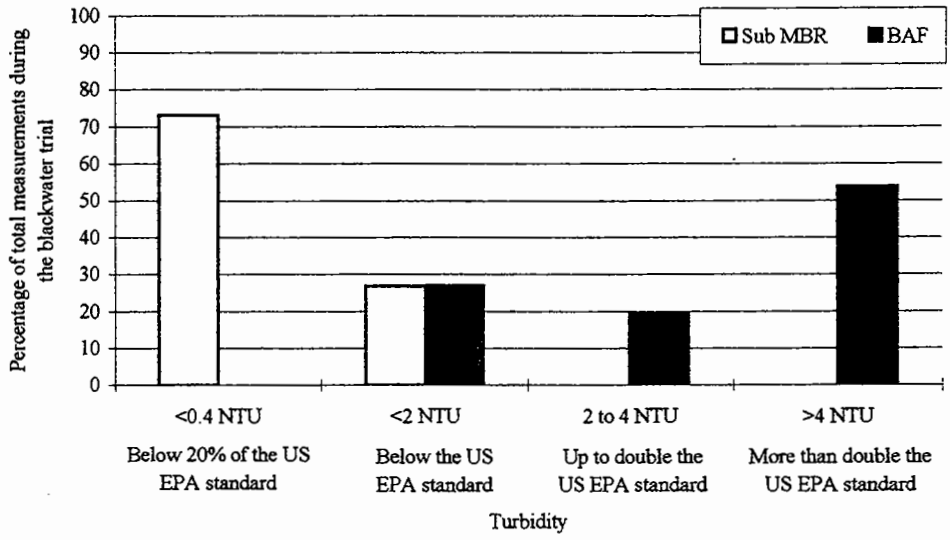


Figure 5.12. Distribution of the effluent turbidity during the blackwater trial with respect to the US EPA water reuse guideline.

5.2.5.3 pH

In most water reuse quality criteria the pH is set between 6 and 9 (Table 2.5, Section 2.3) to avoid long-term contact of any material with acidic or alkaline water. All treatment processes produced effluent with a stable pH at around 7.0 (Table 5.6,

Figure D.18 in Appendix D) and would thus not require chemical adjustment to the desired level. This was expected as the influent pH remained stable despite an increase from 7.0 ± 0.2 in greywater to 7.7 ± 0.2 in blackwater (Table 5.6, Figure D.17 in Appendix D), largely due to the buffering capacity provided by the alkalinity. pH was not measured for the side-stream MBR; the feed pH of the side-stream MBR was assumed to be unalternated by treatment, as with the submerged MBR.

Some stages of the biological treatment are strongly dependent on pH. Nitrification is favoured by a pH range of 7.2-9.0 (Metcalf and Eddy, 1991) and becomes completely inhibited at a pH <5.0 (Wild *et al.*, 1971). A low pH also affects heterotrophic activity, reducing BOD removal. The results from the current study show that nitrification and organic removal were satisfactory or good over the narrow pH range investigated.

5.2.6 Pathogens

5.2.6.1 General

The reuse water quality guidelines chosen for total coliforms and faecal contaminants in this work were those of the UK bathing water standard, such that the lower guideline values (Table 5.1) defined the limit. The UK bathing water standards for both total coliforms and faecal streptococci (Table 2.5, Section 2.3) allow the presence of pathogen in the effluent, in contrast to the US EPA standard where a non-detectable level is required (Table 2.5, Section 2.3). However, it can be argued that such a strict criterion for non-potable reuse would not be necessary if the risk of human contact is low, as is the case in toilet flushing.

5.2.6.2 Total coliforms

The submerged and side-stream MBRs produced effluents of good quality at a mean 6 log total coliform removal from the influents (Table 5.7), with the majority of the samples having a bacteria count below the limit of detection (Figures 5.15-5.17 and D.21 in Appendix D). Hence these processes would meet even the strictest non-

potable water reuse criteria, such as the US EPA standard. The complete retention of solids achievable in membrane bioreactors is thought to include bacteria and viruses (Stephenson *et al.*, 2000). The very good bacteria removal in the current study confirm this theory. The MBRs hence have a major advantage in comparison with systems in which the reduction of the pathogens present in greywater is through disinfection by chlorine, as in the case in a basic two-stage process (Sayers, 1998), or by exposure to UV light, such as in some greywater schemes using rotating biological contactors and fluidised bed reactors as treatment processes (Nolde, 1999). The results from this work also appear to compare favourably with physical membrane filtration of greywater (Le Clech *et al.*, 2000) where only a 3 log total coliform reduction was achieved.

On a few occasions during steady-state operation a higher coliform count than expected occurred in the submerged MBR (Figures 5.12-5.14). The most likely explanation for this is sample contamination in the effluent line, as reported by Ueda and Hata (2000) on a submerged MBR.

Since the BAF does not present an actual physical barrier, the pathogen removal from greywater was unsurprisingly lower (2 log) than that of the MBR systems (Table 5.7). On changing to grey/blackwater and later to blackwater more BAF effluent samples were found to have more than 10 times the total coliform count the UK bathing water standard (Figures 5.12-5.14 and D.21 in Appendix D), whilst the pathogen removal capacity remained at 1-3 log (Table 5.7). In comparison to this, the pathogen rejection of 4-7 log (to the non-detectable level) offered by the submerged and side-stream MBRs alike provides a substantially disinfected treated product water, all but obviating downstream chemical disinfection.

Table 5.7. Bacteriological quality of the influents and effluents during the greywater, *grey/blackwater* and **blackwater** trials. Mean ± standard deviation.

Parameter	Influent	Side-stream MBR influent	Submerged MBR		Side-stream MBR		MABR		BAF	
			Effluent	Removal (log)	Effluent	Removal (log)	Effluent	Removal (log)	Effluent	Removal (log)
Total coliforms (cfu 100^{-1} ml^{-1})	1.47E+06 ± 4.34E+06	3.00E+04 ± 3.35E+04	2.19E+00 ± 5.35E+00	6	1.17E+00 ± 3.88E-01	4	1.96E+04 ±	2	2.04E+04 ± 5.47E+04	2
	4.59E+07 ± 1.12E+08	-	*1.02E+02 ± 3.97E+02*	*7*	-	-	6.18E+04 -	-	*3.01E+04 ± 3.52E+04*	*3*
									*(8.78E+04 ± 1.02E+05)**	*(3)**
	1.96E+07 ± 1.37E+07	-	**2.05E+01 ± 6.86E+01**	**6**	-	-	-	-	**2.24E+06 ± 3.51E+05**	**1**
E.coli (cfu 100^{-1} ml^{-1})	4.22E+03 ± 4.85E+03	2.28E+02 ± 3.08E+02	1.00E+00 ± 0.00E+00	3	1.00E+00 ± 0.00E+00	2	2.01E+0.3 ±	0	2.59E+03 ± 4.41E+03	0
	1.29E+06 ± 1.88E+06	-	*1.00E+00 ± 0.00E+00*	*6*	-	-	4.13E+03 -	-	*1.71E+03 ± 2.44E+03*	*3*
									*(1.04E+04 ± 1.13E+04)**	*(2)**
	3.56E+06 ± 3.37E+06	-	**1.00E+00 ± 0.00E+00**	**6**	-	-	-	-	**4.03E+05 ± 5.86E+05**	**1**
Faecal streptococci (cfu 100^{-1} ml^{-1})	1.16E+03 ± 2.29E+03	-	6.01E+01 ± 3.78E+02	2	-	-	1.65E+03 ±	0	7.81E+02 ± 1.80E+03	1
	1.13E+05 ± 1.03E+05	-	*1.00E+00 ± 0.00E+00*	*5*	-	-	2.59E+03 -	-	*1.26E+03 ± 1.64E+03*	*2*
									*(2.29E+03 ± 1.35E+03)**	*(2)**
	1.53E+05 ± 5.27E+04	-	**1.00E+00 ± 0.00E+00**	**5**	-	-	-	-	**5.41E+03 ± 4.03E+03**	**2**

() when flow to the BAF increased by 50%*

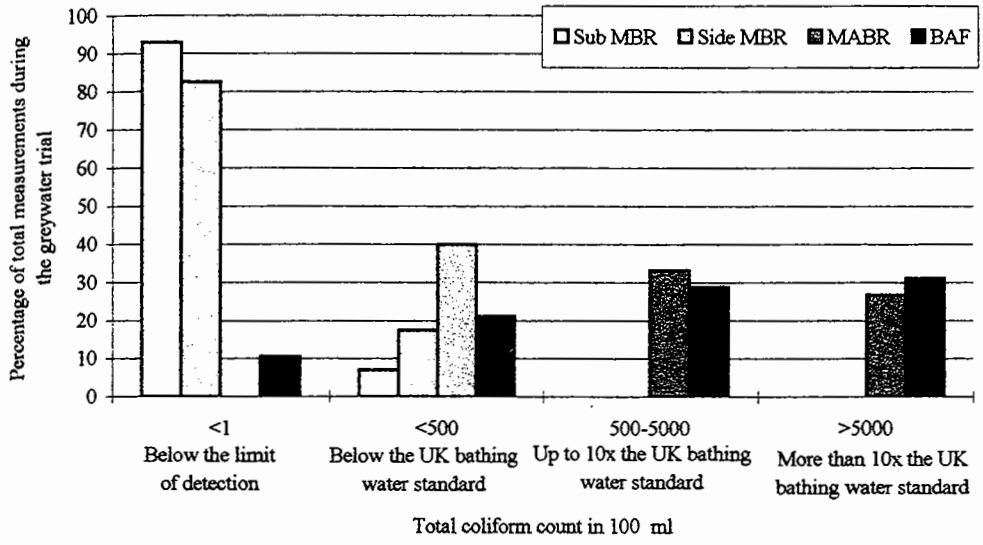


Figure 5.12. Distribution of the effluent total coliforms during the greywater trial with respect to the UK bathing water standard.

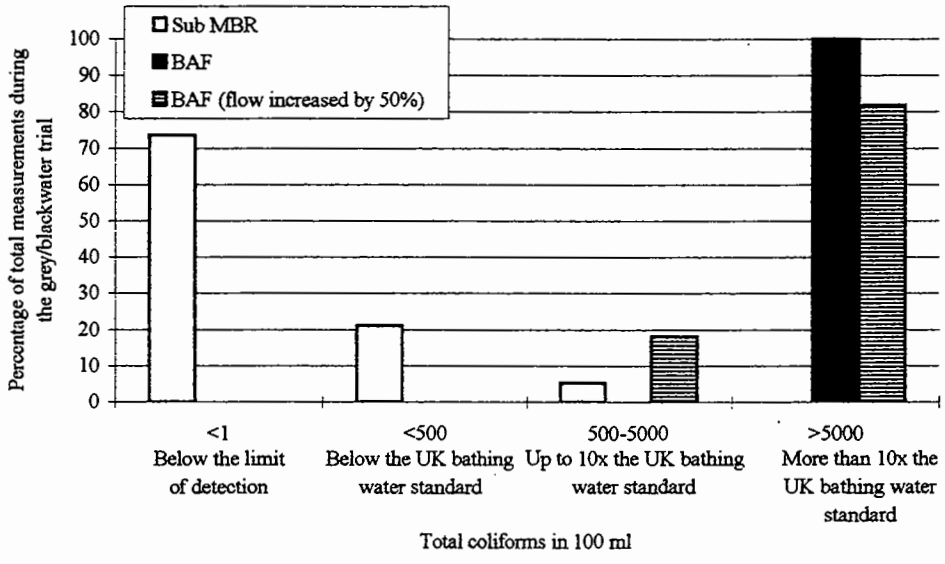


Figure 5.13. Distribution of the effluent total coliforms during the grey/blackwater trial with respect to the UK bathing water standard.

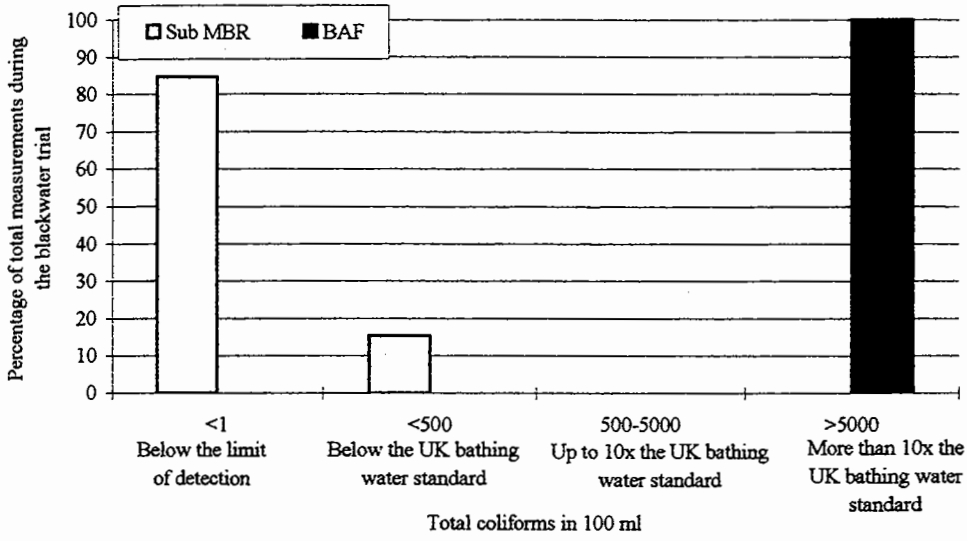


Figure 5.15. Distribution of the effluent total coliforms during the blackwater trial with respect to the UK bathing water standard.

5.2.6.3 Faecal contamination

Faecal streptococci

Faecal contamination in the samples was determined as faecal streptococci and *E.coli* counts, the influent and effluent values of which are shown in Figures D.22-D.23 in Appendix D. The distribution of the faecal streptococci count data in the effluent samples is presented in Figures 5.16-5.18 with respect to the UK bathing water standard. Faecal streptococci counts were not determined from the side-stream MBR samples as *E.coli* was considered a sufficient determinant of faecal contamination during the 32-day greywater trial.

The submerged MBR performed best throughout the trials at a faecal streptococci removal from 2 to 5 log, meeting the standard at all times, whilst the majority of the samples had non-detectable bacteria (Table 5.7, Figures 5.16-5.18 and D.22 in Appendix D). On one occasion at the beginning of the greywater treatment a bacteria count of over 10 times the standard (Figure 5.16) was measured, most likely as a result of sample contamination in the effluent line. The BAF achieved 1-2 log faecal streptococci removal throughout the trials (Table 5.7). The high number of samples

with a bacteria count of more than 10 times that of the standard reflects the absence of an actual physical barrier in the process (Figures 5.18-5.20 and D.22 in Appendix D).

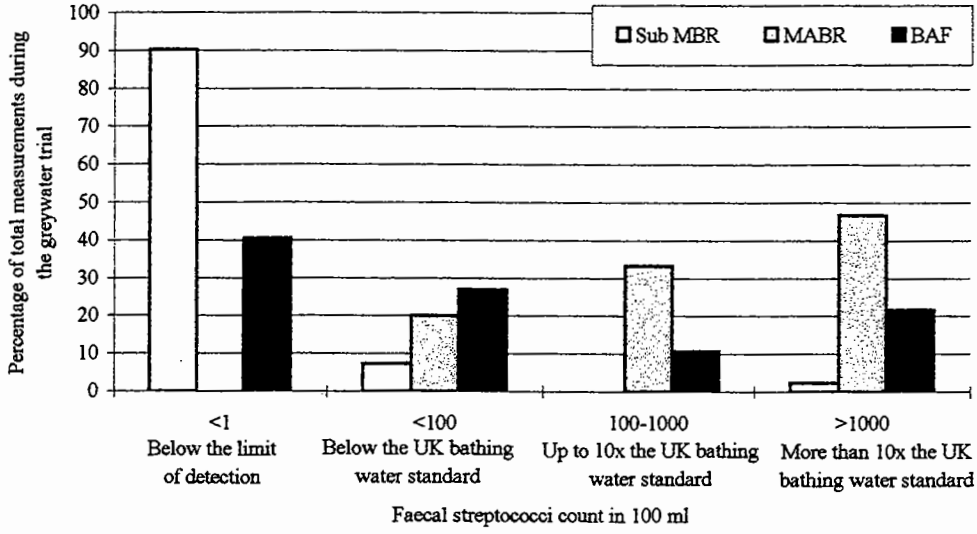


Figure 5.16. Distribution of the effluent faecal streptococci during the greywater trial with respect to the UK bathing water standard.

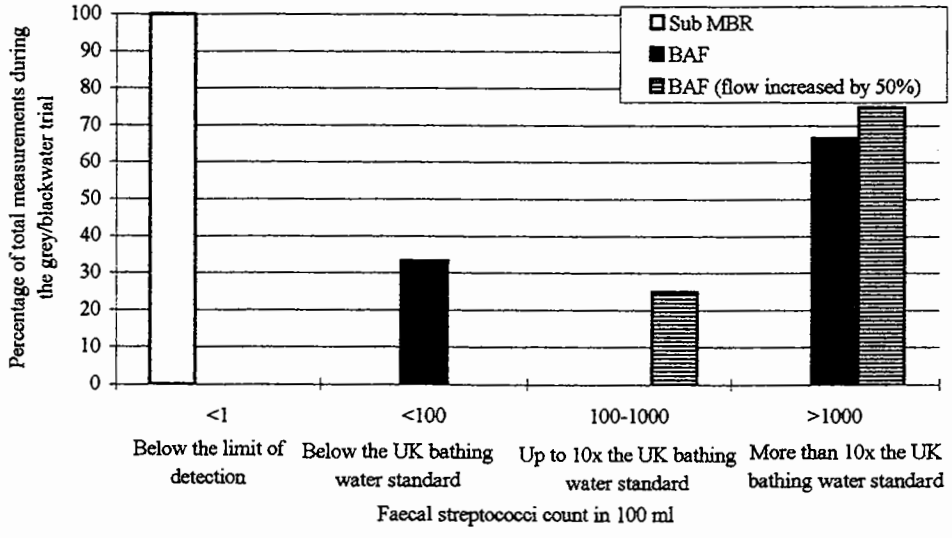


Figure 5.17. Distribution of the effluent faecal streptococci during the grey/blackwater trial with respect to the UK bathing water standard.

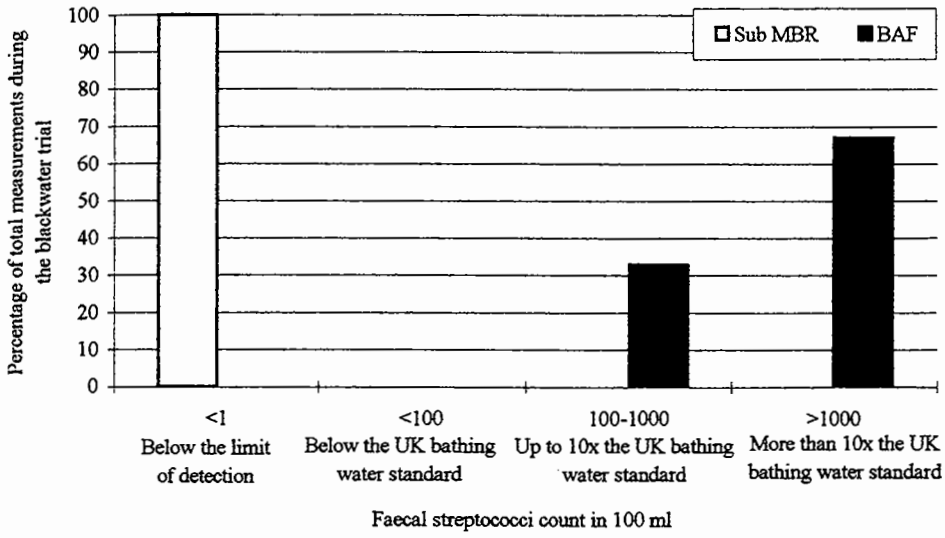


Figure 5.18. Distribution of the effluent faecal streptococci during the blackwater trial with respect to the UK bathing water standard.

E.coli

Most of the existing non-potable water reuse criteria do not include a limit for the *E.coli*. However, in Figures 5.19-5.21 this parameter is compared to the faecal coliform limits of the UK bathing water standard. The submerged and side-stream MBRs produced effluent that was essentially free from the *E.coli* bacteria 100% of the time independent of the influent quality (Table 5.7, Figures 5.19-5.21 and D.23 in Appendix D). The BAF achieved an *E.coli* removal of up to 3 log such that 26% of the samples were below the limit of detection during the greywater trial with the rest of the samples having a significant variation up to 3 log (Figure 5.19). When treating grey/blackwater and blackwater, the fraction of the samples with a higher count increased in the BAF effluent (Figures 5.20-5.21 and D.23 in Appendix D), as was the case for other indicator bacteria used in the current study.

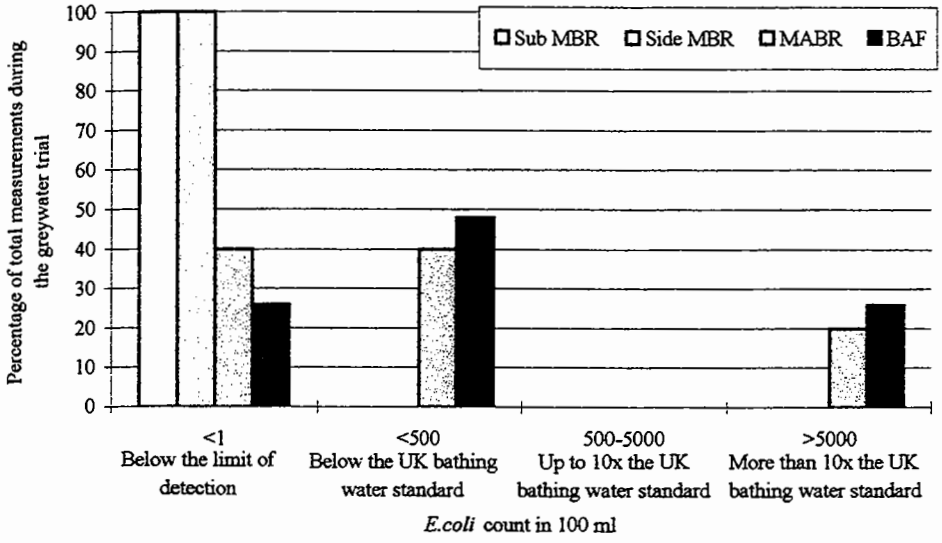


Figure 5.19. Distribution of the effluent *E.coli* during the greywater trial with respect to the UK bathing water standard.

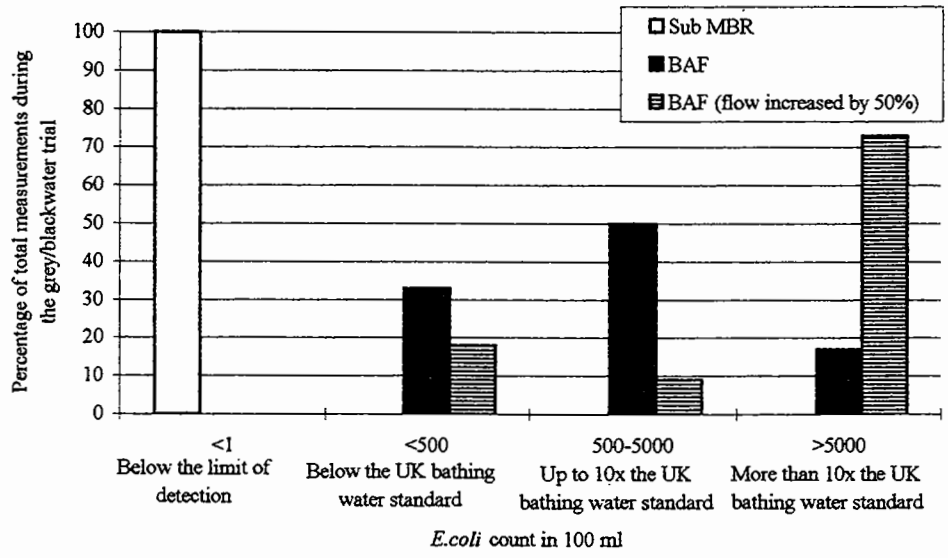


Figure 5.20. Distribution of the effluent *E.coli* during the grey/blackwater trial with respect to the UK bathing water standard.

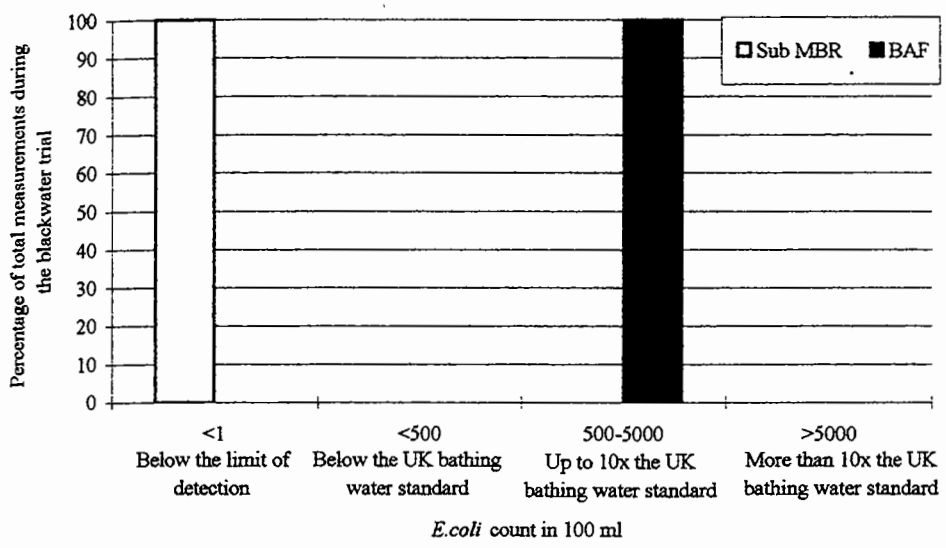


Figure 5.20. Distribution of the effluent *E.coli* during the blackwater trial with respect to the UK bathing water standard.

5.2.7 Mixed liquor of the MBRs

The performance of an MBR is dependent on the concentration and biological behaviour of the sludge in the bioreactor, which is quantified as the MLSS (the mixed liquor suspended solids) and the MLVSS (the volatile component of the MLSS). Food to micro-organism (F:M) ratio relates to the substrate load applied to the MLSS, being an important design parameter for activated sludge processes. In the current work the F:M ratios are presented in terms of the BOD.

The initial MLSS concentration in the submerged MBR was 10 000 mg l^{-1}. Within 19 days from the start-up the solids accumulated mainly in the denitrification units resulted in a mean MLSS of 274 mg l^{-1} and 14 381 mg l^{-1} in the nitrification and denitrification units respectively (Figures D.24-D.25 in Appendix D), with corresponding MLVSS fractions of 83 ± 8% and 73 ± 5%. The accumulation of the solids in the denitrification units was a result of insufficient mixing and nutrient balancing of the influent did not solve this accumulation problem.

Over the 300 days of continuous operation the submerged system appeared to be relatively independent of the largely variable MLSS and F:M ratio (0.04-0.38 d^{-1}), indicating that detrimental effects did not occur during the operating regime. Examples of detrimental effects are more severe membrane fouling than expected (Section 5.2.2.3) or a deterioration in the removal of organic matter. Previously published work (Fan *et al.*, 1999) demonstrated that a variable MLSS (3000-7500 mg l^{-1}) had no effect on membrane fouling during a 300-day operation of a side-stream MBR, and thus supports the findings from this work. Once an equilibrium MLSS had been established in the submerged MBR of the current study, the long sludge age (approaching infinity) and hydraulic retention time (Table 5.3) implied that the process functioned according to the maintenance concept. This is a situation where feed entering the biomass is insufficient to allow cell growth such that all available food is used to maintain the microbiological population (Pirt, 1965). As the feed decreases this balance forces the start of auto digestion bringing the biomass back into equilibrium at a lower MLSS concentration. This produces a stable mixed liquor, which is appropriate to all zero wastage systems. The practical implications of this are the high stable mixed liquors in these systems. However, the particular benefits lie in the long sludge ages allowing the development of specialised micro-organisms to treat the wastewater (Stephenson *et al.*, 2000).

The MLSS in the side-stream MBR dropped from the initial 4900 mg l^{-1} to around 3000 mg l^{-1} over the 32 days of operation (Figure D.26 in Appendix D), indicating food limitation to micro-organism growth. The volatile fraction (67 ± 7%) and F:M ratio (0.01 d^{-1}) were the both lowest of the three MBRs. The long HRT observed resulted in very good organic removal (Sections 5.2.4.1-5.2.4.2).

A combination of a high MLSS and a low F:M ratio was achieved in the modified submerged MBR units. For unit A corresponding values of 13 618 ± 952 mg l^{-1} and 0.04 d^{-1} were measured over 89 days (Figure D.27 in Appendix D). The similar volatile fractions in this (69 ± 3%) and the side-stream configurations indicated similar level of activity. In the unit B the MLSS, F:M and MLVSS over a 5-month

period were 12 232 ± 2431 mg l^{-1}, 0.03 d^{-1} and 73 ± 11% (Figure D.28 in Appendix D). The HRTs (3.3 ± 0.4 h in A and 4.3 ± 0.8 in B) in these modified units were significantly lower than those in the other MBR systems in the current study. At this stage the greywater treatment performance was not regularly monitored as the processes were used for other experiments including intermittent operation trials (Section 6.3) and respirometry work (Section 6.4). Grab samples of the effluent taken during the early part of the steady-state trial were of a very high quality (Table D.1 in Appendix D).

The large variation in the MLSS of the modified unit B resulted from the solids concentration remaining relatively stable for 80 days (MLSS decline of 0.4 mg l^{-1} d^{-1}) and then dropping rapidly at 63 mg l^{-1} d^{-1} for the subsequent 80 days (Figure D.28 in Appendix D). The decline was adhered to the greywater not providing sufficient food for the micro-organisms. This compares well with a situation where a system is not adequately fed due to intermittent operation. Davies *et al.* (1998) reported a submerged MBR operated for two months on a cyclic mode. The daily mode consisted of feeding for 12 h on raw sewage and starvation for 12 h. During the 2-month period the MLSS dropped by two thirds presumably due to endogenous digestion. After the rapid initial drop in the sludge concentration the rate of decline stabilised at a lower level.

Scott *et al.* (1998) reported an MBR successfully operating at a MLSS lower than the design values. This high-strength food industry wastewater was severely nitrogen deficient with a BOD_5:N:P of 1000:1.7:5. The combination of this and the low MLSS (1500 mg l^{-1}) contributed to a limited cell growth rate, which was consequently enhanced by adding ammonia sulphate (300 mg N l^{-1}) to the bioreactor. The removal efficiency (98% COD, 99% BOD_5 and 99% SS) indicated that an indigenous microflora, with a high specific activity for the waste, are first retained by the membrane and then built up in the bioreactor. These findings are supported by the current study, despite the differences in wastewater quality and the MLSS.

DO and temperature

The aeration rates in all three MBRs were similar such that similar dissolved oxygen levels were measured in the side-stream and modified submerged configurations (8.0 mg l^{-1} and 7.4 ± 1.2 mg l^{-1}, respectively). Throughout the trials the DO in the nitrification units of the submerged MBR was maintained at 7.9 ± 1.0 mg l^{-1} by the high air flow to the system. The corresponding value of 2.2 ± 1.0 mg l^{-1} measured in the denitrification units during the grey/blackwater and blackwater trials suggests that the chamber was not operated under anaerobic conditions at all times. Values below 0.5 mg l^{-1} were occasionally measured, however, and the enzyme system needed for the anaerobic bacteria was possibly suppressed by the presence of dissolved oxygen (Metcalf and Eddy, 1991) most of the time.

The temperature in the nitrification and denitrification units of the submerged MBR was 24 ± 4°C and 19 ± 3°C, respectively. For the modified configuration 22 ± 4°C was measured, the variation relating to the seasonal changes in ambient temperatures. Temperature was not determined for the side-stream MBR sludge but was presumed to be similar to that of the effluent (Table 5.6).

5.3 Nutrient addition to enhance performance

5.3.1 Nutrient addition in pilot-scale systems treating greywater

Addition of selected nutrients to generally nutrient-deficient greywater has demonstrated a beneficial effect on the respiration rate and the COD removal of activated sludge (Section 4.3.3). The principle was applied to an operating pilot-scale system, in this case to a BAF as its flow rate is easier to control than that of the submerged MBR. The assessment involved addition of nitrogen and phosphorus which had previously been identified as the key missing components in greywater (Section 4.3.2).

Nutrient addition can be a useful process-enhancement measure on occasions when a poor quality effluent is produced due to a significantly increased organic loading rate to the unit or when a weak wastewater is more difficult to treat by a biological process (Sections 2.2.2 and 5.3.4). Jefferson *et al.* (2000b) reported synthetic soap solution treatment by a BAF. After nutrient addition higher OLRs were be applied to a BAF without a loss of BOD and COD removal efficiency. It was also observed that an upper loading rate limit exists at 1.9 kgBOD m^{-3} d^{-1} and 4.3 kgCOD m^{-3} d^{-1} beyond which nutrient balancing ceases to offer an advantage.

5.3.2 Procedure

The flow rate of the BAF was sequentially increased to the point of general deterioration in the effluent quality after which the daily prepared greywater (Section 3.2.2.2) was nutrient-balanced to investigate process enhancement. It was of particular interest to observe changes in the removal of organic matter at feed flow rates and OLRs higher than those during the previously-described steady-state trials with greywater (Section 5.2). The five-stage procedure was as follows:

- days 1-10: flow at 0.302 m^3 d^{-1},
- days 11-17: flow rate increased by 50%,
- days 18-24: flow rate increased by 100%,
- days 25-31: nutrient solution A (footnote in Table 5.8) added, and
- days 32-35: nutrient solution B (footnote in Table 5.8) added.

Based on relevant work (Jefferson *et al.*, 2000b), a mixture of ammonium sulphate (61.1 mg l^{-1}) and phosphoric acid (0.01 ml l^{-1}) was first used as the nutrient solution (A) in this work, resulting in a rapid decline in the effluent pH to below 7.0 (Table 5.8). The effluent pH increased to the original level on replacing the mixture by that of ammonium sulphate (47.4 mg l^{-1}) and diammonium hydrogen orthophosphate (12.5 mg l^{-1}) (nutrient solution B).

Table 5.8. The performance of the BAF during the nutrient addition trials. Mean ± standard deviation. *(Removal %, * log).*

Parameter	Flow at 0.302 $m^3 d^{-1}$		Flow +50%		Flow +100%		Nutrient *A* added		Nutrient *B* added	
	Influent	Effluent	Influent	Effluent	Influent	Effluent	Influent	Effluent	Influent	Effluent
Flow ($m^3 d^{-1}$)	-	0.302 ± 0.015	-	0.500 ± 0.031	-	0.536 ± 0.033	-	0.524 ± 0.042	-	0.536 ± 0.030
OLR (kgBOD $m^{-3} d^{-1}$)	-	0.59 ± 0.30	-	1.18 ± 0.47	-	1.57 ± 0.30	-	1.13 ± 0.17	-	1.08 ± 0.37
Solids load (kgSS $m^{-3} d^{-1}$)		0.41 ± 0.18		0.50 ± 0.24		1.33 ± 1.30		0.44 ± 0.10		0.55 ± 0.04
BOD_5 (mg l^{-1})	97 ± 49	11 ± 7 *(88 ± 7)*	115 ± 44	16 ± 8 *(87 ± 2)*	144 ± 22	29 ± 11 *(80 ± 7)*	141 ± 41	9 ± 6 *(92 ± 6)*	100 ± 30	6 ± 3 *(93 ± 5)*
tCOD (mg l^{-1})	227 ± 63	29 ± 16 *(88 ± 4)*	237 ± 64	47 ± 22 *(53 ± 11)*	363 ± 136	63 ± 21 *(80 ± 12)*	209 ± 60	26 ± 17 *(88 ± 9)*	220 ± 38	27 ± 12 *(87 ± 6)*
sCOD (mg l^{-1})	69 ± 32	16 ± 13 *(80 ± 15)*	35 ± 20	19 ± 20 *(67 ± 43)*	90 ± 36	38 ± 17 *(58 ± 14)*	57 ± 15	13 ± 10 *(78 ± 22)*	69 ± 32	20 ± 0 *(64 ± 22)*
SS (mg l^{-1})	67 ± 30	10 ± 8 *(81 ± 17)*	49 ± 23	17 ± 9 *(57 ± 29)*	122 ± 117	26 ± 6 *(72 ± 17)*	44 ± 11	5 ± 5 *(91 ± 10)*	51 ± 4	3 ± 3 *(93 ± 7)*
Turbidity (NTU)	-	5.4 ± 3.9	-	8.2 ± 3.5	-	16.1 ± 6.7	-	2.4 ± 1.7	-	1.4 ± 0.9
TN (mg l^{-1})	-	-	-	-	9 ± 10	5 ± 3 (59 ± 2)	13 ± 2	8 ± 1 (46 ± 24)	12 ± 4	7 ± 1 (39 ± 13)
P (mg l^{-1})	-	-	0.45 ± 0.14	0.38 ± 0.04 *(11 ± 36)*	2.94 ± 3.70	2.85 ± 4.03 *(49 ± 73)*	5.04 ± 0.43	3.81 ± 1.05 *(28 ± 19)*	3.69 ± 0.26	2.94 ± 0.56 *(19 ± 18)*
pH (-)	7.5 ± 0.4	7.6 ± 0.2	7.3 ± 0.1	7.6 ± 0.3	7.2 ± 0.1	7.7 ± 0.2	6.9 ± 0.1	6.9 ± 0.3	7.1 ± 0.1	7.1 ± 0.2
DO (mg l^{-1})	-	8.4 ± 1.1	-	6.6 ± 1.2	-	7.6 ± 1.0	-	8.3 ± 0.7	-	8.3 ± 0.8
Temperature (°C)	-	22 ± 3	25 ± 4	22 ± 4	26 ± 6	22 ± 3	22 ± 4	23 ± 3	27 ± 1	24 ± 2
Total coliforms (cfu 100^{-1} ml^{-1})	22626 ± 22684	6890 ± 14904 *(1*)*	11862 ± 4375	41098 ± 39615 *(0*)*	3896 ± 4766	110076 ±104284 *(0)*	753 ± 720	5783 ± 4363 *(0*)*	1464 ± 883	1330 ± 1166 *(0*)*
E.coli (cfu 100^{-1} ml^{-1})	2575 ± 4583	91 ± 26 *(2*)*	3396 ± 7370	7740 ± 14973 *(0*)*	248 ± 331	1056 ± 1856 *(0*)*	117 ± 41	325 ± 504 *(0*)*	140 ± 89	184 ± 188 *(0*)*
Faecal streptococci (cfu 100^{-1} ml^{-1})	1827 ± 1097	970 ± 1207 *(1*)*	470 ± 956	873 ± 1198 *(0*)*	2419 ± 0	1564 ± 1908 *(0*)*	81480 ± 139850	100 ± 0 *(2*)*	1492 ± 1270	78 ± 48 *(2*)*

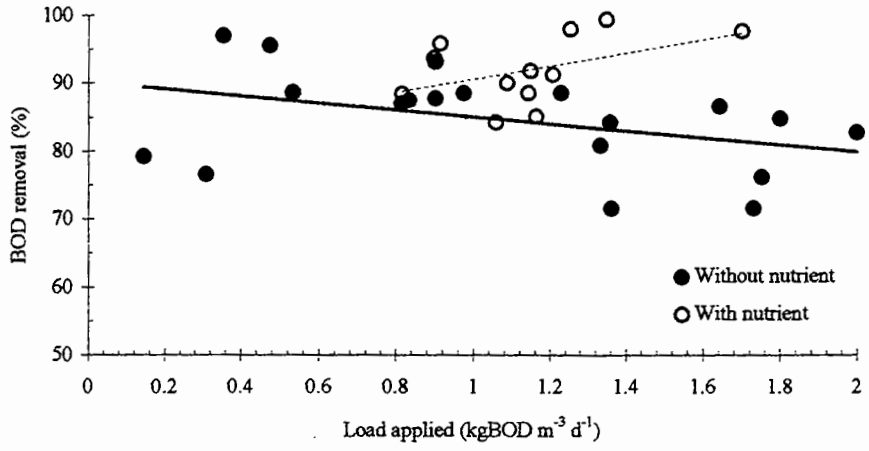
A Ammonium sulphate (61.1 mg l^{-1}) and phosphoric acid (0.01 ml l^{-1}) *B* Ammonium sulphate (47.4 mg l^{-1}) and diammonium hydrogen orthophosphate (12.5 mg l^{-1})

5.3.3 Greywater

The greywater C:TN:P ratio of 500:9:1 changed to 49:3:1 (average of A and B) on adding a nutrient solution. This did not conform to the reference ratios reported in the literature (Section 4.3.1) as well as expected due to the fluctuating feedwater quality. The COD:BOD ratio of 2.76 ± 1.26 in the nutrient-limited greywater dropped to 2.05 ± 0.60 (average of A and B) in the nutrient-balanced greywater. In both types of greywater degradation of organic matter was accelerated both by the precipitation of the solids and the temperature in the feedtank (24 ± 4°C). Both aspects have been discussed in Section 4.2.3. Within 24 h from the preparing the greywater solution reductions of 6-54%, 20-53% and 6-85% were measured in BOD_5, total nitrogen and phosphorus concentrations, respectively. Though greywater degradation patterns are case-specific depending on factors such as storage conditions and greywater quality, similar findings have been reported elsewhere (Section 4.4).

5.3.4 Flow vs.quality

The increased flow rate to the BAF combined with the variable influent BOD_5 first doubled the OLR to 1.18 kgBOD m^{-3} d^{-1} for six days and then increased it by an additional 33% to 1.57 kgBOD m^{-3} d^{-1} over the following six days. This was followed by a decline by about a third to 1.08-1.13 kgBOD m^{-3} d^{-1} during the nutrient addition stages (Table 5.8). During the entire period the process achieved a reasonable though inconsistent BOD removal which improved after nutrient addition (Figure 5.22). At a flow rate of 0.302 m^3 d^{-1} 88% of the biodegradable organic matter in the greywater was removed, with some 11 mg l^{-1} of BOD remaining in the effluent. The increased flow had little impact on the removal whereas the mean concentration in the effluent increased to 16 mg l^{-1} (Table 5.8). After the second step change in the flow rate the removal declined to 80% whilst the mean effluent BOD was 29 mg l^{-1}. Once the nutrients were added to the greywater, the organic removal improved to 92-93% (Figure 5.22, Table 5.8), with the effluent BOD being 6-9 mg l^{-1}.



gure 5.22. BOD removal versus BOD loading rate with and without nutrient addition to greywater.

s shown in Table 5.8, the trend for the tCOD was similar to that of the BOD_5 though ss clear. The 88% tCOD removal at 1.33 ± 0.37 kgCOD m^{-3} d^{-1} dropped to 53% at 32 ± 0.67 kgCOD m^{-3} d^{-1} followed by an increase to 80% in the third stage at a oading rate of 4.17 ± 1.61 kgCOD m^{-3} d^{-1} (Table 5.8). During these stages the ffluent tCOD increased from the 29 mg l^{-1} to 63 mg l^{-1} (Table 5.8). After the nutrient ddition at a mean loading rate of 2.28-2.39 kgCOD m^{-3} d^{-1} the effluent tCOD and emoval recovered to 26-27 mg l^{-1} and 87-88%, respectively (Table 5.8). Possibly due o the solids being more readily filtered out, most of the time the sCOD removal was ower than that of the tCOD. The former gradually declined until the introduction of utrient A, such that the performance nearly recovered to the original level before it lropped again on addition of nutrient B (Table 5.8). The data indicate that the ncreased amount of nutrients available for the micro-organisms can results in mproved removal of organics, even during a relatively short test period. These indings are in good agreement with those of previous nutrient balancing studies Ghyoot *et al.*, 1999; Saunamäki 1994; Tam *et al.*, 1992).

Though the solids load to the unit at 0.41-0.50 kgSS m^{-3} d^{-1} was similar in the first two stages, the removal decreased from 81% to 57% (Table 5.8) due to the increased flow rate. In the third stage a further increase in the flow rate combined with a significantly higher influent solids at a mean loading of 1.33 kgSS m^{-3} d^{-1} lead to a

72% SS removal (Table 5.8), indicating that the biomass was able to utilise the nutrients within the solid matter. After the nutrient additions the solids removal improved to 91-93% with the effluent quality being very good at 3-5 mg l^{-1}. The influent solids at a loading rate of 0.44-0.55 kgSS m^{-3} d^{-1} was similar to that measured at the beginning of the trial (Table 5.8). It is not clear however whether the nutrients enhanced solids removal or not.

Turbidity, being an easily performed test, provided a relatively rapid indicator of the deterioration in the effluent quality (Figure 5.23). It gradually increased from 5.4 to 8.2 NTU with the first step change and then trebled to 16.1 NTU as the flow rate was doubled (Figure 5.23, Table 5.8). On addition of nutrient solution A to the greywater, the mean turbidity dropped rapidly to 2.4 NTU and achieved a stable value of 1.4 ± 0.9 NTU after the addition of nutrient B (Table 5.8).

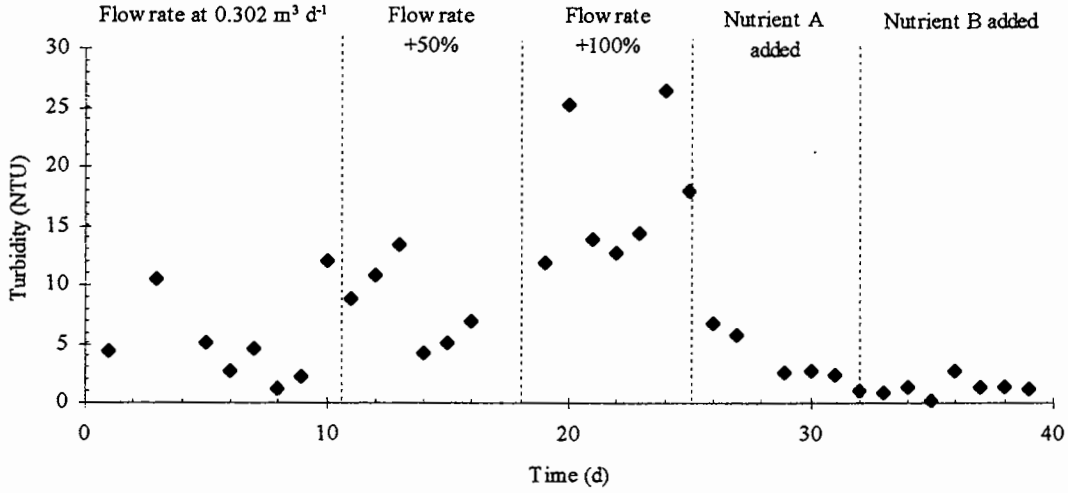


Figure 5.23. BAF effluent turbidity during the nutrient addition trials. Nutrient A = ammonium sulphate and phosphoric acid, nutrient B = ammonium sulphate and diammonium hydrogen orthophosphate.

Although nitrogen and phosphorus were no longer limiting after nutrient balancing, the removal of TN and P was found to decrease (Table 5.8). This may have resulted from the degradation of the greywater and hence the fluctuating influent quality (Section 5.3.3.3). Faecal streptococci removal appeared to increase on nutrient balancing whereas no such trend was observed for total/*E.*coliforms (Table 5.8),

ıough the change cannot be regarded as statistically significant. On the whole this /ork demonstrates the potential to enhance a biological process performance in terms f effluent turbidity and BOD and COD removal by adding macronutrients to reywater. Optimising the process during the nutrient addition may have a key role in voiding variable performance.

Nitrogen and/or phosphorus levels in paper and pulp mill effluents are often too low or successful biological treatment, therefore nutrient addition is necessary (Järvinen, .997; Saunamäki, 1997). The effect of BOD:P ratio on the treatment of paper mill effluent by an activated sludge plant has been studied (Saunamäki, 1994 and 1997). At a VLR of 0.77 kgBOD m^{-3} d^{-1} and a BOD:P ratio of 100:(0.56-1.10) some 91-92% BOD, 74-76% COD and 86-93% SS were removed from the paper mill wastewater (Saunamäki, 1994). Below a BOD:P ratio of 100:0.4 insufficient phosphorus available for the biomass lead to decreased BOD (82%), COD (69%) and SS (67%) removal. These declined rapidly (71-76%, 50-60% and 60-78% respectively) on doubling the loading rate (1.66 kgBOD m^{-3} d^{-1}) despite a beneficial nutrient ratio, suggesting that the loading rate was too high. At another paper mill the phosphorus levels in the effluent were sufficient, but nitrogen at a BOD:N ratio of 100:(3.0-4.5) was added to the effluent prior to the treatment (Saunamäki, 1994). At volumetric loading rates of 0.38-0.77 kgBOD m^{-3} d^{-1} the mean BOD, COD and TN removal were >98%, 84% and 69%.

Adachi and Fuchu (1991) reported a BAF treating aluminium rolling process wastewater to which urea and phosphate were added to equal a BOD:N:P ratio of 100:5:1. The initial BOD, COD_{Mn} and SS levels of 42 mg l^{-1}, 32 mg l^{-1} and 16 mg l^{-1}, respectively were reduced to 5 mg l^{-1}, 20 mg l^{-1} and 7 mg l^{-1} which met the effluent quality criteria. The n-hexane extracts, the main concern in the influent, were reduced from 40 mg l^{-1} to 10 mg l^{-1} such that the target values also in terms of this parameter were met. The effluent TN and TP levels were slightly higher than those in the influent due to a leakage of nutrient salts from the BAF.

4 Operational experience

erational issues are a key factor in recycling technology selection. Despite the reasing number of existing reuse schemes information on problems with greywater atment processes is scarce and often originates from supplementary studies. One of : most common concerns is the inconsistent greywater flow due to water usage tterns (Section 2.2.1). Hence balancing of the storage volume is often required to sure greywater supply to the online treatment processes.

oblems encountered during the steady-state greywater trial of the current work are mmarised in Table 5.9. A logbook on the daily operation of the plant kept during ıe steady-state trials is shown in Table E.1 in Appendix E. During the start-up stage ıe flow to the treatment processes was significantly reduced or entirely prevented by air blocking the tubing. As it was not the major component of the synthetic greywater it was omitted from the analogue after day 5 of the steady-state trial. In-)uilding treatment processes often comprise pre-screening of the greywater to remove arger solid matter such as hair and lint. In particular basic two-stage systems are prone to filter blocking (Christova-Boal *et al.*, 1996; Sayers, 1998) because of the coarseness of the filter and hence may cause inconvenience for the user unless regularly cleaned.

All the treatment processes except for the side-stream MBR were subject to foaming. This was most deleterious in the MABR where it lead to severe complications during the start-up procedure (Section 5.2.2). The BAF foamed occasionally for 2-3 days and during the start-up whilst the submerged MBR foamed on transition to grey/blackwater. However, loss of treatment or biomass was not observed. Foaming is a typical characteristic of aerated fresh greywater and is dependent on the soap or shampoo composition (Section 2.2). It can be prevented by using antifoaming agents (Wheale and Cooper-Smith, 1995), foam blockers or water sprays (Holmes and Dutt, 1999). Basic two-stage processes have been reported (Sayers, 1998) to be subject on long-term to this problem because of the inadequate treatment.

ble 5.9. Summary of operational problems related to greywater treatment.

roblem	Impact on process (action taken)	How to avoid
Iair blocking ıbing	No or low flow, possibility of pumps running dry (hair not used as part of recipe)	Coarse filtration prior to actual treatment
·oaming	Loss of treatment and biomass*	Use of less foaming products, antifoaming agents or foam blockers
Scale build-up in anks	Solids accumulation on tank walls resulting in loss of solids in influent to be treated (mixing increased, tanks brushed)	Increase mixing, brush tanks
Degradation	Lower strength influent	Short storage time prior to treatment, low storage temperature (Section 4.3)

not observed

The storage of greywater lead to scale formation (Section 4.2.3) that was overcome by mproving the mixing and regularly brushing the tanks. An analysis of the scale evealed that inconsistent solids concentration measured in the greywater during the rials could be attributed to the gradual build-up and detachment of the scale on the ank walls. Another major problem was organics degradation (Section 4.2.3), which may have an impact on the biological treatment as demonstrated in Sections 5.2.4-5.2.5.

Backwashing the BAF

The backwash water use was excessive at 22-39% of the total daily flow but it was not subject to optimisation in the current study. The backwash procedure was carried out manually on a daily basis following the breakdown of the backwash program (Section 5.2.5.2). Media loss from the BAF was not measured but it occurred during automatic backwash (actuated by the pressure in the column) possibly as a result of irregular backwashing.

During all three stages of the trial a backwash profile for tCOD and SS was determined from 100 ml grab samples taken during the backwashes carried out on

:ven consecutive days. The patterns are similar and independent of the influent uality (Figures 5.24-5.25), suggesting that large variations did not occur in the ackwash regime. Both parameters show peaks at 2 min after initiating the program)llowed by a sharp decline at 3 min after which the concentration in the washwater radually decreased. As the flow rate to the BAF was increased by 50%, the ercentage of the tCOD and solids removed from the column during backwash ıcreased respectively.

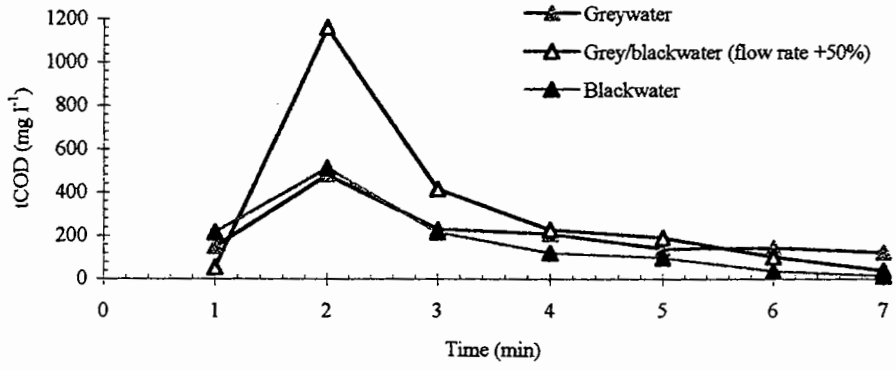


Figure 5.24. BAF backwash profiles for tCOD of the backwash water during the steady-state operation. Each profile is an average of 7 samples.

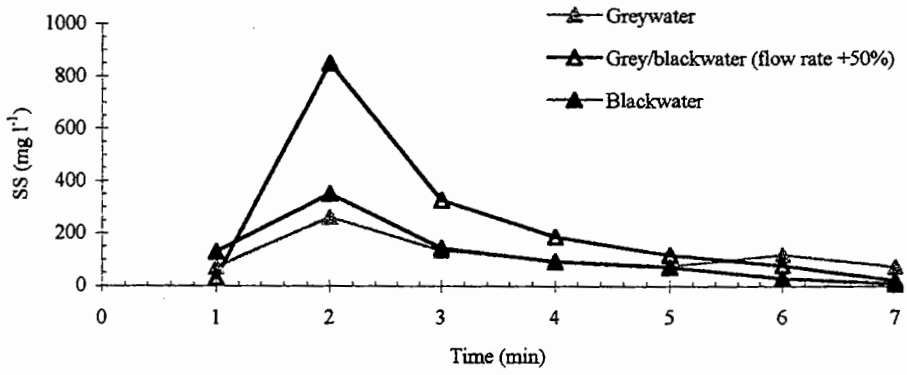


Figure 5.24. BAF backwash profiles for SS of the backwash water during the steady-state operation. Each profile is an average of 7 samples.

5 Discussion

3Rs have been operated at a range of flux rates (Table 2.8, Section 2.4.6.4, Tables 0-5.11) and membrane cleaning frequency. In the current work the submerged BR reached an overall mean flux of 12 l m^{-2} h^{-1} (200 l m^{-2} h^{-1} bar^{-1}) after 76 days ection 5.2.3.3). This was maintained until the end of the 300-day trial.

ıe results reflect published data trends of flux decline and stabilisation during long- :m operation of submerged systems without membrane cleaning in ımestic/municipal wastewater treatment. Chiemchaisri *et al.* (1993) achieved a ean flux of 4.6 l m^{-2} h^{-1} after 30 days of operation and maintained this level for ıother 30 days. Nah *et al.* (2000) operated an MBR at a flux of 10 l m^{-2} h^{-1} at 0.04-06 bar for 150 days – similar to a study carried out by Murakami *et al.* (1999), who :hieved a mean flux rate of 16.6 l m^{-2} h^{-1} over 140 days. Ueda and Hata (1999) eated raw domestic wastewater by a submerged MBR for 371 days. During this :riod the mean flux of 17.6 l m^{-2} h^{-1} (440 l m^{-2} h^{-1} bar^{-1}) was increased to 31.7 l m^{-2} ^{1} (211 l m^{-2} h^{-1} bar^{-1}) for 21 days without resulting in a deteriorated effluent quality.

ünder and Krauth (1998) investigated the steady- and unsteady-state operation of late and hollow fibre modules without membrane cleaning during a 130-day trial. .fter a 30-day start-up period the flux rates for these processes were maintained at 18 m^{-2} h^{-1} (0.08 bar) and 17 l m^{-2} h^{-1} (0.25 bar) over 55 days. Effluent quality from these rocesses was similar (Table 5.13) during design dry weather flow. In the following 5 days the loading rates simulated the variations in the diurnal flow and weather onditions. The maximum flux rates for the plate and hollow fibre modules were 35 l n^{-2} h^{-1} (0.32 bar) and 30 l m^{-2} h^{-1} (0.48 bar) respectively. During this period the :ffluent quality did not significantly differ from that in the steady-state operation. Côté *et al.* (1997), working on a hollow-fibre module, reported a mean flux of around !5-35 l m^{-2} h^{-1} for TMP about 0.02 bar for a backwashed system, backwashing taking ılace several times in an hour for up to 30 s.

Side-stream MBRs are usually operated at higher flux rates than the submerged configurations (Section 5.2.3.3, Tables 5.10-5.11), resulting in more frequent need to clean membranes. In the current study the flux in the side-stream system was variable at 38-64 l m^{-2} h^{-1} with the permeability being 15-26 l m^{-2} h^{-1} bar^{-1} (Section 5.2.3.3). The membrane was not cleaned during the 32-day greywater trial. Similar findings have been reported by Fan *et al.* (1999) on municipal wastewater: a side-stream MBR was operated for 70 days at 71 l m^{-2} h^{-1} without membrane cleaning. Increasing the flux to 143 l m^{-2} h^{-1} lead to a faster membrane fouling and thus the necessity to clean the membrane every 3 to 5 days. However, the increased flux appears not to influence effluent quality if the filtration capacity is regularly restored. A flux range of 75-150 l m^{-2} h^{-1} was found to have no significant impact on the effluent COD and ammonia during a 162-day trial although the wastewater quality was largely variable (Xing *et al.*, 2000). Trouve *et al.* (1994) maintained a flux of 60-80 l m^{-2} h^{-1} at 1 bar after a week's operation for longer a than 15 days without membrane cleaning. In the work by Chaize and Huyard (1990), where the membrane was periodically cleaned, the initial specific flux of around 50 l m^{-2} h^{-1} bar^{-1} declined to around 15 l m^{-2} h^{-1} bar^{-1} in about 30 days and remained at this level for the following 130 days.

The large variation in the reported data (Tables 2.8, Section 2.4.6.4, Tables 5.10-5.11), with reference to the hydraulic resistance (i.e. the inverse specific flux) of the membrane system, can be attributed to a number of factors. The principal component is the mixed liquor properties. Some MBR suppliers (for example, Rhodia/Orelis) design their systems based on dissolved COD, this being considered representative of fouling propensity (Holmes, 2001). Of as great significance is the configuration. Since side-stream systems are operated at around 3-5 times the flux of equivalent submerged systems, the fouling propensity is inevitably higher and so the specific flux lower. This inevitably leads to a requirement for more regular membrane cleaning and/or backwashing. Lastly, the membrane geometry is important. A hollow fibre membrane element, in which water passes from outside to inside the lumen, allows little control of concentration polarisation by promoting turbulence (unlike an in-to-out wide bore tube or channel; Jacangelo and Buckley, 1996). These systems also

require regular backflushing and cleaning, although can operate at a lower flux whilst maintaining the same production rate because of the high membrane surface area to bulk module volume ratio.

Steady-state operation of advanced biological processes is well reported though greywater applications are limited in comparison to sewage (Table 2.8, Section 2.4.6.4, Table 2.11, Section 2.4.6.5, Tables 5.10-5.14). In the current work the volumetric loading rates (VLRs), which increased over the course of the steady-state trial, were lower than in normal operation due to process limitations and low influent strength (Section 5.2.4.1). Experience from operating greywater recycling systems has highlighted that biological processes can achieve a significant BOD removal and a good quality effluent despite a variable quality influent. A rotating biological contactor installed in a multi-occupancy housing reduced BOD_7 of 50-250 mg l^{-1} in greywater to <5 mg l^{-1} (Nolde, 1999). Similar effluent quality was achieved using a fluidised bed reactor in a single family home where the greywater BOD_7 was 70-300 mg l^{-1} (Nolde, 1999). Likewise, a side-stream MBR treating greywater from a large building produced effluent BOD of 1.8-5.5 mg l^{-1} with the influent range being 120-577 mg l^{-1} (Huitorel, 2000). Both MBRs (Günder and Krauth, 1998; Ueda and Hata, 1999) and BAFs (Chudoba and Pujol, 1998) have been reported to tolerate increased hydraulic and organic loading, significantly higher than in the current study, without impairing effluent quality (Section 6.6).

In the current work organic removal by the submerged MBR did not significantly change as a result of the increased volumetric loading rate over the 300 days of operation (Tables 5.4 and 5.13); in this respect treatment of greywater did not differ from that of blackwater. The values for removal of 92-95% COD and 96-98% BOD removal of the submerged MBR compare well with published data on similar systems where a 87-93% COD (Davies *et al.*, 1998; Gander *et al.*, 2000) and a 96-98% BOD removal from screened municipal wastewater has been achieved at relatively low loading rates of 0.03-0.15 kgBOD m^{-3} d^{-1} (Davies *et al.*, 1998; Gander *et al.*, 2000; Martyn *et al.*, 1999). MBRs operated at VLRs of about 2-7 times that of the current

work have resulted in a substantial organics removal from municipal wastewater: 90-99% of the COD at 0.96-3.2 kgCOD m^{-3} d^{-1} (Bouhabila *et al.*, 1998; Buisson *et al.*, 1998; Günder and Krauth, 1998; Peters *et al.*, 1999) and 91-99% of the biodegradable fraction at 0.29-0.66 kgBOD m^{-3} d^{-1} (Futamura *et al.*, 1994; Ishida *et al.*, 1993; Martyn *et al.*, 1999; Peters *et al.*, 1999; Ueda and Hata, 1999).

Published data on side-stream MBRs treating similar influent quality to that of greywater in the current work (Table 5.4) indicate good organic removal from domestic/municipal wastewater: >94% BOD removal at 0.18 kgBOD m^{-3} d^{-1} (Trouve *et al.*, 1994) and >94% COD removal at 0.45-0.55 kgCOD m^{-3} d^{-1} has been achieved (Fan *et al.*, 1998; Chaize and Huyard, 1990; Trouve *et al.*, 1994). Even at significantly higher COD loadings of up to around 3 kgCOD m^{-3} d^{-1} the removal has been shown to remain at around 97% (Xing *et al.*, 2000). Both submerged and side-stream systems typically produce effluent with a BOD of less than 10 mg l^{-1} (Tables 5.12-5.13).

MBRs are commonly operated at MLSS levels of 10 000-20 000 mg l^{-1} (Table 2.8, Section 2.4.6.4, Tables 5.12-5.13) and concentrations as high as 50 000 mg l^{-1} have been reported for some zero sludge wastage processes (Houten and Eikelboom, 1997; Müller *et al.*, 1995). These are significantly higher than the corresponding values for the submerged and side-stream MBRs in the current work, where the biomass growth was limited by the low greywater strength (Section 5.2.7). The MLSS values of around 2000-3000 mg l^{-1} were more typical of a conventional activated sludge process treating domestic wastewater (Gray, 1990). However, reports on well-performing MBRs with a low MLSS of around 1500-4000 mg l^{-1} (Fan *et al.*, 1998; Martyn *et al.*, 1999; Scott *et al.*, 1998; Trouve *et al.*, 1994; Ueda and Horan, 2000) suggest that the wastewater-specific microflora in the bioreactor is a key factor. Low food to micro-organism (F:M) ratios (Tables 5.12-5.13) and relatively high volatile fractions of 60-79% noted for both submerged and side-stream configurations (Nah *et al.*, 2000; Murakami *et al.*, 1999; Trouve *et al.* 1994; Ueda and Hata, 1999; Xing *et al.*, 2000) are comparable with the respective values obtained in the current work (Section 5.2.7).

Table 5.10. Comparison of submerged MBRs in terms of flux, pathogen removal and turbidity.

Influent type	Membrane type/pore size (μm)	Flux ($l\ m^{-2}\ h^{-1}$)	Pathogens [a] total coliforms, [b] faecal streptococci, [c] *E.coli*, [d] faecal coliforms, [e] bacteriophage			Turbidity (NTU)	Reference
			In (cfu in 100 ml)	Out (cfu in 100 ml)	Removal (log)		
Greywater	PF/0.4	18.2 (12.8 equilibrium)	1500000[a] 590000[b] 4200[c]	1[a] 30[b] 1[c]	6[a] 4[b] 3[c]	0.3	Current study
Grey/blackwater	PF/0.4	12.8	45000000[a] 110000[b] 1300000[c]	93[a] 1[b] 1[c]	6[a] 5[b] 5[c]	0.4	Current study
Blackwater	PF/0.4	11.6	18000000[a] 150000[b] 3600000[c]	60[a] 1[b] 1[c]	6[a] 5[b] 5[c]	0.4	Current study
Household	HF/0.1	10/166-250	n/a	n.d[a]	n/a	n/a	Nah *et al.* (2000)
Household	HF/0.03	4.6	n/a	n/a	na	<0.5	Chiemchairsri *et al.* (1993)
Domestic	HF/0.4	16.6	n/a	n.d	n/a	n/a	Murakami *et al.* (1999)
Domestic	HF/0.1	n/a	n/a	0-43[c]	n/a	n/a	Futamura *et al.* (1994)
Raw domestic	HF, PE/0.4	16.6	48000000[a]	6[a]	6[a]	n/a	Ueda and Hata (1999)
Urban	HF, 0.1	30	n/a	n/a	n/a	n/a	Bouhabila *et al.* (1998)
Municipal	HF/200 000*	35 25	56000000[a] 59000000[a]	20[a] 43[a]	6.4[a] 6.1[a]	0.14 0.24	Côté *et al.* (1997)
Municipal	PF/0.4	7.1	680000[b] 11000000[d] 1700[e]	n.d.[b] 1.1[d] 8.8[e]	6[b] 7[d] 3[e]	n/a	Ueda and Horan (2000)
Municipal	WC	2.6-26.5	n/a	n/a	n/a	n/a	Martyn *et al.* (1999)
Municipal	-/0.2	40	n/a	n/a	n/a	n/a	Peters *et al.* (1999)
Municipal	HF/200 000*	n/a	n/a	n/a	6.2[d]	0.25	Buisson *et al.* (1998)
Municipal	P/0.4 HF/0.2	18 17	n/a n/a	<30[a], 0[b], <30[d] 60[a], 0[b], <30[d]	n/a n/a	n/a n/a	Günder and Krauth (1998)
Municipal	PF/0.4	20	n/a	n/a	n/a	n/a	Davies *et al.* (1998)

* Dalton, n.d. not detected

Table 5.11. Comparison of side-stream MBRs in terms of flux, pathogen removal and turbidity.

Influent type	Membrane type/ pore size (μm)	Flux (l m^{-2} h^{-1})	Pathogens [a] total coliforms, [b] *E.coli*, [c]colon bacilli			Turbidity (NTU)	Reference
			In (cfu in 100 ml)	Out (cfu in 100 ml)	Removal (log)		
Greywater	T/4000*	48	30000[a], 420[b]	2.1[a], 1.5[b]	6[a], 3[b]	0.9	Current study
Greywater	P	120	n/a	n/a	4[c]	n/a	Huitorel (2000)
Urban	T, C	71-143	n/a	n/a	n/a	n/a	Fan *et al.* (1998)
Urban	T, C/0.02	75-150	n/a	n/a	n/a	n/a	Xing *et al.* (2000)
Domestic	PF/50 000*	15	n/a	n/a	n/a	n/a	Chaize and Huyard (1990)
Domestic	T, C/0.1	73-104	n/a	n/a	n/a	n/a	Ghyoot *et al.* (1999)
		77-154	n/a	n/a	n/a	n/a	
Municipal	T, C/0.1	60-80	n/a	n/a	n/a	n/a	Trouve *et al.* (1994)

T = tubular, P = plate, C = ceramic, PF = polysulphone, * Dalton

Table 5.12. Comparison of side-stream MBRs in terms of organic, solids and ammonia removal.

Influent type	MLSS (mg l^{-1})/ F:M ratio (d^{-1})	HRT (h)	Loading rate (kg m^{-3} d^{-1})	Wastewater quality [a] COD, [b] BOD, [c] SS, [d] NH_3, [e] COD_{Mn}			Reference
				In (mg l^{-1})	Out (mg l^{-1})	Removal (%)	
Greywater	4044/0.01[b]	67.8	0.10[a], 0.06[b], 0.02[c]	273[a], 181[b], 58[c]	2[a], 1[b], 4[c]	99[a], 100[b], 93[c]	Current study
Greywater	20 000/n/a	1	n/a	89[e], 349[b], 97[c]	12[e], 4[b], n.d.[c]	86[e], 99[b]	Huitorel (2000)
Urban	3000/0.16[a]	15	0.62[a]	388[a]	15[a]	96[a]	Fan *et al.* (1998)
	3000-7500/0.28[a]	7.5	0.55[a]	310[a]	19[a]	94[a]	
Urban	maximum 23 100/ 0.54[a, v]	5	1.38-3.15a	200-800[a], 10-40[c]	9.4[a], 0.2-1.3[d]	97[a], 96[d], 100[c]	Xing *et al.* (2000)
Domestic	8000-10 000/ 0.06-0.10[a]	1-8	0.45-1.5[a]	250-550[a], 200-400[c], 40-80[d]	<30[a], <0.5[d]	>88[a], >99[d]	Chaize and Huyard (1990)
Domestic	40 000-50 000/ 0.02[a]	7.4-14.8	0.9-2.0[a]	130[a]	<1.3[a]	99[a]	Müller *et al.* (1995)
Municipal	2500[v]/0.2[a, v]	24	0.49[a], 0.18[b]	488[a], 182[b], 216[c], 35[d]	20[a], <10[b], n.d.[c], <1[d]	>95[a], >94[b], 100[c], 97[d]	Trouve *et al.* (1994)

[v] volatile, n.d. not detected

Table 5.13. Comparison of submerged MBRs in terms of organic, solids and ammonia removal.

Influent	MLSS (mg l^{-1})/ F:M ratio (d^{-1})	HRT (h)	Loading rate (kg m^{-3} d^{-1})	Wastewater quality [a] COD, [b] BOD, [c] SS, [d] NH_3 In (mg l^{-1})	Out (mg l^{-1})	Removal (%)	Reference
Greywater	1172 nit, 4451 denit/ 0.04[b]	13.6	0.25[a], 0.08[b]	128[a], 41[b], 52[c]	7[a], 1[b], 4[c]	93[a], 96[b], 89[c]	Current study
Grey/blackwater	212 nit, 6888 denit/ 0.33[b]	15.4	0.25[a], 0.07[b], 0.013[c]	159[a], 41[b], 59[c], 9.1[d]	9[a], 1[b], 4[c], 0.1[d]	92[a], 97[b], 90[c], 99[d]	Current study
Blackwater	425 nit, 12 078 denit/ 0.38[b]	18.0	0.43[a], 0.14[b], 0.037[c]	300[a], 110[b], 158[c], 26.5[d]	16[a], 1[b], 3[c], 2.6[d]	95[a], 98[b], 94[c], 89[d]	Current study
Household	6000-14 000/ n/a	10-15	n/a	343[a], 127[b], 128[c], 19.9[d]	11[a], 1[b], 0.6[d]	97[a], 99[b], 100[c], 97[d]	Nah *et al.* (2000)
Household	n/a	1	n/a	60-200[a], 11-17[d] (TKN)	<10[a], 0-4[d] (TKN)	n/a	Chiemchaisri *et al.* (1993)
Domestic	27 900/ n/a	n/a	n/a	98[b], 89[c], 24.7[d] (TN)	0.9[b], n.d[c]	99[b]	Murakami *et al.* (1999)
Domestic	9200/0.03[a]	10.5	0.29[b]	44a, 127[b], 90[c], 16.0[d]	3.9a, 0.8[b], 0[c], 0.6[d]	91[a], 99[b], 100[c], 96[d]	Futamura *et al.* (1994)
Raw domestic	12 900/0.02[a]	13.4	0.245[b]	135[b], 163[c], 20[d]	1.3[b], 0.03[c], 0.3[d]	99[b], 100[c], 99[d]	Ueda and Hata (1999)
Urban	6800/0.45[a]	3.3	3.2[a]	420[a]	42[a]	90[a]	Bouhabila *et al.* (1998)
	8300/0.39[a]	3.3	3.2[a]	450[a]	29[a]	94[a]	
Municipal	5000-15 000/0.28[a]	2	n/a	356[a], 187[b], 118[c], 28[d]	16[a], <5[b], n.d[c], 5.6[d]	96[a], >97[b], >99[c], 80[d]	Côté *et al.* (1997)
	15 000/0.39[a]	9	n/a	482[a], 220[b], 120[c], 39[d]	10[a], <5[b], n.d[c], 0.4[d]	98[a], >98[b], >99[c], 99[d]	
Municipal	2500/0.02[b]	91	0.05[b]	172[b], 29[d]	1.8[b], 0.02[d]	>98[b], >99[d]	Martyn *et al.* (1999)
	5000/0.13[b]	10	0.66[b]	269[b], 38[d]	4[b], 0.09[d]	99[b], >99[d]	
Municipal	16 000/0.03-0.11[b]	4.98	1.76[a], 0.36[b]	365[a], 115[b]	2[a], 10[b]	>99[a], 91[b]	Peters *et al.* (1999)
Municipal	15 000/0.1[a]	7.3	1.2[a]	457[a], 124[c], 37.9[d]	10.5[a], <5[b], n.d.[c]	>97[a]	Buisson *et al.* (1998)
Municipal	12 000-16 000/ n/a	6	0.96[a]	250[a], 125[b], 40-60[d] (TKN)	<20[a], n.d[c], 0[d]	>92[a], >92[b], 100[c]	Günder and Krauth (1998)
	12 000-18 000/ n/a	5	1.23[a]	250[a], 125[b], 40-60[d] (TKN)	<20[a], n.d[c], 0[d]	>92[a], >92[b], 100[c]	
Municipal	10 000-39 000/ 0.03-0.15[b]	4.5	0.03-0.15[b]	469[a], 200[b], 418[c]	61[a], 4[b], n.d[c], 5[d]	>98[a], 98[b], 100[c], 88[d]	Davies *et al.* (1998)

PF = polysulphone, HF = hollow fibre, PE = polyethylene, WC = woven cloth, P = plate, nit = nitrification unit, denit = denitrification unit, n.d. not dectected

In some cases a relatively low organic content, similar to that in greywater, and a low organic loading rate (OLR) has resulted in a deteriorated performance by a BAF. Pujol *et al.* (1992) investigated a range of VLRs on a BAF treating municipal wastewater and found the lowest COD removal of 55% to occur at 0.5 kgCOD m^{-3} d^{-1} (35 mg l^{-1}) and the highest of 88% at a much greater loading rate of 8.0 kgCOD m^{-3} d^{-1} (607 mg l^{-1}; Table 5.14). Peng *et al.* (1995) measured an influent COD of 25-43 mg l^{-1} in a mixture of domestic wastewater and industrial cooling water discharge and achieved a 48-70% COD removal by a two-stage BAF at 0.44-4.8 kgCOD m^{-3} d^{-1} (Table 5.14) From this point of view the BAF in the current study was 'underloaded', though apparently without adversely impacting upon process performance. A reasonably stable removal of COD (82-90%) and BOD (86-94%) was achieved throughout the trials at a mean flow rate of 0.333 m^{-3} d^{-1}, suggesting that the loading rates were sufficient for a good performance; the impact of an increased flow rate is discussed later in this section.

There was little apparent difference between greywater and blackwater treatment, and the best performance was achieved in the grey/blackwater trial (Table 5.14). Previous greywater and sewage studies have shown that a BAF can be challenged with much higher organic loads than in the current study without a significant decline in the treatment performance. 70-90% of the BOD in greywater has been removed at OLRs of 1-4 kgBOD m^{-3} d^{-1} (Adachi and Fuchu, 1991; Jefferson *et al.*, 2000b). In domestic/municipal wastewater treatment the VLRs up to 8 times those of the blackwater in the current study have resulted in 75-94% BOD and 75-90% COD removal (Table 2.11, Section 2.4.6.5, Table 5.14).

In the MBRs the solids are retained by the membrane such that often a complete removal of suspended solids is achieved (Günder and Krauth, 1998; Ueda and Hata, 1999). In the current work both MBRs were subject to regrowth in the effluent pipes, hence the unusually high effluent solids (Table 5.6). In contrast, throughout the trials the turbidity of 0.3-0.4 NTU in the submerged MBR and 0.9 NTU in the side-stream configuration was of a typical quality with reference to 0.14-0.5 NTU reported

hiemchaisri *et al.*, 1993; Côté *et al.*, 1997). The solids removal by the BAF proved from 82% in the greywater trial to 91% in the blackwater trial (Table 5.14) ue to the higher proportion of larger particles present. As much as 94% of the solids grey/blackwater were removed at 0.362 m^3 d^{-1} but an increased flow rate decreased e efficiency by 9% (Table 5.14). This is to be expected with because of a higher ltration velocity, as has been noted by Smith and Brignal (1996) in municipal astewater treatment (Table 5.14). Generally the solids removal in the current study mpares well with 85% in greywater (Adachi and Fuchu, 1991) and 80-95% in unicipal wastewater treatment (Pujol *et al.*, 1992; Rundle, 1996; Smith and Brignal, 996; Wheale and Cooper-Smith, 1995).

he nitrogen and phosphorus levels different markedly between greywater and lackwater (Table 5.5), and they had a direct impact on the performance. As observed the greywater trial (Section 5.2.4.2) nitrification is often not possible when mmonia levels are low and phosphorus removal may be insignificant. On changing ue matrix the process has to adapt to the prevailing conditions. The submerged MBR dapted to the stronger influent after a delay of a few days (Section 5.2.4.2) resulting a complete nitrification in the grey/blackwater trial but exhibited some fluctuation uring the blackwater trial (Table 5.5); the case of the BAF is discussed later. These indings are similar to those for municipal wastewater where 80-99% of ammonia has een removed by MBRs (Côté *et al.*, 1997; Davies *et al.*, 1998; Ishida *et al.*, 1993; Jeda and Hata, 1999).

hosphorus removal of 8-19% from grey/blackwater and blackwater by the ubmerged MBR is in good agreement with that of the 12% removal from municipal astewater reported by Côté *et al.* (1997). The 50% P reduction by the BAF from greywater was attributed to biomass synthesis whereas in the grey/blackwater and blackwater trial the removal (18-29%) occurred also through filtration (Table 5.5). A 35% phosphorus removal from sewage by a BAF through physical filtration without chemical dosing has been reported (Clark *et al.*, 1997). Total nitrogen removal of up to 22% and 48% by the MBR and the BAF, respectively, suggests that some of the

'ganic N remained untreated (Table 5.5). In the MBRs reductions of 79-82% in TN ıd 67-74% TP (Buisson *et al.*, 1998; Ishida *et al.*, 1993; Ueda and Hata, 1999) have ɜen achieved from sewage which had similar nitrogen and phosphorus values to ıose in the current work. Davies *et al.* (1998), who observed a mean TN removal of round 49% from the initial 10-120 mg l^{-1} in sewage, accounted the large variation in ıe effluent ammonia for by the F:M ratio. Nah *et al.* (2000) noted that nitrogen ɜmoval of 40-60% and 80% was achievable at the respective BOD:TN ratios of <1 nd >2.

ɜmith and Brignal (1996) observed a reduced performance of a BAF in terms of BOD ınd ammonia removal as a result of a decreased hydraulic retention time (HRT) and ın increased OLR. As the HRT was decreased by 30% (from 1 to 0.7 h) and the BOD ınd NH_3 loading rates increased by 42% and 50%, the corresponding removal rates ɗeclined by 5% and 4% (Table 5.14). These parameters as well as the SS removal ɗeteriorated significantly as the HRT was decreased to 0.3 h (Table 5.14). In the current work a similar decline (10%) in the BOD removal was observed when the flow rate was increased by 50% in the grey/blackwater trial (Table 5.14). The ammonia removal, however, improved from 72% to 81% as the ammonia load increased by 45%. The reason for the difference in the ammonia removal between these studies is the smaller step change in the loading rate and a longer HRT in the current work, such that the conditions for ammonia removal were more favourable. These conditions improved significantly on changing the feedwater to grey/blackwater and blackwater (Table 5.14).

In the MBRs pathogens are removed by activated sludge treatment and the membrane barrier (Ueda and Horan, 2000). MBRs have been shown to efficiently remove bacteria often near the non-detectable level (Côté *et al.*, 1997; Krauth and Staab, 1993; Ueda and Hata, 1999; Ueda and Horan, 2000) effective removal of viruses has also been demonstrated (Chiemchairsri *et al.*, 1992). Similar results to those reported for domestic/municipal wastewater (Table 5.10) were obtained in the current study where the submerged and side-stream MBRs effluents had non-detectable bacteria

(total/*E.*coliforms and faecal streptococci) 74-100% and 83-100% of the time, respectively. Significant differences in the pathogen removal by the submerged system between greywater and blackwater were not observed (Sections 5.2.6.2-5.2.6.4) .

Mechanisms of bacterial and viral transport through membranes have been studied (Herath *et al.* 1998). It has been suggested that the orientation of the organism when approaching the membrane surface is an important factor. This is the case particularly for organisms that are not uniform in shape. As an organism goes through various growth phases its size changes (Leahy and Sullivan, 1978). Cells become smaller during the transition from the growth phase to the stationary phase. This is because they divide faster than they grow so that they may pass through the membrane when they are smaller than their normal size. This could be a reason for bacteria breakthrough observed in some membrane systems as aftergrowth may occur if the environmental factors such as substrate, pH and temperature are suitable. This seems a likely explanation to the occasional breakthrough and regrowth in the submerged MBR of the current study (Sections 5.2.6.2-5.2.6.3).

Since BAFs are not used primarily for pathogen removal, data on this issue are not widely available. However, the performance can be discussed with relation to conventional sewage treatment where faecal coliform and faecal streptococci may be reduced by about 2.3 log and 1.8 log, respectively (Ueda and Horan, 2000). These are comparable with corresponding data from the current work where the BAF achieved up to a 3 log removal of total/*E.*coliform and faecal streptococci (Table 5.7). The percentage of the samples with non-detectable bacteria decreased with the increased influent strength and the increased flow rate. In the greywater trial 11-41% of the samples contained non-detectable levels of pathogens depending on the indicator bacteria whereas in the grey/blackwater and blackwater trials no counts below 1 cfu in 100 ml were measured (Sections 5.2.6.2-5.2.6.4).

Table 5.14. Comparison of BAFs.

Influent type	Configuration	Media type, size (mm)	HRT (h)	Loading rate (kg m^{-3} d^{-1})	Wastewater quality [a]COD, [b]BOD, [c]SS, [d]NH_3			Reference
					In (mg l^{-1})	Out (mg l^{-1})	Removal (%)	
Greywater	Downflow	Lytag 3-4	3.7	0.84[a], 0.27[b], 0.33[c]	128[a], 41[b], 52[c]	12[a], 4[b], 6[c]	82[a], 89[b], 81[c]	Current study
Grey/blackwater	Downflow	Lytag 3-4	3.3	1.37[a], 0.37[b], 0.49[c], 0.067[d]	159[a], 41[b], 59[c], 9.1[d]	19[a], 5[b], 4[c], 4.1[d]	90[a], 94[b], 94[c], 72[d]	Current study
			2.2	1.49[a], 0.44[b], 0.60[c], 0.097[d]	159[a], 41[b], 59[c], 9.1[d]	22[a], 5[b], 6[c], 1.4[d]	74[a], 84[b], 83[c], 81[d]	
Blackwater	Downflow	Lytag 3-4	4.5	2.03[a], 0.67[b], 1.01[c], 0.170[d]	300[a], 110[b], 158[c], 26.5[d]	57[a], 12[b], 17[c], 0.1[d]	84[a], 86[b], 91[c], 100[d]	Current study
Greywater and cooling tower water (4:1)	Downflow	n/a	n/a	1-4[b]	120[b] 100[c]	20[b] 15[c]	70-90[b]	Adachi and Fuchu (1991)
Municipal	Downflow and upflow	Vitrified clay 2.5-6	n/a	0.5-8.0[a], 0.6-0.7[d]	35-607[a], 10-142[c]	14-99[a], 5-21[c]	55-85[a], 46-87[c], 70-87[d]	Pujol *et al.* (1992)
Municipal and industrial (3:2)	Upflow	Plastic n/a	0.5	0.44-4.6[b]	25-43[a], 12-18[b], 40[c], 3.36[d]	n/a	48-70[a]	Peng *et al.* (1995)
Municipal	Downflow	Expanded shale 3-6	1.3	1.5[b]	131[b], 89[c], 44[d]	10[b], 12[c], 21.5[d]	92[b], 86[c], 51[d]	Wheale and Cooper-Smith (1995)
Municipal/ industrial	Upflow	Silica sand 2	n/a n/a	1.3[b] 3.7[b]	88[b], 110[c], 20.9[d] 284[b], 143[c], 30.8[d]	13[b], 11[c], 11.1[d] 23[b], 27[c], 14.1[d]	85[b], 87[c], 49[d] 91[b], 80[c], 54[d]	Rundle (1996)
Municipal	Downflow	Granular 2-6	0.3	3.9[b], 0.6[d]	n/a	17[b], 32[c], 13.1[d]	75[b], 69[c], 2[d]	Smith and Brignal (1996)
			0.5	2.6[b], 0.4[d]	n/a	13[b], 17[c], 7.9[d]	91[b], 83[c], 52[d]	
			0.7	1.7[b], 0.3[d]	n/a	12[b], 7[c], 1.7[d]	88[b], 93[c], 89[d]	
			1	1.2[b], 0.2[d]	n/a	7[b], 5[c], 1.1[d]	83[b], 95[c], 93[d]	

Chapter 6 Process evaluation - unsteady state

6.1 Introduction

Unsteady-state operation is an important for small-scale in-building recycling systems. Compared to sewage treatment works (STW), which have a large buffer capacity, small-scale plants are more prone to intermittent operation as well as hydraulic and organic shock loads. The continuous operation of greywater recycling systems relies much on the storage of untreated greywater. In this respect work has often focused on balancing the supply and demand with an appropriate greywater storage volume. This has already been covered in Chapter 4.

This part of the work focused on envisaged events that could impact upon the submerged membrane bioreactor (MBR) and the biological aerated filter (BAF), these processes having performed satisfactorily during the steady-state trials (Section 5.2). Both systems were cleaned and the configuration of the submerged MBR was modified (Section 3.3.2.1) following the steady-state operation. After restarting they were run for 1 month prior to the unsteady-state trials. The evaluation was focused on three areas:

- the influence of intermittent operation of feed and air on process recovery with respect to key water quality parameters, including response from an on-line BOD_5 monitor,
- a survey on the behavioural patterns in households as a basis for an impact analysis, in which the effects of the substances on a biological treatment system were assessed using respirometry, and
- the influent transition periods in the steady-state trials (Section 5.2).

6.2 RACOD: Results of evaluation

This part of the work was carried out as a joint work with M. Heinz, a visiting overseas student, who undertook most of the practical work and some of the data analysis. The results are presented in the following.

An on-line sensor for assessing the unit process operation was employed to measure the easily biodegradable fraction of BOD or COD. The effectiveness of the RACOD (Section 3.3.3.2) sensor at different flow rates and for different wastewater strengths (Table 6.1) was first evaluated in comparison to the standard BOD_5 test. The BOD_5 measurements of the feedwater samples taken six times a day confirmed that biodegradation did not significantly occur during 24 h of storage. With the monitor being designed for on-line use, it was of a particular interest to investigate its applicability for batch measurement by changing the feedwater type every two to four days. In the commercial literature on the monitor, RACOD refers to the on-line monitor and RABOD to the readings equivalent to BOD_5, and the same terms are used in this work.

Table 6.1. RABOD and **BOD_5** readings of wastewaters at different feed flow rates. Mean ± standard deviation.

Flow rate (ml min^{-1})	Primary sewage influent (mg l^{-1})	Primary settled sewage effluent (mg l^{-1})	Secondary settled sewage effluent (mg l^{-1})	Tertiary sewage effluent (mg l^{-1})	Synthetic greywater (mg l^{-1})
1	168 ± 15	142 ± 40	13 ± 10	89 ± 20	24 ± 14
	137 ± 47	**122 ± 14**	**12 ± 2**	**5 ± 2**	**47 ± 18**
2	105 ± 81	37 ± 12	14 ± 9	-	18 ± 13
	-	**99 ± 24**	**19 ± 4**	-	**103 ± 24**
3	-	47 ± 12	9 ± 3	9 ± 6	14 ± 4
	-	**76 ± 2**	**29 ± 3**	**5 ± 2**	-

The order in which the municipal wastewaters were tested on the RACOD was expected to affect the response time of the biomass. The tests at the flow rate of 3 ml

min^{-1} were carried out from the weakest wastewater concentration (tertiary effluent) to the second strongest one (primary settled sewage effluent) (Figure F.1 in Appendix F). Though the RACOD distinguished only little difference between the weaker wastewaters, a fast response of the RABOD to the change in the feedwater BOD_5 was observed during the run with primary settled sewage effluent, which represented a significantly stronger wastewater (Table 6.1, Figure F.1 in Appendix F). This trend did not reoccur when the wastewaters were run at 1 ml min^{-1} in the reverse order (i.e. from primary sewage influent to tertiary effluent) (Figure F.3 in Appendix F), indicating a longer lag time of the biomass than in the previous case. From the erroneously high RABOD readings for tertiary effluent (Table 6.1) it was concluded that the recovery time allowed for the biomass was not sufficient for the RACOD to reach expected values at low feedwater concentrations and flow rate. As the feedwater was changed from tertiary effluent to secondary settled sewage, the lag phase was expected to be short. This was confirmed by the relatively good correlation of the RABOD with BOD.

Synthetic greywater (Matrix no.1, Table 3.1, Section 3.3.1) was tested at feed flow rates from 1 ml min^{-1} to 3 ml min^{-1}. Low correlation of the RABOD readings with the BOD_5 implied either the slow response pattern described earlier or a substance interference with the dissolved oxygen probe. The best results were obtained at 1 ml min^{-1} (Table 6.1, Figure F.4 in Appendix F).

The low reproducibility of the RABOD readings in these trials suggests that using a single RABOD value as an indication of the actual BOD_5 of a sample could be misleading. The results further indicate that a relationship between the feedwater strength and the flow rate exists and that the rate of biomass recovery should be taken into account.

The factors for calculating the actual BOD_5 in a sample shown in Table 6.2 are based on the mean values of RABOD and BOD_5. An example of the correlation between these two parameters is shown in Figure 6.1. These results should be interpreted

carefully as in some cases the collected data at the selected settings was small and not as representative as in the others. The values for dilute feedwaters appeared to match the feedwater BOD_5 better at higher flow rates than at lower ones (Table 6.1, Figures F.1-F.3 in Appendix F). The flow rate of 3 ml min^{-1} was selected as the effluents from the submerged MBR and the BAF were expected to mostly have a relatively low BOD in the intermittent operation trials (Sections 6.3.3-6.3.5).

Table 6.2. Correlation factor between RABOD and BOD_5 at different sample flow rates for different feedwater types (BOD_5 = factor x RABOD).

	Sample flow rate (ml min^{-1})		
Feedwater type	1	2	3
Primary sewage influent	0.900	-	-
Primary settled sewage effluent	1.126	2.666	1.619
Secondary settled sewage effluent	0.885	1.661	3.053
Tertiary sewage effluent	0.051	-	1.571
Synthetic greywater	1.728	5.650	-

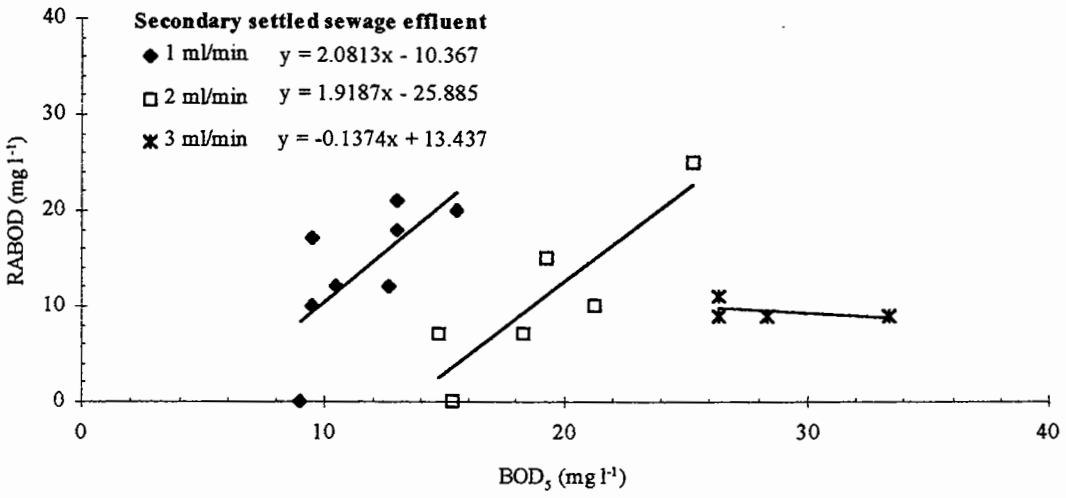


Figure 6.1. Correlation between RABOD and BOD_5 at different flow rates for secondary settled sewage effluent.

It appears that at the selected settings this on-line sensor is a poor indicator for weak solutions. However, it may be suitable for monitoring the BOD quality of the effluent for reuse as a relative indicator of deteriorated treatment capacity of a process rather than an absolute surrogate measurement of BOD.

6.3 Intermittent operation

6.3.1 Procedure

The sensitivity of the biological processes to intermittent operation was evaluated by turning off feed, air or both to the submerged MBR and the BAF for a predetermined time. The test periods from 30 min to 8 h represented incidents of a equipment breakdown or a power failure, assuming restoration of normal operation would take place within these time periods. Longer time periods were not tested as this would have risked the inactivation of the biomass. In such a case the treatment processes would have been subjected to a time-consuming restart. The longest feed-off trial at 3.5 weeks represented an extended holiday of the occupants.

The entire range of tests was carried out on the BAF, a plug-flow reactor, where the organic loadings disperse slowly and hence the effects of shocks may be distinguishable. Under normal circumstances in the submerged MBR completely mixed conditions prevail and organic shocks are diluted. The mixed liquor volume was considered important for the normal operation of the process (Sections 6.3.2 and 6.3.4) and therefore the longest feed-off and power failure tests were undertaken on the MBR. The recovery time of a process was determined on resuming feed and/or air flow: it is the period taken for the effluent quality to reach either the appropriate water quality standard or the effluent baseline value measured prior to the trial.

Sampling was conducted at 12.5 min, 30 min, 1 h, 2 h, 4 h, 8 h, 16 h, 24 h, 32 h and 48 h (and 72 h if necessary) after the particular supply to the treatment process was restored. Turbidity was used as the determining parameter for the length of monitoring. BOD_5, a more meaningful indicator of the treatment performance, was not chosen due to the long duration of the analysis. Sampling was continued until the effluent turbidity attained either the level that had prevailed prior to the experiment or 2 NTU (the US EPA standard). In the case of the 3.5-week feed-off trial the processes

were monitored for two days after the feed supply was restored. It was assumed that in the case of a longer recovery time, necessary precautions would be taken to avoid a loss of process performance. The RACOD on-line monitor was used primarily to investigate its potential to detect changes in the effluent BOD_5. The key water quality and operational parameters were measured. The greywater quality is shown in Tables F.1-F.2 in Appendix F.

6.3.2 Feed-off trials

When the submerged MBR is not fed, the mixed liquor level in the tank declines to a level just above the membranes. Below this level the membrane dries out and no longer functions. Over the period of the 8 h trial the mixed liquor level dropped to the membrane level the flow rate (Table F.2 in Appendix F); in this case the permeate flux was limited by the declining mixed liquor level rather than the much higher pressure provided by header device to the unit. The mixed liquor recovered to the original level about 20 min after the feed supply to the process was restored. In the 3.5-week trial the effluent lines of the submerged MBR were closed to maintain the liquid level in the nitrification chamber. In the case of the BAF the effluent valve was closed to prevent the water draining from the column.

In both tests the filtration performance of the submerged MBR was maintained. Low turbidity (Figure 6.2 and F.10 in Appendix F) and SS (Figures F.15-F.16 in Appendix F) levels in the treated water were recorded following restart. An example of the effluent turbidity transients (Figure 6.2) demonstrates the difference between the process types. The membrane system compares favourably with the BAF, a non-barrier system, which exhibits increased effluent turbidity in the first 2 h after restoring the greywater flow after a feed-off time of 8 h. The values reach the baseline in the subsequent few hours. The recovery times for the BAF turbidity were similar (Figure 6.3), despite the variable effluent quality in the trials, from 30 min to 4 h feed-off time (Figures F.3 and F.6-F.9 in Appendix F). Thus it seems that these time periods represented the time taken by the biomass to decay to a level at which the

performance in terms of effluent turbidity changes. On restoring the flow in the 3.5-week test the impaired filtration of the BAF resulted in high effluent turbidity and solids, which did not recover in 2 days (Figures F.10 and F.16 in Appendix F).

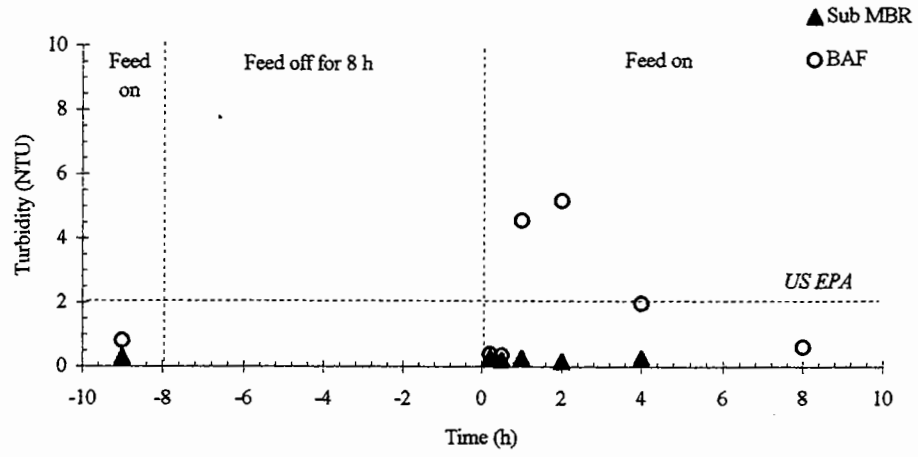


Figure 6.2. Turbidity transient of the submerged MBR and BAF effluents in the 8 h feed-off trial.

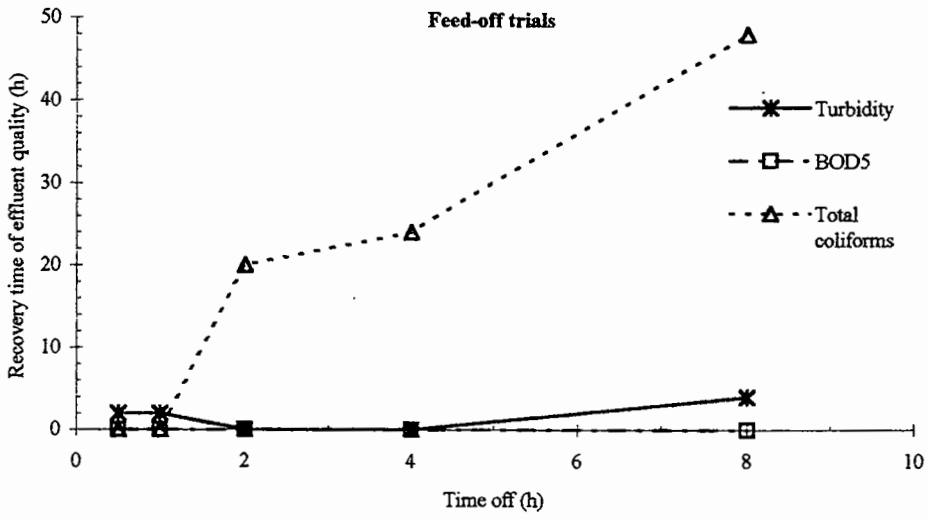


Figure 6.3. BAF recovery time in terms of turbidity, BOD_5 and total coliforms after the feed-off trials.

n the 8 h trial the first effluent sample from the submerged MBR had a BOD_5 of 9 ıg l^{-1} (Figure 6.4) despite the weak greywater at a mean BOD_5 of 33 mg l^{-1} (Table F.2 ı Appendix F). This resulted from hydraulic shock imported by restarting. On estoring the flow the HRT was around 2.6 h, about half of the mean residence time

3 h) measured once the liquid level in the bioreactor had recovered, and too low a sidence time for significant biodegradation to take place. In such cases the material sorbed on the sludge floc has insufficient time to be stabilised (Gray, 1990). The rformance rapidly recovered such that within 30 min the effluent BOD_5 was around mg l^{-1} (Figure 6.4). In the 3.5-week feed-off trial the full mixed liquor volume of the stem was combined with a low influx of relatively strong greywater (174 mg l^{-1}). his resulted in an effluent BOD_5 of 0-4 mg l^{-1}, demonstrating the buffer capacity of e process (Figure F.21 in Appendix F).

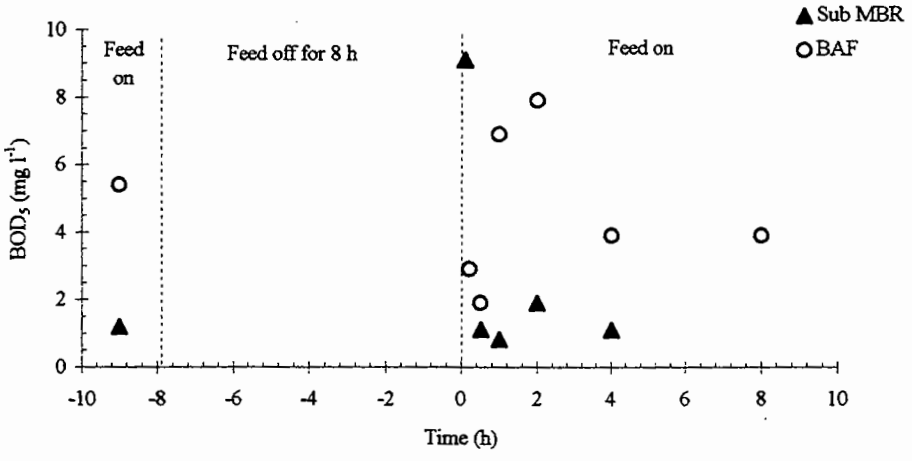


Figure 6.4. BOD_5 transient of the submerged MBR and BAF effluents in the 8 h feed-off trial.

Following the feed-off periods for the BAF of up to 8 h the mean influent was 122-165 mg l^{-1} (Table F.1 in Appendix) and the effluent BOD_5 remained <10 mg l^{-1} (Figures 6.4 and F.17-F.20 in Appendix F). The relatively stable organic removal and the effluent quality indicate that, as sufficient amounts of oxygen were available, the heterotrophs were almost unaffected by the temporary shortage of nutrients. This was not the case, however, when the influent flow to the process had been off for 3.5 weeks. The process did not recover to the water quality standard level within 2 days (Figure F.21 in Appendix F), suggesting that the biomass could not be revived within this time.

The effluent pH was relatively stable in most tests (Tables F.3 and F.5-F.7 in Appendix F). When the greywater supply to the submerged MBR was turned off for 3.5 weeks, the pH in the nitrification chamber dropped. The value of 3.8 measured in the first sample recovered to around 7.0 within 24 h of restoring the feed flow (Table F.3 in Appendix F). Adverse effects as a result of the low pH were not observed. In cases where pH of the mixed liquor is below 6.0 bacteria are inhibited and fungi begin to dominate causing bulking in activated sludge (Gray, 1989). Therefore a drop in pH may be a potential problem if the greywater supply to a submerged MBR is unavailable for a long period of time.

In both trials the submerged MBR efficiently removed total/*E.*coliforms and faecal streptococci from the greywater to below the limit of detection (Figures F.26-F.27, F.32-F.33 and F.38-F.39 in Appendix F). The steady-state greywater trials on the BAF had demonstrated that a sufficient coliform removal by this process is not achieved for non-potable reuse without additional treatment (Section 5.2.5). The unsteady-state trials showed the BAF to be relatively intolerant to hydraulic shocks, leading to long recovery times (Figures 6.3 and F.22-F.32 in Appendix F). The 3.5-week cessation to the greywater supply combined with a high total coliform count in the influent resulted in a poor bacteriological quality of the BAF effluent over a period of 2 days (Figure F.33 in Appendix F).

Mixed liquor of the MBR

The mixed liquor quality of the modified submerged MBR (unit B) prior to the 3.5 weeks feed-off period is discussed in Section 5.2.7. During the 3.5-week feed-off trial MLSS and MLVSS levels of 8193 ± 424 mg l^{-1} and 76 ± 4% respectively, were measured for in the submerged MBR (Figure 6.5). The slow rate of solids decrease over a long period of time without influent indicates the robustness of the process in this respect. 15 min after the greywater supply to the unit was restored, foaming occurred with an increasing intensity, such that after 4 h the air flow was reduced from 20 to 5 l min^{-1} for the following 44 h after which it was increased to 10 l min^{-1}. On restoring the flow the MLSS increased rapidly over the first 8 h (Figure 6.5), after

which the flocs became increasingly entrained in the foam. A mean MLSS of 8158 ± 516 mg l^{-1} and a corresponding MLVSS of 74 ± 1% were measured over a 2-day period. This monitoring period was to investigate the almost immediate effects of restoring the greywater flow rather than long-terms effects. In this work foaming was observed; it could, in some cases, lead to a loss of sludge or treatment capacity. The impact of insufficient substrate is discussed in Section 6.6.

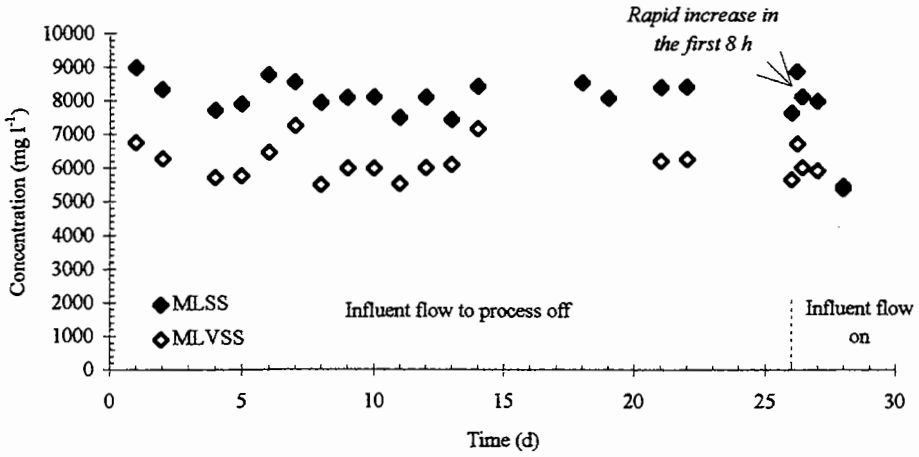


Figure 6.5. MLSS and MLVSS of the submerged MBR nitrification unit during and after the 3.5 week feed-off period.

6.3.3 Air-off trials

Air-off trials were not carried out on the submerged MBR for two reasons. Firstly, it would risk irreversible inactivation of the entire bacteria population in the mixed liquor or severe deterioration in the filtration performance, demanding a time-consuming start-up protocol (Section 3.2.3.1) to be re-initiated. The decline of DO in the MBR mixed liquor determined from three 60 ml grab samples had shown a drop in 44 min from a mean 7.6 mg l^{-1} to below the critical oxygen concentration of 1.0 mg l^{-1} (Figure F.5 in Appendix F). However, it was not considered that a significant deterioration in the biological process performance would take place over this time period and hence it was not tested. For reasons regarding air scouring on the membrane and mixing the submerged MBR is operated at higher DO levels than a

conventional activated sludge plant (Section 2.4.5). Generally 1-2 mg l^{-1} of DO is sufficient for heterotrophic microbial activity, though optimum growth is dependent on sufficient essential nutrients and trace elements (Gray, 1989), as discussed in Section 4.3. The metabolism becomes oxygen-limited for carbonaceous oxidation and nitrification below the respective minimum critical oxygen concentrations of 0.5 and 2.0 mg l^{-1} (Gray, 1989).

Tests from 30 min to 8 h were carried out on the BAF. During the air-off periods samples were taken at 20 min intervals for up to 3 h and subsequently at 30 min or 1 h intervals. After the air flow was restored, the sampling frequency was as listed in Section 6.3.1.

The trend for the turbidity transient was similar in all air-off trials (Figures 6.6 and F.40-F.43 in Appendix F). An example of the BAF effluent turbidity in the 8 h trial is shown in Figure 6.6. It rapidly increases when the air flow to the process is off, and on resuming the air supply the turbidity reaches its peak value. The decline generally decreases exponentially to a value above the US EPA limit of 2 NTU. The recovery times, i.e. the times taken to the effluent quality to reach the values measured prior to air-off period or the water quality standard (Figure 6.7), suggest that air-off periods longer than 4 h lead to 2-3 times longer restoration to the normal effluent quality.

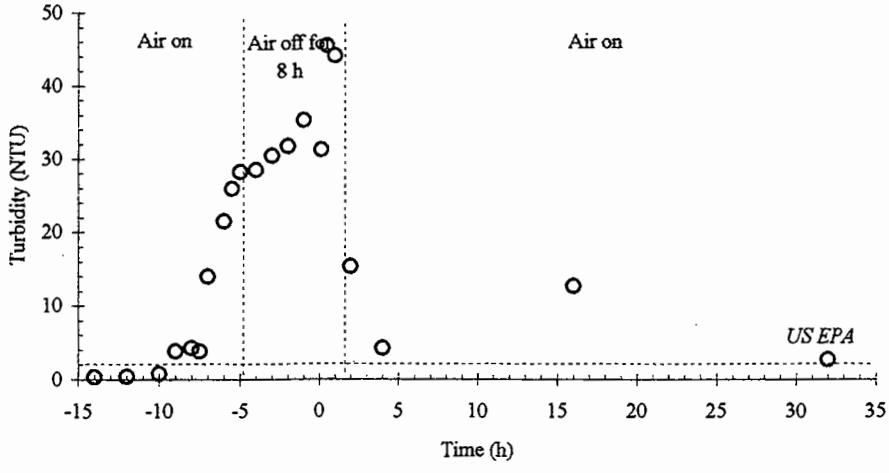
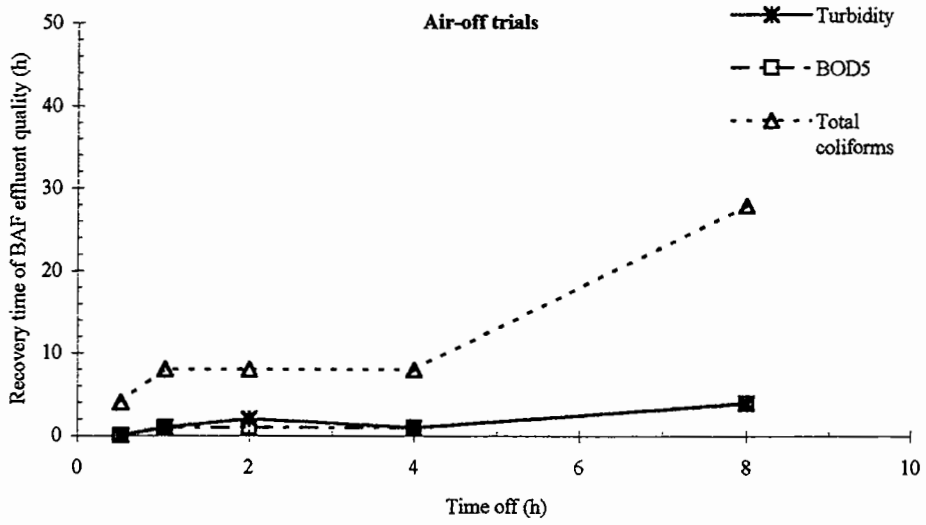


Figure 6.6. Turbidity transient of the BAF effluent before, during and after air supply off for 8 h.

)ue to the relatively low influent solids concentration (Table F.1 in Appendix) and he scatter in the effluent values (Figures F.44-F.48 in Appendix F), interpretation of ecovery time data is difficult. In the tests of up to 2 h the highest effluent SS values ıccurred when the air flow to the BAF was restored (Figures F.44-F.46 in Appendix '). The highest SS values were observed during the air-off period in the 4 and 8 h rials (Figures F.47-F.48 in Appendix F).



'igure 6.7. BAF recovery time in terms of turbidity, BOD_5 and total coliforms after the air-off trials.

The dissolved oxygen of >4.6 mg l^{-1} measured in the BAF effluent (Tables F.5-F.7 in Appendix F) may have resulted from oxygen entrapment in the medium, suggesting hat the heterotrophs were not limited by the oxygen concentration. However, the BOD_5 transients (Figures 6.8 and F.49-F.52 in Appendix F) show that the values ncreased during the air-off periods. For example, during the 8 h air-off period the effluent BOD_5 increased three-fold and remained at this level for 2 h after resuming he air flow (Figure 6.8). This was followed by a rapid drop to the baseline value of around 15 mg l^{-1} by 4 h following resumption of the air flow. The subsequent peak recorded at 16 h was likely to result from some fluctuation in the solid load to the system at the time. The reasons for the poor BOD removal during the air-off periods, despite recorded high aqueous DO levels, may reflect the heterogeneity of the system: solution DO levels may not reflect the levels within the attached biofilm itself. It has been noted in other studies (Pearce, 1996) that conditions in media bed are not only

plug-flow in vertical direction but that local conditions may also significantly vary in lateral direction. Variable oxygen transfer efficiency during air flow rate fluctuation has been shown (Pearce, 1996) to result in a changeable effluent quality.

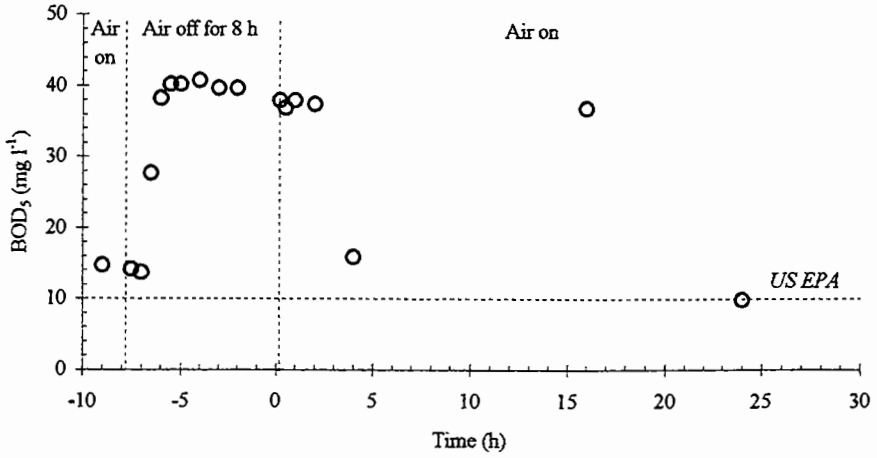


Figure 6.8. BOD_5 transient of the BAF effluent before, during and after air supply off for 8 h.

Total coliform and faecal streptococci counts of the BAF effluent varied from below the limit of detection to 7 log (Figures F.53-F.57 and F.63-F.67 in Appendix F). The recovery times of total coliforms increased with the increasing length of the test: air-off periods of 30 min to 4 h resulted in recovery times of 4-8 h, and after an 8 h test the recovery time was 28 h (Figure 6.8). This suggests that at between 4 and 8 h of an air-off period there is a critical point of operation at which the pathogen removal is deteriorated for a longer period of time. The same trend was observed for *E.coli* (Figures F.58-F.62 in Appendix F).

6.3.4 Feed and air-off trials

Interruption to both feed and air flow simulated an electrical power failure. The intermittent operation trials undertaken so far had shown the length of a test and the greywater quality to be important factors in determining the robustness of the process. Since the influent quality of greywater is difficult to control, only the length of the power failure was investigated in these trials. The BAF had been shown to be more sensitive to intermittent operation, hence tests from 2 to 8 h were chosen for the

ystem. The effluent lines were closed so as to prevent the column from draining luring the power failure. Due to the sludge volume effects (Section 6.3.2), an 8 h test vas carried out on the submerged MBR with the effluent lines left open. During the irst 3 h the initial DO of 8.9 mg l^{-1} in the mixed liquor dropped to 0.4 mg l^{-1} (Table '.3 in Appendix F). Consequently, the dissolved oxygen was below the critical ›xygen concentration for the remaining 5 h. The mixed liquor level in the submerged ⁄IBR recovered to the original level in less than 30 min after the power supply was estored.

n the submerged MBR the stable flux, low turbidity (Figure 6.9) and non-detectable ›acteria in the effluent (Figures F.77, F.80 and F.83 in Appendix) indicated that nembrane fouling had not occurred during the power failure. This is an important inding in terms of robustness and reliability of this process for the specific duty of in-›uilding water recycling. Generally the rate of aeration is critical for the submerged ⁄IBR as the lack of air scouring on the membrane surface can lead to a more rapid ›ouling layer formation (Section 2.4.5) and hence to a low flux and a poor ›acteriological quality of the effluent.

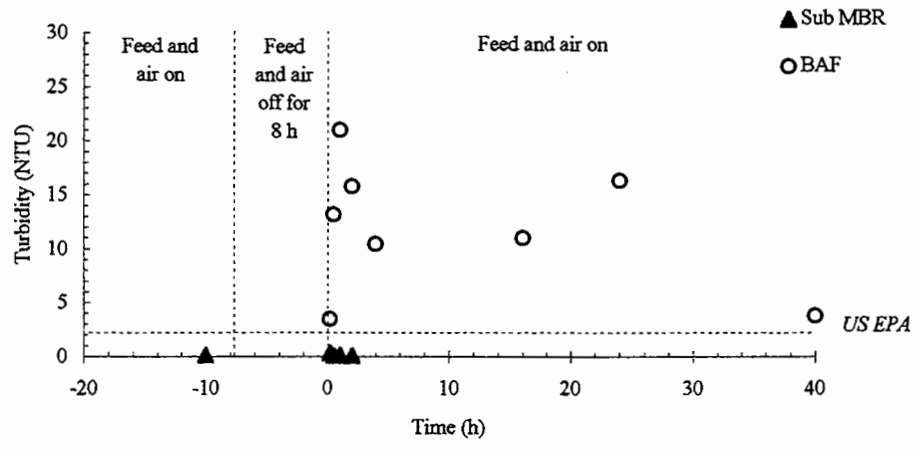


Figure 6.9. Turbidity transient of the submerged MBR and BAF effluents in the 8 h feed- and air-off trial.

In the submerged MBR the initial effluent BOD_5 of 1 mg l^{-1} increased six-fold over the first 3 h of the power failure, recovering to the initial level 30 min after the feed

and air supply had been resumed (Figure 6.10). To some extent the BOD removal appeared to be affected by two factors. Firstly, by the long period without feed and oxygen, and secondly by the relatively strong greywater (mean BOD_5 of 191 mg l^{-1}). Nevertheless the impact of an 8 h power failure on the biological stage of the process was insignificant with respect to the existing water reuse quality criteria.

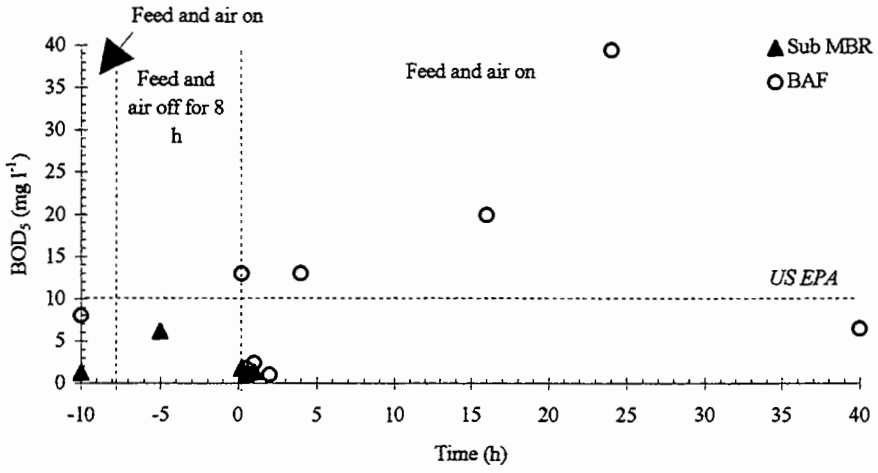


Figure 6.10. BOD_5 transient of the submerged MBR and BAF effluents in the 8 h feed- and air-off trial.

The 2 h feed- and air-off period had little effect on solids and organic removal by the BAF (Figures F.70 and F.73 in Appendix F). The increased effluent values and recovery times for all the parameters (Figures 6.9-6.11 and F.68-F.82 in Appendix F) measured in the subsequent tests were a result of longer test times and higher influent values (Table F.1 in Appendix F). Examples of this are turbidity and BOD_5 shown in Figures 6.9 and 6.10, respectively. Turbidity values and the recovery times in the 8 h tests were up to 10 times those in the shorter power failure trials (F.68-F.69 in Appendix F). The same was observed for the BOD_5 values which were up to 3 times higher than those in the shorter trials (F.73-F.74 in Appendix F). The results suggest that the treatment performance becomes limited both by substrate and oxygen after a power failure longer than 4 h.

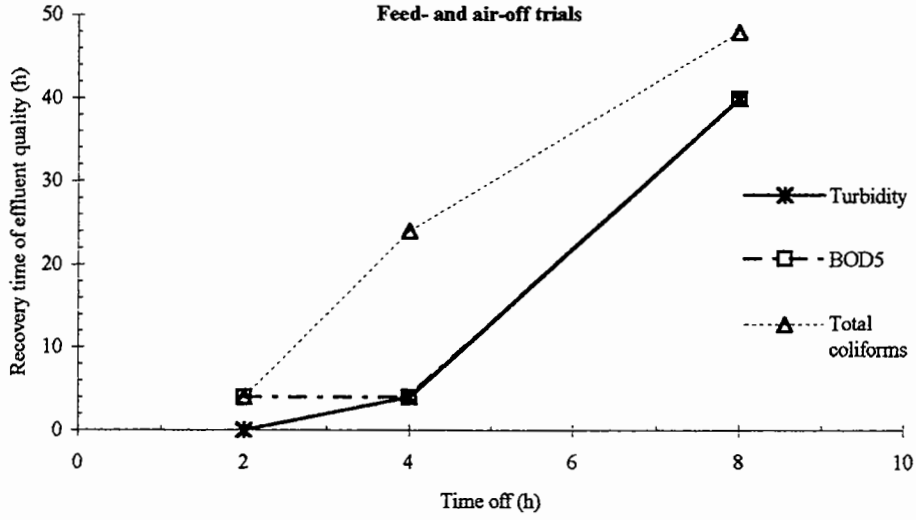


Figure 6.11. BAF recovery time in terms of turbidity, BOD_5 and total coliforms after the feed- and air-off trials.

6.3.5 RACOD vs.BOD_5

The failings of the RACOD as an online monitor for greywater became evident over the course of the feed-off trials. The peak RABOD readings did not correlate with those of the measured BOD_5 levels (Figures F.84-F.89 in Appendix F). This was also noted in all power failure simulation experiments for both treatment processes investigated as well as in the air-off tests on the BAF. The RACOD instrument appears to be much more suited to monitoring high-strength wastes such as raw, primary and secondary sewage effluent or COD-laden industrial wastes. In the current experiments, the preponderance of negative readings casts considerable doubt over the applicability of this instrument for the specific duty of treated greywater quality monitoring.

6.4 Survey on behavioural patterns in households

6.4.1 Survey method

A questionnaire (Table G.1 in Appendix G) on consumers' behavioural patterns was distributed to 520 people of all age, gender and social groups in the UK. The aim of the survey was to determine the following:

- which substances listed in the questionnaire were discharged into greywater sources,
- at what frequency the substances were discharged into a drain, and
- which of the substances were perceived as being harmful to environment.

341 replies were used as the basis for the respirometry work (Sections 3.3.2.2 6.4.2 and 6.4.3.2). Correlation between the age categories (<25, 26-40 and >41 years), the gender of the consumers and the questionnaire results (Tables G.3-G.4) was analysed (Section 6.4.3). As the questionnaire did not specifically address household occupancy or type, it was not known whether the persons responded for all the occupants in the household or for themselves only. The survey questionnaire was designed and distributed by B. Jefferson, a research officer in the department. The results are interpreted in the current work.

6.4.2 Analysis of the respirometry work

The impact of the substances was assessed by the specific oxygen uptake rate (SOUR) of the sample where the rate of oxygen uptake by biomass is divided by the biomass concentration. The results shown in Table G.2 in Appendix G are given as a percentage of the control value as described in Section 4.3.2. The critical concentration (c_{crit}) of each substance was determined as the concentration beyond

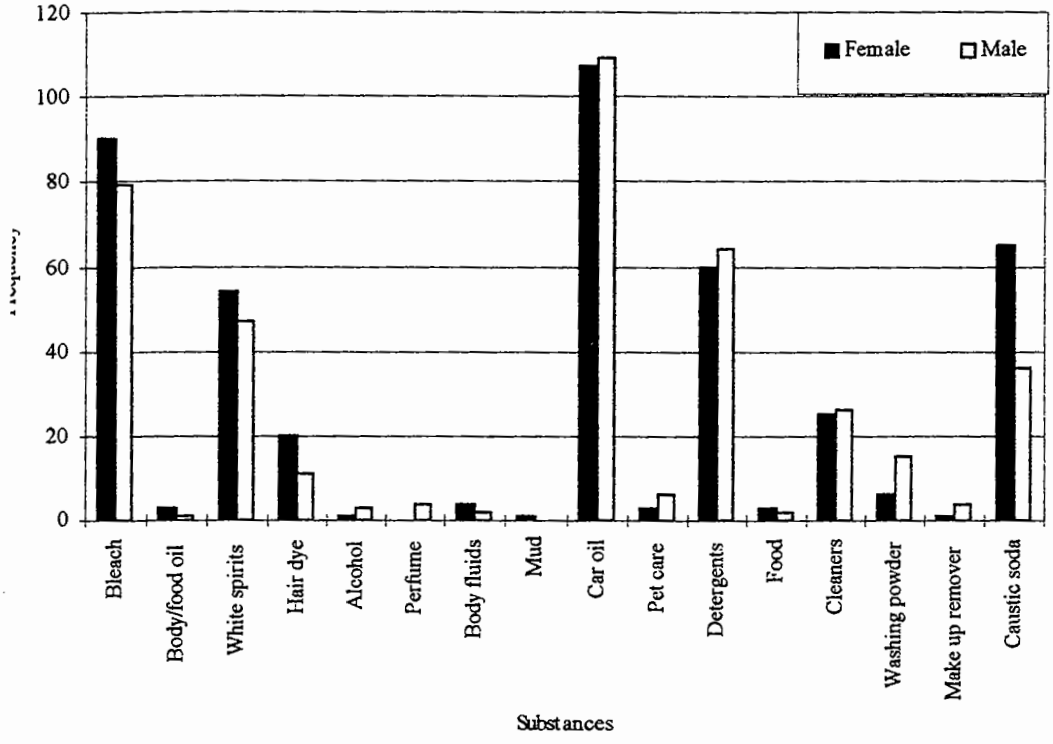
which the SOUR of the sample dropped below 100%, leading to reduced process performance.

6.4.3 Results

6.4.3.1 Survey

Perceived data

Bleach, car oil, detergents, caustic soda and white spirit were perceived to be the most harmful to the environment (Figure 6.12, Table G.3 in Appendix G). These substances are harmful to humans. Substances that are routinely used for direct physical human contact or indigestion, such as make-up remover and vegetable oils, were considered to be the least harmful to the environment (Figure 6.12, Table G.3 in Appendix G). No distinctive correlation between age and substance was found (Table G.3 in Appendix G): only in the case of caustic soda was a significant difference between female and male respondents found, with females judging it more detrimental to the environment than the males (Figure 6.12, Table G.3 in Appendix). All three female age groups and the males over 26 years of age perceived car oil as being the most harmful substance of those listed in the survey, whilst the younger males perceived bleach most deleterious (Table G.3 in Appendix G). Nearly twice as many males over 26 years of age than those younger than this age limit ranked detergents a damaging factor to the environment.

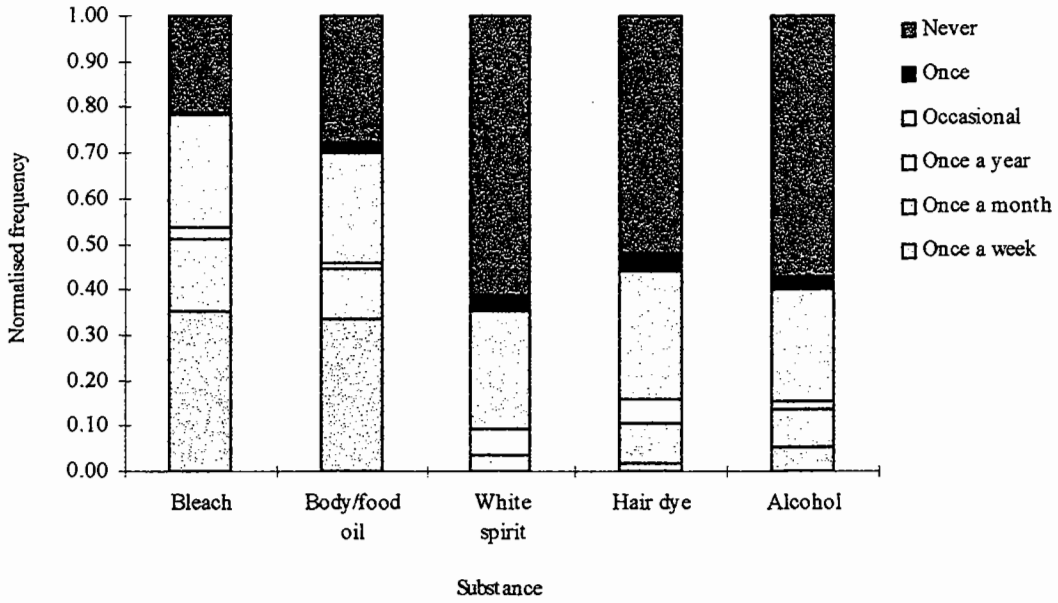


ıre 6.12. Substances perceived to be most harmful to the environment.

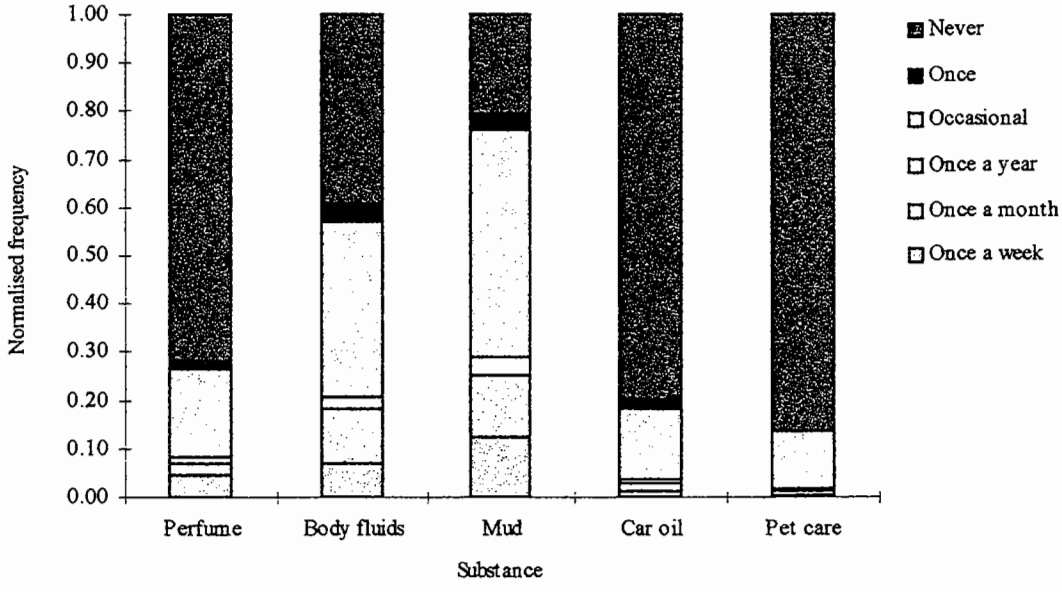
ːcharge frequency

normalised frequency of different substances being discharged with greywater is ːsented in Figures 6.13-6.15. The perceived deleterious nature of the substance did t appear to affect the frequency with which bleach, body and body/food oils - all ıssified as "harmful" - were discharged into a sink. Less frequent discharges in the se of white spirit, caustic soda and car oil are obviously linked to the limited ıusehold uses of these substances. For example, dominantly used as a solvent, white irit was 'occasionally' or 'never' used by the respondents whereas cleaners were scharged with the greywater mostly 'once a week' (Figures 6.13 and 6.15, Table G.4 Appendix G). Patterns between discharge frequency, gender and age may have ːen related to household occupancy and individual behaviour, the overall trend being ıdependent of gender. An exception to this appears to be bleach which the older ːspondents perceived harmful to the environment and yet discharged it more than the ounger people (Tables G.3-G.4 in Appendix G). People over 26 years of age also ischarged cleaners and food more often than the younger ones. The replies referring

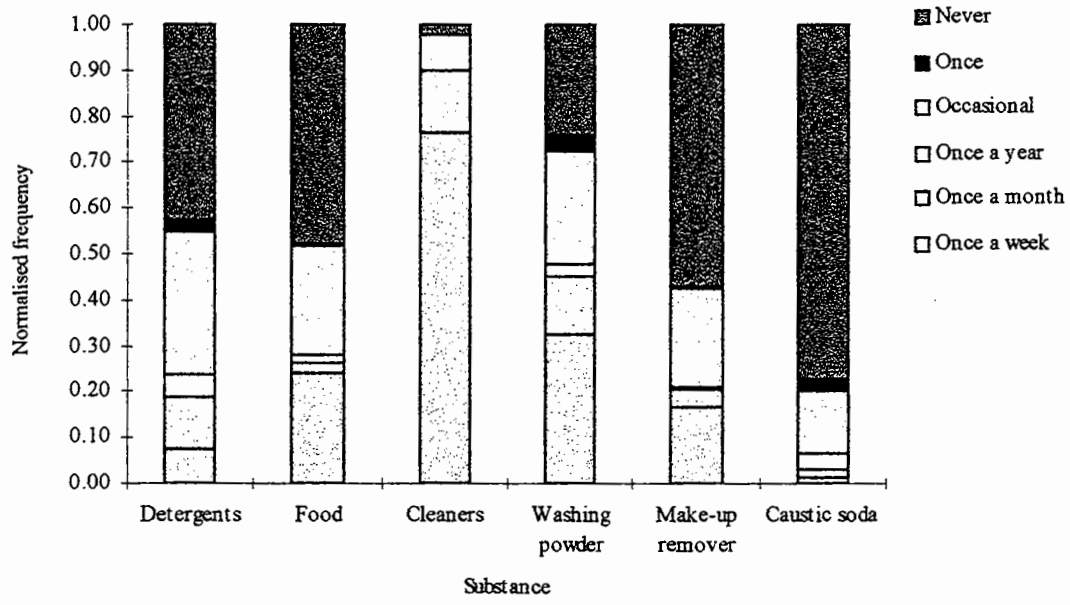
perfume, make-up remover and hair dye were independent of gender, suggesting at the respondents replied on the part of the household rather than just themselves dividually in these cases.



gure 6.13. Normalised frequency of bleach, body/food oil, white spirit, hair dye and alcohol being scharged in greywater.



igure 6.14. Normalised frequency of perfume, body fluids, mud, car oil and pet care products scharged in greywater.



gure 6.15. Normalised frequcncy of detergents, food, cleaning products, washing powder, make-up nover and caustic soda discharged in greywater.

4.3.2 Respirometry

his work assessed the impact of the identified pollutants on the recycling technology ther than the chemical mechanisms on the biomass such that possible pathways ere not investigated. The respirometry results were interpreted by the changes on the ygen uptake and COD removal as follows:

stimulation indicated by increased COD removal and increased or unchanged oxygen uptake,

metabolic inhibition indicated by decreased COD removal and oxygen uptake,

uncoupling of metabolism by decreased COD removal and increased oxygen uptake; anabolism is inhibited whereas catabolism is not, and

chemical reactions indicated by increased COD removal and decreased oxygen uptake; increased adsorption of COD-imparting substrate components onto the bacterial cell walls.

Effects

For most substances tested the SOUR increased with increasing substance concentration to a point where the substance, due to its chemical composition, was no longer was beneficial to the biomass sustainability resulting in a decline in its respiration rate (Figure 6.16). The concentration at which this occurred was defined as the critical concentration (c_{crit}) of the substance. A summary of these values shown in Table 6.3 demonstrates the nature of the added substance in terms of its use. Household cleaner such as bleach and high-strength chemicals such as car oil, caustic soda and perfume reduced cell growth at low concentrations and hence had low c_{crit} values (Table 6.3, Figure 6.16). A beneficial effect of readily assimilable nutrient matter on the SOUR was expected at all concentrations but adequate mixing of the samples during the respirometer test may have been hindered by the high MLSS at some "food" concentrations (based on soup). High doses of washing powder and hair dye resulted in foaming at such an intensity that the respirometry test could not be completed. This indicates that under such circumstances a biological system could be subjected to problems leading to a significant loss of performance.

Table 6.3. Critical concentrations (c_{crit}) of the test substances as defined by the specific oxygen uptake rate (SOUR) with respect to biomass.

Substance	c_{crit} (ml l^{-1})	Substance	c_{crit} (ml l^{-1})
Bleach	1.4	Pet shampoo	36
Caustic soda	4.5	Bathroom cleaner	43
Vegetable oil	<10	Food	60
Perfume	12	White spirits	95
Car oil	15	Alcohol	130
Washing powder	24	Make-up remover	320
Hair dye	>26	Urea	positive effect at all concentrations
Carpet cleaner	30		(0.25-6.2 ml l^{-1}) tested

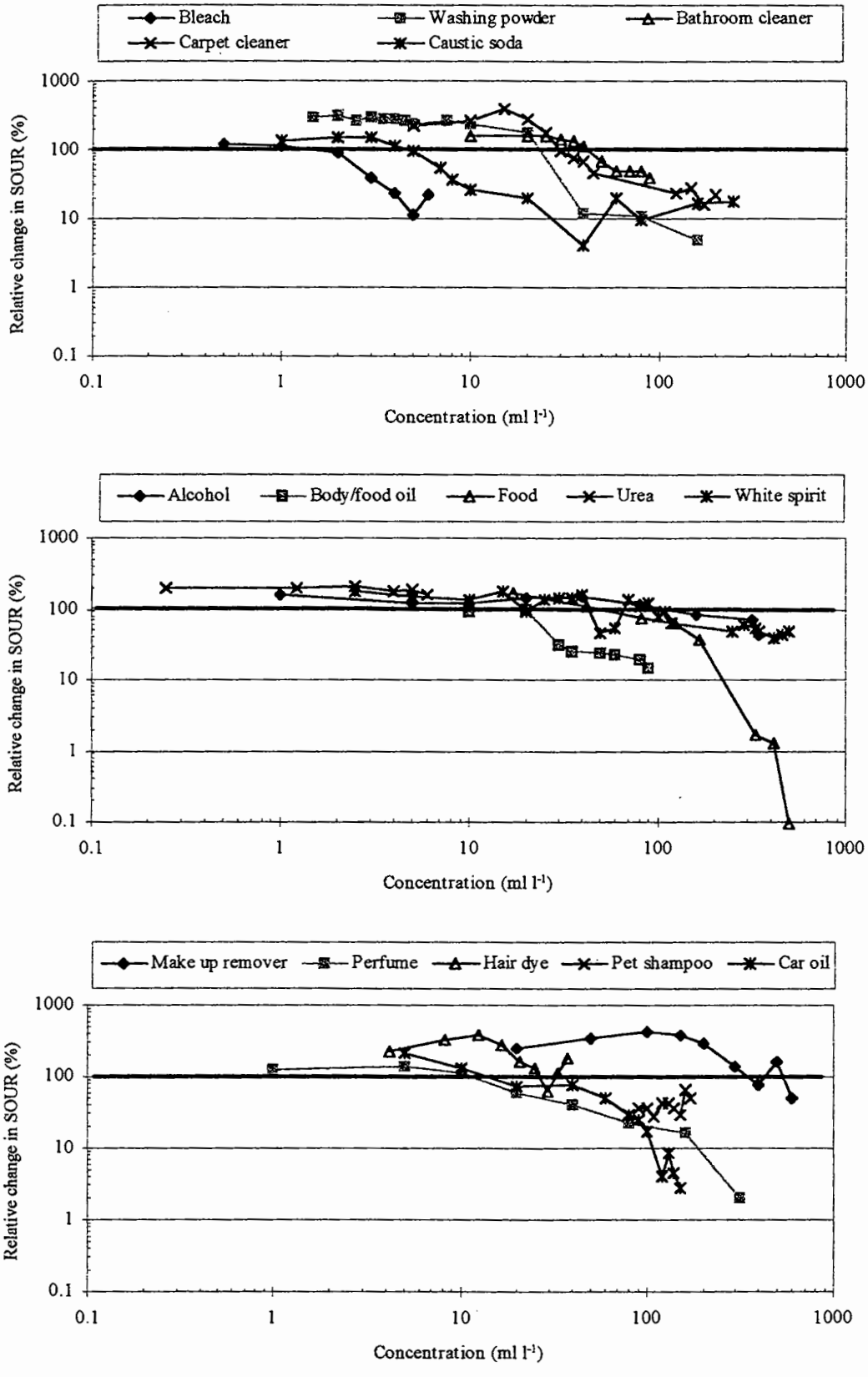


Figure 6.16. Relative change in the specific oxygen uptake rate of the test substances. The weighted line at 100% represents the control without a substance addition.

The large variability of the COD measurements (Tables G.5-G.12 in Appendix G) complicated the general interpretation of the effect of a substance on the biomass, such that it was not sensible to plot c_{crit} values for this parameter. It appears that in most cases accidental addition of the test substances would increase respiration of the activated sludge up to their respective c_{crit} (Figure 6.16, Tables G.5-G.12 in Appendix G) though only in half of these cases was improved COD removal observed at relatively low concentrations. Uncoupling of metabolism occurred in the bleach and caustic soda samples up to their respective c_{crit} after which the effects indicated metabolic inhibition (Tables G.7 and G.10 in Appendix G). Carpet cleaner, alcohol, pet shampoo and car oil were also stimulatory up to their respective c_{crit} values after which they exhibited chemical reactions (Tables G.5, G.8 and G.12 in Appendix G.12). Make-up remover, hair dye, washing powder and perfume were also stimulatory at low concentrations, either inhibiting metabolism or uncoupling and inhibiting metabolism at higher doses (Tables G.9-G.11 in Appendix G).

In some cases the changes in COD removal and respiration rates were not distinct, suggesting that a number of effects were taking place after addition of the test substances. For example, both metabolic inhibition and chemical reactions were observed in the biomass to which vegetable oil was added (Table G.7 in Appendix G). The variable results for urea indicated uncoupling of metabolism as increased respiration rates and decreased COD removal rates were observed with reference to those of the controls (Table G.6 in Appendix G). As most results presented here were not indicative of clear patterns, it should be emphasised that under these circumstances they should be regarded as observations only.

pH

The pH of the biomass significantly increased in the samples to which hair dye and caustic soda were added, such that values of up to 9.8 and 11.4 respectively were measured (Figure 6.17, Tables G.9-G.10 in Appendix G). The former reached pH 9 at a pollutant dose of 25 ml l^{-1} whereas the latter had already exceeded 9.5 at a dose of 1 ml l^{-1}, though in both cases the pH appeared to change little at doses above 20 ml l^{-1}.

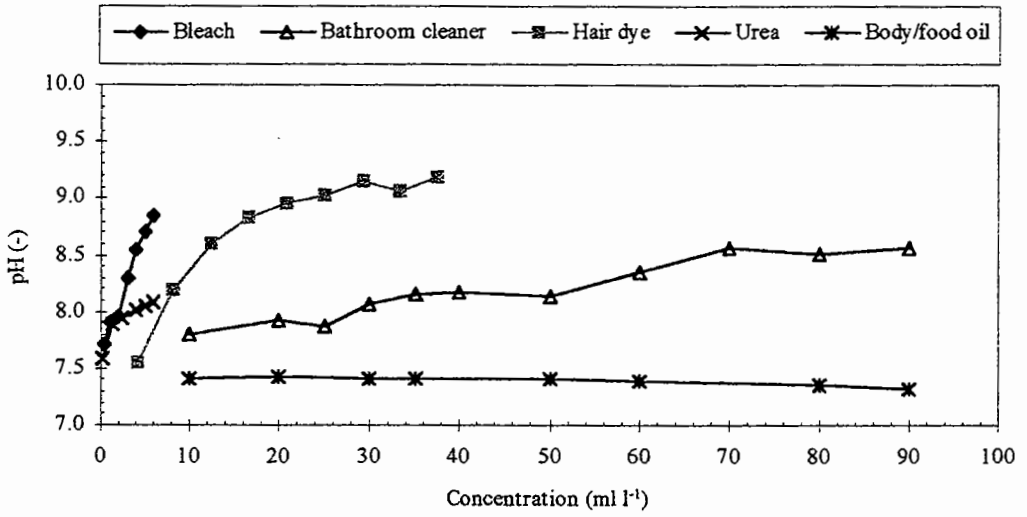
Smaller immediate changes to alkaline pH were observed in the biomass samples with bleach, bathroom cleaner and washing powder (Figure 6.17, Tables G.5, G.7 and G.10 in Appendix G). A significant drop in the pH to 4.7 was observed in the biomass after the addition of food at the highest dose and to a lesser extent following the pet shampoo doses (Figure 6.17, Tables G.9 and C.12 in Appendix G). Various concentrations of vegetable oil, white spirit, make-up remover and alcohol had relatively little if no impact on the pH of the sample (Figure 6.17, Tables G.5, G.7 and G.11-G.12 in Appendix G). Activated sludge plants are generally operated at a particular pH range to maximise the treatment efficiency, an example being the strongly pH-dependent nitrification process (Section 5.2.4.3). The observed changes in the pH indicate that if some substances were discharged in large volumes, deterioration of specific microbiological genera in activated sludge could be expected.

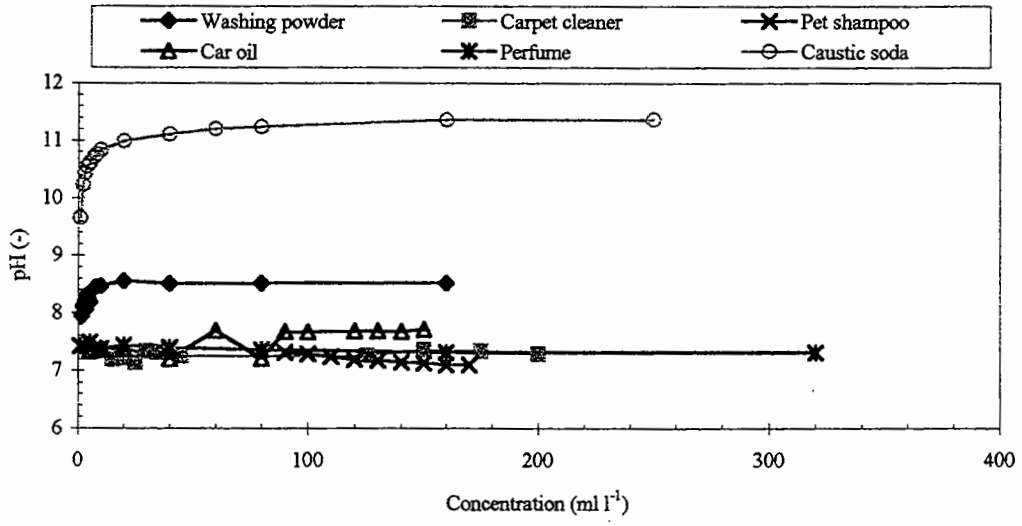
As greywater contains little nitrogen, the critical concentrations in terms of pH were defined by the ideal pH range of 6.5-8.5 for carbonaceous oxidation (Gray, 1989) rather than for nitrification (Section 5.2.4.3). Figure 6.17 and Table 6.4 illustrate that BOD removal would be affected by most substances at some of the tested concentrations. Similarly, several substances remained in the ideal pH range. When the c_{crit} values in terms of oxygen uptake rate (Table 6.3) and pH are compared, it can be seen that for caustic soda, washing powder and hair dye the c_{crit} for the pH is lower and hence a more crucial factor than that for the respiration rate.

Table 6.4. c_{crit} of the substances as defined by pH in the biomass (ideal range 6.5-8.5).

Substance	c_{crit} (ml l^{-1})	Substance	c_{crit} (ml l^{-1})	Substance	c_{crit} (ml l^{-1})
Caustic soda	<1	Bathroom cleaner	67	Vegetable oil	n/a at 10-90 ml l^{-1}
Bleach	4.0	Pet shampoo	114	Make-up remover	n/a at 17-500 ml l^{-1}
Hair dye	14	Alcohol	240	Perfume	n/a at 1-320 ml l^{-1}
Washing powder	14	Car oil	n/a at 5-150 ml l^{-1}	Urea	n/a at 0.25-6.20 ml l^{-1}
Food	40	Carpet cleaner	n/a at 5-200 ml l^{-1}	White spirit	n/a at 2.5-500 ml l^{-1}

n/a not available





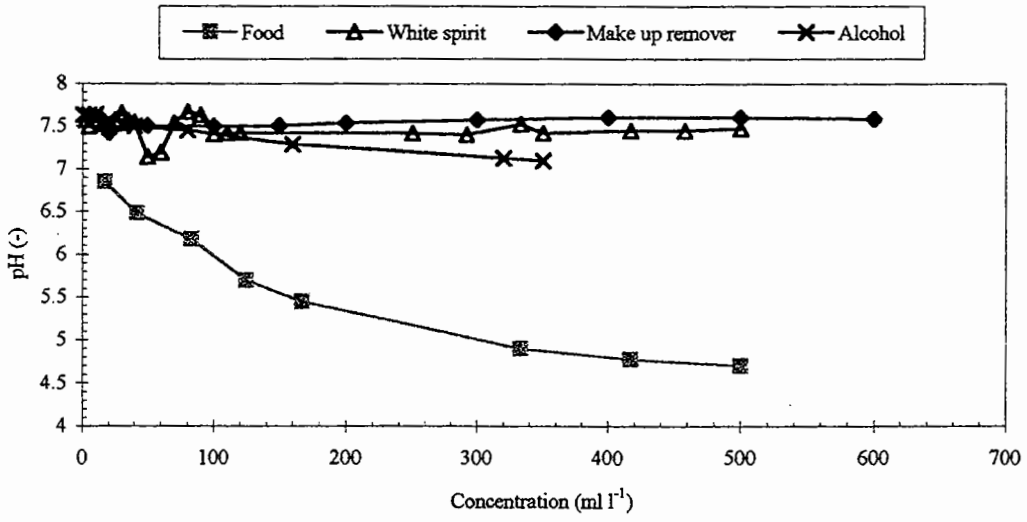


Figure 6.17. Change in the pH of the biomass with reference to the concentration of the substance.

6.5 Effects of change in feed

6.5.1 Overall performance

Changeover during the steady-state trials involved emptying the feed tanks of the old feedwater, filling them up with the new one, i.e. grey/blackwater or blackwater, and restarting normal operation regime after a maximum off-line period of 2 h. An equilibration period of four days was assumed for the submerged MBR, given its long HRT (Table 5.4, Section 5.2.4.1). The key water quality parameters were measured 12.5 min, 30 min, 1 h, 2 h, 4 h, 8 h, 24 h following the step change in concentration at the start of the changeover, followed by every 24 h for the subsequent 3 days. Influent and effluent quality data are shown in Table H.1 in Appendix H.

Greywater to grey/blackwater

The large sludge volume combined with the low influx of feedwater to the submerged MBR was expected to largely buffer the changed influent quality (Section 6.3.2). On changing the feedwater to grey/blackwater the organic loading rate increased by 50% to 0.12 ± 0.05 kgBOD m^{-3} d^{-1}, which is lower than many reported values (Table 2.8, Section 2.4.6.4). The submerged MBR achieved a substantial organic removal, producing effluent with a BOD_5 of 1-2 mg l^{-1} (Figure 6.18, Table H.1 in Appendix H). For such a process unusually high turbidity and SS peak values at 0.8 NTU and 6 mg l^{-1}, respectively, strongly suggest contamination or regrowth in the effluent lines. A breakthrough of total coliforms up to 1 log occurred three times, recovering to the non-detectable level within the first 24 h (Figure 6.18, Table H.1 in Appendix H). *E.coli* remained at a non-detectable level and pH at around neutral during the changeover period.

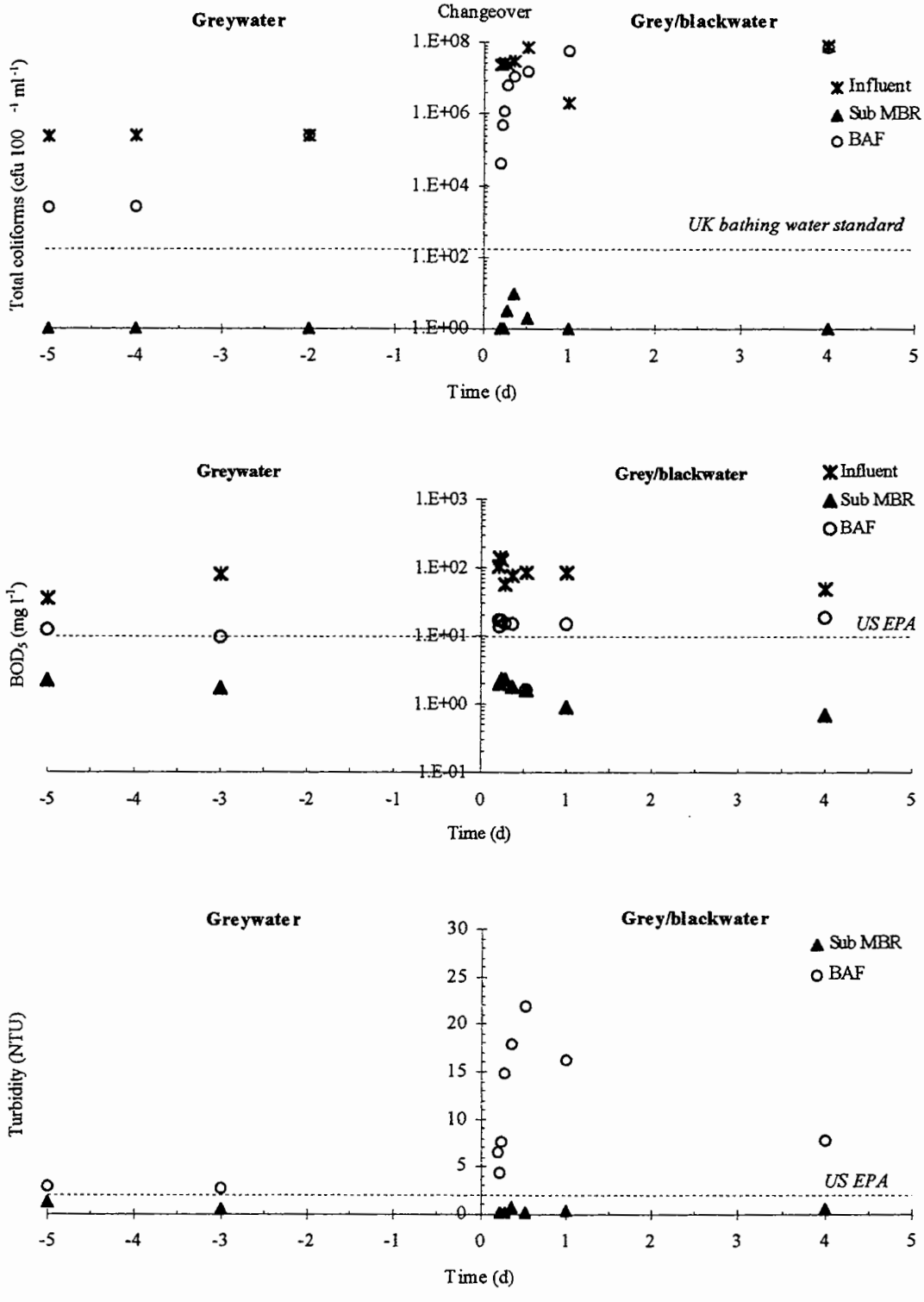


Figure 6.18. Influent, submerged MBR and BAF quality in terms of BOD_5, turbidity and total coliforms before and after the changeover from greywater to grey/blackwater.

The transition to grey/blackwater on the BAF resulted in a substantial loss of reatment capacity over an extended period of time (4 days). This was observed in the neasured parameters as a rapid increase followed by a slow decrease to a stable level ıigher than that prior to the changeover (Figure 6.18). The influent BOD_5 increased he OLR by 199% leading to a mean value of 0.59 ± 0.20 kgBOD m^{-3} d^{-1}, which is ow compared to the reference values in literature (Table 2.11, Section 2.4.6.5). The effluent BOD_5 remained relatively stable at mostly >10 mg l^{-1} over the 4-day period, with a maximum value of 19 mg l^{-1} (Table H.1 in Appendix H, Figure 6.18). The increased solids load in the grey/blackwater resulted in SS of 4-47 mg l^{-1} and turbidity 4.3-21.9 NTU in the BAF effluent (Table H.1 in Appendix H, Figure 6.18). Though significant bacteria removal by the BAF, a non-barrier process, was not expected (Section 5.2), the influent transition resulted in severely impaired bacteria removal by this process such that effluent total/*E.*coliform counts up to 7 log, the same as in the influent were measured. The treatment capacity of the BAF appeared to improve between days 1 and 4, however (Figure 6.18, Table H.1 in Appendix H).

None of the key parameters in the BAF effluent recovered to the former baseline or the water quality standards during the 4-day changeover period. Due to the significantly changed influent quality the recovery times were longer than those in the 2 h feed-off trial described in Section 6.3.3, such that direct comparison between these two trials cannot be made.

Grey/blackwater to blackwater

The OLR to the submerged MBR increased by 171% to 0.19 ± 0.04 kgBOD m^{-3} d^{-1} on transition to blackwater (Table H.1 in Appendix H). With relation to the long HRT and the sludge volume, the biological treatment by the unit remained fairly unaffected. The effluent BOD_5 ranged from below the detection limit up to 3 mg l^{-1}. Impaired filtration was observed as repeated breakthrough of bacteria (Figure 6.19, Table H.1 in Appendix H). The bacteriological quality of the effluent did not recover to the non-detectable level within the four days, this was possibly due to regrowth in the effluent pipes caused by the initial temporary failure of the membrane barrier and

aintained by the low effluent BOD. Also a temporary rise in the *E.coli* count to 1 g was also recorded. Though the submerged MBR effluent turbidity was low at <0.3 TU at all times, the high suspended solids peak of 13 mg l^{-1} after 30 min suggests mporary regrowth in the effluent pipes.

he performance of the BAF was similar to that recorded during the changeover from eywater to grey/blackwater though generally much higher values were measured Table H.1 in Appendix H) as a result of the substantially increased OLR, which had ıcreased by 122% to 0.91 ± 0.18 kgBOD m^{-3} d^{-1}. The increased solids concentration ı the blackwater produced peak values for BOD_5 and turbidity of 42 mg l^{-1} and 43.3 TU respectively - twice as high as those measured during the first changeover period Figure 6.19, Table H.1 in Appendix H). As grey/blackwater and blackwater had otal/*E.*coliform counts of the same magnitude, there were no significant changes in ıe bacteria removal. The effluent bacteria count was variable, ranging from 3 to 7 og (Figure 6.19, Table H.1 in Appendix H). Though some samples for BOD_5, solids nd turbidity were below the appropriate quality standards, neither these parameters ıor total coliform count recovered to the former baseline or the existing water quality riteria within four days after the step change.

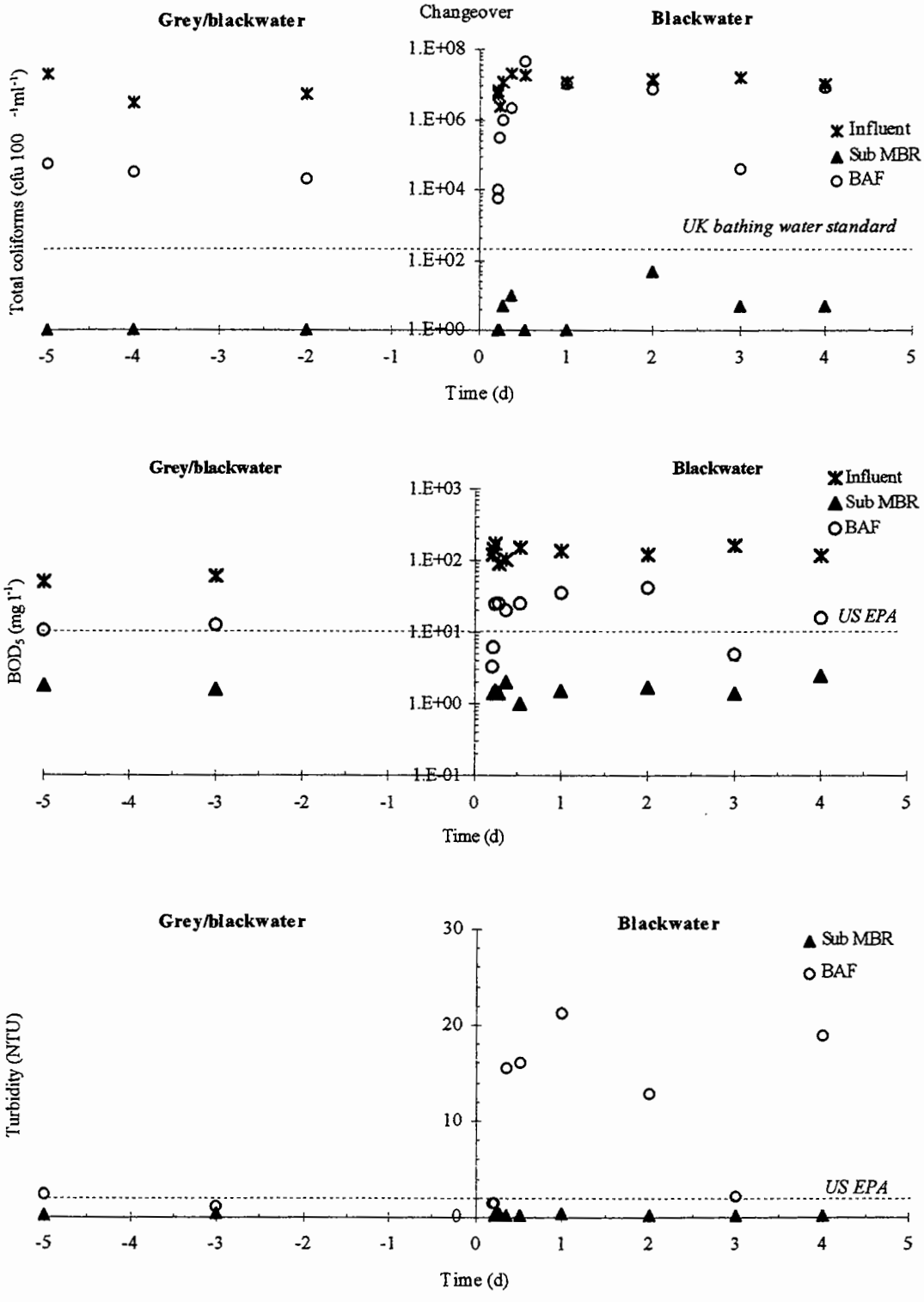


Figure 6.19. Influent, submerged MBR and BAF quality in terms of BOD_5, turbidity and total coliforms before and after the changeover from grey/blackwater to blackwater.

.6 Discussion

xperience in domestic recycling schemes has shown that the very small water :mand at weekends and holiday periods creates operational problems for biological eatment plants and raises the cost of reclaimed water (Kiya and Aya, 1991). owever, information on underloading is scarce for biological systems for which ıore often results from steady-state operation (Tables 2.8 and 2.11, Section 2.4.6.5 ıd Table 5.14, Section 5.5) or overloading (Chudoba and Pujol, 1998; Günder and rauth, 1998; Ueda and Hata, 1999) are documented. Generally process performance uring intermittent operation appears to be most influenced by the duration of the lectrical power, feed or air supply being off and the influent strength. Overall onclusions drawn from the results (Sections 6.3-6.5) relate only to environmental ıctors; absolute process efficiency is likely to be influenced by process design and peration parameters such as media or membrane selection, aeration rate and rate of hange of feedwater quality. Relative trends noted are, however, likely to be valid cross a range of conditions.

oth MBRs (Günder and Krauth, 1998; Ueda and Hata, 1999) and BAFs (Chudoba nd Pujol, 1998) treating municipal wastewater have been reported to cope with ıcreased flows and organic loads without producing impaired quality effluent. ünder and Krauth (1998) increased the flux of plate and hollow fibre MBR modules rom 18 l m^{-2} h^{-1} to 35 l m^{-2} h^{-1} and from 17 l m^{-2} h^{-1} to 30 l m^{-2} h^{-1}, respectively for a 5-day period. The effluent quality did not significantly differ from that in the steady-tate operation (Tables 5.12-5.13, Section 5.5). Similarly, Ueda and Hata (1999) bserved no deterioration in the filtration performance of a submerged MBR as a esult of increased inflow (1.5- to 3-fold). The volumetric loading rate was increased rom 0.215 kgBOD m^{-3} d^{-1} to 0.364-0.574 kgBOD m^{-3} d^{-1} for 58 days. During this eriod the mean flux ranged from 15.9 to 31.7 l m^{-2} h^{-1}. Three-stage BAFs consisting f carbon removal, nitrification and denitrification stages have been found to be elatively robust at variable loadings in municipal wastewater treatment (Chudoba nd Pujol, 1998). The influent, pre-treated by lamella settling for solids and soluble

hosphorus removal, was fed on to the BAF first at 12 m^3 h^{-1}. The flow rate was radually increased to 25 m^3 h^{-1} and maintained at this level for several weeks. The ffluent quality met the European Economic Community (EEC) standards of 135 mg $^{-1}$ COD, 35 mg l^{-1} SS and 10 mg l^{-1} TN at the maximum loading rates of 17.8 kgCOD n^{-3} d^{-1}, 1.2 NH_4-N m^{-3} d^{-1} and 2.8 $kgNO_x$-N m^{-3} d^{-1} applied to the three stages of the rocess. Above a flow rate of 28 m^3 h^{-1} the carbon removal stage became overloaded uch that some of the COD was removed by the nitrification unit. This resulted in a emporary decrease in the nitrification capacity as heterotrophs colonised the bottom of the nitrification filter. A short-term (6 h) overloading at 32 m^3 h^{-1} had no effect on he quality of the effluent whereas an increase to 40 m^3 h^{-1} for 6 h lead to a fraction of the COD being partially removed by the nitrification stage. The recovery of normal peration after these trials, however, was almost immediate as the heterotrophic acteria on the bottom of the nitrification column was replaced by the autotrophic acteria after backwashing the BAF. Adachi and Fuchu (1991) noted that under ormal circumstances the BAF performance recovered relatively quickly from high oncentrations of toxic matter, acids and alkaline. The authors attributed the rapid ecovery to the adaptability of the biofilm.

ixed-film reactors have been found to cope with organic shock loadings better than ydraulic shock loadings especially when using structured media due to the openness of the packing and the complete mixing of the liquid (Gray, 1993; Hamoda *et al.*, 1987). This stability of the biomass has been attributed to the use of regular ackwashing (Gray, 1993). Regular backwashing also improves biomass stability in granular media BAFs, however, an increase in the nominal flow results in an increased amount of mixing, causing increased effluent solids levels as the solids in the column are transported (Tschui *et al.*, 1994).

Opposite views on the robustness of fixed-film reactors during hydraulic and organic shock loads have been expressed by many researchers. According to one theory (Woolard, 1997), shock loads to continuous flow biofilm reactors, such as BAFs, cause stratification and uneven distribution of the biomass, limiting the ability of the

reactor to respond. It has also been suggested (Wilderer *et al.*, 1993) that species, population size and activity stratification takes place in plug-flow fixed-bed biofilm reactors along the reactor length in the direction of the flow. On increasing the feedwater concentration or flow rate some of the components are transported to zones in the reactor where the required micro-organisms are not present in adequate numbers and this results in a deteriorated performance. Generally BAFs treat organic shock loads by gradually reducing the concentration (Chen and Cheng, 1994) such that high loadings may lead to breakthrough of organic matter (Ruffer and Rosenwinkel, 1984). Le Tallec *et al.* (1997) found that even if the amount of autotrophic bacteria is the same in two BAFs treating primary settled sewage effluent, the amount of efficient bacteria is not the same if one of the filters is fed with a variable load and the other at a constant load. The variability of applied ammonia loads affected the filter performance, especially in terms of instantaneous effluent quality. In a constantly loaded reactor the maximum effluent ammonia concentration was <1 mg l^{-1} whereas in a reactor under variable loading conditions it was nine-fold. In an in-building recycling scheme such a variability in the effluent quality could occur if the ammonia load to a BAF was constantly or significantly changed. Such substantial changes could take place if the wastewater type (greywater, blackwater) was either suddenly changed (Section 6.5) or any type of wastewater from a building was treated immediately without balancing in an equalisation tank.

Some work has been carried out on underloading sequencing batch reactors (SBRs) located in areas where fluctuating influent flows are likely to occur, for example due to changes in the number of occupants. Wett *et al.* (1998) reported domestic wastewater treatment by an SBR with a maximum load of 120 pe in a tourist resort area. Prior to the measurements the process was underloaded with wastewater flow of 10 pe for a week. This was increased to 60 pe when the 3-day measurement period was begun. The low flow to the process resulted in a poor effluent quality and a MLSS as low as 270 mg l^{-1}, indicating a great loss of active biomass over a 10-day period. Production of humic substances as a result of biological degradation was observed as yellow-brown colour in the effluent. The BOD in the influent was

quickly used up by the heterotrophs, resulting in a COD:BOD ratio of 12:1 in the reactor.

One of the concerns during long periods without feeding is the decrease in the MLSS and active biomass, as demonstrated by the above SBR example. This is caused by a number of processes: decay, maintenance, endogeneous respiration, lysis and predation (van Loosdrecht and Henze, 1998). These occur more rapidly under aerobic conditions than anoxic or anaerobic conditions (Roslev and King, 1995; Siegrist *et al.*, 1999). Thus it is of interest to define the process limits in terms of the maximum period without feeding. These are influenced by the mixed liquor condition and the influent quality. Akunna and Jefferies (1998) simulated a situation of a family going away for the weekend using an SBR treating domestic wastewater. The feeding with raw sewage was curtailed for 52 h, and on resuming the influent flow the effluent BOD and SS were greater than during normal operation. Since the starvation period was relatively short, the poor effluent quality was attributed to the slow response by the micro-organisms to the new condition. The recovery time of less than two days supports this assumption. Both SBR studies (Akunna and Jefferies, 1998; Wett *et al.*, 1998) show that the biology may be compromised during intermittent feeding and hence attention must be paid to such occurrences for in-building water recycling duties. In the current work the submerged MBR was robust without feeding for up to 3.5 weeks, especially when the mixed liquor volume in the reactor was maintained at the normal level by closing the effluent lines (Section 6.3.2). Both biological and physical treatment of the process remained unaffected by the cessation to feed as well as power supply. After a long period (3.5 weeks) without feed foaming of the sludge may occur (Section 6.3.2), leading to impaired treatment performance due to loss of biomass. The equalising effect of the bioreactor volume has been previously reported on (Davies *et al.*, 1998). These authors, reporting on an MBR intermittently fed with raw domestic sewage, noted that at a daily cycle of 12 h of feeding (at 20 l m^{-2} h^{-1}) and 12 h of starvation, an effluent BOD of 4 ± 3 mg l^{-1} at a 98% removal rate was achieved.

quent shifting of activated sludges between feast and famine conditions has been nd (Chiesa *et al.*, 1985; Chiesa and Irvine, 1985; Chudoba *et al.*, 1973) to be ?ctive in controlling excessive growth of filamentous bacteria. Therefore short-n unsteady-state conditions, if properly selected and controlled, are an effective l to maintain long-term quasi-steady-state conditions (Irvine *et al.*, 1997), ticularly in systems where filamentous growth may be a problem. Physiological d genetic interactions between different organisms are complex and not entirely ar, however, co-operation, synergism, symbiosis as well as predator-prey-ationships are to be expected (Irvine *et al.*, 1997). Under different conditions fferent groups of organisms are switched 'on' or 'off': at high nutrient concentrations e-living organisms contribute more to degradation whereas at low nutrient ncentrations degradation is achieved by biofilms and microbial aggregates. The tter could be the case in greywater systems. Once the system is well established it ill tolerate a certain range of oscillations which it can mitigate. Nutrient depletion ring a few hours is more likely to increase both the uptake efficiency and the range f utilised substrates rather than decrease activity (Irvine *et al.*, 1997). This seems a kely explanation to the robustness of the submerged MBR during the feed-off eriods.

longside an adequate amount of substrate sufficient oxygen is required for optimum erformance of a biological process. Low oxygen supply to activated sludge plants nay occur, for example, during high loading periods and result in anaerobic onditions (Gray, 1990). Increased temperature enhances BOD removal which ncreases oxygen demand, resulting in depletion in oxygen levels. These lead to poor ettleability of the sludge and filamentous growth. Chuang *et al.* (1997) found that a variation of DO between 1 and 2 mg l^{-1}, the concentration range required for heterotrophic activity (Gray, 1989), had insignificant effects on the performance of a combined activated sludge-biofilm process. However, biological activity reduces and eventually ceases as the available DO for the biomass declines below the minimum critical concentration required (Section 6.3.3). Due to the relative increase in the ammonia the population of oxidising heterotrophs decreases at a higher rate than that

f autotrophs in activated sludge stored without aeration and feeding (Morgenroth *et* ., 1998). In a study by Morgenroth *et al.* (1998) sludge was still active with 55% of ıe initial volumetric ammonia oxidation rate after 6 weeks of storage without ɜration. Idle periods up to 6 days on an SBR were found not to affect nitrogen ɜmoval. These findings are important for intermittently operated package plants eating domestic sewage where ammonia removal is necessary. In the case of reywater treatment the fate of the heterotrophs is more crucial due to low nitrogen oncentrations generally found in greywater. However, if a recycling system was mployed to treat blackwater, the activity of the nitrifiers would also be important for fficient performance.

Vilén and Balmér (1998) investigated the short-term effects of DO on the turbidity of ıe supernatant of activated sludge. Turbidity increased up to two-fold during an air-ff period of 2 h and decreased to the original level in 2 h after the air flow was ɜsumed. When the influent flow was turned off during an anaerobic period the ırbidity increased two-fold. In both cases the pH followed the same pattern as the ırbidity; the increase in pH was attributed to decreased biological activity and a educed production of CO_2 and/or denitrification. The authors noted that the change n pH may have an effect on the surface properties of the sludge flocs, thus affecting he adsorption properties as well as floc stability. Increased turbidity was adhered to a ɔmbination of loss of adsorptive capacity and floc dispersion. In another study Starkey and Karr, 1984) turbidity increase was observed only after 10 h from lecreasing the DO from 5 to 0.4 mg l^{-1}. The difference between the observations in hese studies is likely to result from factors such as organic loading rate, temperature ınd wastewater composition (Wilén and Balmér, 1998). Palmgren *et al.* (1998) found hat oxygen limitation (<0.1 mg l^{-1}) lowers the bacteria cell surface hydrophobicity vhen the cells grow very slowly or not at all. A lower hydrophobicity may decrease ɜolid-liquid separation efficiency in wastewater treatment plants where bacteria ypically have such low growth rates (Palmgren *et al.*, 1998). In the current work the ɜffluent turbidity values did not notably change as a result of feed-off period (8 h) or ɔower failure (8 h) due to the membrane barrier in the submerged MBR. The effluent

H increased from the initial 7.6 to 8.0 during the first 3 h of the power failure and ecreased to 7.8-7.9 on resuming the feed and air flow (Table F.3 in Appendix F), orroborating findings by Wilén and Balmér (1998).

earce (1996) studied the oxygen transfer efficiency (OTE) of a downflow BAF using ineral media (3.3 mm). At the volumetric loading rates (1.0, 1.7 and 2.3 kgBOD m^{-3} $^{-1}$) tested the effluent ammonia levels remained below 2 mg l^{-1} at an air velocity 5-10 n h^{-1} but increased nearly six-fold when the air rate was reduced to below 5 m h^{-1}. At hese low air velocities the average bulk liquid DO of 3-5 mg l^{-1} was well above the equired concentration for nitrification. This suggests that the bulk liquid DO neasurement may not be the most appropriate for DO and process control as bulk iquid DO may not be the controlling factor in overall oxygen transfer. In the current tudy, where OTE values were not determined, the BAF effluent quality deteriorated luring air-off periods even though the effluent DO remained relatively high (>4.6 mg $^{-1}$, Section 6.3.3). This suggests variable DO levels and oxygen transfer efficiency in he column.

Air-off tests were not carried out on the submerged MBR for operational reasons. However, the results from the simulated 8 h power failure demonstrate (Section 6.3.4) that although the mixed liquor becomes oxygen-limited the bacteria are quickly revived such that there is no significant effect on biological performance. Though this type of incident may not have a long-lasting impact on the biology, the membrane may be more vulnerable to such an incident. It has been shown (Ueda and Hata, 1999) that an aeration failure to an MBR may lead to a rapid accumulation of a fouling layer on the membrane surface and thus increased filtration resistance if the filtration is continued. In the current study membrane fouling as a result of a power failure was not observed even though filtration was continued to the lowest possible level, i.e. just above the membranes, in this configuration. A likely reason is a slow build-up of a foulant layer prior to the trials combined with a low flow.

short-term unavailability of greywater did not have an impact on BAF effluent BOD_5 and turbidity (Section 6.3.2). Longer periods of intermittent operation of feed, air and power led to increased effluent values and prolonged recovery times. Though a reduction in bacteria removal was not expected, any type of intermittent operation appeared to result in a fluctuation in the performance. This was observed as high pathogen counts and often a wide range of recovery times (Figures 6.3, 6.7 and 6.13). The results suggest that a power failure is most deleterious for the BAF, which becomes limited by both nutrients and oxygen (Section 6.3.4).

Key determinants of treated water quality for the BAF effluent during intermittent operation are shown in Table 6.5. The recovery times based on the trials are indicative only, since this parameter is influent quality dependent. Although effects of deteriorated effluent quality downstream were not investigated, it is clear that disinfection would be required in any event due to the bacteriological content of the effluent.

Table 6.5. Recovery times of the major contaminants of the BAF effluent as a result of intermittent operation of feed and/or air.

Contamination	Feed off (30 min - 8 h)	Air off (30 min -8 h)	Feed and air off (2-8 h)
Total coliforms	0-48 h	4-24 h	4-48 h
Faecal streptococci	0-40 h	2-28 h	0-24 h
Solids	0-4 h	0-24 h	0-4 h
Turbidity	-	0-4 h	0-40 h
BOD	-	0-4 h	4-40 h
	3.5 weeks off - no recovery of any parameter in 48 h		

Significant changes in the influent quality are often reported in terms of shock loads or step changes, rather than actual changes in the influent itself. In the current work the influent transitions (Section 6.5) were combined with a relatively short period without feed due to the changeover procedure. With reference to the feed-off trials (Section 6.3.2), it is evident that the submerged MBR was not affected by the cessation of feed whereas this was not the case for the BAF. The respective mean

s at 0.12-0.19 kgBOD m^{-3} d^{-1} and 0.59-0.91 kgBOD m^{-3} d^{-1} for the submerged . and the BAF were lower than in similar processes (Tables 2.8 and 2.11, ons 2.4.6.4-2.4.6.5) such that this type of unsteady-state operation did not aps represent a significant organic shock load to the systems. The results (Section show that a change in the feedwater quality of the submerged MBR did not have npact on the effluent quality. In the current work a breakthrough of bacteria wed by a regrowth in the effluent pipes was observed. None-the-less, the eriological quality still met the UK bathing water standard. In a similar event the effluent quality more often than not exceeded the appropriate standards for the water reuse quality parameters of turbidity, BOD_5 and total coliforms (Figures 3-6.19), demonstrating the need for substantial post-treatment of the BAF effluent at least 4 days after an influent changeover. This limited robustness to step-nging places serious limitations of the BAF process for the duty of domestic water ycling.

addition to intermittent operation, the greywater recycling system must be tolerant abrupt changes in feedwater treatability and/or toxicity caused by occasional charge of specific compounds. Little information on the impact of such a scenario currently available partly due to many reuse schemes collecting greywater clusively from the sources less susceptible to such changes (baths, showers and ndbasins; Section 2.2.2). However, it is possible that such occurrences take place d are reported as foaming problems or deteriorated performance rather than directly lated to spiking of a specific substance to greywater. The survey results (Section 4.3.1) suggest that such events do occur in households. Large reuse systems have a fficient buffer volume to dilute such chemical shocks such that correspondingly rger volumes of specific onerous pollutants can be discharged without producing gnificant problems. It is more likely that small and medium sized recycling systems ould be susceptible to occasional discharge of onerous substances.

is important to investigate the potential effect of shock loads before launching a reywater recycling system. A series of tests were carried out by Surendran and

Wheatley (1998) on a biological process prior to initiation of the actual reuse scheme : Loughborough University (Section 2.5.3.2). A biofilter process was shock loaded y adding three types of cleaner, shower gel, laundry detergent and dishwasher etergent to the synthetic greywater so as to give a ten-fold increase in the COD. The rongly biocidic dishwasher detergent inhibited the biotreatment unit for several days hilst the other detergents were not found to influence performance for such a rolonged period. The findings from the substance addition respirometry work Section 6.4.3.2) suggest that the deteriorated performance in all tests by Surendran nd Wheatley (1998) occurred not only because of the high COD and the chemical ffects of the detergents but also because of the likely change in the pH of the reywater. In the subsequent full-scale operation at Loughborough University reywater from cleaners and kitchen sinks were not collected because of the amount f detergents and disinfectants, hence problems related to these substances affecting iological treatment were avoided.

Jolde (1999) noted that common personal hygiene products and household cleaning hemicals, as well as the occasional use of medicinal baths, do not cause problems in roperly functioning greywater treatment systems in which carrier materials are used. his is to be expected, given that many such substances are used at relatively low oncentrations and possibly diluted in a storage tank prior to treatment. The oncentration ranges tested on grab samples of activated sludge in the current work Section 6.4.3.2) revealed, however, that biological activity is reduced by certain hemicals; the critical concentration of the substance, i.e. the concentration at which ne specific oxygen uptake rate decreases below that measured for the control, was ound to decrease from make-up remover (320 ml l^{-1}) to household detergents (30-43 nl l^{-1}) and to household bleach (1.4 ml l^{-1}).

As stated before, many biological systems in recycling schemes are usually operated t steady state so as to avoid problems related to decreased biological activity. The esults show that the submerged MBR compares favourably with the BAF when greywater is treated on a discontinuous basis and on occasions when the influent

quality significantly changes. Therefore the former process has an added reliability factor essential for an in-building recycling duties. It can thus be concluded that the submerged MBR is suitable for unsteady-state operation, whereas the BAF relies on substantial post-treatment.

Chapter 7 Summary and conclusions

7.1 Experimental summary

7.1.1 Greywater characteristics

Greywater grab samples from baths, showers and handbasins in households demonstrated a large variation in chemical, physical and biological quality, generally being low in nutrient concentrations. The pathogen levels were the most variable of the parameters analysed regardless of the greywater source.

The composition and strength of greywater as well as the storage conditions had a significant impact on the dynamic behaviour of greywater. Real greywater began to degrade in 1-2 days whereas its synthetic analogue demonstrated a lag time of several days. Pollutant concentrations in stirred samples declined faster than those stored under quiescent conditions. Greywater samples taken from the base of the quiescent storage tank exhibited increased suspended solids (SS), chemical oxygen demand (COD) and biochemical oxygen demand (BOD) levels compared to the samples from the top of the tank. This was attributed to settling of organic matter during storage. Pathogen levels (total coliforms, *E.coli* and faecal streptococci) in quiescent real greywater increased by 2-5 log over the first 2-4 days of storage and persisted at that level until the end of the 18-day experiment.

Macro- and micronutrient addition to activated sludge treating real and synthetic greywaters was found to have potential to enhance process performance, with the COD removal improving in 86% of the cases. The higher variability in tests with real greywater than in a synthetic matrix was accounted for by the nature of the greywater.

'.1.2 Technical performance: steady state

teady state performance of the treatment processes was evaluated for a range of imulated domestic wastewaters: greywater (169 days), grey/blackwater (67 days) and lackwater (64 days). Over the course of these stages the volumetric loading rates VLRs) to the processes increased with increased influent strength. However, the peration at recommended rates (the BAF and the MABR), process limitations (the MBRs) and the influent quality resulted in the loading rates being significantly lower han those under normal operation. Despite this the process performance was generally good at the loading rates applied. The most pronounced difference between greywater and blackwater was the depleted nutrient levels in the former. The mean greywater ammonia loads to the submerged MBR and the BAF of 0.001kgNH_3-N m^{-3} d^{-1} and 0.002-0.170 kgNH_3-N m^{-3} d^{-1}, respectively increased to 0.037 kgNH_3-N m^{-3} d^{-1} and 0.170 kgNH_3-N m^{-3} d^{-1} in the blackwater trial. This improved ammonia removal of the processes.

The mean VLRs of the submerged MBR during the 300-day trial were 0.07-0.14 kgBOD m^{-3} d^{-1}, 0.25-0.43 kgCOD m^{-3} d^{-1} and 0.09-0.21 kgSS m^{-3} d^{-1}. The effluent quality was very good regardless the influent quality at a mean BOD_5, tCOD and SS removal of >96%, >92% and 89%, respectively. The turbidity was less than 0.4 NTU whilst a corresponding 6 log, 4-5 and 3-6 log total coliform, faecal streptococci and *E.coli* removal to the non-detectable level was achieved.

The initial flux of 60 l m^{-2} h^{-1} (1000 l m^{-2} h^{-1} bar^{-1}) in the submerged MBR declined to a stable 12 l m^{-2} h^{-1} (200 l m^{-2} h^{-1} bar^{-1}) over a 76-day period in the greywater trial. During the grey/blackwater and blackwater trials the respective mean flux rates were 12.8 l m^{-2} h^{-1} (213 l m^{-2} h^{-1} bar^{-1}) and 11.6 l m^{-2} h^{-1} (193 l m^{-2} h^{-1} bar^{-1}). The system was successfully operated without membrane cleaning for 300 days, an encouraging result with respect to robustness in long-term use.

reywater treatment by the side-stream MBR resulted in a very good quality effluent:)0% BOD_5, 99% tCOD and 93% SS removal efficiencies were achieved at 0.06 gBOD m^{-3} d^{-1},0.10 kgCOD m^{-3} d^{-1} and 0.02 kgSS m^{-3} d^{-1}. respectively. The effluent ssentially disinfected at a 6 and 2 log total coliform and *E.coli* removal to the non-etectable level. The effluent turbidity was less than 0.9 NTU. The side-stream ystem was operated at a flux range of 38-64 l m^{-2} h^{-1} (15-26 l m^{-2} h^{-1} bar^{-1}). The iembrane module was not cleaned for 32 days in this case.

he mean VLRs of the BAF during the trials were 0.27-0.67 kgBOD m^{-3} d^{-1}, 0.84-.03 kgCOD m^{-3} d^{-1} and 0.33-1.01 kgSS m^{-3} d^{-1}. The BAF achieved a good overall emoval of BOD_5, tCOD and SS at >86%, >82% and >81%, respectively. Both the athogen levels (up to a 3 log removal) and turbidity (mean 2.0-12.7 NTU) reflected he absence of a physical barrier. After the first 201 days of operation at a mean flow of 0.345 m^3 d^{-1} the flow rate was increased by 50% for 32 days in the grey/blackwater rial. This resulted in improved ammonia removal due to increased ammonia load but educed carbonaceous and solids removal efficiency because of the increased iltration velocity. The flow rate was returned to its original level (mean 0.309 m^3 d^{-1}) at the beginning of the blackwater trial.

Operational problems limited the assessment of the MABR to a relatively short-term greywater trial at 0.225 m^3 d^{-1}. The process achieved 72% BOD_5, 31% tCOD and 66% SS removal at the respective loading rates of 0.83 kgBOD m^{-3} d^{-1}, 3.73 kgCOD m^{-3} d^{-1} and 1.39 kgSS m^{-3} d^{-1}. Up to a 2 log reduction in total/*E.*coliforms and faecal streptococci was achieved. The mean effluent turbidity was 6.6 NTU.

Mixed liquor suspended solids (MLSS) levels of both MBRs declined over the course of the study. The submerged MBR was subject to solids accumulation in the denitrification units (14 381 mg l^{-1}) because of insufficient mixing, resulting in a very low MLSS in the nitrification unit (274 mg l^{-1}). The volatile fraction of the MLSS (MLVSS) in these units was 73% and 83% respectively. The largely variable MLSS and food to micro-organism (F:M) ratio of 0.04-0.38 d^{-1} (BOD) appeared not to have a

:trimental effect on the performance. In the side-stream MBR the initial MLSS of)00 mg l^{-1} declined to around 3000 mg l^{-1} over the 32-day greywater trial. The fLVSS and F:M ratio were 67% and 0.03 d^{-1} respectively.

'he compliance of the processes in terms of BOD, turbidity and total coliforms Figure 7.1) with respect to the appropriate water reuse quality standards was variable epending on the process type. The submerged and side-stream MBRs met the criteria 00% of the time with the exception of some effluent total coliform counts in the ubmerged system during the blackwater trial. The best BAF performance in terms of 3OD and turbidity was achieved during the grey/blackwater trial at the recommended low rate.

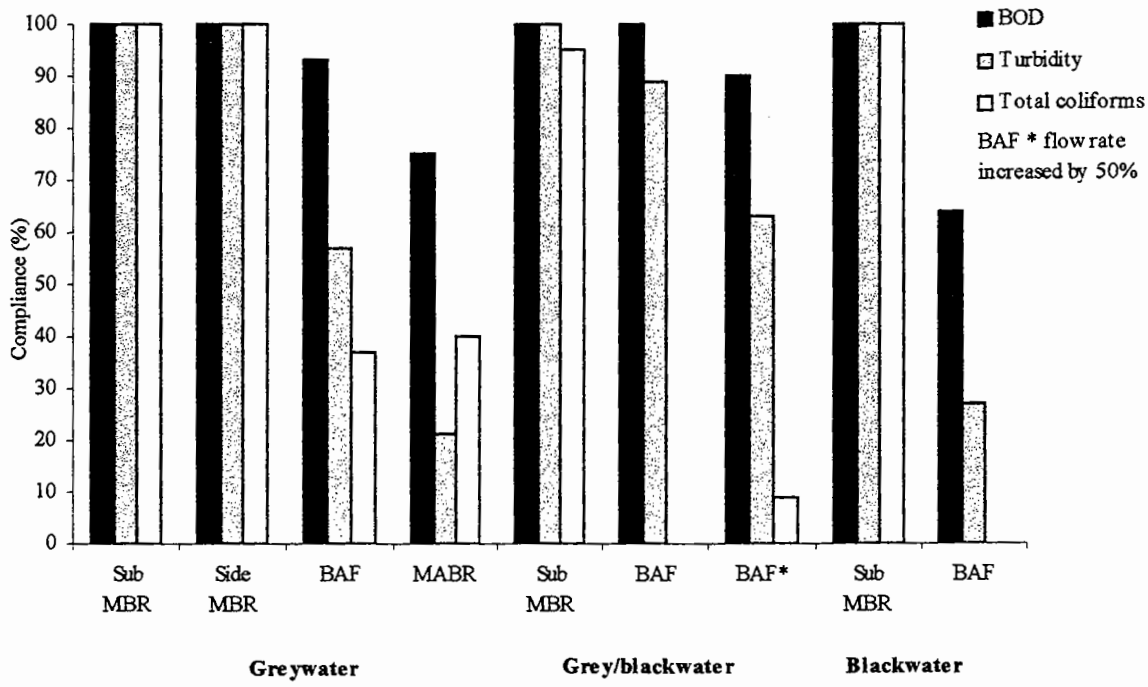


Figure 7.1. Compliance of the treatment processes in steady state in terms of the key water quality parameters. The standards: US EPA for BOD (10 mg l^{-1}) and turbidity (2 NTU), UK bathing water standard for total coliforms (500 cfu in 100 ml).

7.1.3 Technical performance: unsteady state

On step changing the influent from greywater to grey/blackwater and further from grey/blackwater to blackwater the submerged MBR effluent quality did not significantly deteriorate from that attained under steady-state operation. Regrowth of bacteria was observed in the effluent lines after both step changes, however, the effluent values were below the UK bathing water standard (500 cfu in 100 ml for total coliforms).

In the case of the BAF similar dynamic step changes in influent quality lead to a loss of performance in terms of all key water quality parameters (BOD_5, SS, turbidity, total coliforms and faecal streptococci). In some cases the recovery times to normal operation (effluent quality) exceeded 4 days. Generally the performance during the step changes was similar though higher effluent values were measured in the transition from grey/blackwater to blackwater due to higher influent values.

Up to a 3.5-week cessation to feed and an 8-hour to feed and air supplies had little effect on the biological and filtration performance of the submerged MBR treating greywater. The relatively stable MLSS of 8193 ± 424 mg l^{-1} (76 ± 4% MLVSS) under feed-limited conditions for 3.5 weeks indicated the robustness of the performance in this respect. On resuming the influent flow to the process sludge foaming occurred over a 2-day monitoring period.

Intermittent flow of feed and/or air to the BAF led to significant deterioration in effluent quality, such that recovery took up to 48 h (over 2 days in the case of 3.5-week feed-off trial). The most detrimental type of intermittent operation was simulated power failure during which the BAF became limited both by substrate and oxygen.

The behavioural patterns of 341 people were evaluated in terms of: a) the perception of household substances being harmful to environment, and b) the type and frequency

of substances discharged into greywater sources. The substance perceived to be most harmful to the environment by the surveyed population were car oil (63% of females and 64% of males) and bleach (53% of females and 46% of males). Cleaning products were the substances most frequently ('once a week' by 77% of females and 57% of males) discharged into drain. The concept of a substance being deleterious to environment did not affect the frequency with which some substances were discharged with greywater. No distinctive correlation was found between substance and age of the respondent.

Laboratory experiments based on the survey suggested that substances occasionally or frequently found in greywater may reduce the optimum performance of a biological process through decreased oxygen uptake and COD removal rate. Most of the 15 substances tested exhibited increased specific oxygen uptake rate of the biomass with increasing substance concentration to a point where the substance was no longer beneficial to the biomass sustainability. The most deleterious substances were bleach and caustic soda, which had a negative impact on the biomass at respective concentrations of 1.4 ml l^{-1} and 4.5 ml l^{-1}. The least detrimental substances to biomass were alcohol (130 ml l^{-1}) and make-up remover (320 ml l^{-1}). Urea had a beneficial effect at the relatively small concentration range (0.25-6.2 ml l^{-1}) tested.

7.1.4 Overall performance and process selection matrix

Based on the results from this work, a summary of the key technology selection aspects for greywater treatment is presented in Table 7.1. All four processes described represent small-footprint systems of a modular design and require screening of the influent as a pre-treatment as well as a supply of either air (the submerged and side-stream MBRs and the BAF) or pure oxygen (the MABR) as an ancillary service.

Biological treatment of greywater is necessary for a substantial removal of pollutants so as to alleviate problems associated with regrowth of organics. The submerged and side-stream MBRs achieve not only a significant BOD removal but also meet other

existing key water quality criteria (SS, turbidity, total coliforms, faecal streptococci; Figure 7.1), with the MF and UF membranes essentially providing disinfection. However, any downstream process would be required to maintain a residual level of disinfection during storage and distribution, favouring additional chemical dosing. Results from the grey/blackwater and blackwater trials indicate adequate performance of the submerged MBR for the duty of in-building recycling of both greywater and blackwater. The BAF achieves a good removal of organics, generally producing effluent quality ranging from fair to good. Greywater treatment by the MABR is of limited efficacy. Consequently, substantial post-treatment in terms of filtration and disinfection is necessary for the product waters from both non-barrier processes, i.e. the MABR and the BAF, so as to meet the most stringent of the existing water reuse quality standards.

Another benefit of the MBRs is the low sludge production, which compares favourably to both the BAF and the MABR, both of which require regular backwashing in order to restore their treatment capacity. Though occasional membrane replacement is necessary for the submerged and side-stream MBRs, the former can be operated for long periods without membrane cleaning whilst the latter requires regular chemical cleaning.

Greywater recycling schemes may be subject to unsteady-state operation due to replacements, maintenance, breakdown of equipment and holiday periods. The submerged MBR performance was relatively unaffected by intermittent introduction of feed or feed and air. By contrast, the results from the BAF demonstrate that a substantial polishing of the effluent would be required after such incidents.

In conclusion, the most viable scheme for in-building recycling according to this study comprises a submerged MBR with downstream chemical disinfection.

Table 7.1. Summary of technology selection for greywater recycling.

Assessment criteria	Submerged MBR	Side-stream MBR	BAF	MABR
Product water quality	Very good	Very good	Good/fair	Fair
Sludge	No excess sludge	No excess sludge	Daily backwash	Regular backwash
Compliance in steady state*	Yes	Yes	No	No
Ancillary services	Air	Air	Air	Pure oxygen
Pre-treatment	Screen	Screen	Screen	Screen
Post-treatment	Disinfection	Disinfection	Filtration + disinfection, sludge disposal	Filtration + disinfection, sludge disposal
Downtime/ maintenance	Occasional membrane replacement	Chemical cleaning, membrane replacement	Backwashing, media replacement	Backwashing
Suitability for unsteady state (compliance*)	Yes	-	No (unless substantial post-treatment)	-
Other			Excessive foaming at start-up	Unsuitable for greywater treatment

* water reuse quality criteria

7.2 Recommendations for further work

The following investigations are recommended for further work:

- The current work has investigated pilot-scale treatment processes for in-building wastewater recycling. Larger process units and recycling schemes may be selected on the basis of previous studies which have shown them to be more cost-effective than smaller ones. The scale-up issues remain an area of investigation as operation of large-scale biological processes, especially during unsteady state, may result in different results than those presented in this study.

- Wastewater-specific flora is a key factor in wastewater treatment, not the least in that of greywater. Further work is required to investigate if a greywater system requires different microgenera to that of municipal wastewater in order to yield optimum performance.
- Since the loading rates applied to the treatment processes were relatively low in the current work, further work is required so as to assess the effects of more significant organic and hydraulic shock loads. In the current work the impact of substances potentially harmful to biomass was evaluated with grab samples of activated sludge. Further work is necessary to determine the effects of chemical shocks to a pilot-scale biological process.
- Tests in the unsteady state in the current work were carried out one after another after the recovery of the submerged MBR and the BAF to normal operation. Therefore biology was, on a regular basis, subject to stress conditions of intermittent operation of feed and/or air. The impact of restressing biology requires further work so as to investigate the behaviour of micro-organisms during and after such incidents, i.e. if the micro-organisms behave according to hysteresis when recovering from unsteady state or if their condition is impaired by these incidents.

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Table A.1. Domestic water reuse schemes.

AUTHOR, YEAR, LOCATION	TYPE OF WASTEWATER	REUSE APPLICATION	TREATMENT PROCESS	NOTES
Ackerman *et al.* 1981 USA	Laundry wastewater	Laundry water	Laundry wastewater collected into equalisation basin and passed through cartridge filter. Mixed media filter insufficient for turbidity and suspended solids removal. Flocculants tested for improved colour removal and resistance to chlorine. Anion exchange system employed to limit total dissolved solids and to remove phosphates from the wastewater.	(1) Because of variable water demand at commercial laundries100% recycling system not favourable. Thus 75% recycle with top up with potable water chosen. (2) COD exceeded desired aim (50 mg l^{-1}), but no deterioration in textile characteristic or cleanliness observed. (3) Pilot-scale trials extended into full scale trials without major problems. Parameters chosen during pilot-scale experiments did not change later. (4) Bleach removed by activated carbon adsorption. (5) At flow rate of 0.08 m min^{-1} through carbon column, nominal detention time of 20 minutes sufficient to remove detergents, COD and chlorine. (6) 0.15 g of ion exchange resin effective to treat 1 litre of wastewater at optimum flow rate of 0.08 m min^{-1}.
Adachi and Fuchu 1991 Japan	Domestic wastewater from a building (120 m^3 d^{-1}) and cooling tower blowdown water (30 m^3 d^{-1})	Toilet flushing	Screening and pH adjustment with sulphuric acid. Oil removed prior to BAF process (diameter 1680 mm, bed height 2000 mm). Polishing by sand filtration and ozonation process. Algicide charged into cooling tower for biogrowth control.	(1) Estimated water savings 40%. (2) Advantages: • compactness • facilitated maintenance • excellent organic matter and suspended solid removal (final effluent: <5 mg l^{-1} BOD of and

				<1 mg l^{-1} SS) • capable of meeting load fluctuations • suitable for industrial use (3) Disadvantages: • effectiveness of biofilm deteriorates if toxic matter present (4) Algicide caused deterioration of transparency of effluent from BAF process. Post BAF-treatment solved problem, and no unfavourable effect of water quality noted. (5) Charging nutrient salts may lead to excessive generation of slime (can be solved by light shielding).
Asano *et al.* 1981 Japan	Wastewater from an apartment complex Greywater	Toilet flushing, cleansing water for garbage collection areas, and ornamental pond and stream Toilet flushing	Treatment trains: screening, aerated grit chamber, comminutor, extended aeration activated sludge process, sedimentation, alum coagulation, sedimentation and dual media filtration. About 75-80 % of effluent discharged after chlorination. Remaining 20-25% of effluent subjected to on-site wastewater reclamation and reuse systems. Intake water withdrawn from between media filtration and chlorination stages and treated with ozone. Greywater treatment: ultrascreen, activated sludge process, sedimentation, clarification, flocculation and coagulation. Collected secondary effluent mixed with cooling water blowdown water. Mixture pumped through high rate filter and chlorinated, then mixed with backwash water from filter and stored.	(1) Reclaimed water quality: BOD_5 3-30 mg l^{-1} COD 5-40 mg l^{-1} ABS 0-2 mg l^{-1} SS trace 30 mg l^{-1} n-hexane extract 0-8 mg l^{-1} (2) Treated greywater quality: TP 2.5 mg l^{-1} BOD_5 2 mg l^{-1} COD 5.2 mg l^{-1} TN 1 mg l^{-1} SS 1 mg l^{-1} ABS 0.2 mg l^{-1} pH 7.6 (3) Problem areas identified: • finding rational methods of enforcing use of lower quality water on selected water reclamation district, • incorporating reclaimed water into traditional water supply - wastewater receiver responsibilities of municipality

				(4) Concluded that uniform accounting methods for pricing reclaimed water essential.
Burrows *et al.* 1991 USA	Greywater from showers	Shower water	(1) Microfiltration cartridges with nominal pore size of 0.2 μm, (2) Powdered activated carbon and coagulation/flocculation/ sedimentation followed by diatomaceous earth filtration. Three different kinds of flocculant used: aluminium sulphate, cationic polymer and cationic and anionic polymer with addition of sulphuric acid to adjust pH.	(1) Both systems capable of meeting standards for recycle of field shower water. (2) Microfiltration system resulted in 75% removal of TOC and over 99% removal of turbidity. (3) Advantages: • bacteria and parasite removal in absence of chemical disinfection • turbidity reduction superior • progressive build-up of TDS characteristic of coagulation process avoided chemical supply requirements minimal (3) Second PAC-based system resulted in 70% reduction of TDS, but ≥ 1 mg l^{-1} of activated carbon needed.
Calhoun 1979 USA	Wastewater from commercial complex	Toilet flushing	Approximately half of wastewater treated with the *Cycle-Let* system (activated sludge biological treatment process) and recycled back to toilet flushing. Blackwater treatment systems consisted of collection system including specially designed toilet and urinals, waste treatment process, ultrafiltration for solids/liquid separation. Polishing in three stages: • activated carbon for colour and odour removal • ultraviolet light-generated ozone for disinfection • storage and pressurising for return to toilets and urinals for flushing	The project satisfied both economic and environmental objectives.

Christova-Boal *et al.* 1996 Australia	Greywater	Toilet flushing Irrigation	Three houses were retrofitted to reuse greywater for garden watering and toilet flushing. One house designed with greywater system incorporated prior to construction.	(1) Greywater quality studies revealed that water derived from the bathroom, shower and hand basin preferred for garden irrigation. Greywater considered as "dilute sewage", had to be of adequate quality to prevent build-up of undesirable materials in cistern or its operating components. (2) Laundry water considered more suitable for toilet flushing, disinfection necessary prior to reuse.
Gerba *et al.* 1995 USA	Greywater from residence	Toilet flushing Irrigation	Residential single family house retrofitted to recycle greywater for irrigation and toilet flushing. *System 1* Water hyacinths and sand filtration Greywater pumped through two tanks (aquacells) containing water hyacinths where mean retention time was six days. Tanks aerated to reduce odours. From second tank greywater distributed to sand filter (0.38 m in depth and surface area of 3 m^2). Water then collected to storage tank for reuse in irrigation or toilet flushing. No mechanical raking or manipulation of sand filter *System 2* Water hyacinths, copper ion disinfection and sand filtration System 1 modified to provide active recirculation of greywater in storage tank to first aquacells containing water hyacinths. Swimming pool purity unit providing copper ion for disinfection installed in recirculation line prior to return of greywater to aquacell. Overflow pipe attached to second aquacell to bypass sand filter	(1) All systems met State of Arizona regulations for nitrates, turbidity, and suspended solids reduction. System 1 overall best greywater treatment system evaluated. However, System 4 more effective in colder climates where hyacinths do not function as well. Treatment efficacy: *System 1* Faecal coliforms Log_{10} 3.5 ± 1.0 cfu 100^{-1} ml^{-1} Total coliforms Log_{10} 4.7 ± 0.8 cfu 100^{-1} ml^{-1} pH 6.7 ± 0.6 NO_3-N 1.5 ± 1.4 mg l^{-1} SS 16.8 ± 13.3 mg l^{-1} Turbidity 3.9 ± 2.4 NTU BOD 3.7 ± 1.2 mg l^{-1} *System 2* Faecal coliforms Log_{10} 4.4 ± 1.0 cfu 100^{-1} ml^{-1} Total coliforms Log_{10} 5.8 ± 0.9 cfu 100^{-1} ml^{-1} pH 7.7 ± 0.2 NO_3-N 1.4 ± 1.0 mg l^{-1} SS 4.7 ± 5.7 mg l^{-1} Turbidity 3.6 ± 3.7 NTU *System 3* Faecal coliforms Log_{10} 4.6 ± 0.5 cfu 100^{-1} ml^{-1} Total coliforms Log_{10} 7.0 ± 1.0 cfu 100^{-1} ml^{-1} pH 7.8 ± 0.2 NO_3-N 2.3 ± 2.0 mg l^{-1}

			System 3 Copper ion disinfection and sand filtration and *System 4* Copper/silver ion disinfection Further modified from previous system by removing hyacinths and covering aquacells with Styrofoam to prevent evaporation. System 3 used copper and silver disinfection. *System 5* 20-μm cartridge filtration Aquacells and sand filter removed and replaced with 20-micron nominal porosity cartridge filter as primary source of greywater treatment. All greywater passed through this filter with no by-pass to storage tank	SS 9.3 ± 6.7 mg l^{-1} Turbidity 8.8 ± 3.5 NTU *System 4* Faecal coliforms Log_{10} 4.4 ± 0.7 cfu 100^{-1} ml^{-1} Total coliforms Log_{10} 5.8 ± 0.6 cfu 100^{-1} ml^{-1} pH 7.8± 0.1 NO_3-N 2.0 ± 0.8 mg l^{-1} SS 7.0 ± 6.8 mg l^{-1} Turbidity 2.9 ± 3.0 *System 5* Faecal coliforms Log_{10} 5.2 ± 1.4 cfu 100^{-1} ml^{-1} Total coliforms Log_{10} 6.3 ± 1.0 cfu 100^{-1} ml^{-1} pH 7.8 ± 0.3 NO_3-N 2.6 ± 0.7 mg l^{-1} SS 7.7 ± 5.3 mg l^{-1} Turbidity 6.7 ± 8.1 NTU (2) Additional disinfection probably needed to reduce concentration of pathogenic microorganisms present in greywater.
Huitorel 2000 Japan, Europe	(1) Wastewater from buildings and large hotels (2) Septic tank wastewater	(1) Toilet flushing Floor washing (2) N/A	(1) *Ubis* system: Wastewater screened twice prior to bioreactor and UF where concentrate is recirculated. Residence time one hour. Influent quality: COD_{KMn} 89.1 mg l^{-1} BOD 349 mg l^{-1} SS 96.5 mg l^{-1} n-hexane extract 11.7 mg l^{-1} MBAS 6.5 mg l^{-1} Colon bacilli group 18.5 $x10^4$ pp/cc Chemical cleaning of membranes every 45 days. (2) *Asmex* system: As above but several bioreactors. Effluent further treated by activated carbon and dephosphoration resins. Residence time 6-9 days.	(1) Advantages: • compact • simple operation and maintenance • copes with changes in organic load • low running and energy costs • high effluent quality Effluent quality: COD_{KMn} 12 mg l^{-1} BOD 3.7 mg l^{-1} SS n.d. n-hexane extract <1 mg l^{-1} MBAS 0.3 mg l^{-1} Colon bacilli group n.d. More than 45 Ubis processes in Japan by early 2000. (2) Advantages: • compact, small bioreactor volume

			Influent quality: BOD 13 500 mg l^{-1} COD_{KMn} 7000 mg l^{-1} SS 21 000 mg l^{-1} TN 5000 mg l^{-1} TP 330 mg l^{-1} pH 8.0 Bacterium 105-107 pp/cc	• treats heavily loaded influents without dilution • elimination of organics and phosphates • sand filter not required • high effluent quality Effluent quality: BOD 10 mg l^{-1} COD_{KMn} 10 mg l^{-1} SS 0 mg l^{-1} TN 10 mg l^{-1} TP 1 mg l^{-1} pH 6.5-7.5 Colour 10 Bacterium n.d. 35 Asmex processes in Japan by early 2000.
Irwin 1994 USA	Wastewater from office buildings (p.e. 3200)	Irrigation of ornamental ponds Toilet flushing	Treatment process: • screening • aerated biological system (suspended growth) • UF (pore size 0.005 µm) • activated carbon • UV	Effluent quality: BOD 5 mg l^{-1} TSS 5 mg l^{-1} Turbidity 0.2 NTU Total coliforms <2.2 cfu in 100 ml More than 80 industrial applications using this treatment process by mid-1990's.
Ishida *et al.* 1995 Japan	Domestic wastewater	Domestic reuse	The *Kubota* process: Screened (50 mm), degritted wastewater passed to aerated equalisation tank followed by (optional) denitrification tank and fine screen (1 mm). Flat plate polyolefine membranes (pore size 0.4 µm) submerged in nitrification tank. Part of mixed liquor returned from nitrification tank to denitrification tank. Final effluent collected by permeate suction pump. Excess sludge removed through valve between denitrification and nitrification units. Influent quality: pH 7.3 SS 153 mg l^{-1} BOD 176 mg l^{-1} COD 79 mg l^{-1} TN 29.6 mg l^{-1} NH_4-N 22.4 mg l^{-1} NO_2-N 0.2 mg l^{-1} NO_3-N 0.1 mg l^{-1} TP 3.7 mg l^{-1} Cl^- 51.5 mg l^{-1} *E.coli* and heterotrophes not	(1) Effluent quality: pH 7.1 SS <1 mg l^{-1} BOD 1.5 mg l^{-1} COD 6 mg l^{-1} TN 5.4 mg l^{-1} NH_4-N 0.1 mg l^{-1} NO_2-N 4.6 mg l^{-1} NO_3-N 0.1 mg l^{-1} TP 1.2 mg l^{-1} Cl^- 53.6 mg l^{-1} *E.coli* and heterotrophes not detected (2) 30% reduction in sludge production and improved running costs achieved, installation area 25-50% smaller than that of conventional plant. (3) Plant operated for 3 years without problems.

			analysed Operational parameters: • MLSS 12 000-18 000 mg l^{-1} • volumetric BOD loading 0.32-0.63 kg $m^{-3}d^{-1}$ • BOD-MLSS loading 0.025-0.042 kg $kg^{-1}d^{-1}$ • total nitrogen-MLSS loading 0.004-0.007 kg $kg^{-1}d^{-1}$ • hydraulic retention time 7.6-11.4 hr • sludge retention time 25-40 days • temperature 18.5-22.0°C • flux 0.5 m^3 $m^{-2}d^{-1}$	
Kiya *et al.* 1991 Japan	Wastewater from high-rise buildings		(1) Outside of public sewage works service area Two-stage biological treatment with final polishing by rapid sand filtration and disinfection. (2) Inside public sewage works service area Direct ultrafiltration membrane separation of raw wastewater.	(1) System produced high quality effluent but the unit could not be turned off at weekends. (2) Membrane separated activated sludge process could not remove low-molecular-weight soluble organic matter but unit could be turned off at weekends. (3) Very low sludge generation; high bacterial quality product water.
Lewinger *et al.* 1987 USA	Wastewater effluent from municipal treatment plant	Toilet and urinal flushing	Study to evaluate feasibility of dual distribution system in high-rise office building. Reclaimed wastewater supplied by treatment plant provided tertiary treatment through in-line coagulation and dual media filtration followed by disinfection.	(1) Reclaimed water clear, no noticeable difference in colour between the reclaimed water and domestic water, no odour, and maximum COD of 50 mg l^{-1}, slightly corrosive - but no more than domestic water. (2) Dye injection recommended for reclaimed water to differentiate it from domestic water. (3) Life cycle cost of supplying reclaimed water to at least 50% of high-rises in area less expensive than purchasing and distributing domestic water.

Maeda et al. 1996 Japan	Secondary effluent from municipal treatment plant	Toilet flushing	Rapid mono-media upflow sand filters as advanced treatment for secondary effluent. Filters: • total capacity 450,000 m^3 d^{-1} in total • filtration rate 200 m^3 d^{-1} • backwashing daily Reclaimed water disinfected by dosage of sodium hypochlorite, then distributed through distribution system. Each of 19 high-rise buildings had reclaimed water storage tank. Reclaimed water pumped to roof top tank to distribute to toilets.	(1) Initial bad odour and clogging of plumbing eliminated by improving facilities and operation. (2) Reclaimed water quality: pH 6.9-7.5 SS 0 mg l^{-1} BOD 1.0-4.3 mg l^{-1} COD_{Mn} 5.3-9.2 mg l^{-1} Turbidity 0-1 NTU TN 9.6-16.0 mg l^{-1} TP 0.48-1.00 mg l^{-1} Total coliform n.d. in 100 ml Combined chlorine residual 0.2-1.5 mg l^{-1}. Appearance from slightly yellow to clear. Odour from weak mould odour to strong sewage odour.
Murakami *et al.* 1989 Japan	Municipal treatment plant effluent Domestic wastewater	Toilet flushing	The processes used widely in Japan: • activated sludge process • attached growth biological processes (submerged biological filters and rotating biological contactors • the above and additional treatment (filtration, chemical clarification, ozonation and carbon adsorption)	Described treatment systems have successfully been used in Japan.
Naranjo *et al.* 1993 USA	Wastewater from offices, shopping centres, commercial and public facilities	Toilet and urinal flushing Landscape irrigation	Four-stage treatment (*Cycle-Let recycling system*): • anoxic and aerated biological process • UF • activated carbon • UV Mixed liquor was seeded with coliphage MS-2, rotavirus SA-11 and poliovirus type 1 to evaluate virus removal of the system. Additionally samples collected for naturally occurring animal viruses after completion of tests.	Process capable of producing enteric virus free product water. Test viruses were not found after UF stage (99.9997% removal) and any viruses passing membrane were eliminated to well below detection limits by UV.

Neal 1996 Australia	Greywater	Toilet flushing Garden watering	Two commercial systems for domestic water reuse: *System 1* Four-stage treatment • anaerobic treatment • aeration • settlement • chlorination (chlorine tablets) Treated water collected into buffer tank. *System 2* Intermittent cycle, extended aeration activation sludge process with effluent discharged to buffer tank. System half size of conventional plant; no initial anaerobic stage so reducing odours. Denitrification occurred in anoxic conditions; less energy required due to intermittent aeration.	(1) Two commercial systems technically successful: residents more aware of their water balance. Storage tanks gave household owners control over times when irrigation occurred. (2) Greywater studies set following as minimum treatment: gross screening, aeration (digestion/natural flocculation) sedimentation, disinfection (if secondary level of physical contact possible during reuse) (3) Potential reuse of greywater investigated by setting up five treatment processed using composite samples from households. (4)Treatment processes sampled over 3-week-period: • storage at typical winter temperature • storage at typical summer temperature • aeration at typical winter temperature • aeration at typical summer temperature • trickle filter at typical winter temperature
Nolde 1999 Germany	Greywater	Toilet flushing	Two systems for domestic water reuse: System 1 for multi-occupancy scale (70 pe) • sedimentation • multi-stage rotating batch contactor • vertical-flow sand filter with reed bed • UV disinfection System 2 for single family home • two fluidised-bed reactors	(1) Systems 1 and 2 have been operational since 1989 and 1995, respectively, without problems. (2) Treated water quality meets the German standards for reuse.

Phagoo and Côté 2000 USA	Wastewater from toilets and food court	Toilet flushing Some discharged to environment	Screening, anoxic zone (with ethanol addition), aerobic zone, *ZeeWeed* membrane. *Plant A influent* BOD 408 mg l^{-1} TSS 226 mg l^{-1} TKN 117 mg l^{-1} *Plant B influent* BOD 102 mg l^{-1} TSS 190 mg l^{-1} TKN 100 mg l^{-1} Permeate for toilet flushing filtered through activated carbon and disinfected by UV.	Operational since 1997. *Plant A effluent* BOD <2.0 mg l^{-1} TSS <5.0 mg l^{-1} NH_4-N 0.9 mg l^{-1} TN 12.1 mg l^{-1} *Plant B effluent* BOD <2.0 mg l^{-1} TSS <5.0 mg l^{-1} NH_4-N 0.3 mg l^{-1} TN 10.3 mg l^{-1}
Sayers 1998 UK	Greywater from single homes	Toilet flushing	Basic two-stage treatment consisting of a filter and disinfection.	Trials conducted for 2 years on various sites highlighted need to carry out maintenance in order to avoid operational problems. Detailed case study is presented in Section 2.5.3.4.
Shammas *et al.* 1994 USA	Primary municipal wastewater effluent Activated sludge (AS) effluent Rotating biological contactor (RBC) effluent Trickling filter (TF) effluent Lagoons effluent	Not specifically identified	Results of five applications using compact flotation-filtration tertiary treatment "Sandfloat". Unit combined predose coagulation flocculation, dissolved air flotation and rapid granular filtration. Flotation-filtration tank was 1.5 m in diameter, 1.8 m deep and 2.10 m high. Average processing time less than 15 minutes. Pilot plant design capacity of 300 m^3 d^{-1} of wastewater flow. Chemical dosages of 20-120 mg l^{-1} alum / 0.25-2 mg l^{-1} of non-ionic polymer across range of effluent types.	Removal efficiencies from the treatment plants: *Primary wastewater treatment* - 20 mg l^{-1} alum and 2 mg l^{-1} non-ionic polymer: 93% turbidity, 97% TSS, 88% BOD, 69% COD, 99% P and 98% total coliforms *AS plant*: 92% TSS and 91% turbidity *RBC* : BOD, COD, TSS and total coliform removal all superior to that of secondary clarifier and sand filter *TF* - 12 mg l^{-1} alum and 0.25 mg l^{-1} anionic polymer: 91-93% TSS, BOD and P

				Lagoon - 10 mg l^{-1} alum and 1 mg l^{-1} anionic polymer: 79% TSS and 56% BOD
Smith *et al.* 1999b UK	Greywater (handbasins, 120 m^3 d^{-1}) rainwater (max. 100 m^3 d^{-1}) and groundwater (600 m^3 d^{-1}) from the Millenium Dome	Toilet flushing	Greywater treatment: BAF Rainwater treatment: Reed bed/lagoon Groundwater treatment: H_2O_2 and GAC After these stages all water is combined and treated: • UF • RO • pH • disinfection • pH adjustment	Detailed case study presented in Section 2.5.3.3.
Tchobanoglous *et al.* 1998 USA	Wastewater from office building	Toilet flushing, garden irrigation	Treatment (at full occupancy 98 m^3 d^{-1}): • pre-treatment • biological oxidation • ultrafiltration • activated carbon • UV disinfection • chlorination	Operational since 1992. Performance: • influent BOD_5 600 mg l^{-1}, TSS 600 mg l^{-1}, TN 150 mg l^{-1} • effluent BOD_5 5 mg l^{-1}, TSS 5 mg l^{-1}, TN 4 mg l^{-1}, turbidity 0.15 NTU and TC < 2 MPN in 100 ml
Young *et al.* 1994 USA	Municipal treatment plant effluent	Toilet flushing in commercial buildings	Tertiary treated wastewater produced by in-line coagulation and dual media filtration followed by disinfection. This effluent quality had been used for irrigation, now considered for reuse in high-rise buildings. Activated sludge process with ultrafiltration for solid-liquid separation. Pre-treatment (screening, chemical clarification, or combination of these processes) + membrane process (ultrafiltration or reverse osmosis) [+ supplemental treatment (carbon adsorption)]. This type of treatment used only for greywater.	(1) Reclaimed water usage successfully extended in area (additional high-rise buildings, single family residential lots and cooling towers for commercial buildings). (2) When the treatment plant off-line in winter months some odour and colour problems observed - solved by using bag cartridge filters on-site. (3) 25% of domestic water and 75% of reclaimed water for toilet flushing. (4) Flow equalisation and chlorine disinfection needed.

able B.1. Calibration of the feed pump at different speeds. Mean ± standard deviation of 20 readings.

	Pump speed							
	1	2	3	4	5	6.25	7	8
low rate min^{-1})	3.734 ± 0.020	4.756 ± 0.018	5.881 ± 0.020	7.283 ± 0.033	9.094 ± 0.067	11.296 ± 0.069	14.064 ± 0.039	17.434 ± 0.137

able B.2. Brands and ingredients of the test substances used in the survey respirometry work.

est substance	Brand	Ingredients
leach	Tesco's bleach 1314C	Amongst other ingredients less than 5% of sodium hypochlorite
ood oil	Tesco's sunflower oil Sainsbury's pure vegetable oil	Sunflower oil Vegetable oil
White spirit	Great Mills' white spirit	n/a
Hair dye	L'Oréal Excellence Creme hair colourant	Colourant: aqua, cetearyl alcohol, propylene glycol, deceth-3 laureth-12, ammonium hydroxide, oleth-30, lauric acid, hexadimethrine chlorine, glycol distearate, polyquaternium-22, silica dimethyl silylate, ethanolamine, pentasodium pentetate, sodium metabisulfite, resorcinol, carbomer, p-phyenylenediamine, t-butyl hyroquinone, m-aminophenol, 2,3-diaminophonoxoxyethanol HCl, hydroxybenzomorpholine, parfum (C3418/2) Developer: aqua, hydrogen peroxide, cetearyl alcohol, tridiceth-2 carboxamide mea, ceteareth-30, glycerin, pentasodium pentetate, sodium stannate, tetrasodium pyrophosphate (C11174/1)
Alcohol	Ethanol	Ethanol (diluted to 40%) AnalaR, BDH Merck, Poole, UK
Perfume	Joe Bloggs Juice Female eau de toilette spray	Alcohol denat, parfym, propylene glycol
Body fluids	Urea	14.8 mg l^{-1} urea (based on body fluid analogue references from literature in which urea is the biggest single component) AnalaR, BDH Merck, Poole, UK
Mud	Great Mills flower compost	n/a
Car oil	Castrol super T high quality mineral based 2 stroke motorcycle engine oil	n/a
Pet shampoo	Bob Martin flea shampoo for dogs and puppies under 12 weeks' of age	Pyrethrins equivalent to 0.0475% w/w as Pyrethrum extract B.P. (Vet), piperonyl butoxide B.P (Vet) 0.49% w/w
Carpet cleaner	Vanish shampoo - carpet and upholstery stain remover	Amongst other ingredients less than 5% of polycarboxylates, anionic surfactants and non-ionic surfactants
Bathroom cleaner	Tesco bathroom cleaner 1252C	Amongst other ingredients preservative and less than 5% of amophoteric surfactants and phosphonate
Washing powder	Ariel washing liquid (automatic heavy-duty liquid detergent with biological action)	Less than 5% of phosphonates and cationic surfactants, 5-15% of nonionic surfactants and soap, 15-30% of anionic surfactants, and enzymes and optic brighteners

Make-up remover	L'Oréal Plénitude gentle make-up remover	Aqua, hexylene glycol, disodium cocoamphodiacetat, panthenol sodium laureth-8 sulfate, sodium laureth sulfate, triethanoamine, allantoin, quartenium-15, sodium oleth sulfate, magnesium laureth sulfate, magnesium laureth-8 sulfate, magnesium oleth sulfate, methylparaben, sodium benzoate, chlorohexidine dihydrochloride, disodium EDTA, parfym
Caustic soda	dp washing soda crystals	n/a
Food	Tesco's fresh tomato and basil soup	Vegetable stock (water, salt, dextrose, onion powder, flavourings, hydrogenated vegetable oil, yeast extract, sugar, mushroom extract, spices, leek extract, citric acid, herb, herb extract), tomato purée, tomatoes (18%), onion, carrot, single cream (3%), sugar, vegetable oil, modified maize starch, red wine vinegar, orange juice concentrate, salt, basil, garlic purée, black pepper, parsley

Table C.1. Shower, handbasin and bath water samples collected from households showing greywater source, products used and gender of donor.

Sample number	Greywater source	Products used	Gender of donor
1	Shower	Shampoo, conditioner, soap	Female
2	Shower	Shampoo, shower gel	Female
3	Shower	Liquid soap	Female
4	Shower	Bubble bath, shampoo, soap	Female
5	Shower	Shampoo, shower gel	Female
6	Shower	Anti-dandruff shampoo, shower gel, shaving oil	Male
7	Shower	Shampoo + conditioner	Female
8	Shower	Shampoo, soap	Male
9	Shower	Shampoo	Female
10	Shower	Shampoo, shower gel, shaving gel	Female
11	Shower	Liquid soap	Female
12	Shower	Shampoo, shower gel	Female
13	Shower	Shampoo, shower gel	Female
14	Shower	Shampoo, shower gel	Female
15	Shower	Shampoo, shower gel	Female
16	Shower	Shampoo, shower gel	Female
17	Shower	Shampoo, conditioner, soap	Female
18	Shower	Shampoo, conditioner, shower gel	Female
19	Shower	Shampoo, shower gel	Female
20	Shower	Shampoo, shower gel	Male
21	Shower	Shampoo, conditioner, shower gel	Female
22	Shower	Soap	Female
23	Shower	Liquid soap	Female
24	Shower	Shower gel	Female
25	Shower	Shampoo, shower gel, facial scrub	Female
26	Shower	Shampoo, shower gel, face wash	Female
27	Shower	Shampoo, shower gel	Male
28	Shower	Shampoo, conditioner, shower gel	Female
29	Shower	Shampoo, conditioner, soap, body scrub	Female
30	Shower	Shampoo, shower gel, make-up remover	Female
31	Shower	Shampoo, liquid soap	Female
32	Shower	Shampoo, shower gel	Female
33	Shower	Shampoo, shower gel	Male
34	Shower	Shampoo + conditioner	Female
35	Shower	Shampoo + conditioner, soap	Female
36	Shower	Shower gel	Female
37	Shower	Shampoo, shower gel	Male
38	Shower	Shampoo + conditioner, shower gel	Female
39	Shower	Shampoo, conditioner, soap	Female
40	Shower	Shampoo, conditioner, soap	Female
41	Shower	Shampoo, conditioner, soap	Female
42	Shower	Liquid soap	Female
43	Shower	Shower gel, soap	Female
44	Shower	Shampoo, conditioner, soap	Female
45	Shower	Shampoo, conditioner, soap	Female
46	Shower	Shampoo, conditioner, soap	Female
47	Shower	Shampoo + conditioner	Female
48	Shower	Shampoo, shower gel, soap	Female
49	Shower	Shampoo, conditioner, soap	Female
50	Shower	Shampoo, conditioner, soap	Female
51	Shower	Liquid soap	Female
52	Shower	n/a	Male
53	Shower	n/a	Male
54	Shower	n/a	Male

	Shower	n/a	Male
	Shower	n/a	Male
	Shower	n/a	Female
	Shower	n/a	Female
	Shower	n/a	Female
	Shower	n/a	Female
	Shower	n/a	Female
	Shower	n/a	Female
	Shower	n/a	Female
	Shower	n/a	Female
	Shower	n/a	Female
	Handbasin	n/a	Female
	Handbasin	n/a	Male
	Handbasin	n/a	Male
	Handbasin	n/a	Female
	Handbasin	Bubble bath	Female
	Handbasin	Make-up remover	Female
	Handbasin	n/a	Female
	Handbasin	Make-up remover, soap	Female
	Handbasin	Make-up remover	Female
	Handbasin	Make-up remover	Female
	Handbasin	Facewash, shampoo	Male
	Handbasin	Bath foam	Male
	Handbasin	Soap	Female
	Handbasin	Make-up remover	Female
	Handbasin	Soap, toothpaste	Female
1	Handbasin	Shampoo, soap, facewash	Male
2	Handbasin	Soap	Male
3	Handbasin	Shampoo, shower gel	Male
4	Handbasin	Shaving gel	Male
5	Handbasin	Soap	Male
6	Handbasin	Soap	Male
7	Handbasin	Soap	Male
8	Handbasin	Shampoo, shoer gel	Male
9	Handbasin	Facewash, shoap	Male
0	Bath	Shampoo, bath foam, shaving gel	Female
1	Bath	Shampoo, shower gel, bubble bath	Male
2	Bath	Shampoo + conditioner, soap	Female
3	Bath	Bubble bath, soap	Female
94	Bath	Soap	Female
95	Bath	Shampoo, facewash	Male
96	Bath	Soap	Female
97	Bath	Bath foam, shampoo, soap, facial scrub	Female
98	Bath	Liquid soap	Female
99	Bath	Soap	Female
100	Bath	n/a	Female
101	Bath	n/a	Male
102	Bath	n/a	Male
Repeat	Bath	Shampoo, soap, bubble bath	Male
Repeat	Shower	Shampoo, shower gel	Female

Table C.2. Shampoos and soaps used in the product effect experiment.

Price range	Product	Ingredients
High	Neutrogena shampoo	Aqua, ammonium lauryl sulfate, cocamide DEA, cocamidopropyl betaine, glycerin, imidazolidinyl urea, methylparaben, prolylparaben, citric acid, parfum
High	Dove bar soap	Sodium cocoyl isethionate, stearic acid, sodium tallowate, aqua, sodium isethionate, coconut acid, sodium stearate, cocoamidopropyl betaine, parfum, sodium palm kernelate, sodium chloride, trisodium EDTA, zinc stearate, tetrasodium etidronate, CI77891
Moderate 1	Pantene shampoo for normal hair	Aqua, ammonium laureth sulfate, ammonium lauryl sulfate, glycol distearate, sodium chloride, sodium citrate, cocamide MEA, dimethicone, panthenol, panthenyl ethyl ether, cetyl alcohol, stearyl alcohol, sodium benzoate, citric acid, ammonium xylenesulfonate, DMDM hydatoin, tetrasodium EDTA, parfum
	Pantene conditioner for normal hair	Aqua, cyclomethicone, cetyl alcohol, quatemium-18, steramidoporpyl dimethylamine, dimethicone, steraryl alcohol, panthenol, panthenyl ethyl ether, hydroxyethylcellulose, ceteraryl alcohol, polysorbate 60, glyceryl stearate, citric acid, EDTA, DMDM hydatoin, methylparaben, prolylparaben, parfum
Moderate 2	Pantene 2 in 1 shampoo with conditioner for normal hair	Aqua, ammonium laureth sulfate, ammonium lauryl sulfate, glycol distearate, sodium chloride, sodium citrate, cocamide MEA, dimethicone, panthenol, panthenyl ethyl ether, cetyl alcohol, stearyl alcohol, sodium benzoate, citric acid, ammonium xylenesulfonate, DMDM hydatoin, tetrasodium EDTA, parfum
Moderate (for 1 and 2)	Imperial Leather bar soap	Sodium tallowate, sodium palm kernelate, aqua, parfum, citric acid, titanium dioxice, pentasodium pentetate, cellulose gum, BHT, CI11680, CI74260
Low	Almond shampoo	n/a
Low	Tesco's peach perfume bar soap	Sodium tallowate, sodium palm kernelate, aqua, glycerin, parfum, sodium chloride, tetrasodium etridronate, tetrasodium EDTA, CI77891, CI12490

Table C.3. *Microbial nutrient requirements and concentrations present in real and synthetic greywaters.*

Nutrient	Reported requirements (mg l^{-1})	Role of trace metal[a]	Concentration detected (mg l^{-1})	
			Real greywater	Synthetic greywater
N, P, S	15, 3, 1 (minimum)[b]	-	5.00, 1.37, 16.3	5.00, 0.047, 17.5
Ca	0.4-1.4	Cell transport systems and osmotic balance in all bacteria. Bridging anionic extracellular polymers aiding flocculation. Increase growth rates. Requirements and effects vary. Interacts with other metals.	47.9	47.0
K	0.8- >3.0	Cell transport systems and osmotic balance in bacteria.	5.79	3.96
Fe	0.1-0.4	Growth factor in bacteria, fungi and algae. Adsorbed in proportion to the concentration available. Electron transport in cytochromes. Synthesis of catalase, peroxidase and aconitase. Ion reduction for floc formation.	0.017	0.009
Mg	0.4-5.0	Enzyme activator for a number of kinases and phosphotransferase in heterotrophic bacteria.	5.29	5.02
Mn	0.01-0.5	Activates bacterial enzymes. Often interchangable with magnesium in kinase reactions. Lower affinity for binding sites than other metals but still can inhibit metabolism at 1 mg l^{-1}.	0.04	0.02
Cu	0.01-0.5	Bacterial enzyme activator required in trace quantities. Can inhibit metabolism. Chelates other substances, reducing their toxicity.	0.006	0
Al	0.01-0.5	Not known. Affects the species found in sludge (Annaka, 1977).	0.003	0
Zn	0.1-0.5	Bacterial metallic enzyme activator of carbonic anhydrase and carboxypeptidase A. Dissociable on active sites of enzymes. Stimulates cell growth. Toxic at 1 mg l^{-1}, especially to protozoa. Can exacerbate toxic effects of other metals and inhibit metabolism.	0.03	0
Mo	0.2-0.5	A common limiting nutrient (Grau, 1991).	0	0
Co	0.1-5.0	Bacterial metallic enzyme activator. Dissociable on active sites of enzymes. Activates carboxypeptidase for synthesis of vitamin B_{12} (cyanobalamin) but otherwise toxic. Can inhibit metabolism.	0	0

[a] Burgess *et al.*, 1999a

[b] From COD:N:P ratio, Beardsley and Coffey (1985)

Table C.4. Respiration and COD removal rates of activated sludge fed with real and synthetic greywater.

Nutrient dose	Real greywater				Synthetic greywater			
	Oxygen uptake		COD removal		Oxygen uptake		COD removal	
	Rate (kg/kgMLSS/d)	Repeatability (%)	Rate (kg/kgMLSS/d)	Repeatability (%)	Rate (kg/kgMLSS/d)	Repeatability (%)	Rate (kg/kgMLSS/d)	Repeatability (%)
N, P, Co	1.1	26.7	0.2	49.0	1.7	-11.5	0.8	-23.3
C, P, Cu	1.1	27.8	0.3	26.5	1.4	4.2	0.7	0.3
N, P, Fe	0.9	32.1	0.3	34.2	2.2	15.5	1.0	3.2
N, P, Mo	1.2	25.6	0.2	42.7	2.5	16.5	1.3	-26.3
N, P, Zn	1.3	22.9	0.4	21.6	1.6	2.4	0.8	2.2
N, P, Al	0.9	31.3	0.4	76.2	1.5	16.3	0.7	3.1
Co	0.5	-	0.1	-	0.8	-3.1	0.4	2.1
Cu	0.4	-4.0	0.1	1.5	0.6	2.7	0.3	-1.1
Fe	0.6	0.9	0.1	-7.7	1.0	-4.4	0.7	-1.1
Mo	0.6	-	0.1	-	1.0	2.8	0.4	1.7
Zn	0.7	0.9	0.1	14.3	1.0	5.8	0.5	1.6
Al	0.6	-	0.1	-	1.0	1.2	0.5	-5.7
P	1.0	5.7	0.3	5.5	1.2	4.0	0.6	3.8
N	0.9	0.8	0.2	0.8	1.4	-7.5	0.7	4.3

Table C.5. Paired t-tests performed on respiration and COD removal data.

	Oxygen uptake rate	COD removal rate	Normalised oxygen uptake	Normalised COD removal
Real cf.synthetic greywater	df = 14 t stat = -5.47 t crit = 1.08 No significant difference	df = 14 t stat = -8.23 t crit = 1.08 No significant difference	df = 13 t stat = 2.27 t crit = 1.08 Significant difference	df = 13 t stat = 3.40 t crit = 1.08 Significant difference
N/P-balanced cf.N/P-limited greywater	df = 11 t stat = 7.34 t crit = 1.09 Significant difference	df = 11 t stat = 4.54 t crit = 1.09 Significant difference	df = 11 t stat = -0.02 t crit = 1.09 No significant difference	df = 11 t stat = -0.03 t crit = 1.09 No significant difference

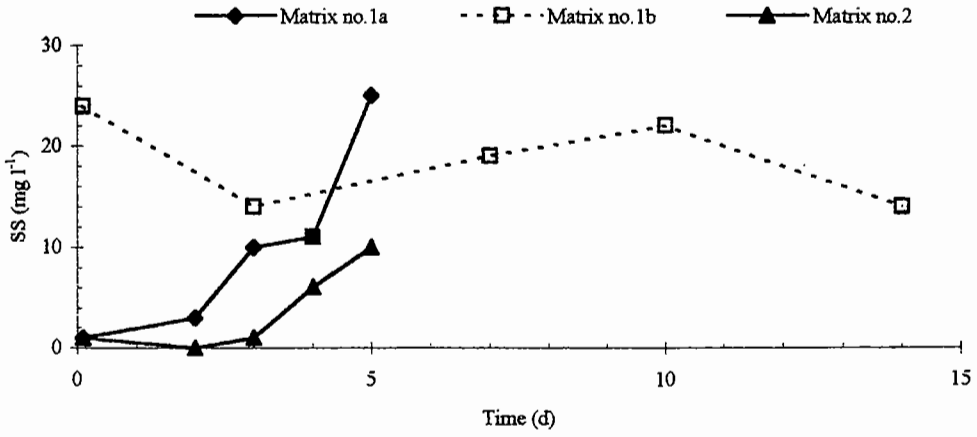


Figure C.1. SS transient of air-mixed small-scale greywater batches (Matrices no.1 and no.2) over 5 and 14 days.

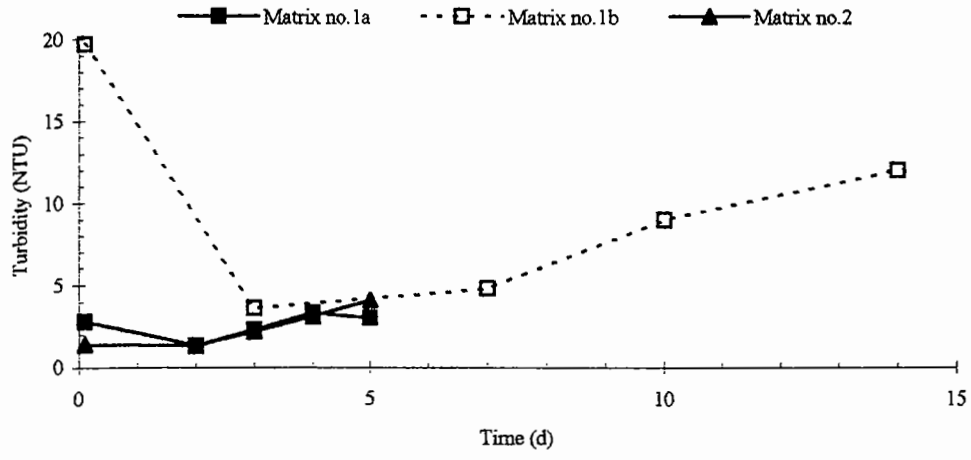


Figure C.2. Turbidity transient of air-mixed small-scale greywater batches (Matrices no.1 and no.2) over 5 and 14 days.

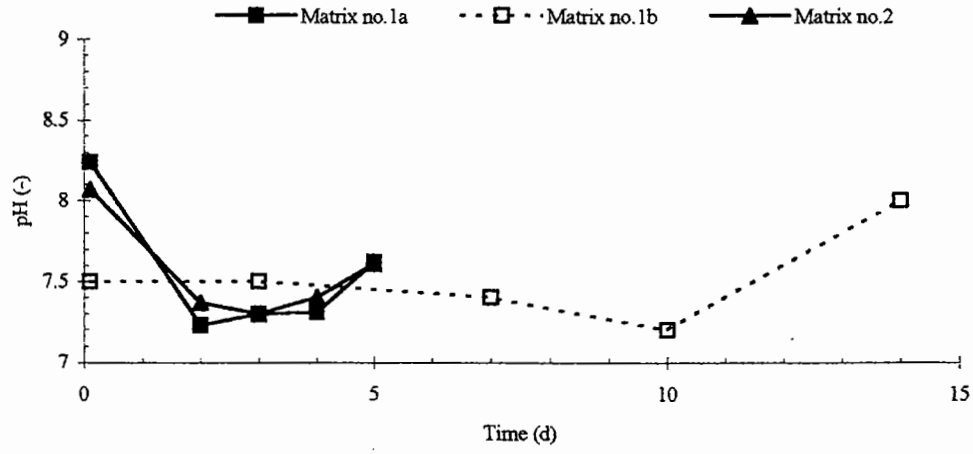


Figure C.3. pH transient of air-mixed small-scale greywater batches (Matrices no.1 and no.2) over 5 and 14 days.

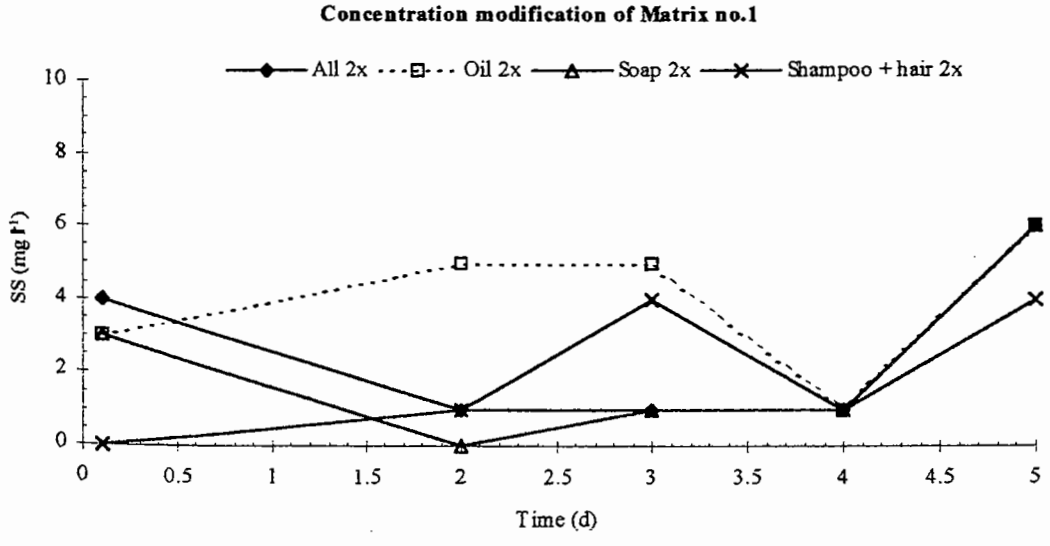


Figure C.4. SS transient of air-mixed small-scale modified greywater batches (Matrix no.1) over 5 days.

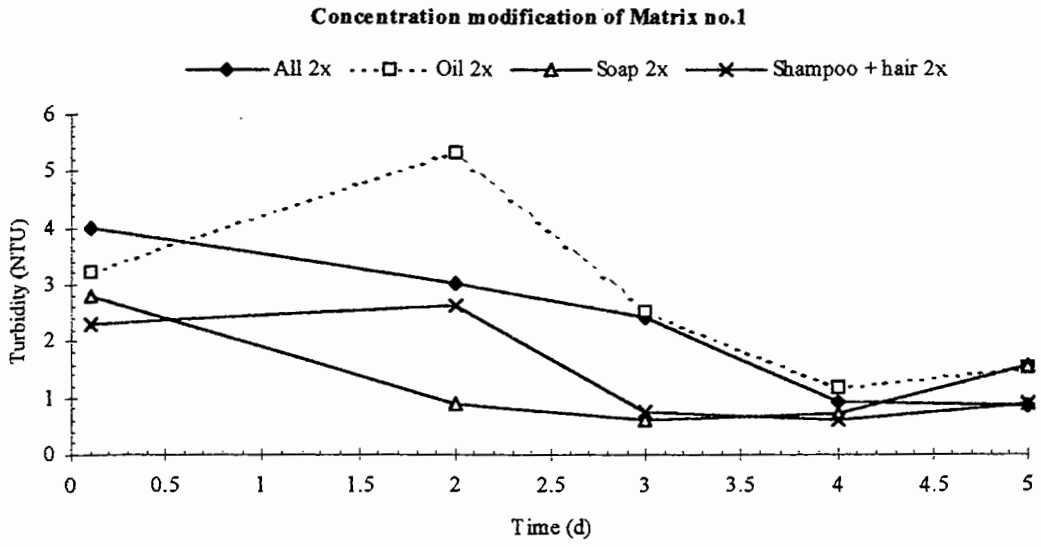


Figure C.5. Turbidity transient of air-mixed small-scale modified greywater batches (Matrix no.1) over 5 days.

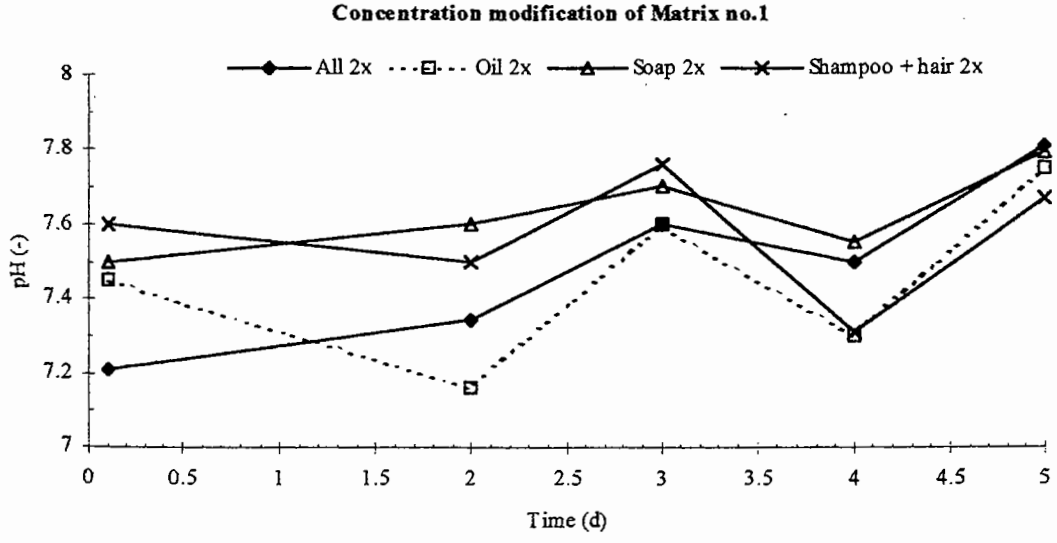


Figure C.6. pH transient of air-mixed small-scale modified greywater batches (Matrix no.1) over 5 days.

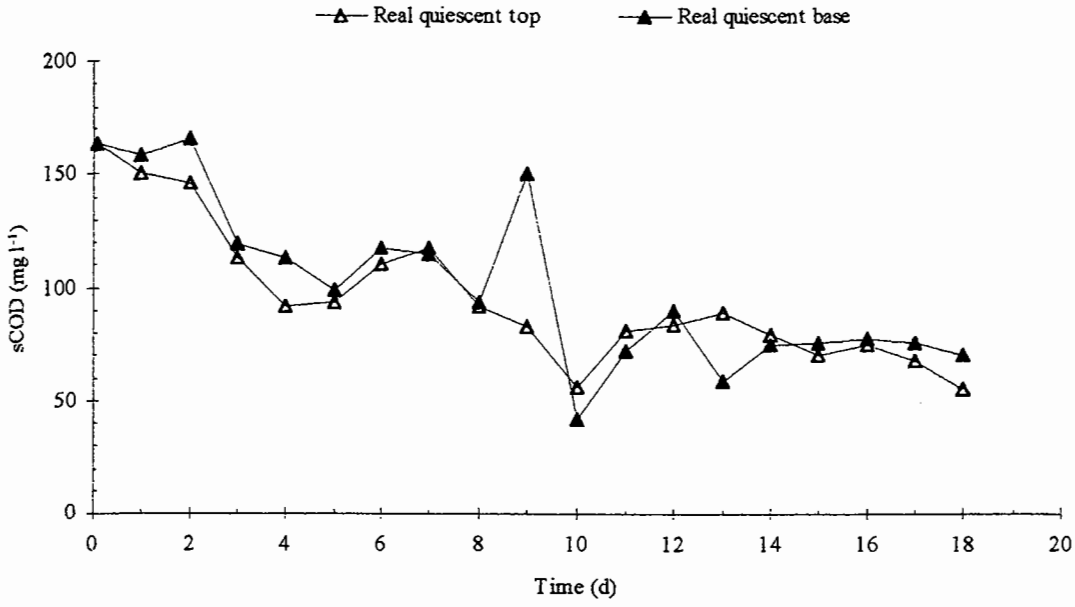


Figure C.7. sCOD degradation of real greywater stored under quiescent condition.

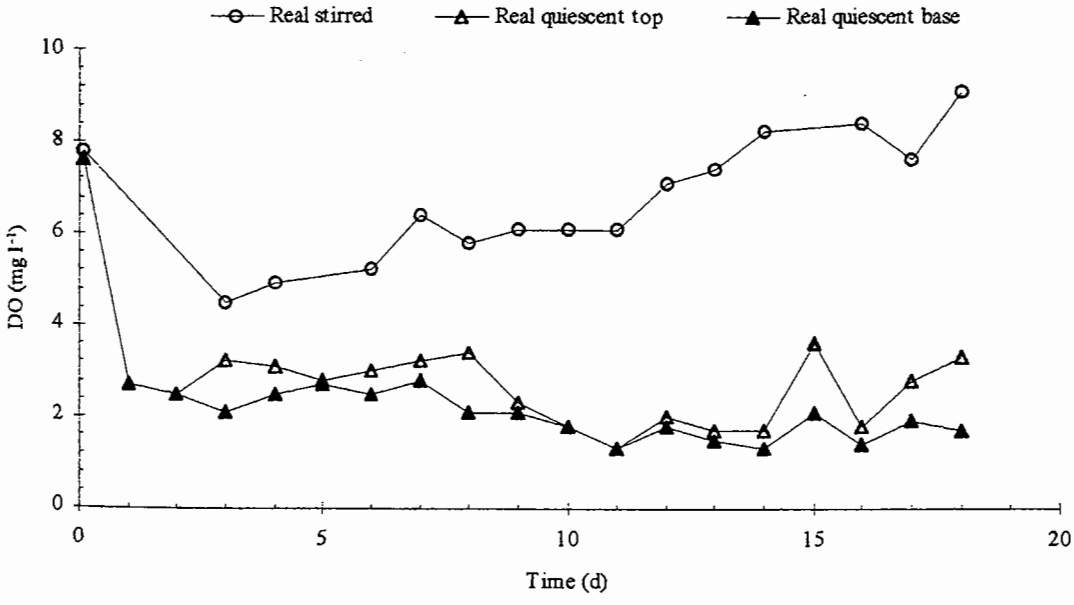


Figure C.8. DO transient of real greywaters stored under stirred or quiescent conditions.

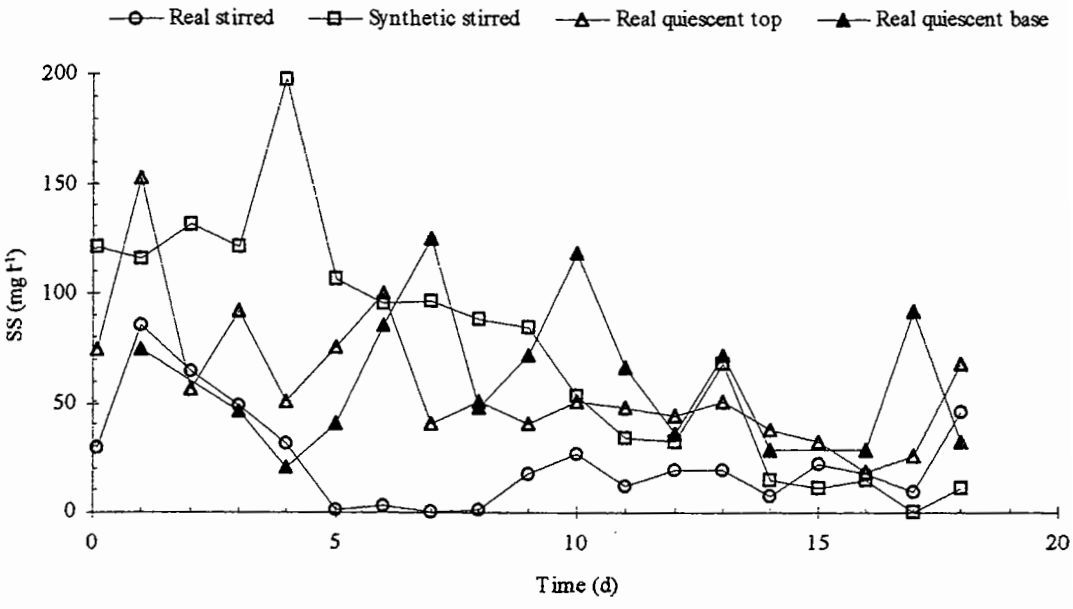


Figure C.9. SS transient of real and synthetic greywaters stored under stirred or quiescent conditions.

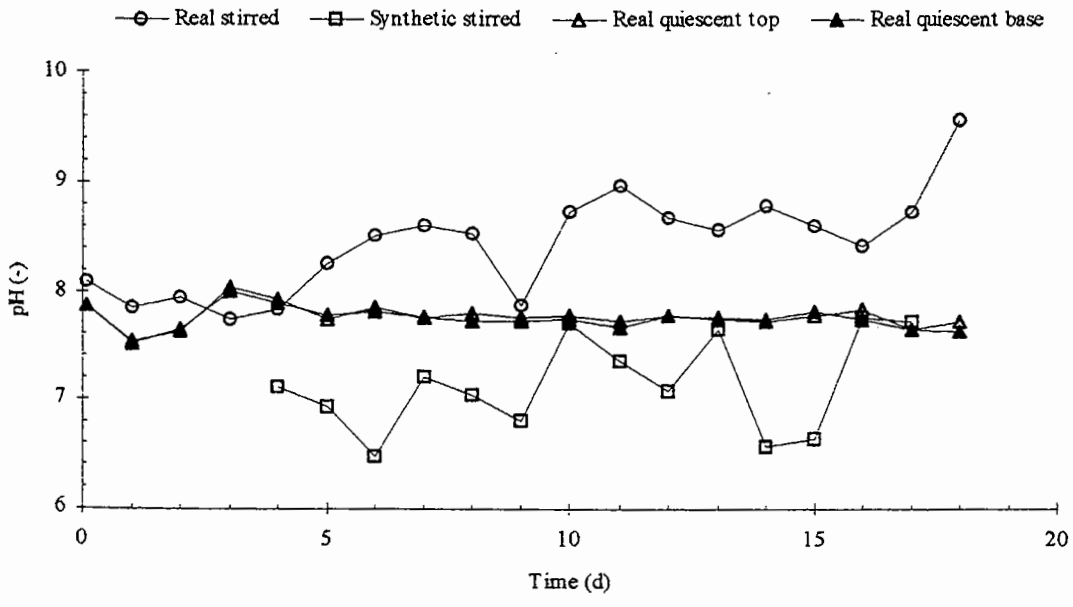


Figure C.10. pH transient of real and synthetic greywaters stored under stirred or quiescent conditions.

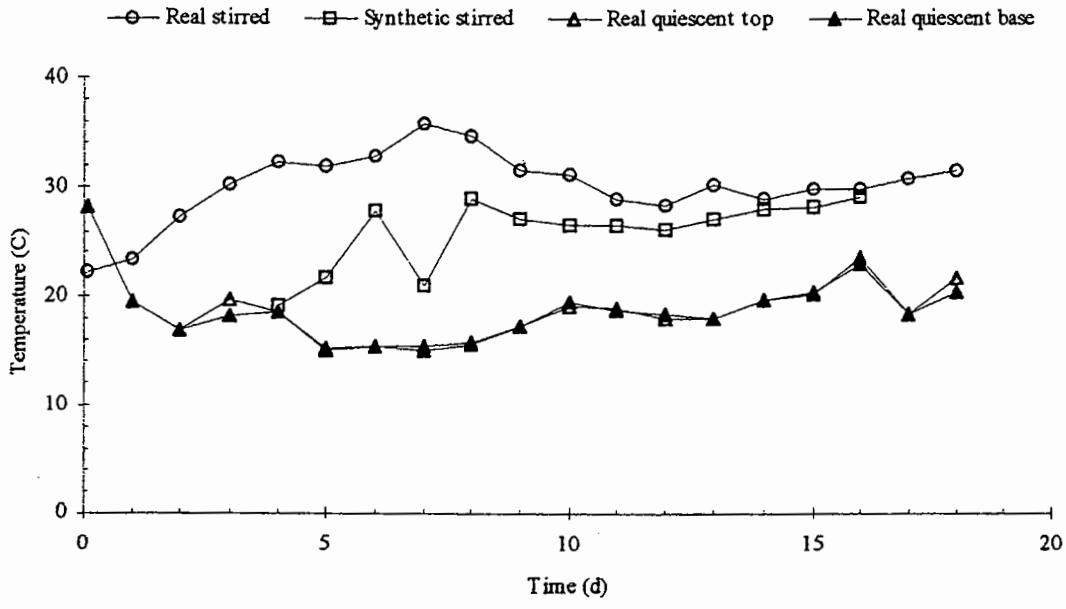


Figure C.11. Temperature transient of real and synthetic greywaters stored under stirred or quiescent conditions.

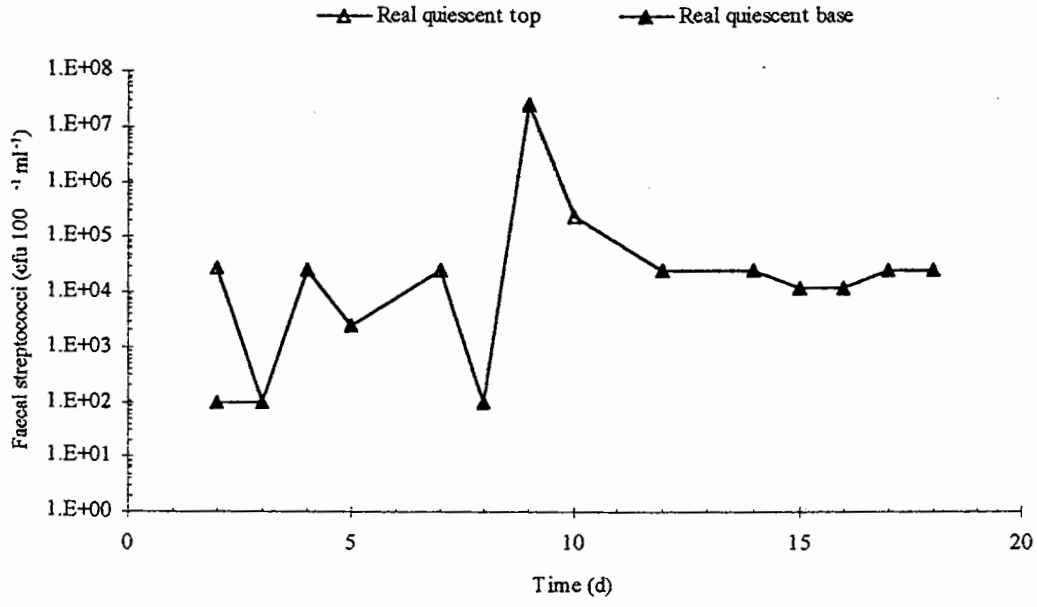


Figure C.12. Faecal streptococci transient of real greywater stored under quiescent condition.

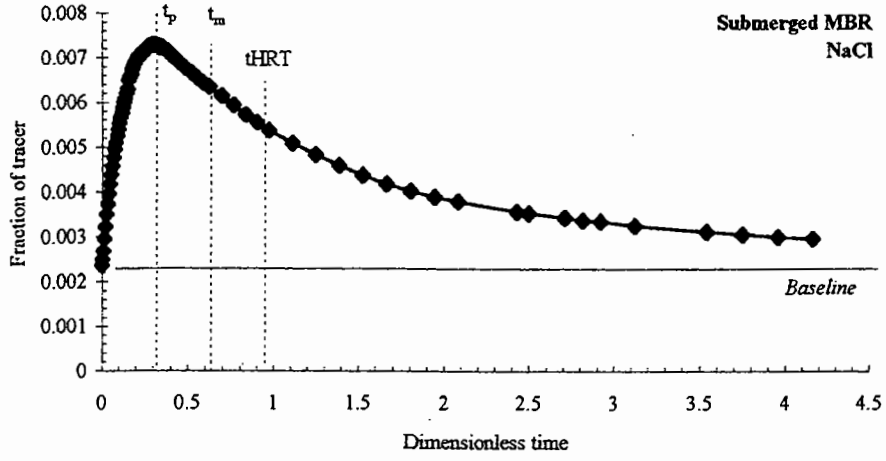


Figure D.1. The residence time distribution of the submerged MBR.

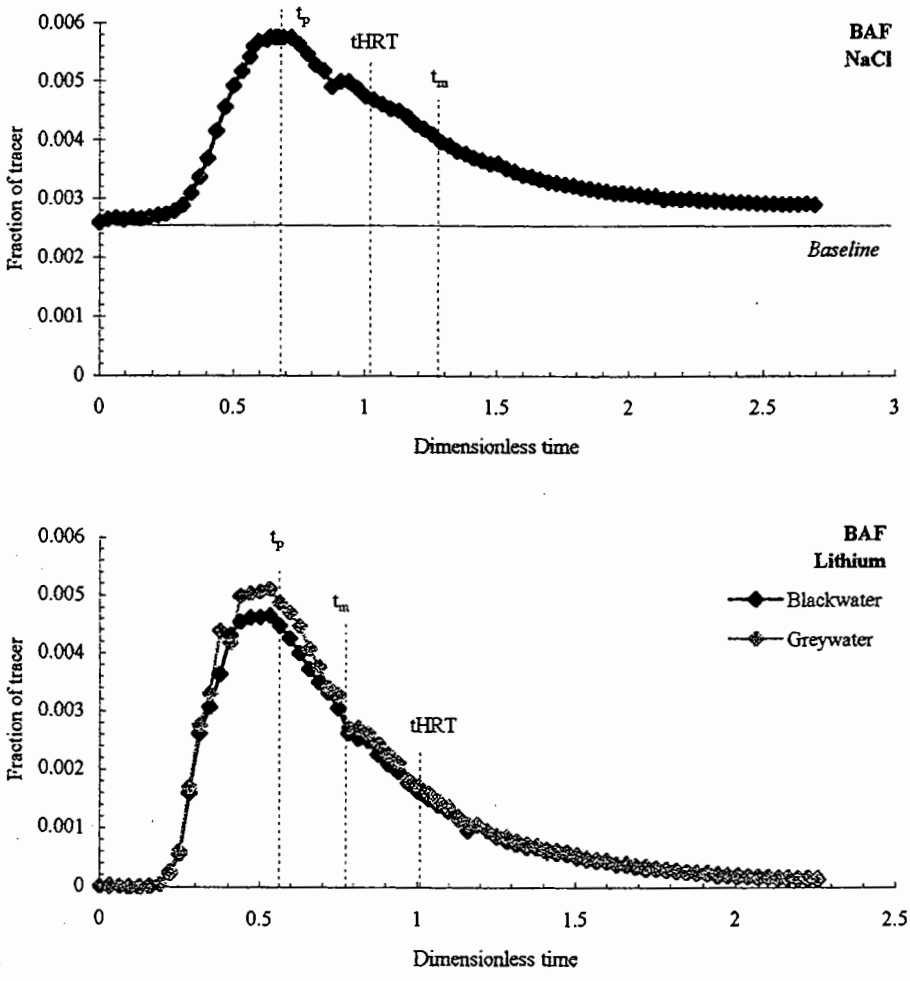


Figure D.2. The residence time distribution of the BAF.

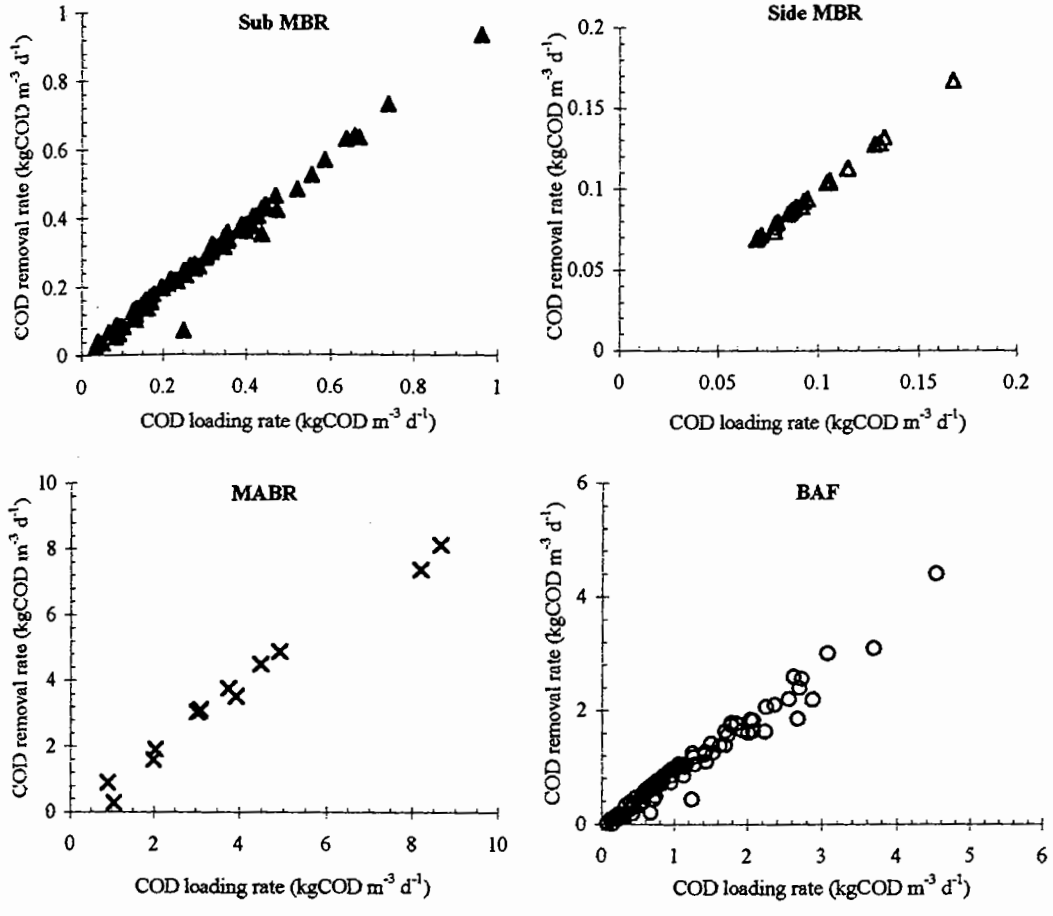


Figure D.3. Loading rate versus removal rate in terms of the COD.

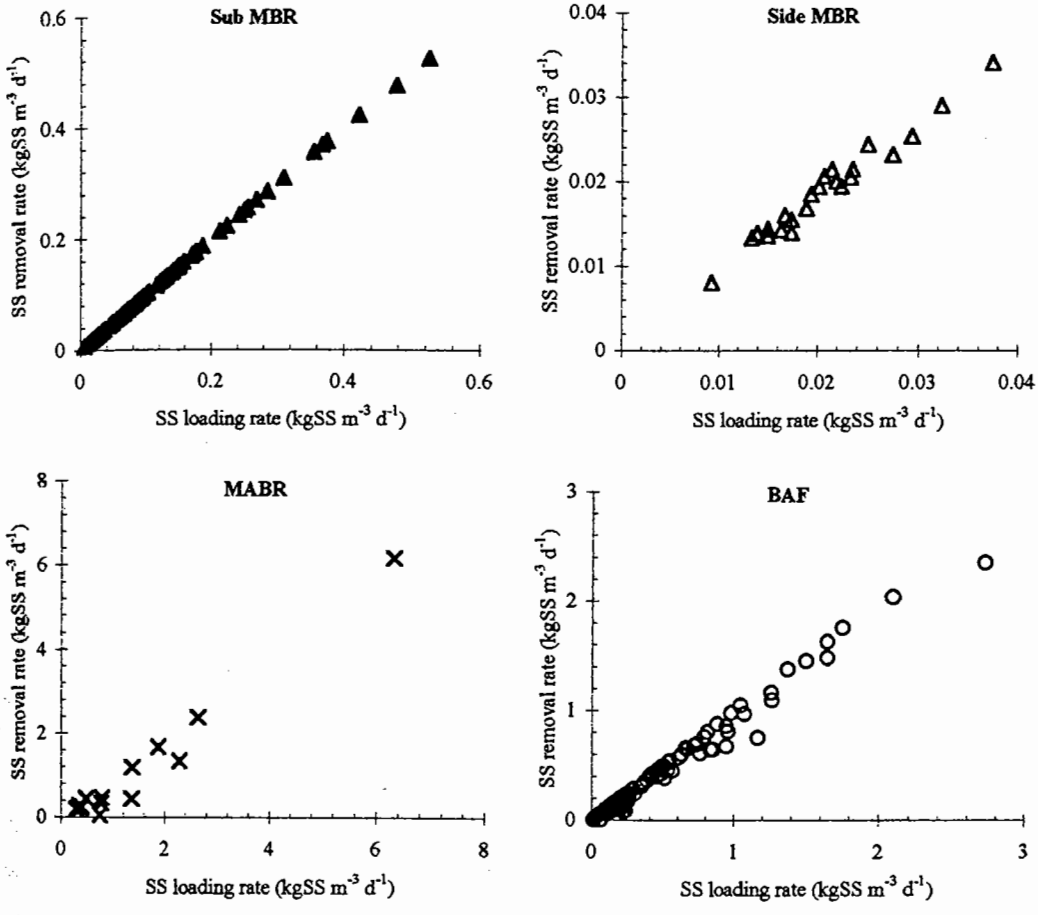


Figure D.4. Loading rate versus removal rate in terms of the SS.

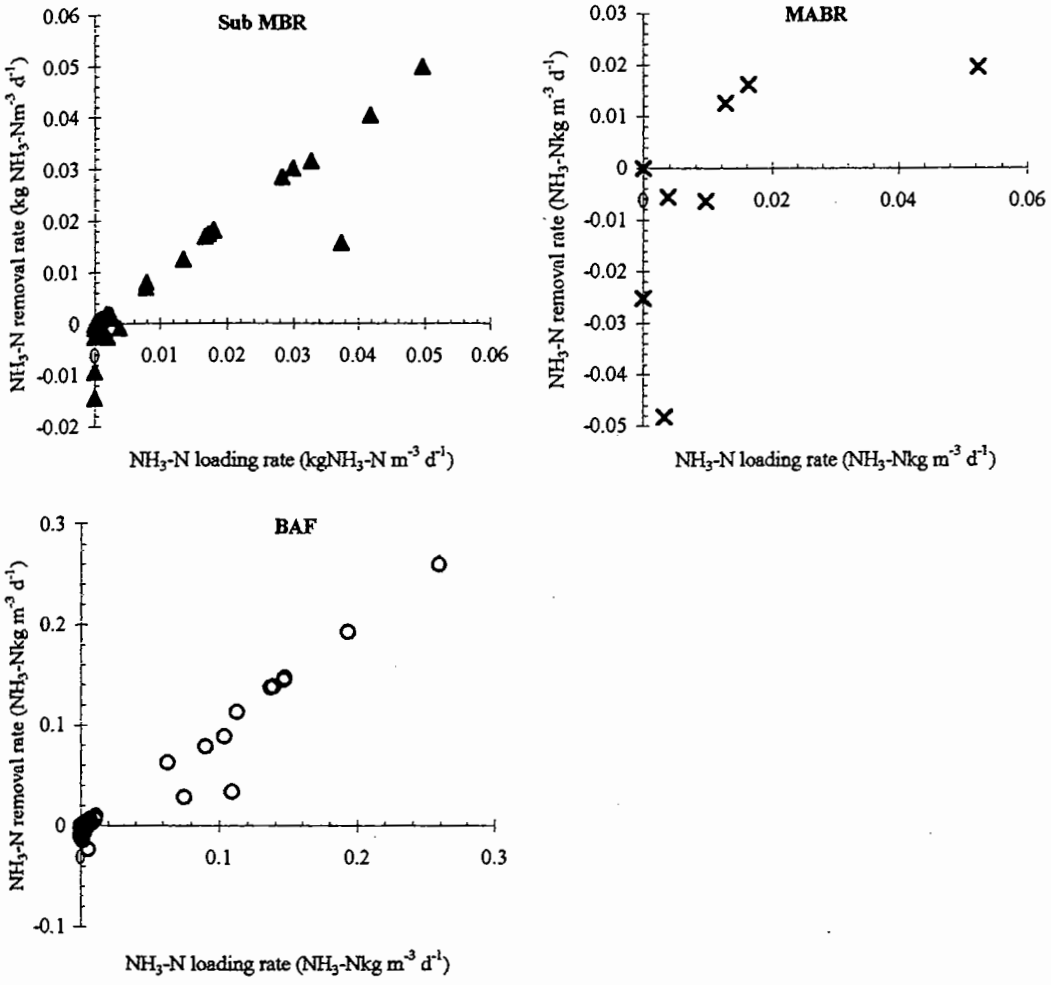


Figure D.5. Loading rate versus removal rate in terms of the ammonia.

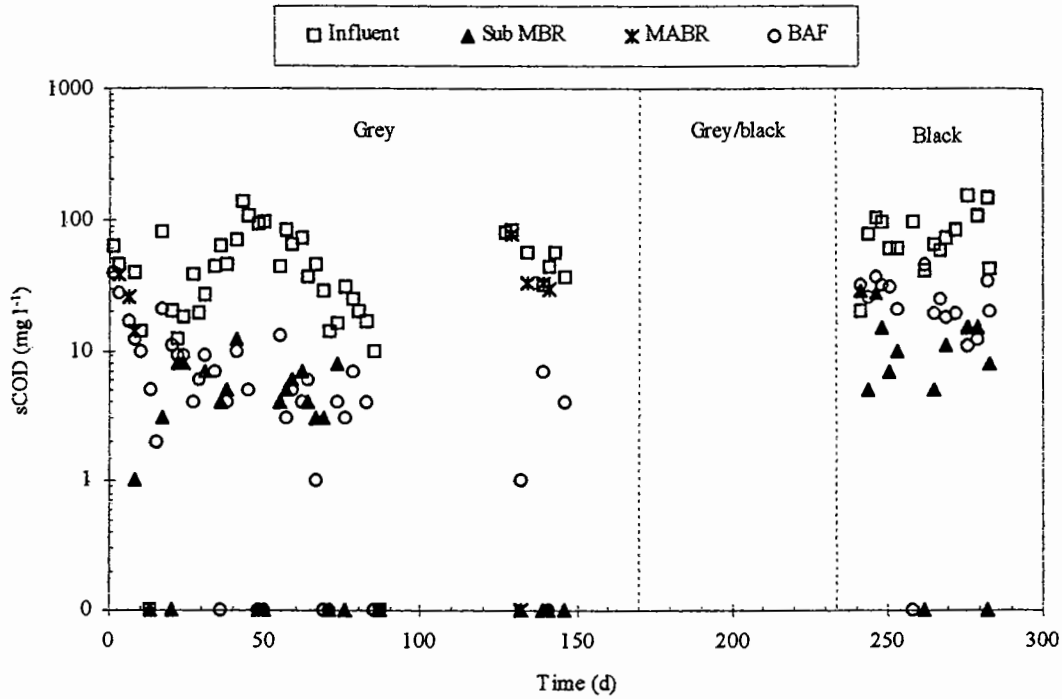


Figure D.8. sCOD transient during the steady-state trials.

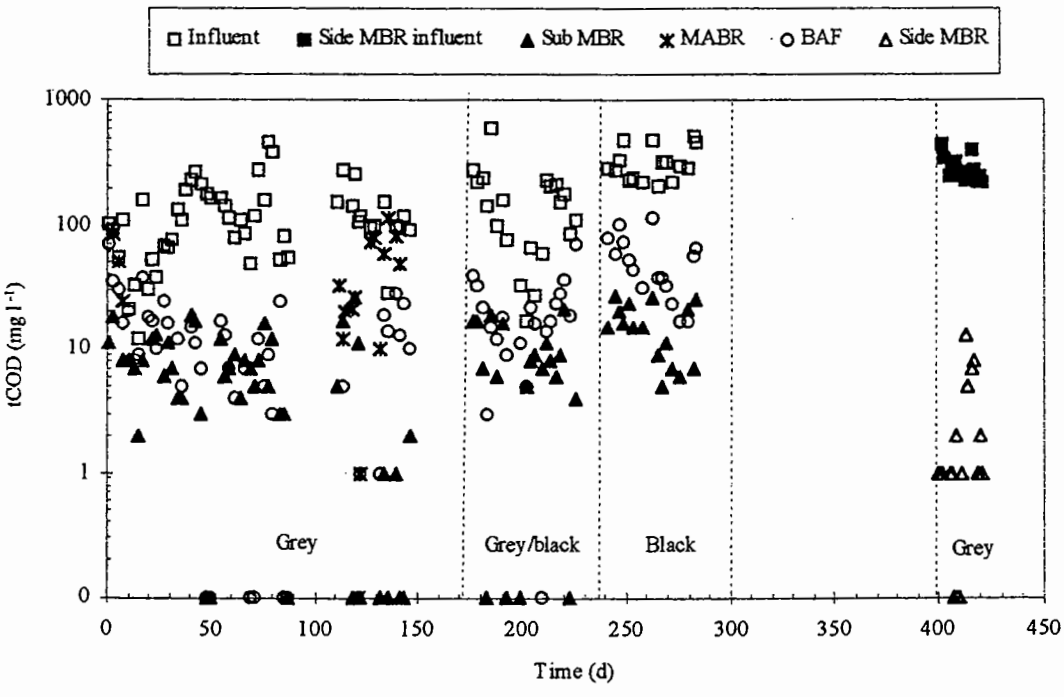


Figure D.9. tCOD transient during the steady-state trials.

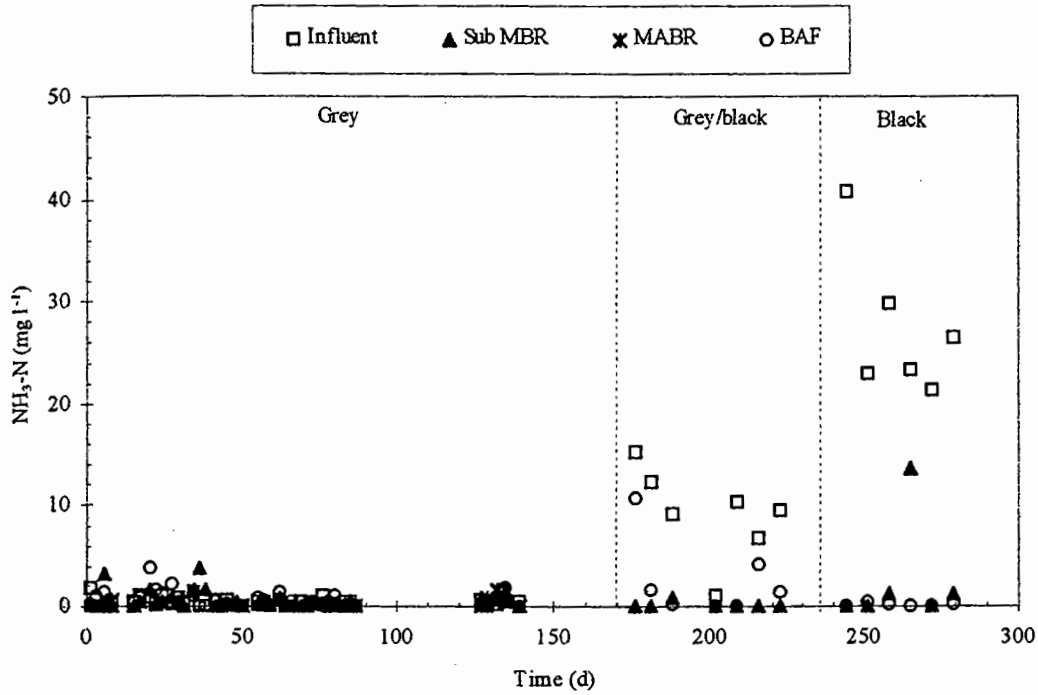


Figure D.10. Ammonia transient during the steady-state trials.

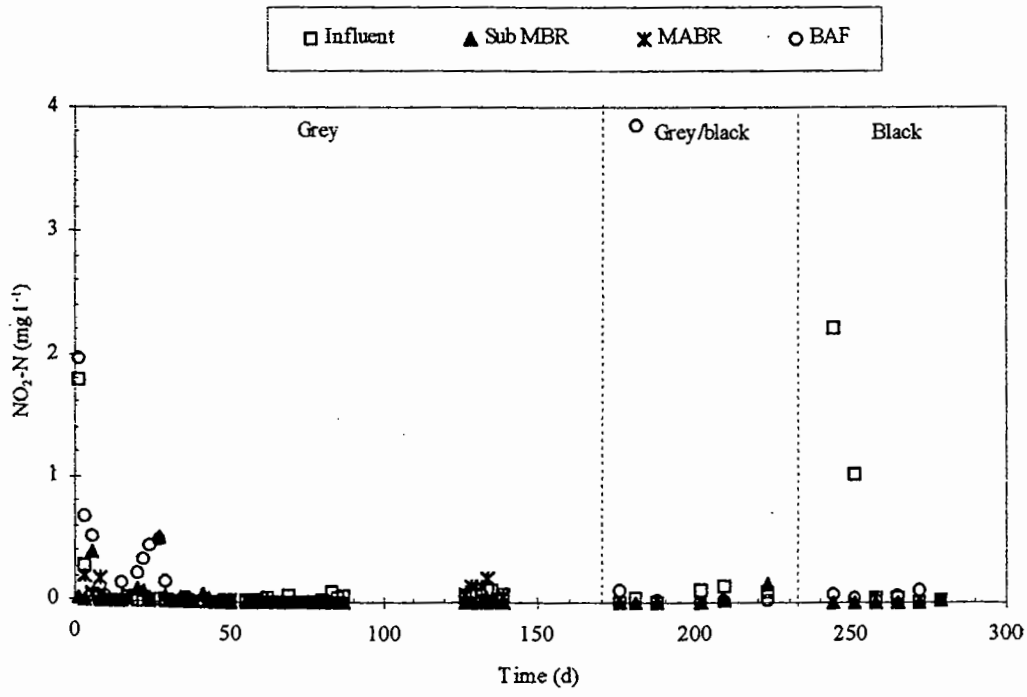


Figure D.11. Nitrite transient during the steady-state trials.

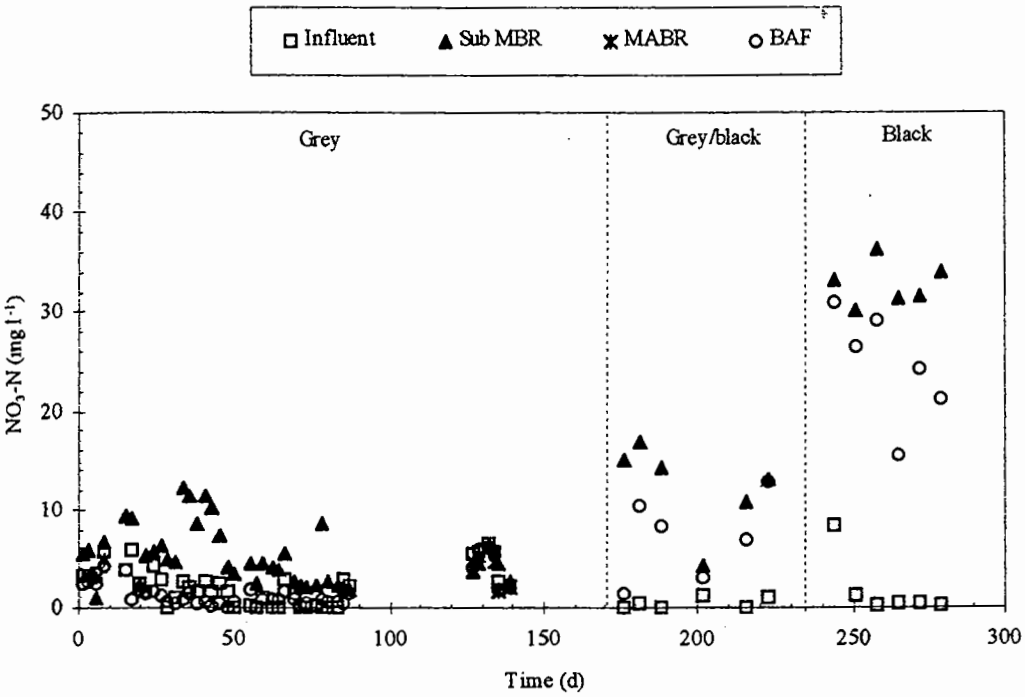


Figure D.12. Nitrate transient during the steady-state trials.

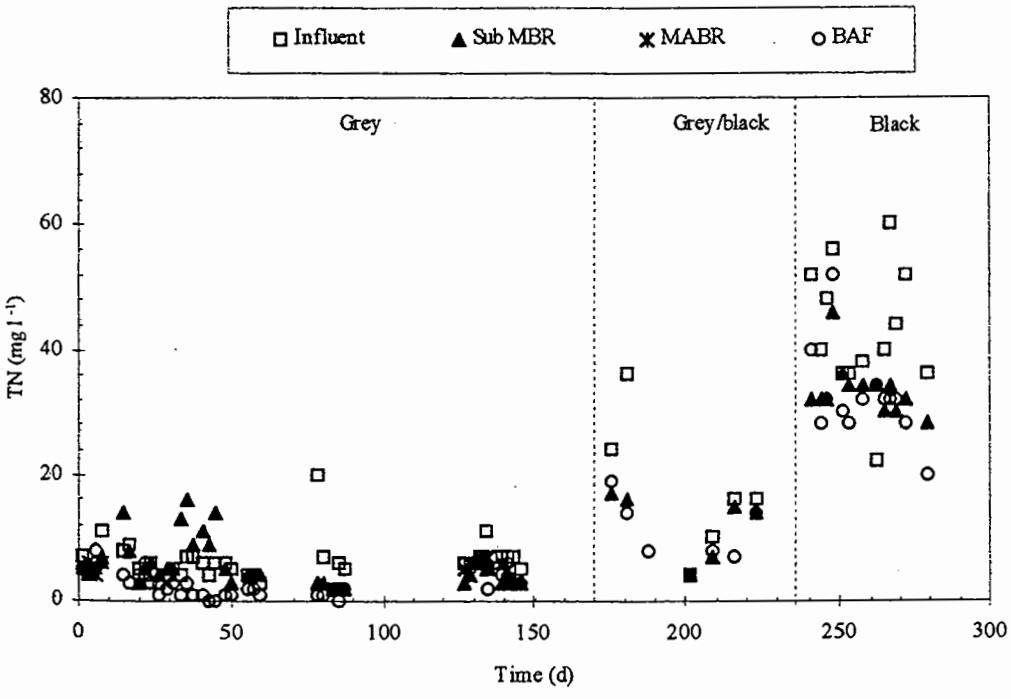


Figure D.13. Total nitrogen transient during the steady-state trials.

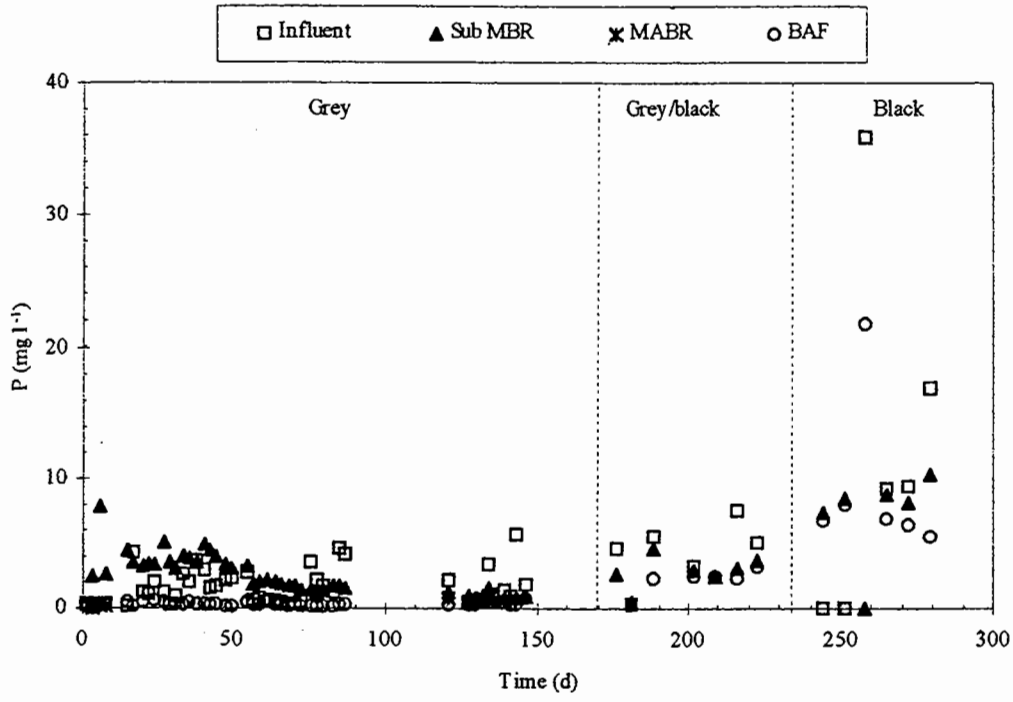


Figure D.14. Phosphorus transient during the steady-state trials.

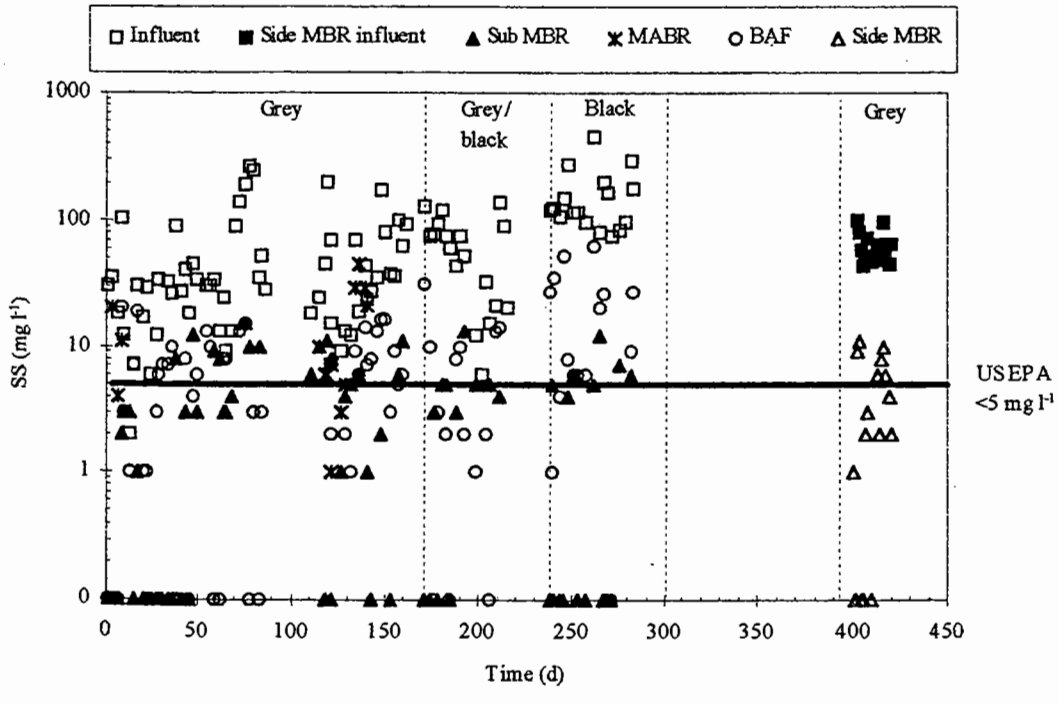


Figure D.15. SS transient during the steady-state trials showing the US EPA standard.

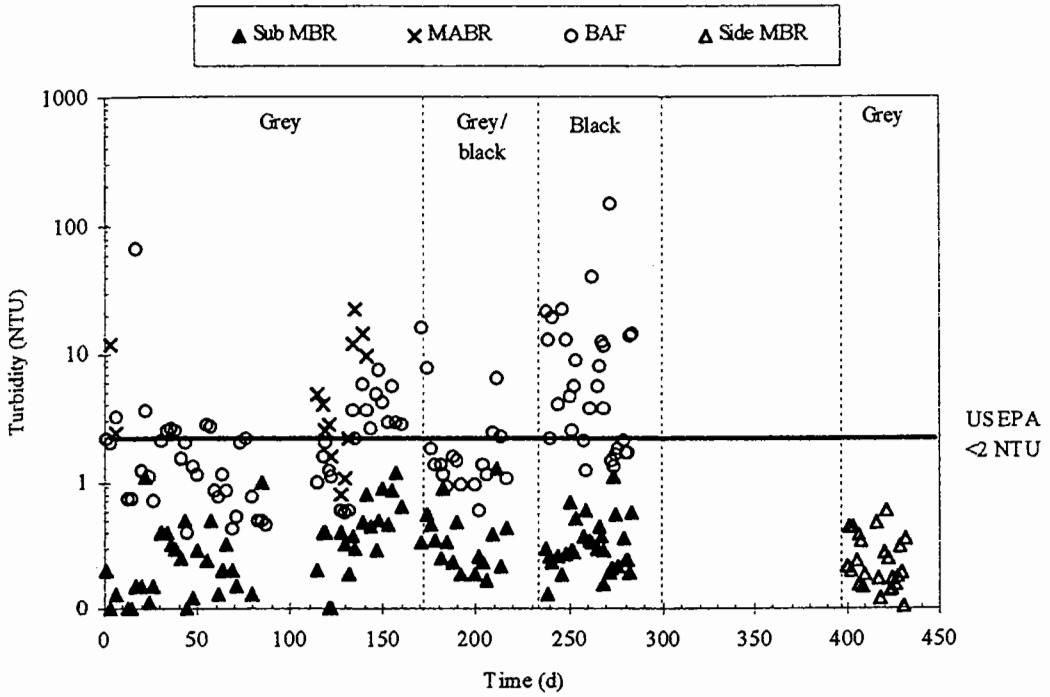


Figure D.16. Effluent turbidity during the steady-state trials showing the US EPA standard.

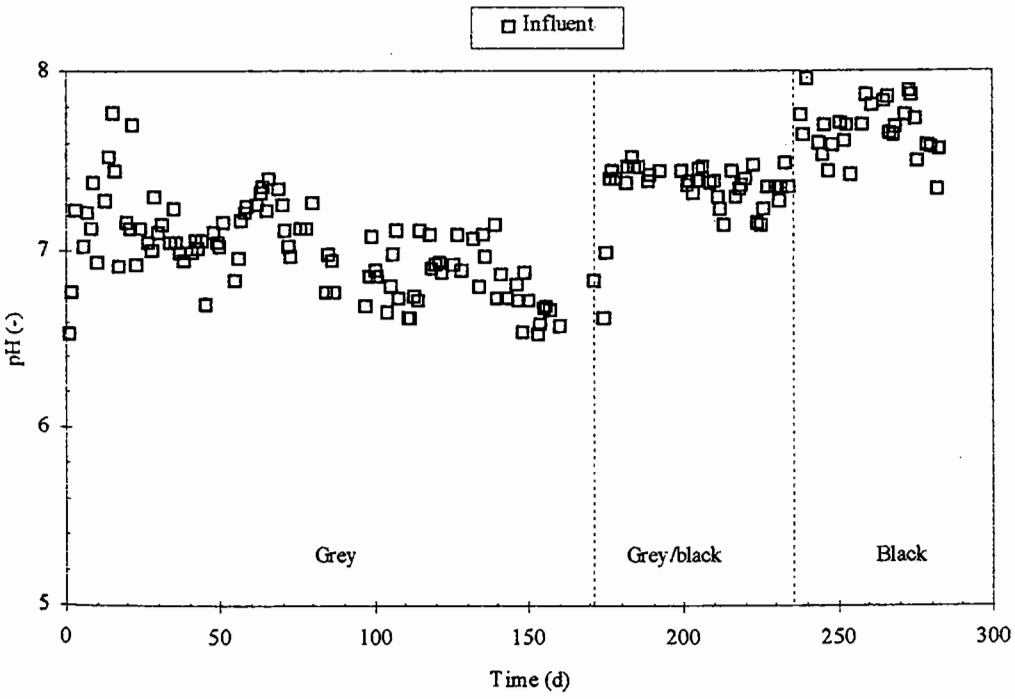


Figure D.17. Influent pH during the steady-state trials.

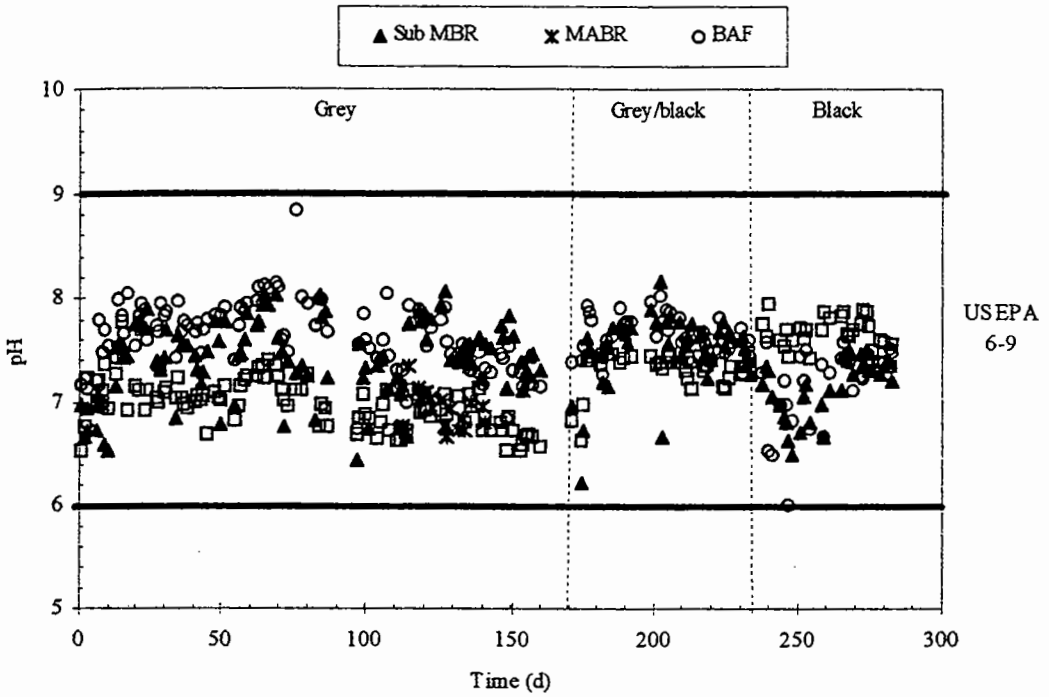


Figure D.18. Effluent pH transient during the steady-state trials showing the US EPA standard.

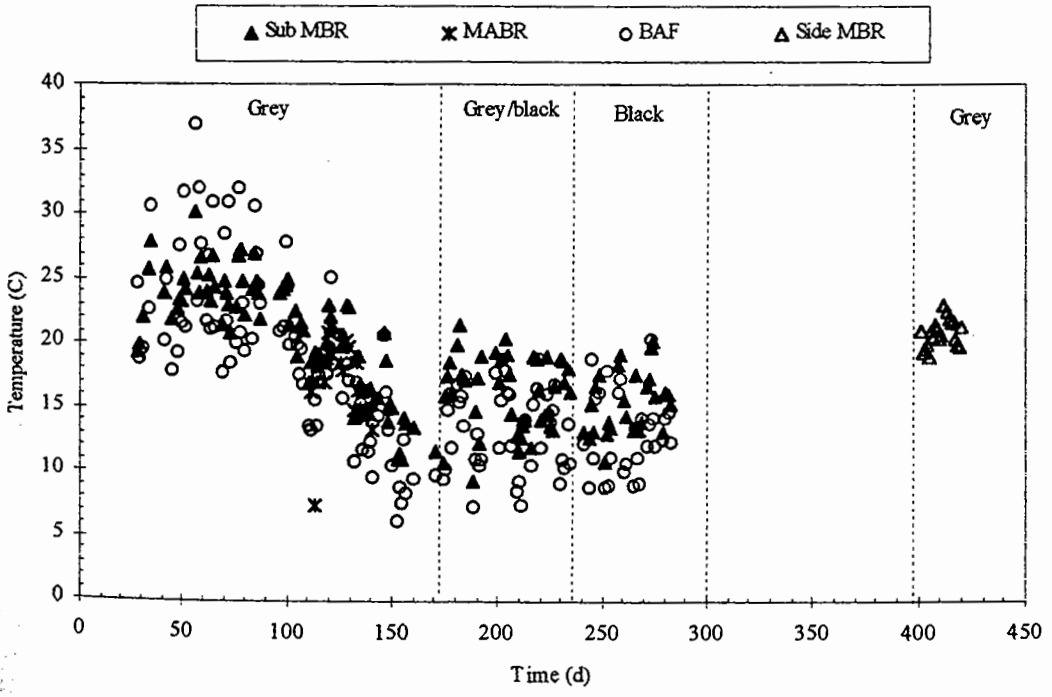


Figure D.19. Effluent temperature during the steady-state trials.

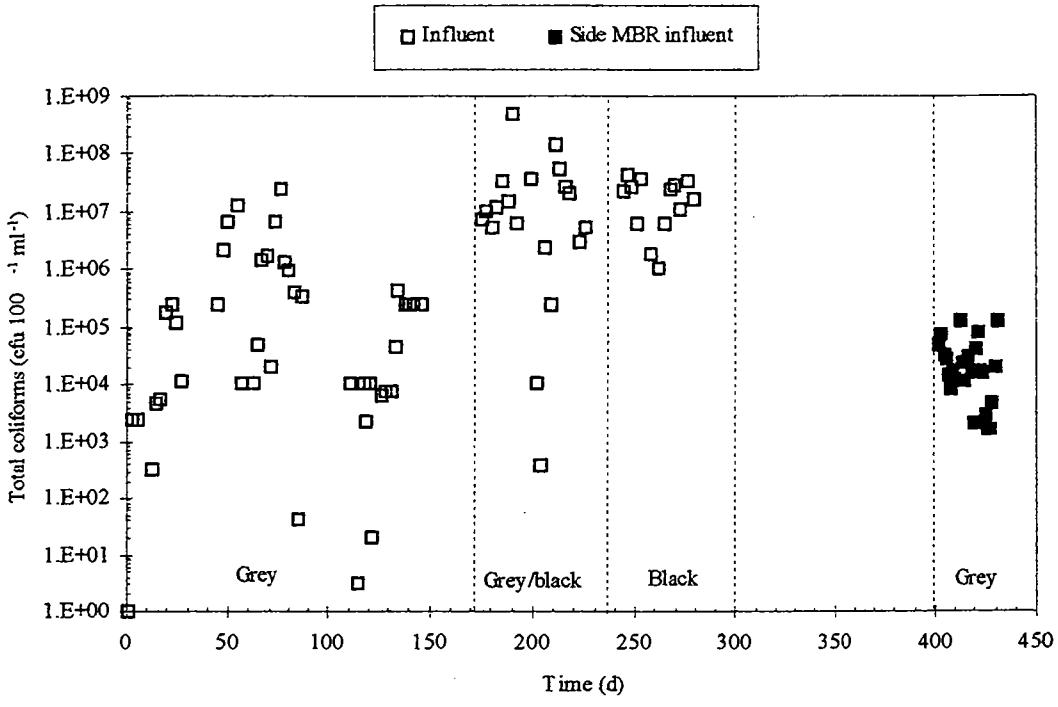


Figure D.20. Influent total coliform transient during the steady-state trials.

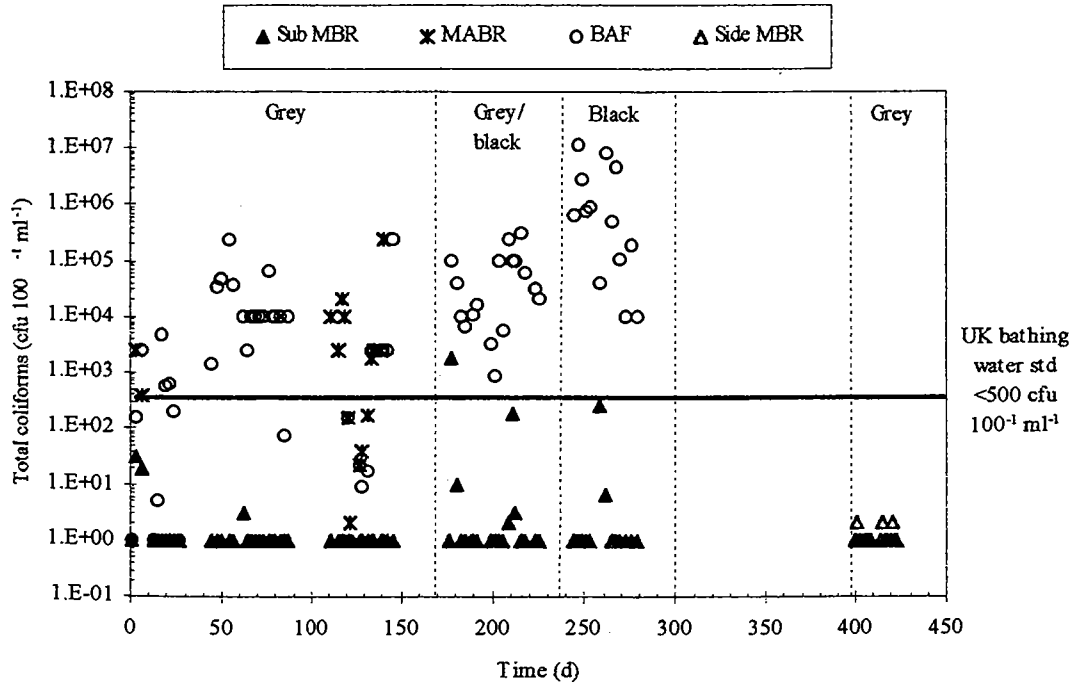


Figure D.21. Effluent total coliforms during the steady-state trials showing the UK bathing water standard.

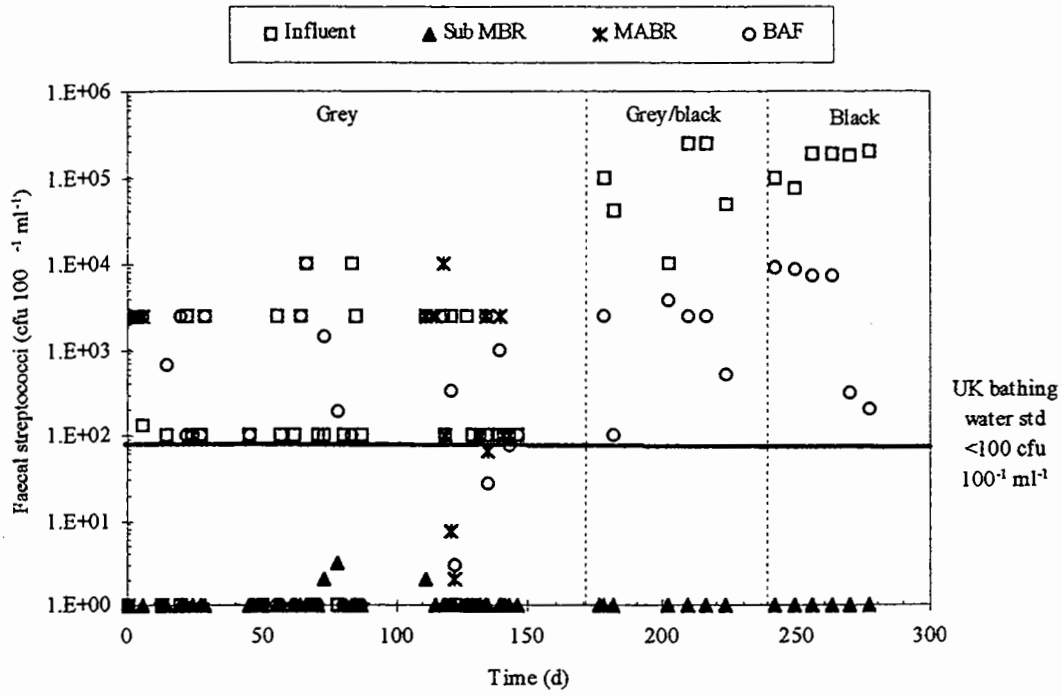


Figure D.22. Faecal streptococci transient during the steady-state trials showing the UK bathing water standard.

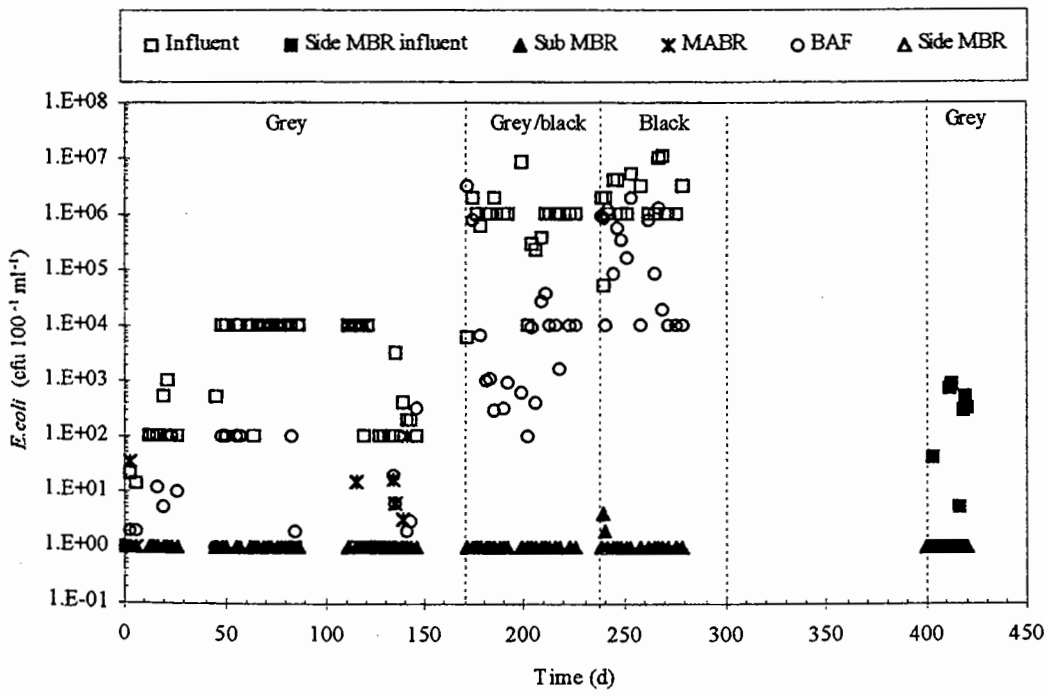


Figure D.23. *E.coli* transient during the steady-state trials.

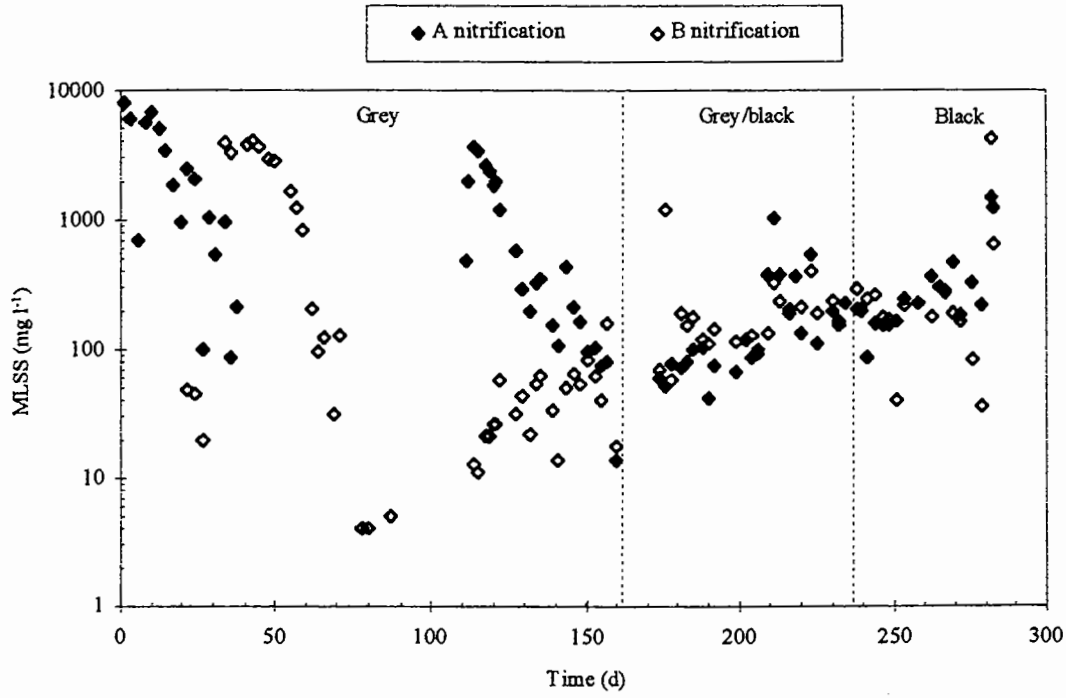


Figure D.24. MLSS in the submerged MBR nitrification units during the steady-state trials.

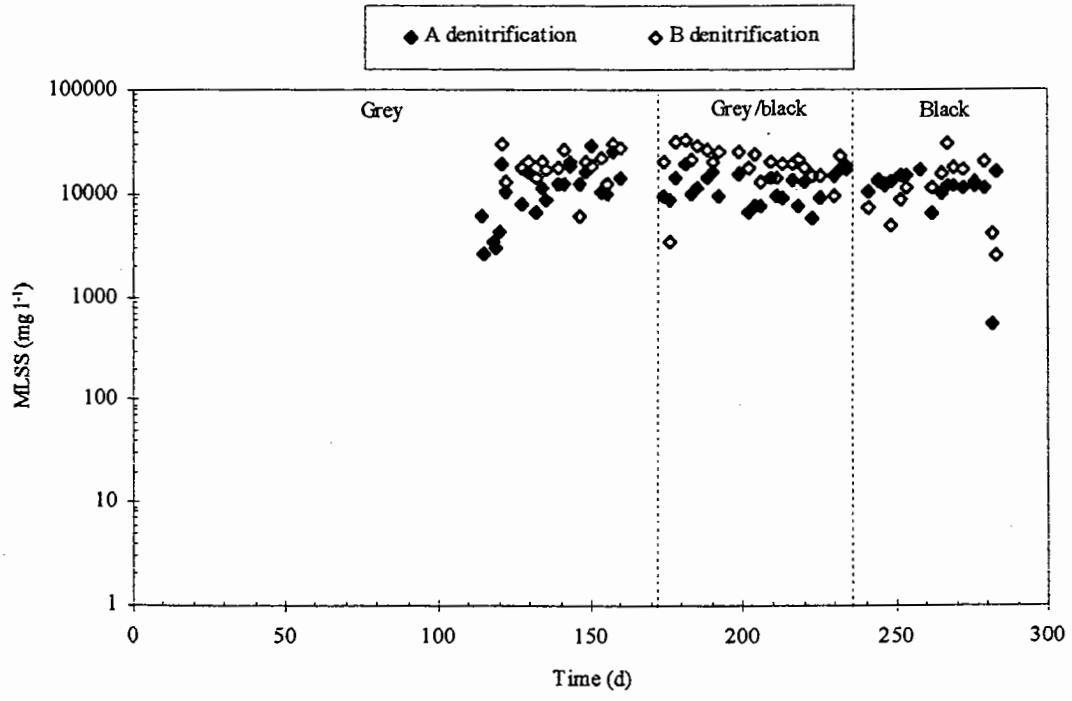


Figure D.25. MLSS in the submerged MBR denitrification units during the steady-state trials.

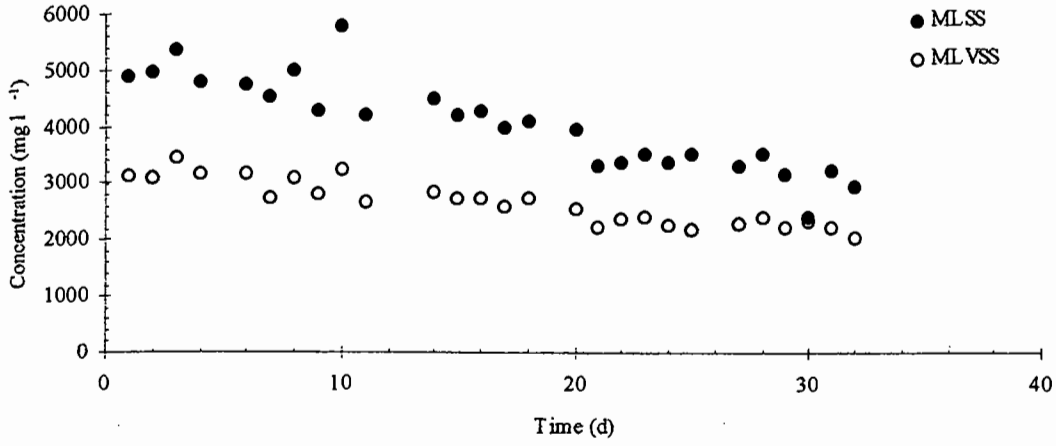


Figure D.26. MLSS and MLVSS of the side-stream MBR during the greywater trial.

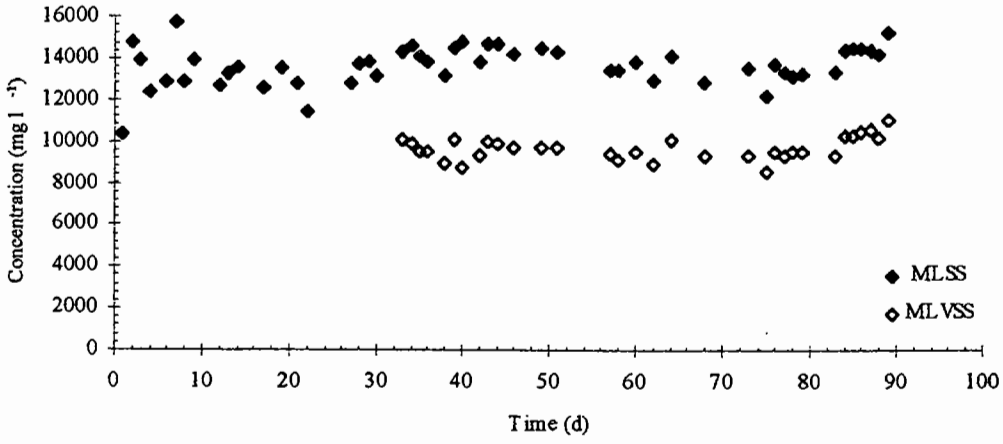


Figure D.27. MLSS and MLVSS of the modified submerged MBR unit A during the greywater trial.

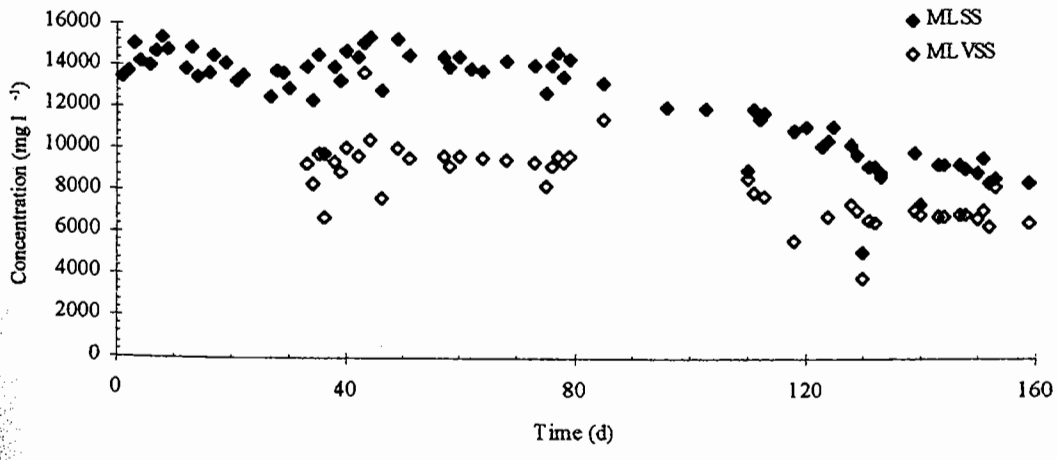


Figure D.28. MLSS and MLVSS of the modified submerged MBR unit B during the greywater trial.

Table D.1. Performance of the modified submerged MBR (unit B) during the steady-state operation with greywater. Mean ± standard deviation.

Parameter	Influent	Submerged MBR	Removal (%)
Flow ($m^3 d^{-1}$) [a]	-	0.138 ± 0.028	
OLR (kgBOD $m^{-3} d^{-1}$) [b]	-	0.67± 0.36	-
BOD_5 (mg l^{-1}) [b]	81 ± 44	1 ± 1	99 ± 2
tCOD (mg l^{-1}) [c]	177 ± 60	7 ± 7	96 ± 4
SS (mg l^{-1}) [c]	55 ± 25	2 ± 1	95 ± 6
Turbidity (NTU) [b]	-	0.2 ± 0.1	-
TN (mg l^{-1}) [d]	5.2 ± 2.7	1.8 ± 1.5	65 ± 24
P (mg l^{-1}) [d]	0.6 ± 0.2	0.4 ± 0.0	36 ± 16
pH (-) [b]	7.4 ± 0.2	7.7 ± 0.1	-
DO (mg l^{-1}) [b]	-	7.8 ± 1.0	-
Temperature (°C) [b]	27 ± 4	22 ± 4	-
Total coliforms (cfu 100^{-1} ml^{-1}) [d]	38936 ± 58440	1 ± 0	4 log
E.coli (cfu 100^{-1} ml^{-1}) [d]	745 ± 1492	1 ± 0	2 log
Faecal streptococci (cfu 100^{-1} ml^{-1}) [d]	7283 ± 11326	1 ± 1	3 log

[a] 82 days [b] 43 days [c] 20 days [d] 5 days

Table E.1. *Operational logbook during the steady-state trials.*

Day	Date	General	Sub MBR	MABR	BAF
Start-up	02.06.98	• Monopump delivered water at 6.86, 6.75 and 6.80 l/min averaging 408 l/h	• Start-up with activated sludge, recirculated and aerated for 24 h	• Start-up with 10% activated sludge, recirculated and aerated for 24 h	
	03.06.98	• Greywater (in 500 l): 16 g bar soap, 160 ml shampoo, ¼ tsp vegetable oil, pinch of hair and 300 ml tertiary effluent		• Foaming	
	04.06.98	• General check-up		• Foaming	• Start-up with greywater
	05.06.98	• General check-up	• Sampling	• Sampling	• Sampling
	06.06.98	• General check-up			
	07.06.98	• General check-up			
	08.06.98	• General check-up	• Sampling	• Early results showed no treatment > dismantled	• Sampling
	09.06.98	• General check-up	• Sampling • Level in chamber A -30 cm and chamber B -20 cm > feed flow rate too low • Mixing pump B leaks from pump head > disconnected		
	10.06.98	• General check-up	• Sampling • Feed pump changed to that of sidestream bioreactor > Marprene tubing changed from bore size 6.4 mm to 8.0 mm • Hair blocked tube connector on feed pump resulting in insufficient flow		• Sampling
	11.06.98	• General check-up • Feed had strange, oily smell (monopump leaks or has run dry?) > main feed tank	• Sampling • Header tank emptied due to oily smell	• Start-up with 10% activated sludge > leaks in probe holder > recirculation pump and feed pump switched off but oxygen	• Sampling

		emptied > fresh feed made		supply left on	
	12.06.98	• General check-up	• Sampling	• Sampling • Leakage on probe holder located and glued. Pumps switched back on.	• Sampling
	13.06.98	• General check-up		• Probe holder leaked > pumps switched off	
	14.06.98	• General check-up		• Silicone applied to leakage point on probe holder	• Hair blocked feed line
	15.06.98	• General check-up	• Sampling		• Sampling
	16.06.98	• General check-up	• Sampling		• Sampling
1	17.06.98	• General check-up • First day of steady state	• Sampling		• Sampling
2	18.06.98	• General check-up • Automatic feed system on	• Sampling		• Sampling
3	19.06.98	• General check-up	• Sampling	• Recirculation partially closed to improve biofilm formation	• Sampling
4	20.06.98	• General check-up			
5	21.06.98	• General check-up • Due to several blockages caused by hair in greywater, hair no longer used in recipe			
6	22.06.98	• General check-up	• Sampling • Temporarily no air supply	• Sampling	• Sampling
7	23.06.98	• General check-up	• Sampling	• Sampling	• Sampling • No flow > tubing tightness on pump head checked > flow recovered
8	24.06.98	• General check-up	• Sampling	• Sampling	• Sampling
9	25.06.98	• General check-up	• Sampling • Mixing pump B leaks from pump head due to worn seals > electrician contacted for	• Sampling • Dismantled as no treatment achieved	• Sampling

			disconnecting pump from control panel • Mixing in chamber A only	
10	26.06.98	• General check-up	• Sampling • Mixing in chamber A only	• Sampling
11	27.06.98	• General check-up	• Mixing in chamber A only	
12	28.06.98	• General check-up	• Mixing in chamber A only	
13	29.06.98	• General check-up	• Sampling • Mixing in chamber A only	• Sampling
14	30.06.98	• General check-up	• Sampling • Mixing in chamber A only	• Sampling
15	01.07.98	• General check-up	• Sampling • Mixing in chamber A only, level in A lower than in B (tubing partially blocked?)	• Sampling
16	02.07.98	• General check-up	• Sampling • Mixing in chamber A only, level in A -10 cm of B	• Sampling
17	03.07.98	• General check-up	• Sampling • Mixing in chamber A only, level in A lower than in B	• Sampling • Backwashed, afterwards effluent green (algae in pipes?)
18	04.07.98	• General check-up	• No mixing in A + B (mixing pump A leaks from pump head due to worn seals > electrician contacted for disconnecting pump from control box)	
19	05.07.98	• General check-up	• No mixing in chambers A + B	
20	06.07.98	• General check-up	• Sampling • No mixing in chambers A + B; header tank mixing on	• Sampling
21	07.07.98	• General check-up	• Sampling • No mixing in chambers A + B • Mixing pump A repaired (seal	• Sampling

			replaced) > electrician contacted > pump connected back to control box (to repair A sooner than B because of higher MLSS and thus possibly better operation)	
22	08.07.98	• General check-up	• Mixing in chamber A only	• Sampling • Backwashed after sampling
23	09.07.98	• General check-up	• Sampling • Mixing in chamber A only	• Sampling
24	10.07.98	• General check-up	• Sampling • Mixing in chamber A only • Mixing pump B repaired (seal replaced) and connected to control panel	• Sampling
25	11.07.98	• General check-up		
26	12.07.98	• General check-up		
27	13.07.98	• General check-up	• Sampling	• Sampling
28	14.07.98	• General check-up • Power cut (1.5 h)	• Sampling	• Sampling
29	15.07.98	• General check-up	• Sampling • Overflow blocked or had airlock resulting in flood. Effluent lines closed and feed pump switched off overnight.	• Sampling
30	16.07.98	• General check-up	• Both chambers fed with 250 ml of greywater • Effluent lines opened to allow mixed liquor level settle • Feed pump switched on at 100 rpm, then down to 90 rpm, overflow tubing cleaned with water and directed to drain	

			outside. Additional fitting attached to overflow to prevent further problems.		
31	17.07.98	• General check-up	• Sampling		• Sampling
32	18.07.98	• General check-up			
33	19.07.98	• General check-up			
34	20.07.98	• General check-up	• Sampling		• Sampling
35	21.07.98	• General check-up • Due to low COD in feed, greywater doubled in strength (in 500 l: 32 g bar soap, 320 ml shampoo ½ tsp vegetable oil and 600 ml tertiary effluent)	• Sampling	• Start-up with greywater • Foaming • Sampling	• Sampling
36	22.07.98	• General check-up	• Sampling	• Recirculating feed in loop (feed pump not on)	• Sampling
37	23.07.98	• General check-up	• Sampling • Chamber B valve to anaerobic chamber opened to improve mixing > foaming increased significantly > valve closed after 1 h • Additional sampling of sludge in anaerobic and aerobic units of chamber A for MLSS and MLVSS measurements for solids profile	• Recirculating feed in loop (feed pump not on)	• Sampling
38	24.07.98	• General check-up	• Sampling • B +10 cm membrane level (inlet to anaerobic chamber blocked overnight?) > tubing cleaned > flow restored	• Recirculating feed in loop (feed pump not on)	• Sampling
39	25.07.98	• General check-up		• Recirculating feed in loop (feed pump not on)	

40	26.07.98	• General check-up		• Recirculating feed in loop (feed pump not on)	
41	27.07.98	• General check-up	• Sampling • B level -20 cm	• Recirculating feed in loop (feed pump not on)	• Sampling • Very low flow (backwashed just before sampling?), which increased later
42	28.07.98	• General check-up	• Sampling • B level -20 cm	• Recirculating feed in loop (feed pump not on)	• Sampling
43	29.07.98	• General check-up	• Sampling	• Dismantled because of poor biofilm formation and no treatment (COD almost same as for influent)	• Sampling • Backwashed
44	30.07.98	• General check-up	• Sampling		• Sampling
45	31.07.98	• General check-up	• Sampling		• Sampling • Flow initially very low > tubing on feed pump tightened > flow recovered • Foaming
46	01.08.98	• General check-up			
47	02.08.98	• General check-up			
48	03.08.98	• General check-up	• Sampling		• Sampling • Foaming
49	04.08.98	• General check-up	• Sampling • Overflow from header tank connected to feed tank		• Sampling • Foaming
50	05.08.98	• General check-up	• Sampling		• Sampling • Foaming
51	06.08.98	• General check-up	• Sampling		• Sampling
52	07.08.98	• General check-up	• Sampling • Water around reactor A water jacket (leak?)		• Sampling
53	08.08.98	• General check-up			
54	09.08.98	• General check-up			

55	10.08.98	• General check-up	• Sampling	• Sampling
56	11.08.98	• General check-up	• Sampling	• Sampling
57	12.08.98	• General check-up	• Sampling	• Sampling
58	13.08.98	• General check-up	• Sampling • Chamber B covered with black polyethylene to prevent algae growth (some work still done about mixing in unit A so it still uncovered) • Chamber A water jacket emptied for leak detection	• Sampling
59	14.08.98	• General check-up	• Sampling	• Sampling
60	15.08.98	• General check-up	• Sampling	• Sampling
61	16.08.98	• General check-up		
62	17.08.98	• General check-up		
63	18.08.98	• General check-up	• Sampling	• Sampling
64	19.08.98	• General check-up	• Sampling	• Sampling
65	20.08.98	• General check-up	• Sampling	• Sampling
66	21.08.98	• General check-up	• Sampling	• Sampling
67	22.08.98	• General check-up	• Chamber A water jacket emptied because of continuous leak	
68	23.08.98	• General check-up		
69	24.08.98	• General check-up	• Sampling	• Sampling
70	25.08.98	• General check-up	• Sampling	• Sampling
71	26.08.98	• General check-up	• Sampling	• Sampling
72	27.08.98	• General check-up	• Sampling	• Sampling
73	28.08.98	• General check-up	• Sampling	• Sampling
74	29.08.98	• General check-up	•	
75	30.08.98	• General check-up	•	
76	31.08.98	• General check-up	• Sampling	• Sampling

					• Backwashed for 1 h before sampling, afterwards effluent greenish
77	01.09.98	• General check-up	• Sampling		• Sampling
78	02.09.98	• General check-up	• Sampling		• Sampling
79	03.09.98	• General check-up	• Sampling		• Sampling
80	04.09.98	• General check-up	• Sampling		• Sampling
81	05.09.98	• General check-up			
82	06.09.98	• General check-up			
83	07.09.98	• General check-up	• Sampling		• Sampling
84	08.09.98	• General check-up	• Sampling		• Sampling
85	09.09.98	• General check-up	• Sampling		• Sampling
86	10.09.98	• General check-up	• Sampling		• Sampling
87	11.09.98	• General check-up	• Sampling		• Sampling
88	12.09.98	• General check-up			
89	13.09.98	• General check-up			
90	14.09.98	• No sampling during this week - greywater prepared by technicians			
91	15.09.98	• No sampling			
92	16.09.98	• No sampling			
93	17.09.98	• No sampling			
94	18.09.98	• No sampling			
95	19.09.98	• No sampling			
96	20.09.98	• No sampling			
97	21.09.98	• General check-up	• Sampling		• Sampling
98	22.09.98	• General check-up	• Sampling • No mixing in header tank		• Sampling
99	23.09.98	• General check-up	• Sampling • No mixing in header tank	• Oxygen line checked for leakages with Teepol > none detected > water recirculated in the loop	• Sampling

100	24.09.98	• General check-up	• Sampling • No mixing in header tank	• Oxygen leak located > module emptied of water • Valve on oxygen regulator came off > taken to office to have its condition evaluated/ repaired or exchanged > to find another regulator from workshop/welding group	• Sampling
101	25.09.98	• General check-up • Power cut (5 h)	• Sampling • No mixing in header tank and A		• Sampling
102	26.09.98	• General check-up	• No mixing in header tank and A		
103	27.09.98	• General check-up	• No mixing in header tank and A		
104	28.09.98	• General check-up	• Sampling • No mixing in header tank and A		• Sampling
105	29.09.98	• General check-up	• Sampling • No mixing in header tank • Mixing in A	• Start-up by recirculating 10% sludge (collected from Cotton Valley same day; 10 l of sludge kept in container for feeding rig and fed daily with raw primary sewage influent)	• Sampling
106	30.09.98	• General check-up	• Sampling • No mixing in header tank	• Diluted sludge recirculated in loop	• Sampling
107	01.10.98	• General check-up	• Sampling • No mixing in header tank	• Diluted sludge recirculated in loop	• Sampling
108	02.10.98	• General check-up	• Sampling • No mixing in header tank	• 200 ml of diluted sludge injected to loop and recirculated	• Sampling
109	03.10.98	• General check-up	• No mixing in header tank	• 200 ml of diluted sludge injected to loop and	

				recirculated	
110	04.10.98	• General check-up	• No mixing in header tank	• 200 ml of diluted sludge injected to loop and recirculated • On-line with greywater supply at 10 rpm	
111	05.10.98	• General check-up	• Sampling • Due to mixing problems in A, effluent from that chamber analysed separately to see any differences between it and effluent in general • No mixing in header tank	• Though pH was unlikely to be problem for operation in this case, decided to try controlling pH of greywater so as to optimise the process • pH probe not properly calibrated > to recalibrate	• Sampling
112	06.10.98	• General check-up	• Sampling • No mixing in header tank • Chamber A emptied as poor mixing thought to be resulting from unattached air stone. Air stone in its place. Sludge emptied into barrel and completely mixed after sample of sludge taken. • To monitor MLSS and MLVSS in all four chambers. • Sludge fully mixed and A back on-line. • Sampling effluent A for COD, SS and turbidity measurements • Slight foaming	• Dosing pump on-line with 0.2 M sodium bicarbonate increased pH to 7.99	• Sampling
113	07.10.98	• General check-up • Automatic feed system not on > float switch had not activated > feed system turned on to manual and feed tank	• Sampling • No mixing in header tank	• Sampling • pH controlled with 0.2 M sodium bicarbonate	• Sampling

		topped up (ratio 1 greywater : 4 water) > electrician contacted > actuated float switch operation checked in water > does not work • Electrician checked power connection > scale on float switch brushed off > automatic feed system worked and turned on to automatic > from now on necessary to brush feed tank and float switch regularly (part of general check-up)			
114	08.10.98	• General check-up	• Sampling • No mixing in header tank	• Sampling	• Sampling
115	09.10.98	• General check-up	• Sampling • No mixing in header tank	• Sampling	• Sampling
116	10.10.98	• General check-up	• No mixing in header tank		
117	11.10.98	• General check-up	• No mixing in header tank		
118	12.10.98	• General check-up	• Sampling • No mixing in header tank	• Sampling	• Sampling
119	13.10.98	• General check-up • Samples of scale on tank walls	• Sampling • No mixing in header tank	• Sampling	• Sampling
120	14.10.98	• General check-up	• Sampling • No mixing in header tank	• Sampling • Recirculation flow rate turned down to improve biofilm growth	• Sampling
121	15.10.98	• General check-up • Samples of scale on tank walls	• Sampling • No mixing in header tank	• Sampling	• Sampling
122	16.10.98	• General check-up • Monopump delivered greywater at 6.15 l/min, 6.15	• Sampling • No mixing in header tank	• Sampling	• Sampling

		l/min and 6.17 l/min, averaged 369 l/h • Samples of scale on tank walls			
123	17.10.98	• General check-up • Samples of scale on tank walls	• No mixing in header tank		
124	18.10.98	• General check-up	• No mixing in header tank		
125	19.10.98	• General check-up • T-junction on mixing pump in feed tank removed to prevent scale building up on tank walls	• No mixing in header tank		
126	20.10.98	• General check-up	• Sampling • No mixing in header tank	• Sampling	• Sampling
127	21.10.98	• General check-up	• Sampling • No mixing in header tank	• Sampling	• Sampling
128	22.10.98	• General check-up • Automatic feed system keeps topping up too much > mixing pump causes too much turbulence to activate float switch	• Sampling • No mixing in header tank	• Sampling • pH probe recalibrated and buffer concentration increased from 0.2 M to 1.2 M sodium bicarbonate to increase effect of each dose	• Sampling
129	23.10.98	• General check-up	• Sampling • No mixing in header tank	• Sampling	• Sampling
130	24.10.98	• General check-up	• No mixing in header tank		
131	25.10.98	• General check-up	• Mixing pump A leaked > no mixing in A • No mixing in header tank		
132	26.10.98	• General check-up	• Sampling • No mixing in header tank • Mixing pump A leakage point in plastic valve, which was removed > mixing pump A on after sampling	• Sampling	• Sampling
133	27.10.98	• General check-up	• Sampling	• Sampling	• Sampling

		• Feed system switches on at any time and not only when required (feed tank half full) > probably due to excessive motion of mixing pump that prevents scale formation but activates float switch too often > switched from auto to manual for better control and to prevent wasting feed • All pipes checked but no real scale observed > scaling problem specific to tank • Tank connector to blackwater tank broken > awaiting dry weather to determine best course of action	• No mixing in header tank	• Oxygen cylinder needs changing	• Manually backwashed, media loss significant >possibly media trapped in pipes and prevents backwashing
134	28.10.98	• General check-up • Feed system on manual due to poor float switch operation	• Sampling • No mixing in header tank	• Sampling • Oxygen cylinder changed	• Sampling
135	29.10.98	• General check-up • Feed system on manual due to poor float switch operation	• Sampling • No mixing in header tank	• Sampling • pH buffer changed from 1 M to 0.5 M sodium bicarbonate (high concentration of carbon in bioreactor may result in high COD in effluent if unit does not remove COD effectively)	• Sampling
136	30.10.98	• General check-up • Feed system on manual due to poor float switch operation	• Sampling • No mixing in header tank	• Sampling • Oxygen cylinder seemed empty > valve on cylinder fully opened > 170 bar left	• Sampling
137	31.10.98	• General check-up • Feed system on manual due to poor float switch operation	• No mixing in header tank		

138	01.11.98	• General check-up • Feed system on manual due to poor float switch operation	• No mixing in header tank		
139	02.11.98	• General check-up • Feed system on manual due to poor float switch operation • Soft soapy scale on walls of greywater concentrate tank, small volume of greywater foams	• Sampling • No mixing in header tank	• Sampling	• Sampling
140	03.11.98	• General check-up • Feed system on manual due to poor float switch operation	• Sampling • No mixing in header tank	• Sampling • Suspected leak in module > one side of module looks clean, another dirty > backwashing tried but pump did not work > options: a) to backwash with clean water, b) to dismantle rig and to check amount of biomass in fibres > recirculation rate turned down to improve better distribution of oxygen > oxygen flow turned down to 35	• Sampling
141	04.11.98	• General check-up • Feed system on manual due to poor float switch operation	• Sampling • Header tank emptied > soft scale on walls > sample of greywater in tank taken for COD and SS > feed pump left at 60 rpm to fill up tank overnight • No mixing in header tank	• Sampling	• Sampling
142	05.11.98	• General check-up • Feed system on manual due to poor float switch operation	• Header tank full but chambers low > inlet tubing cleaned > better flow	• Decommissioned because of air leak > reactor believed not to be operating properly > no	• Media found in pipes and outlet line > column refilled with 3" of media

		• No sampling due to operational problems	• No mixing in header tank	oxygen in fibres acting as surface for growth > check membrane bundle and retreat with heat and dye to seal fibres properly > samples of biofilm taken for microscopy	• Manually backwashed to settle media bed and backwash water sampled at t = 2, 4, 6, and 7 min for COD and SS • Backwash procedure: 1. Air 22 l/min and water 20 l/min at t=0 2. 1 min air at 22 l/min and water at 20 l/min 3. Air off at 5.5 min 4. Water off after 7 min (at this stage leak in backwash pipes detected) • Decided to repeat procedure for backwash profile • Barrel (200 l) set up for monitoring backwash frequency > valve from backwash line to barrel (valve or line leaks)
143	06.11.98	• General check-up • Feed tank cleaned with brush and T-junction put back on mixing pump > feed system switched from manual to automatic	• Sampling • Mixing in header tank		• Sampling
144	07.11.98	• General check-up			
145	08.11.98	• General check-up			
146	09.11.98	• General check-up • Feed tank brushed with broom, deposits on wall as expected (though tank brushed every 1-2 d)	• Sampling		• Sampling • Backwash barrel 1/3 full (barrel not checked during weekend, so could have backwashed any time since Friday afternoon) > emptied

147	10.11.98	• General check-up	• Sampling	• Sampling • Backwash barrel ¾ full > emptied
148	11.11.98	• General check-up	• Sampling	• Sampling • Backwash barrel ¾ full > emptied
149	12.11.98	• General check-up • Feed tank cleaned with broom • Feed system switched from auto to manual. T-piece on mixing pump removed to improve scale removal in feed tank.	• Sampling • Header tank mixer switched off because of poor performance • Sample of surface layer in header tank taken for COD and SS • Submersible pump with T-piece installed in header tank. Mixing improved and aerated contents >T-junction to decrease turbulence. Pieces for T-junction glued and left to dry > pump not on	• Sampling
150	13.11.98	• General check-up • Feed system switched from manual to auto	• Sampling • Header tank mixer off	• Sampling • Backwash barrel 1/3 full > sample taken for SS
151	14.11.98	• General check-up • Feed system had overfilled feed tank > motion of mixing pump had activated float switch so many times that no greywater concentrate left > feed system from auto to manual > T-piece on mixing pump put back on	• No mixing in header tank • Header tank cleaned with brush	
152	15.11.98	• General check-up • Feed system on manual		

		control • Feed tank cleaned with brush • Sample of scale in feed tank taken			
153	16.11.98	• General check-up • Feed system on manual at speed 6 ¼ . Feed tank cleaned with brush.	• Sampling • No mixing in header tank		• Sampling
154	17.11.98	• General check-up • Feed system pipes and centre of the monopump frozen due to cold night • Mono pump disconnected and emptied of water > hot water pumped through for 5 min to check its operation > the pump run for one feed cycle (30 min) to warm pump up • Frost protection urgent > electrician contacted (available tomorrow) > wires attached on feed lines and around centre of monopump > tried to disconnect pump failed because of electric wiring > pump emptied of water and covered with insulation jackets > the electric valve board covered with insulation jackets • Feed system on manual. T-junction on mixing pump in feed tank removed	• Sampling • Mixing in header tank without T-piece on the pump		• Sampling • Backwash barrel ½ full > emptied
155	18.11.98	• General check-up • Feed system on manual	• Sampling	• Fibres checked > left to air dry overnight	• Sampling

156	19.11.98	• General check-up • Feed system on manual • The frost protection wired up (not on control panel)	• Sampling	• Sampling
157	20.11.98	• General check-up • Feed system switched from manual to automatic. T-junction put back on mixing pump. To monitor automatic feed system over weekend.	• Sampling • Feed pump off at control panel (reason unknown) > level in chambers -30 cm > feed pump switched on at 90 rpm for 20 min to recover level and then turned down to 60 rpm, later to 40 rpm (normal flow rate)	• Sampling • Manually backwashed; backwash water sampled at t = 1, 2, 4, 6 min for COD and SS (total duration 6 min)
158	21.11.98	• General check-up		
159	22.11.98	• General check-up		
160	23.11.98	• General check-up • Power to control panels off for unknown reason	• Sampling	• Sampling • Foaming • Manually backwashed; backwash water sampled at t = 1, 2, 3, 4, 5, 6 min for COD and SS (total duration 7 min) • Effluent flow very poor after backwashing but enough sample for COD and SS collected
161	24.11.98	• General check-up	• Sampling	• No effluent flow > feed did not to pass pump head, air in feed line > Marprene tubing at both ends checked for blockages, some 1 cm^2 size algae found in feed line > feed flowed freely when tubing disconnected from tubing connector > Marprene tubing changed, no effect > feed pump switched off

					overnight so that problem can be solved at day light
162	25.11.98	• General check-up • Frost protection wired up to control box • Tube connector glued onto blackwater tank and left to dry overnight	• Sampling • Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes • Effluent sample taken for particle size analysis		• Feed pump switched off
163	26.11.98	• General check-up • Blackwater tank filled up ¾ full with water to check if tube connector leaks > no leaks • Barrel as overflow to blackwater tank, tube connector glued onto tank and left to dry overnight • T-junction taken off mixing pump to improve mixing	• Recirculation pump A leaks from the pump head > leaks less when pump on > pump not switched off • Effluent sample taken for particle size analysis	• Fibre bundle immersed in water and air let through > section 4A leaks > bundle left to air dry overnight before gluing bundle onto fibre holder	• Feed pump switched off
164	27.11.98	• General check-up	• Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes • Level in chambers -20 cm > inlet tubing squeezed to improve flow and to check for blockages • Effluent sample taken for particle size analysis		• Feed pump switched off
165	28.11.98	• General check-up			
166	29.11.98	• General check-up			
167	30.11.98	• General check-up • General check-up • Blackwater overflow checked	• Level in chambers -30 cm > inlet tubing squeezed to improve flow and to check for		• Feed pump switched off

		with water > works • Feed tank brushed > high deposits on walls due to low intake by Sub MBR	blockages		
168	01.12.98	• General check-up • Feed tank brushed • Blackwater system fixed	• Level in chambers -15 cm > inlet tubing squeezed	• Reconnected to start-up with diluted sludge > water in meter allowed gas through > not connected to rig > fibres taken to lab to dry • Fibres checked in water tank outside > airline fitting leaks • Backwash pump does not work > probably fuse in control panel	• Feed pump fixed after blockage located in feed pipe > flow restored
169	02.12.98	• General check-up	• Level in chambers -10 cm > inlet tubing squeezed • Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes		
170	03.12.98	• General check-up • Changeover to greywater/blackwater mix > main feed tank and Sub MBR header tank emptied and filled with new feed • Sampling at 12.5 min, 30 min, 1h, 2 h, 4 h, 8 h and 24 h	• Sampling (intensive) • Foaming 1 h after start-up		• Sampling (intensive) • Backwash switched from auto to manual
171	04.12.98	• General check-up	• Sampling 24 h from start-up • Foaming		• Sampling 24 h from start-up • Backwash switched from manual to auto > backwashed straight away > backwash valve on column opened so that daily monitoring of

				backwash frequency possible
172	05.12.98	• General check-up		• Had backwashed
173	06.12.98	• General check-up		
174	07.12.98	• General check-up	• Sampling and analysing same parameters plus dissolved oxygen as on intensive day • Foaming	• Sampling and analysing same parameters plus dissolved oxygen as on intensive day • Had backwashed - 5 l of water in barrel
175	08.12.98	• General check-up	• Sampling • Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes	• Sampling • Had backwashed - barrel full
176	09.12.98	• General check-up	• Sampling	• Sampling • No evidence of backwash (barrel empty), but level in effluent holding tank low. Effluent may not always flow to barrel.
177	10.12.98	• General check-up	• Sampling • Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes	• Sampling • Had backwashed - effluent tank level low
178	11.12.98	• General check-up	• Sampling	• Sampling
179	12.12.98	• General check-up		• Had backwashed - barrel half full
180	13.12.98	• General check-up		
181	14.12.98	• General check-up • Blackwater tank level low (tank half full), no overflow	• Sampling	• Sampling • Had backwashed - barrel half full
182	15.12.98	• General check-up • Valve on blackwater	• Sampling • Effluent lines flushed with	• Sampling • Had backwashed - 10 l in

		submersible pump line in primary settling tank turned up to increase flow to feed tank • Blackwater flow to blackwater tank constant and overflow to drain > valve turned down slightly to prevent fast flow • Feed tank cleaned with brush	diluted disinfectant and rinsed with water to prevent growth in pipes		barrel and effluent tank level low
183	16.12.98	• General check-up • Sampling • Valve on blackwater line from the primary settling tank to blackwater tank was turned down slightly to prevent fast flow			
184	17.12.98	• General check-up	• Sampling • Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes		• Sampling • Had backwashed - barrel full
185	18.12.98	• General check-up	• Sampling		• Sampling
186	19.12.98	• General check-up			
187	20.19.98	• General check-up			
188	21.12.98	• General check-up	• Sampling	• Assembled > oxygen leak in module > rig dismantled and oxygen line connector refitted > rig assembled > recirculation line snapped broken > rig dismantled and broken parts of recirculation line taken to lab to be glued	• Sampling • Had backwashed - 10 l in barrel
189	22.12.98	• General check-up	• Sampling • Effluent lines flushed with diluted disinfectant and rinsed		• Sampling • Had backwashed - barrel ½ full

			with water to prevent growth in pipes	
190	23.12.98	• General check-up	• Sampling	• Sampling
191	24.12.98	General check-up	• Sampling • Level in chambers -30 cm > inlet tubing squeezed (level in anoxic tanks too low to give truthful DO reading thus DO not measured)	• Sampling • Had backwashed - barrel ½ full
192	25.12.98	• General check-up	• Sampling	• Sampling • Had backwashed
193	26.12.98	• General check-up		
194	27.12.98	• General check-up	• Level in chambers -50 cm > inlet tubing cleaned with brush and water > reactors filled with feed	• Had backwashed - barrel 1/10 full
195	28.12.98	• General check-up		
196	29.12.98	• General check-up		
197	30.12.98	• General check-up		• Had backwashed - barrel ¾ full
198	31.12.98	• General check-up		
199	01.01.99	• General check-up	• Sampling	• Sampling • Had backwashed - barrel full
200	02.01.99	• General check-up		
201	03.01.99	• General check-up • The valve on blackwater submersible pump line in primary settling tank turned up to increase flow to feed tank (there was no overflow from blackwater tank)	• Sampling	• Sampling • Had backwashed - barrel full
202	04.01.99	• General check-up • The valve on the blackwater submersible pump line in	• Sampling • Level in chambers -10 cm > nearly flooding through hole	• Sampling • Backwashed three times • Feed pump increased from 20

		primary settling tank was turned up to increase flow to tank (there was hardly flow from blackwater tank). Valve had previously been turned down instead of up. • Net on blackwater inlet tubing (in blackwater tank) removed, flushed with water, then put back on	above overflow tubing in header tank > overflow tubing outside checked for blockages > solids found in tubing	rpm to 30 rpm to increase flow by 50% > effluent flow measured 390 ml/min
203	05.01.99	• General check-up • Submersible pump (main sewage supply to processes) in primary settling tank was off > switch turned on and off without success > pump taken out of tank, flushed with water and immersed in water barrel without successful start > plug put into another socket and still no power > no power in box > facilities manager contacted	• Sampling	• Sampling
204	06.01.99	• General check-up • Submersible pump (main sewage supply to the rigs) in primary settling tank replaced with another submersible pump > blackwater flow to blackwater tank	• Sampling	• Sampling • Had backwashed - barrel full
205	07.01.99	• General check-up	• Sampling	• Sampling • Had backwashed – barrel full
206	08.01.99	• General check-up	• Sampling	• Sampling • Had backwashed – barrel ½

				full
207	09.01.99	• General check-up		
208	10.01.99	• General check-up		• Had backwashed – barrel ½ full
209	11.01.99	• General check-up	• Sampling	• Sampling • Had backwashed – barrel full
210	12.01.99	• General check-up • Main feed tank cleaned with brush	• Sampling • Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes	• Sampling
211	13.01.99	• General check-up	• Sampling • Level in chambers -40 cm > inlet tubing disconnected and emptied before connecting back to reactors > reactors filled with feed • Header tank cleaned with brush • White particles in effluent during sampling (residue after cleaning effluent lines?)	• Sampling • Had backwashed – barrel ¼ full • Backwashed again before sampling
212	14.01.99	• General check-up	• Sampling • Effluent lines flushed with diluted disinfectant and rinsed with water to prevent growth in pipes • Effluent examined under microscope > dead bacterial mass. Nematode identified > to contact technician about taking photos of bacteria and then change tubing. Next time to use sodium hypochlorite	• Sampling

			instead of general disinfectant to avoid surfactants and detergents.	
213	15.01.99	• General check-up	• Sampling • Level in chambers -40 cm > inlet tubing disconnected and brushed before connecting back to reactors	• Sampling • Had backwashed – 10 l in barrel
214	16.01.99	• General check-up • Blackwater tank level low > valve adjusted to improve flow	• Level in chambers -40 cm > inlet tubing squeezed	• Had backwashed
215	17.01.99	• General check-up		• Backwashed
216	18.01.99	• General check-up	• Sampling • Level in chambers -40 cm > inlet tubing squeezed	• Sampling
217	19.01.99	• General check-up • Greywater tank, blackwater tank and main feed tank brushed	• Sampling • Samples of effluent, all units in A and B and header tank taken for the microscopic analysis • Header tank brushed	
218	20.01.99	• General check-up	• Sampling • Level in chambers -40 cm > inlet tubing squeezed	• Sampling
219	21.01.99	• General check-up • Blackwater tank level low > valve adjusted to improve flow	• Sampling • Level in chambers -40 cm > overflow also blocked and thus influent flooded through small hole above overflow in header tank > inlet tubing cleaned • Header tank brushed • Air flow turned down from 20 l/min to 5 l/min to monitor DO levels in tanks and effluent	• Sampling • Backwashed

220	22.01.99	• General check-up	• Sampling • Recycle on A opened to anoxic tank to improve flow	• Sampling • Trap cleaned and 5" of media added into column
221	23.01.99	• General checkup	• Level in chambers -30 cm > inlet tubing squeezed	• Had backwashed - 5 l in barrel
222	24.01.99	• General checkup	• Level in chambers -30 cm > inlet tubing squeezed	• Poor effluent flow • Had backwashed - barrel ¼ full
223	25.01.99	• General check-up • Greywater and main feed tank brushed	• Level in chambers -40 cm > inlet tubing squeezed • Samples for TC/EC/FS and MLSS taken in early evening • Plastic lid glued onto hole on header tank	• Sampling • Backwash switched from auto to manual (air scour, wash water and wash pump valves)
224	26.01.99	• General check-up	• Sampling	• Sampling • Attempt to manually backwash > backwash water came through effluent pipe to effluent collection tank > probable reason that air scour and wash water valves on 'auto' and wash pump on 'run' • Backwashed; samples taken at t = 1, 2, 3, 4, 5, 6 and 7 min for SS, COD and CST (total duration 8 min)
225	27.01.99	• General check-up	• Sampling	• Sampling
226	28.01.99	• General check-up • Main feed tank and greywater tank brushed	• Sampling • Header tank brushed	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min)
227	29.01.99	• General check-up	• Sampling • Header tank mixer and	• Sampling • Decided to manually backwash

			recirculation pump B wired up to control panel no.2 > more accurate power readings	daily until changeover to blackwater > more consistent backwash profile • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min). Photo of samples taken.
228	30.01.99	• General check-up	• Level in chambers -20 cm > inlet tubing squeezed	• Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min)
229	31.01.99	• General check-up	• Level in chambers -10 cm > inlet tubing squeezed	• Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min)
230	01.02.99	• General check-up	• Sampling • Level in chambers -10 cm > inlet tubing squeezed	• Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity every 10 min for 1 h after backwashing
231	02.02.99	• General check-up • Submersible pump in primary settling tank lowered	• Sampling • Header tank overflowed because of blockage > air turned up to 25 l/min to prevent cake formation • Overflow tubing on header tank blocked > cleaned > effluent lines closed overnight to have a better chance to monitor level in reactor settle • Air flow turned down to 20	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing

			l/min	
232	03.02.99	• General check-up	• Sampling • Feed pump switched on after overflow tubing on header tank flushed with water, effluent lines opened • Effluent lines flushed with hypochlorite and rinsed with water to prevent growth in pipes	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing
233	04.02.99	• General check-up	• Sampling • Level in chambers -10 cm > inlet tubing disconnected and cleaned	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing
234	05.02.99	• General check-up	• Sampling	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Efluent samples for turbidity taken every 10 min for 1 h after backwashing • Flow rate turned down to 220 ml/min and backwash mode switched from manual to auto > backwashed
235	06.02.99	• General check-up		
236	07.02.99	• General check-up	• Header tank flooded > overflow tube cleaned with water and directed to drain	• Had backwashed -5 l in barrel

			rather than to main feed tank	
237	08.02.99	• General check-up • Changeover to blackwater > main feed tank (50 l of grey/black mix left in tank) and Sub MBR header tank (10 l of mix left in tank) emptied and filled with new feed • Start of blackwater run (sampling at 12.5, 30 min, 1 h, 2 h, 4 h, and 8 h) • T-piece on mixer in main feed tank taken off to improve mixing	• Effluent lines disinfected with hypochlorite and rinsed with water • Sampling (intensive) • Barrel set up outside shed as overflow tank for header tank > tubing from bottom tap to drain	• Sampling (intensive)
238	09.02.99	• General check-up	• Sampling at 24 h and 30 h from start of run	• Sampling at 24 h and 30 h from start of run
239	10.02.99	• General check-up	• Sampling at 48 h from start of run	• Sampling at 48 h from start of run • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 minutes) • Backwashed automatically
240	11.02.99	• General check-up • Main feed tank level very low > no flow to rigs > tank filled up • Greywater tank emptied of concentrate and cleaned with water	• Sampling	• Sampling • Both columns on > make note on power readings
241	12.02.99	• General check-up	• Sampling	• Sampling
242	13.02.99	• General check-up		• Had backwashed - 5 l in barrel
243	14.02.99	• General check-up		• Had backwashed - 5 l in barrel

244	15.02.99	• General check-up	• Sampling	• Sampling • Had backwashed - 10 l in barrel • No flow through column > tubing on pump head tightened > flow
245	16.02.99	• General check-up	• Sampling	• Sampling • Had backwashed - 5 l in barrel
246	17.02.99	• General check-up	• Sampling	• Sampling • Had backwashed - 5 l in barrel
247	18.02.99	• General check-up	• Sampling	• Sampling • Had backwashed - 5 l in barrel
248	19.02.99	• General check-up • T-piece on mixer in main feed tank taken off to improve mixing • Feed system switched from manual to auto	• Sampling	• Sampling • Flow rate turned down to 220 ml/min to see if effluent quality improves as loading rate is decreased
249	20.02.99	• General check-up		• Had backwashed - 5 l in barrel
250	21.02.99	• General check-up		• Had backwashed - 5 l in barrel
251	22.02.99	• General check-up	• Sampling	• Sampling • Had backwashed - 5 l in barrel
252	23.02.99	• General check-up		• Sampling • Had backwashed - 5 l in barrel
253	24.02.99	• General check-up	• Sampling	• Sampling • Had backwashed - 10 l in barrel
254	25.02.99	• General check-up	• Sampling	• Sampling
255	26.02.99	• General check-up		
256	27.02.99	• General check-up		
257	28.02.99	• General check-up		
258	01.03.99	• General check-up	• Sampling	• Sampling

				• Had backwashed - 10 l in barrel
259	02.03.99	• General check-up	• Sampling	• Sampling • Very low flow (20 ml/min instead of 220 ml/min - something blocks pipe) > manually backwashed, backwash water through effluent pipe > suspected that actual backwashing did not occur
260	03.03.99	• General check-up		• Flow had recovered
261	04.03.99	• General check-up	• Sampling	• Sampling • Had backwashed - 10 l in barrel
262	05.03.99	• General check-up	• Sampling	• Sampling
263	06.03.99	• General check-up		
264	07.03.99	• General check-up		
265	08.03.99	• General check-up	• Sampling	• Sampling • Had backwashed – 5 l in barrel
266	09.03.99	• General check-up	• Sampling	• Sampling
267	10.03.99	• General check-up	• Sampling	• Sampling
268	11.03.99	• General check-up	• Sampling • Header tank cleaned with brush	• Sampling • Had backwashed - 5 l in barrel
269	12.03.99	• General check-up	• Sampling	• Sampling • One column running since afternoon (make note to power readings).
270	13.03.99	• General check-up	• Sampling	• Sampling
271	14.03.99	• General check-up	• Sampling	• Sampling
272	15.03.99	• General check-up	• Sampling	• Sampling • Hardly any flow. Inlet checked

				by lifting feed inlet tubing on top of column, unable to take lid off. Attempt to manually backwash resulted in backwash water coming through effluent pipe (no air at all). Then it backwashed automatically twice. Flow first 260 ml/min and then turned down to 220 ml/min. • Backwash mode switched from auto to manual > to monitor backwash water and effluent turbidity for 7 d.
273	16.03.99	• General check-up	• Sampling • Header tank cleaned with brush	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing • Both columns running (note to power readings)
274	17.03.99	• General check-up • Submersible pump in main feed tank does not work > to contact electrician	• Sampling • Feed pump overheated and swapped to another Watson-Marlow, 40 rpm (fault in control box or fuse? neither SS MBR nor Sub MBR plug worked) > to have pump checked by either supplier or Cranfield workshop and control box by electrician	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing

275	18.03.99	• General check-up	• Sampling	• Sampling • Effluent from columns soapy because of other greywater experiment run on second column, which does not treat it efficiently (effect on backwash quality?) • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing
276	19.03.99	• General check-up • Electrician checked control box no. 1 and found no fault in fuses for SS MBR, Sub MBR and submersible pump > another submersible pump to be wired for using as mixer in main feed tank	• Sampling	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing • Second column not running (note on power data)
277	20.03.99	• General check-up		• Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 hr after backwashing
278	21.03.99	• General check-up		• Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST total

				duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing
279	22.03.99	• General check-up • Submersible pump wired in for use in main feed tank. Not switched on as t-piece to reduce turbulence needs to be fitted.	• Sampling	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST (total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing • Second column running
280	23.03.99	• General check-up • T-piece on submersible pump fitted and pump switched on	• Sampling	• Sampling • Backwashed; samples taken at t = 2, 3, 4, 5, 6, 7 and 8 min for SS, COD and CST total duration 8 min) • Effluent samples for turbidity taken every 10 min for 1 h after backwashing
281	24.03.99	• General check-up		
282	25.03.99	• General check-up	• Sampling	• Sampling
283	26.03.99	• General check-up	• Sampling	• Sampling • Had backwashed - 10 l in barrel • Antifoaming agent used in second column > chemical also in effluent tank (possible effect on backwash) > to separate effluent flows of two columns
284	27.03.99	• General check-up		
285	28.03.99	• General check-up		

287	30.03.99	• General check-up	• Dismantled and membranes cleaned	• Running with blackwater in order to collect effluent for RO studies (not part of this work)

NB. This log book relates to the operation of the pilot plant only and does not include greywater storage trials, additional experiments or laboratory analyses.
Submerged MBR - level in chambers (e.g. -10 cm i.e. 10 cm below the ideal mixed liquor level in chamber)
Submerged MBR consisted of two sets of chambers, marked as A and B in text

Table F.1. BAF influent quality during the intermittent operation trials. Mean ± standard deviation.

	Feed off						Air off					Feed and air off		
Parameter	30 min	1 h	2 h	4 h	8 h	3.5 weeks	30 min	1 h	2 h	4 h	8 h	2 h	4 h	8 h
BOD_5 (mg l^{-1})	165 ± 0	141 ± 0	122 ± 0	162 ± 0	138 ± 0	148 ± 14	58 ± 0	135 ± 0	40 ± 0	67 ± 0	78 ± 0	36 ± 0	63 ± 0	96 ± 0
SS (mg l^{-1})	74 ± 45	43 ± 18	48 ± 0	62 ± 0	77 ± 7	43 ± 12	27 ± 0	36 ± 0	32 ± 0	44 ± 0	36 ± 0	36 ± 0	11 ± 0	90 ± 0
pH (-)	8.0 ± 0.2	6.9 ± 0.0	7.2 ± 0.1	-	6.5 ± 0.0	7.3 ± 0.2	7.3 ± 0.0	-	-	7.3 ± 0.0	7.6 ± 0.0	7.5 ± 0.0	7.3 ± 0.0	7.1 ± 0.0
TC (cfu 100^{-1} ml^{-1})	8600 ± 0	970 ± 0	45700 ± 0	2310 ± 0	2380 ± 0	12729 ± 16211	6970 ± 0	980 ± 0	8600 ± 0	3270 ± 0	9800 ± 0	6300 ± 0	200 ± 0	9320 ± 0
E.coli (cfu 100^{-1} ml^{-1})	100 ± 0	100 ± 0	100 ± 0	100 ± 0	100 ± 0	134 ± 116	730 ± 0	100 ± 0	100 ± 0	100 ± 0	100 ± 0	100 ± 0	100 ± 0	100 ± 0
FS (cfu 100^{-1} ml^{-1})	100 ± 0	100 ± 0	10 ± 0	10 ± 0	200 ± 0	7 ± 3	1733 ± 0	10 ± 0	1986 ± 0	2419 ± 0	2419 ± 0	10 ± 0	10 ± 0	2419 ± 0

TC = total coliforms, FS = faecal streptococci

Table F.2. Submerged MBR influent quality in the intermittent operation trials. Mean ± standard deviation.

	Feed off		Feed and air off
Parameter	8 h	3.5 weeks	8 h
BOD_5 (mg l^{-1})	33 ± 0	174 ± 2	191 ± 0
SS (mg l^{-1})	34 ± 1	51 ± 9	142 ± 0
pH (-)	7.5 ± 0.0	7.5 ± 0.0	7.4 ± 0.0
Total coliforms (cfu 100^{-1} ml^{-1})	100 ± 0	7905 ± 4658	740 ± 0
E.coli (cfu 100^{-1} ml^{-1})	100 ± 0	101 ± 85	1 ± 0
Faecal streptococci (cfu 100^{-1} ml^{-1})	1 ± 1	1264 ± 1634	1 ± 0

Table F.3. pH, temperature and DO of the submerged MBR effluent in the intermittent operation trials.

	Feed off 8 h			Feed off 3.5 weeks			Feed and air off for 8 h		
Time	pH	T	DO	pH	T	DO	pH	T	DO
h)	(-)	(°C)	(mg l^{-1})	(-)	(°C)	(mg l^{-1})	(-)	(°C)	(mg l^{-1})
Before	7.9	21	8.1	-	-	-	7.6	27	8.3
During	-	-	-	-	-	-	8.0	19	7.6
0.1	-	-	-	4.0	9	10	-	-	-
0.208	7.5	18	8.1	-	-	-	7.8	18	7.4
0.5	7.6	18	8.1	-	-	-	7.8	19	8.2
1	7.7	19	8.3	-	-	-	7.8	18	8.2
2	7.8	18	8.5	-	-	-	7.9	18	8.6
4	7.8	18	8.0	3.9	10	10	-	-	-
8	-	-	-	3.8	12	10	-	-	-
12	-	-	-	-	13	10	-	-	-
24	-	-	-	7.2	11	10	-	-	-
48	-	-	-	6.3	13	10	-	-	-

Table F.4. Flow rates of the submerged MBR in the intermittent operation trials.

Time (h)	Feed off 8 h (m^3 d^{-1})	Feed off 3.5 weeks (m^3 d^{-1})	Feed and air off 8 h (m^3 d^{-1})
Before	0.090	-	0.081
During	-	-	0.049
During			0.036
0.208	0.072	-	0.078
0.5	0.086	-	0.085
1	0.085	0.098	0.086
2	0.091	-	0.086
4	0.089	0.089	-
8	-	0.094	-
12	-	0.094	-
24	-	0.089	-
48	-	0.094	-

Table F.5. pH, temperature and DO of the BAF effluent in the feed off trials from 30 min to 3.5 weeks.

	pH (-)						Temperature (°C)						Dissolved oxygen (mg l^{-1})					
Time (h)	30 min	1 h	2 h	4 h	8 h	3.5 weeks	30 min	1 h	2 h	4 h	8 h	3.5 weeks	30 min	1 h	2 h	4 h	8 h	3.5 weeks
Before	7.8	8.0	7.9	8.1	7.7	-	19	33	24	23	22	-	9.3	6.7	7.9	7.9	8.6	-
0.208	7.5	7.5	7.4	7.4	-	-	23	32	26	22	18	-	8.8	6.5	7.4	8.0	8.6	-
0.5	7.8	7.8	7.9	8.0	-	-	26	32	27	22	16	-	8.1	6.8	7.8	8.7	8.0	-
1	7.8	7.6	8.1	7.7	-	7.2	26	33	28	22	16	9	7.9	7.0	7.8	8.5	9.4	7.4
2	7.6	8.0	8.1	7.9	-		29	33	26	23	16	-	7.6	6.3	7.8	8.4	9.5	-
4	7.6	7.6	8.1	7.5	7.8	7.5	26	28	22	21	15	18	7.0	7.0	7.7	8.3	9.8	9.7
8	-	7.8	8.1	8.0	-	7.6	-	23	22	18	15	23	-	8.1	8.2	8.8	9.6	8.2
12	-	-	-	-	-	7.5	-	-	-	-	-	16	-	-	9.0	-	-	10.0
16	-	-	8.2	7.2	-	-	-	-	18	18	-	-	-	-	-	9.0	10.0	-
20	-	-	-	7.6	-	-	-	-	-	21	-	-	-	-	-	8.9	-	-
24	-	-	-	7.4	-	7.7	-	-	-	22	-	9	-	-	-	8.4	-	10.0
32	-	-	-	-	7.8	-	-	-	-	-	-	-	-	-	-	-	4.7	-
40	-	-	-	7.4	-	-	-	-	-	22	-	-	-	-	-	8.6	-	-
48	-	-		-	7.9	7.8	-	-	-	-	-	9	-	-	-	-	8.3	10.0

Table F.6. pH, temperature and DO of the BAF effluent in the air off trials from 30 min to 8 h.

	pH (-)					Temperature (°C)					Dissolved oxygen (mg l^{-1})				
Time (h)	30 min	1 h	2 h	4 h	8 h	30 min	1 h	2 h	4 h	8 h	30 min	1 h	2 h	4 h	8 h
Before	7.7	6.9	7.7	7.9	8.3	30	32	27	24	16	7.4	6.3	7.6	8.2	8.8
During	7.4	6.9	7.6	8.0	7.8	31	33	27	25	17	7.0	6.3	7.4	7.7	8.3
During	-	6.6	7.5	7.7	8.0	-	34	27	26	17	-	5.8	6.9	6.7	7.9
During	-	-	7.5	7.6	7.8	-	-	27	26	18	-	-	6.3	6.4	6.7
During	-	-	7.4	7.4	7.7	-	-	27	27	20	-	-	5.9	6.3	6.1
During	-	-	7.4	7.4	7.6	-	-	27	27	20	-	-	5.8	6.1	5.6
During	-	-	-	7.4	7.6	-	-	-	28	20	-	-	-	5.8	5.2
During	-	-	-	7.4	7.6	-	-	-	29	20	-	-	-	5.3	5.2
During	-	-	-	7.4	7.6	-	-	-	30	20	-	-	-	5.2	5.0
During	-	-	-	-	7.6	-	-	-	29	20	-	-	-	5.2	5.1
During	-	-	-	7.4	7.6	-	-	-		20	-	-	-	-	4.7
0.208	7.7	7.5	7.3	7.2	7.5	32	34	27	30	21	6.3	5.3	5.7	5.0	5.0
0.5	7.8	7.6	7.7	7.3	7.6	34	33	27	31	21	6.7	6.0	6.9	4.8	7.0
1	8.0	7.5	7.8	7.2	7.5	34	32	27	32	20	6.6	6.4	7.2	4.9	7.7
2	7.7	7.7	7.8	7.5	7.7	35	34	26	33	21	6.7	6.3	7.5	5.0	8.1
4	7.9	8.0	7.6	7.8	7.9	36	31	24	32	20	6.5	7.0	7.7	6.4	-
8	-	8.1	8.0	8.0	8.1	-	-	-	26	17	-	7.0	7.7	6.5	8.5
24	-	-	-	7.7	8.1	-	-	-	33	28	-	-	-	7.1	9.6

Table F.7. pH, temperature and DO of the BAF effluent in the feed and air off trials from 2 to 8 h.

	pH (-)			Temperature (°C)			Dissolved oxygen (mg l^{-1})		
Time (h)	2 h	4 h	8 h	2 h	4 h	8 h	2 h	4 h	8 h
Before	7.8	7.6	7.6	20	23	17	9.1	8.5	9.6
0.208	7.7	7.5	7.5	28	34	35	6.9	6.3	4.5
0.5	7.8	7.6	7.4	26	31	34	7.0	6.8	4.7
1	7.8	7.7	7.7	26	33	35	7.1	6.5	4.6
2	7.8	7.7	7.3	27	35	35	7.1	6.2	4.5
4	7.7	7.8	7.8	27	32	29	7.5	7.0	5.2
7	-	7.7	-	-	27	-	-	-	7.0
16	-	-	7.7	-	-	18	-	-	-
24	-	7.8	7.2	-	-	36	-	10.0	5.5
40	-	-	-	-	-	19	-	-	8.5
48	-	-	-	-	-	34	-	-	5.6

Table F.8. Flow rates ($m^3\ d^{-1}$) of the BAF in the intermittent operation trials.

	Feed off						Air off					Feed and air off		
Time	30 min ($m^3\ d^{-1}$)	1 h ($m^3\ d^{-1}$)	2 h ($m^3\ d^{-1}$)	4 h ($m^3\ d^{-1}$)	8 h ($m^3\ d^{-1}$)	3.5 weeks ($m^3\ d^{-1}$)	30 min ($m^3\ d^{-1}$)	1 h ($m^3\ d^{-1}$)	2 h ($m^3\ d^{-1}$)	4 h ($m^3\ d^{-1}$)	8 h ($m^3\ d^{-1}$)	2 h ($m^3\ d^{-1}$)	4 h ($m^3\ d^{-1}$)	8 h ($m^3\ d^{-1}$)
Before	0.271	0.245	0.374	0.288	0.274	-	0.300	0.291	0.276	0.297	0.259	0.248	0.253	0.282
During	-	-	-	-	-	-	0.271	0.276	0.271	0.285	0.282	-	-	-
During	-	-	-	-	-	-	-	0.276	0.268	0.285	0.271	-	-	-
During	-	-	-	-	-	-	-	-	0.268	0.285	0.288	-	-	-
During	-	-	-	-	-	-	-	-	0.276	0.294	0.294	-	-	-
During	-	-	-	-	-	-	-	-	0.268	0.291	0.302	-	-	-
During	-	-	-	-	-	-	-	-	-	0.300	0.300	-	-	-
During	-	-	-	-	-	-	-	-	-	0.297	0.279	-	-	-
During	-	-	-	-	-	-	-	-	-	0.297	0.288	-	-	-
During	-	-	-	-	-	-	-	-	-	0.300	0.84	-	-	-
During	-	-	-	-	-	-	-	-	-	0.297	0.285	-	-	-
0.208	0.291	0.276	0.282	0.314	0.213	-	0.300	0.291	0.299	0.259	0.271	0.251	0.294	0.308
0.5	0.274	0.308	0.262	0.317	0.259	-	0.271	0.294	0.291	0.276	0.294	0.251	0.294	0.302
1	0.317	0.297	0.294	0.315	0.271	0.317	0.294	0.271	0.294	0.305	0.274	0.256	0.288	0.302
2	0.314	0.288	0.308	0.288	0.285	-	0.268	0.300	0.297	0.300	0.282	0.253	0.282	0.302
4	0.288	0.265	0.276	0.279	0.259	0.288	0.259	0.294	0.271	0.285	0.268	0.251	0.300	0.294
8	-	0.251	0.288	0.259	0.228	0.271	-	0.300	0.256	0.300	-	-	0.262	0.282
12	-	-	-	-	-	0.262	-	-	-	-	-	-	-	-
16	-	-	0.271	0.265	-	-	-	-	-	0.285	0.274	-	-	-
24	-	-	-	0.294	0.209	0.266	-	-	-	0.300	-	-	-	0.294
32	-	-	-	0.251	0.300	-	-	-	-	-	-	-	-	-
40	-	-	-	0.294	-	-	-	-	-	0.302	-	-	-	0.265
48	-	-	-		0.251	0.253	-	-	-	-	-	-	-	0.285

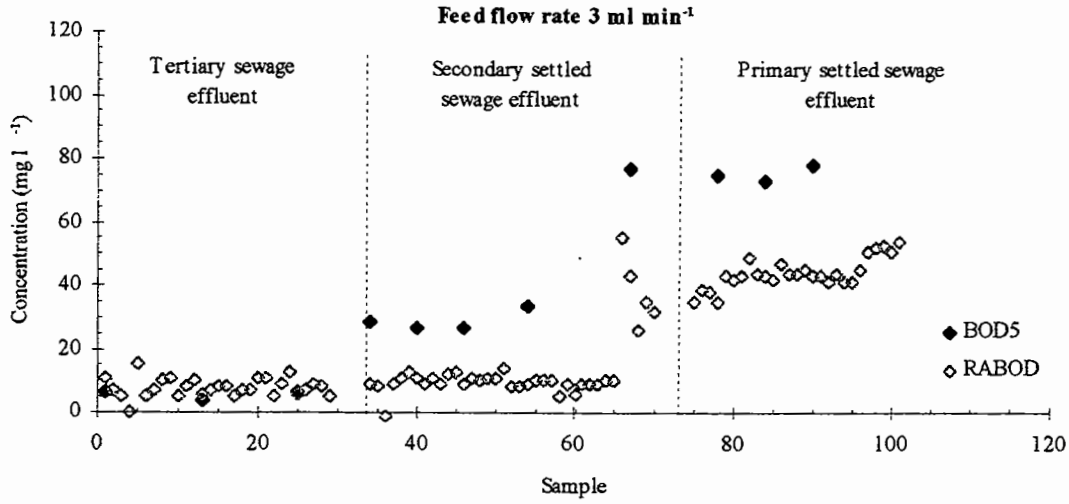


Figure F.1. Correlation between RABOD and BOD_5 from tertiary sewage effluent to primary settled sewage effluent at a feed flow rate of 3 ml min^{-1}.

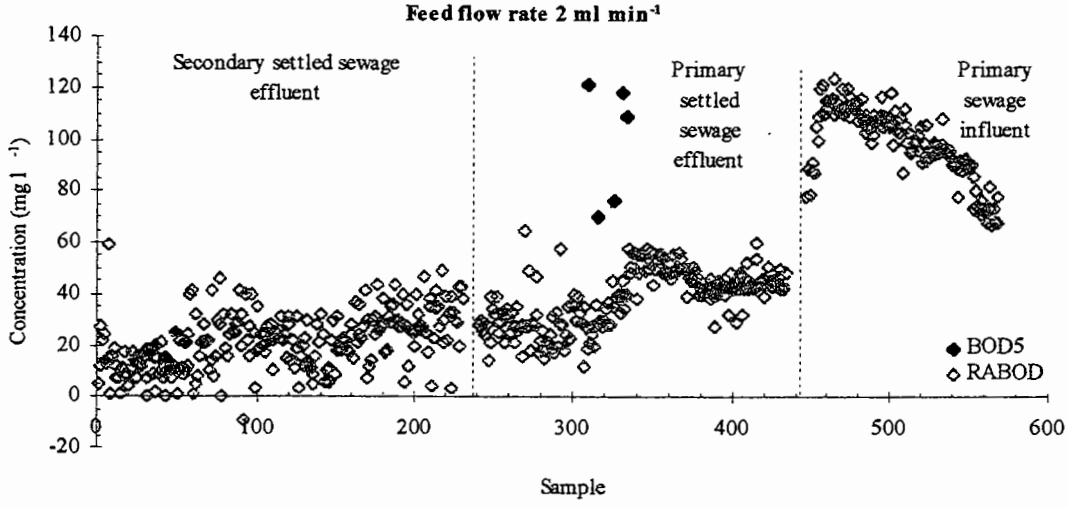


Figure F.2. Correlation between RABOD and BOD_5 from secondary settled sewage effluent to primary sewage influent at a feed flow rate of 2 ml min^{-1}.

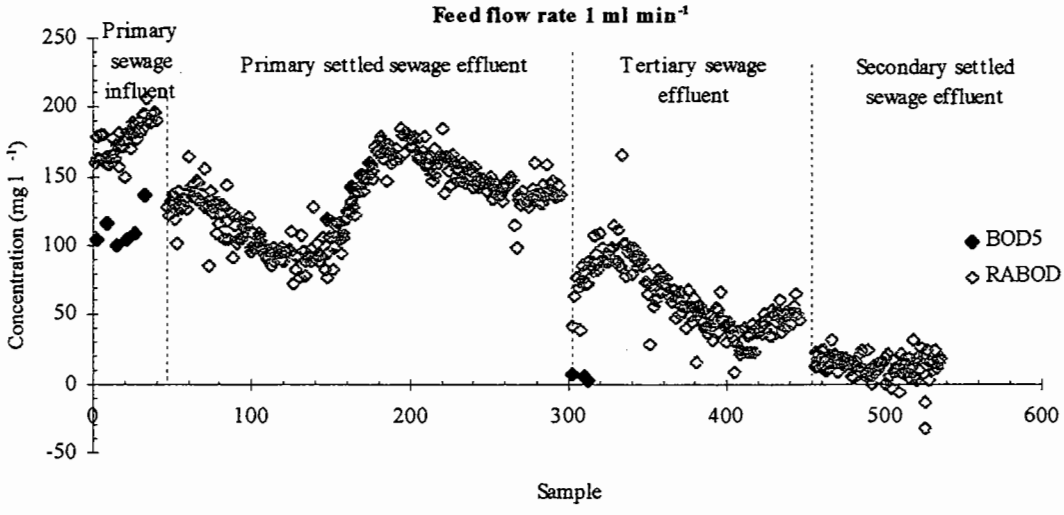


Figure F.3. Correlation between RABOD and BOD_5 from primary sewage influent to secondary settled sewage effluent at a feed flow rate of 1 ml min^{-1}.

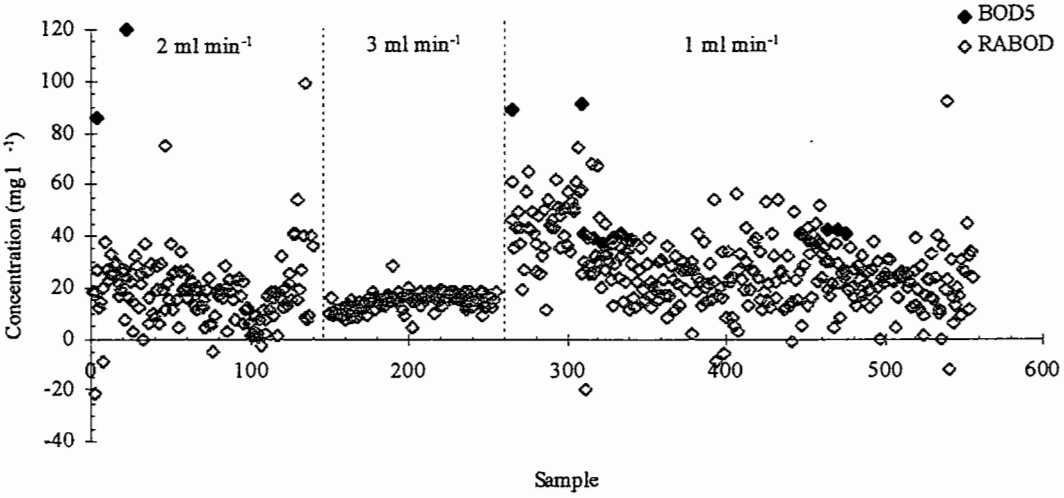


Figure F.4. Correlation between RABOD and BOD_5 of synthetic greywater at feed flow rates of 1-3 ml min^{-1}.

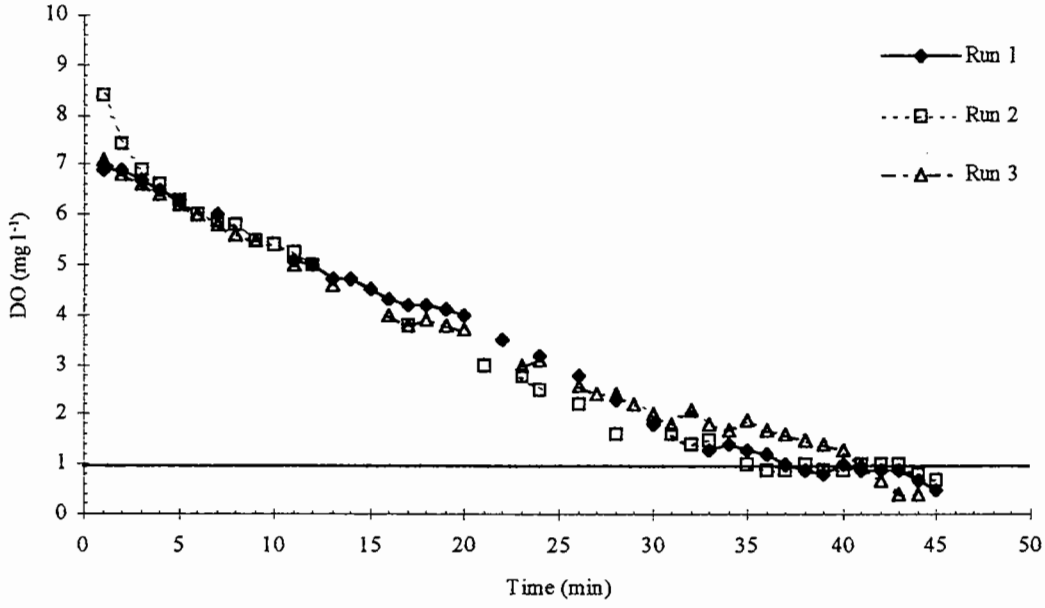


Figure F.5. Dissolved oxygen decline in the submerged MBR nitrification unit after air turned off.

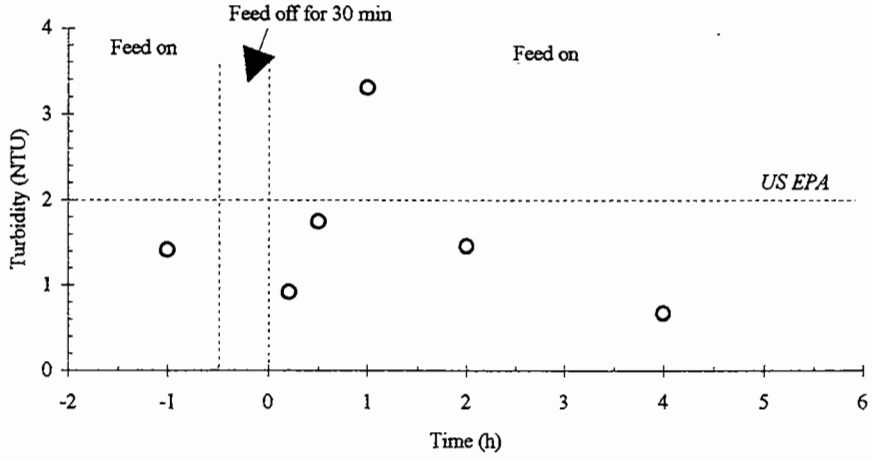


Figure F.6. BAF effluent turbidity in the 30 min feed-off trial.

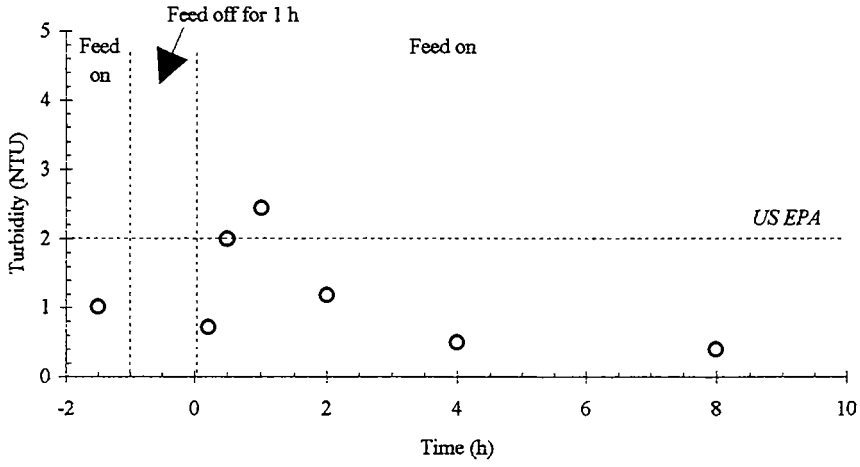


Figure F.7. BAF effluent turbidity in the 1 h feed-off trial.

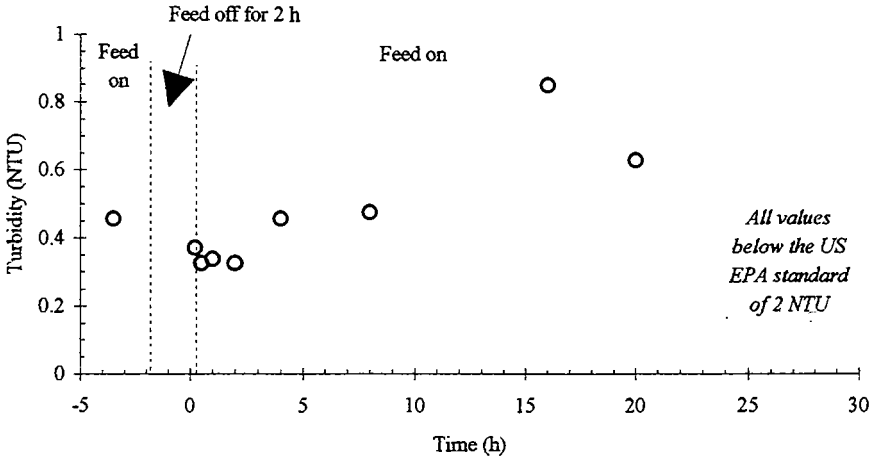


Figure F.8. BAF effluent turbidity in the 2 h feed-off trial.

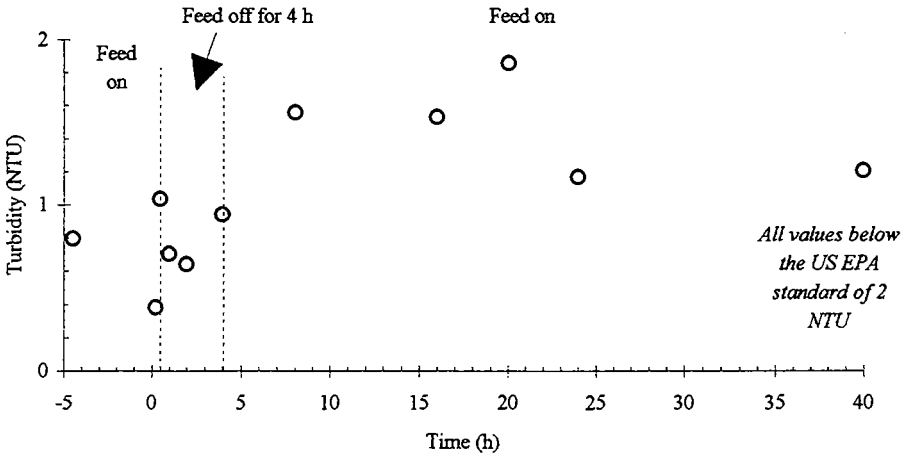


Figure F.9. BAF effluent turbidity in the 4 h feed-off trial.

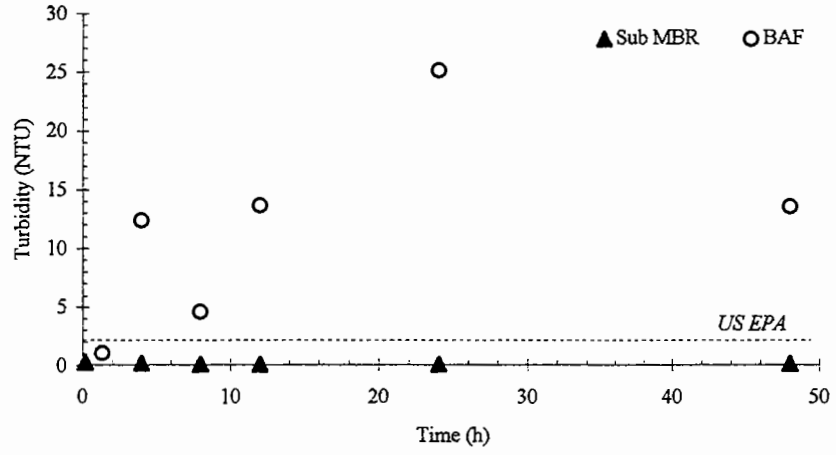


Figure F.10. Submerged MBR and BAF effluent turbidity after the 3.5-week feed-off trial.

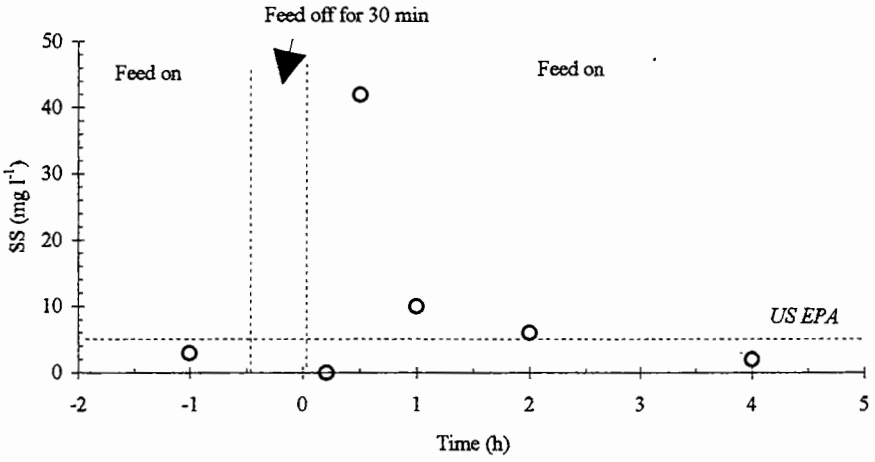


Figure F.11. BAF effluent solids in the 30 min feed-off trial.

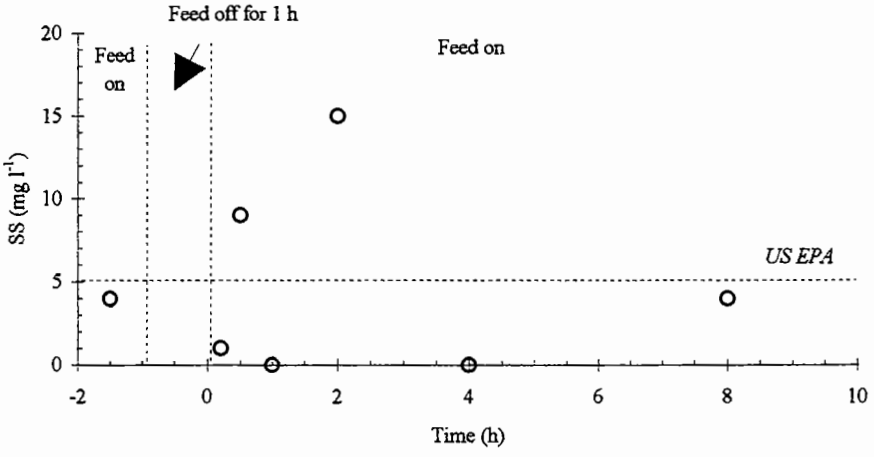


Figure F.12. BAF effluent solids in the 1 h feed-off trial.

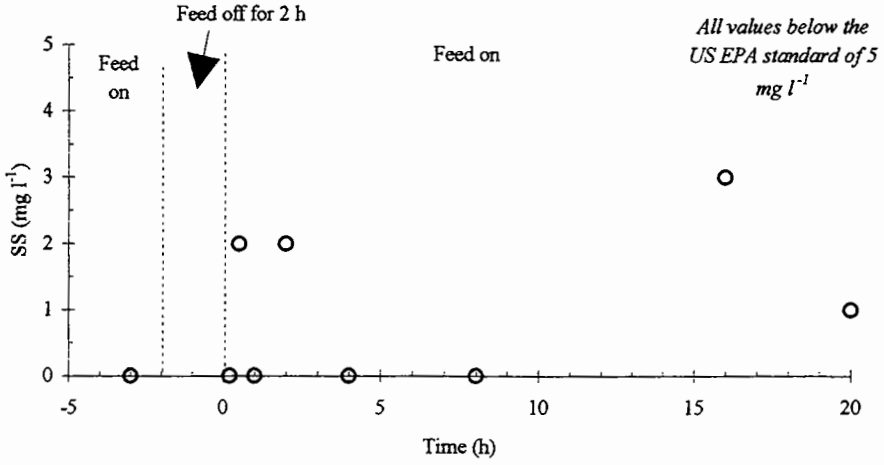


Figure F.13. BAF effluent solids in the 2 h feed-off trial.

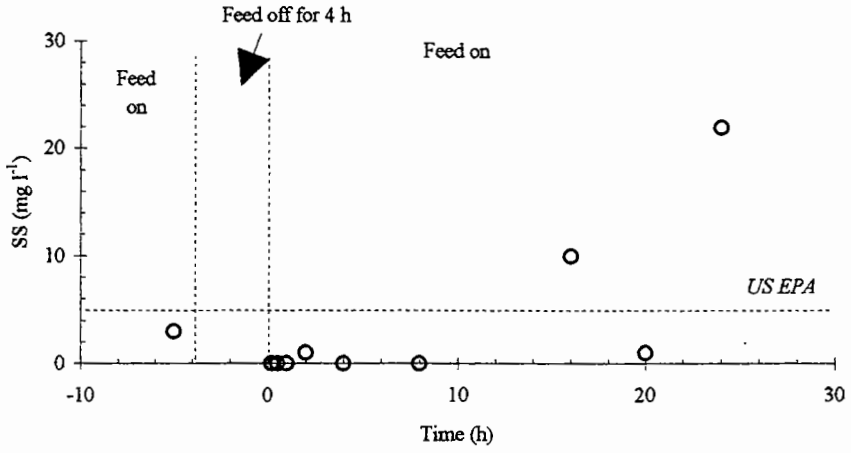


Figure F.14. BAF effluent solids in the 4 h feed-off trial.

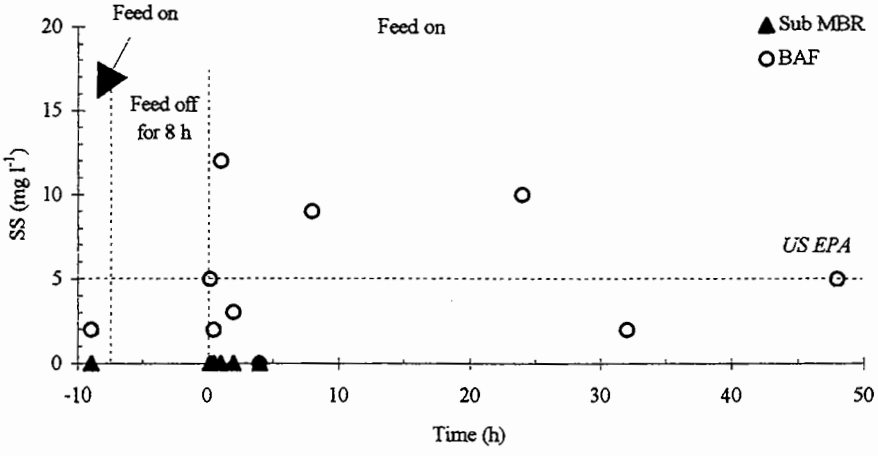


Figure F.15. Submerged MBR and BAF effluent solids in the 8 h feed-off trial.

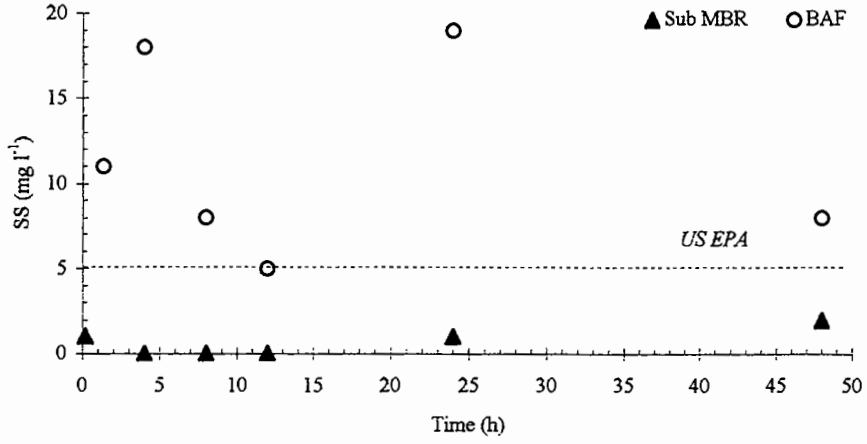


Figure F.16. Submerged MBR and BAF effluent solids after the 3.5-week feed-off trial.

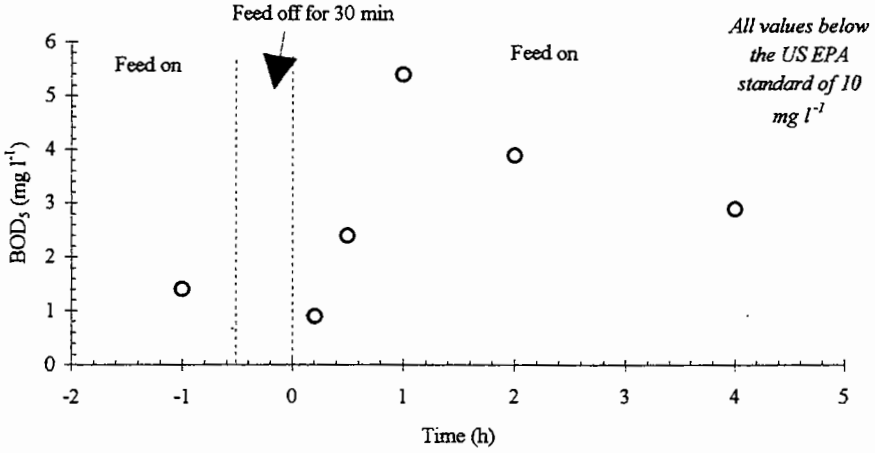


Figure F.17. BAF effluent BOD in the 30 min feed-off trial.

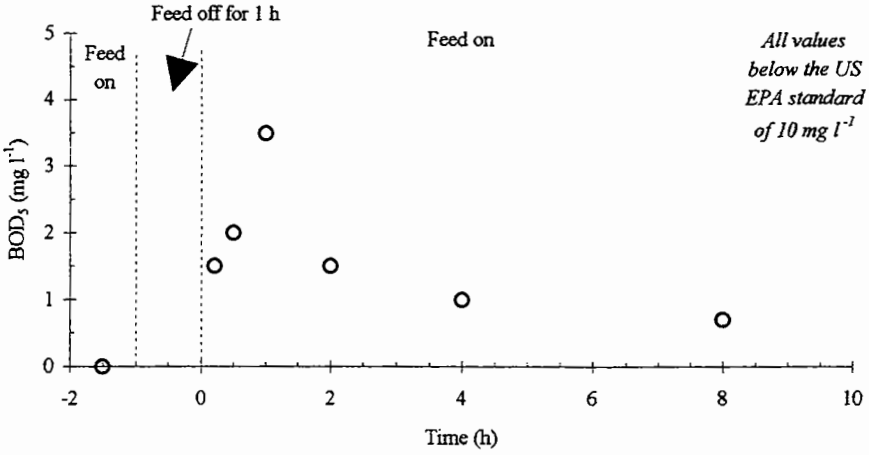


Figure F.18. BAF effluent BOD in the 1 h feed-off trial.

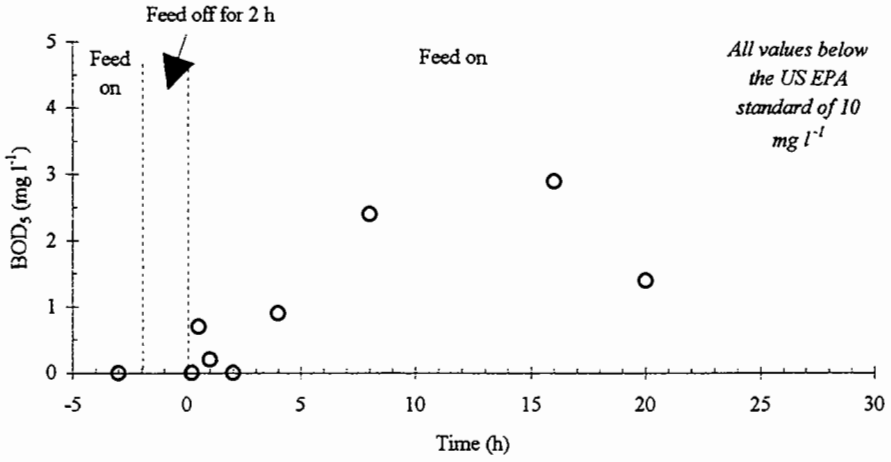


Figure F.19. BAF effluent BOD in the 2 h feed-off trial.

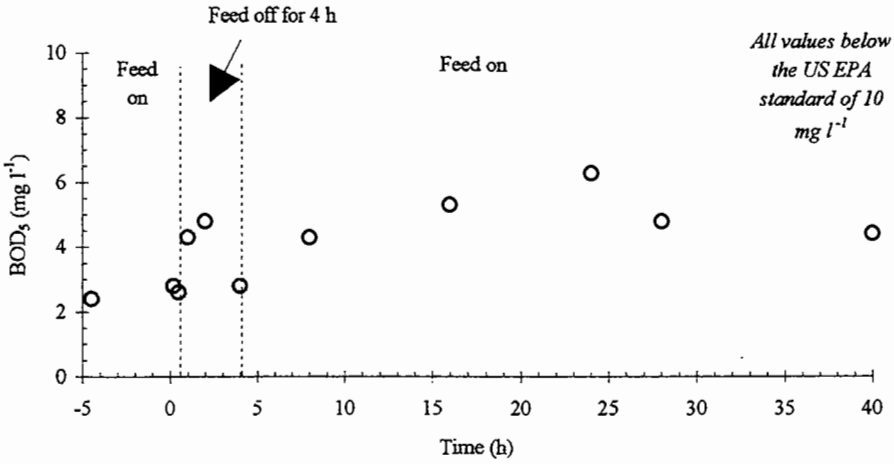


Figure F.20. BAF effluent BOD in the 4 h feed-off trial.

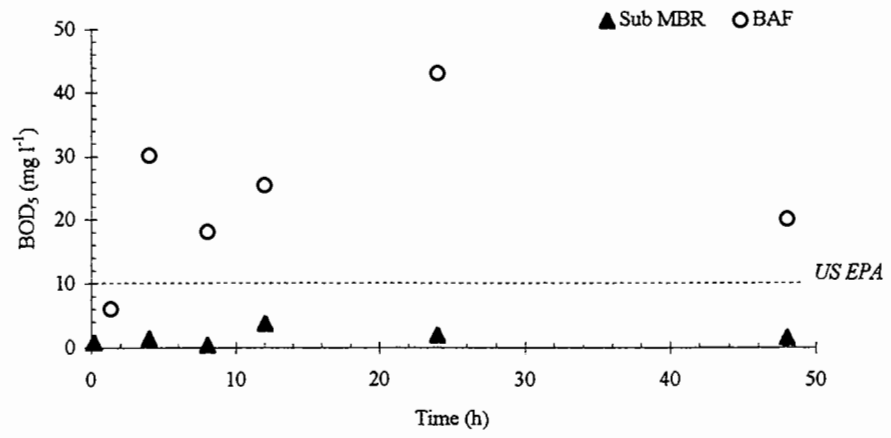


Figure F.21. Submerged MBR and BAF effluent BOD after the 3.5-week feed-off trial.

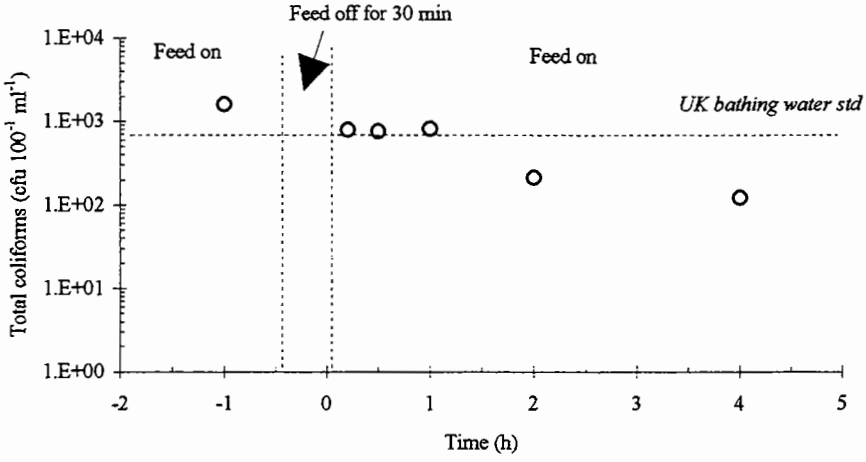


Figure F.22. BAF effluent total coliforms in the 30 min feed-off trial.

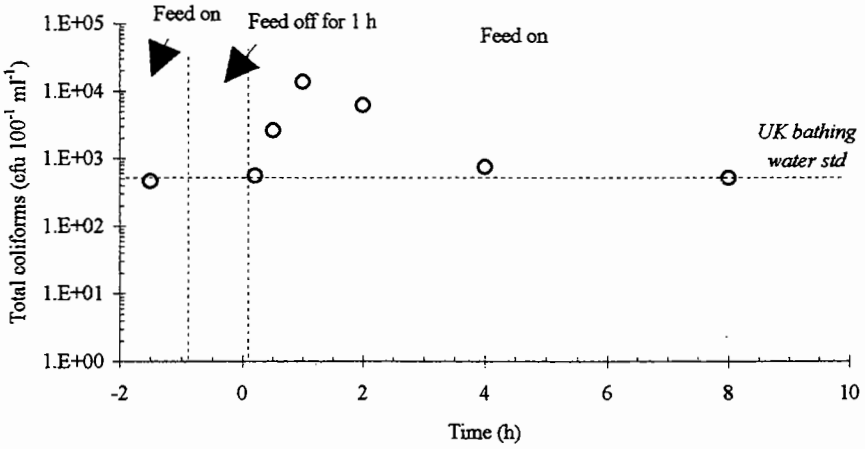


Figure F.23. BAF effluent total coliforms in the 1 h feed-off trial.

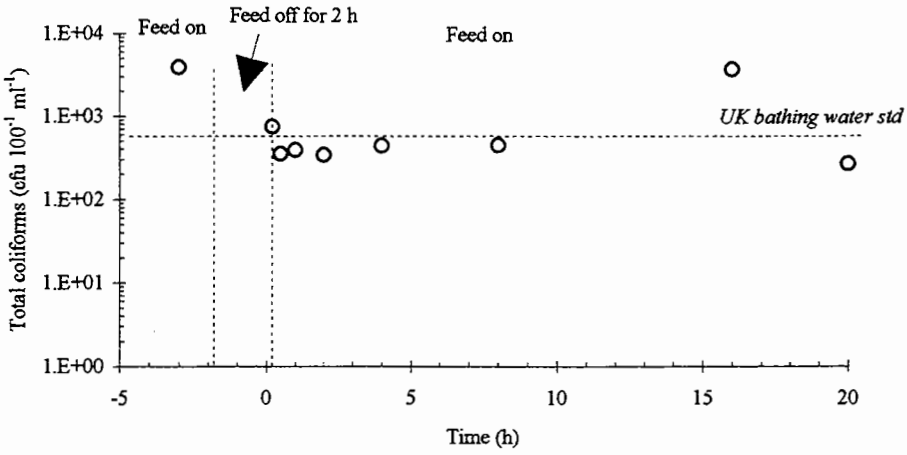


Figure F.24. BAF effluent total coliforms in the 2 h feed-off trial.

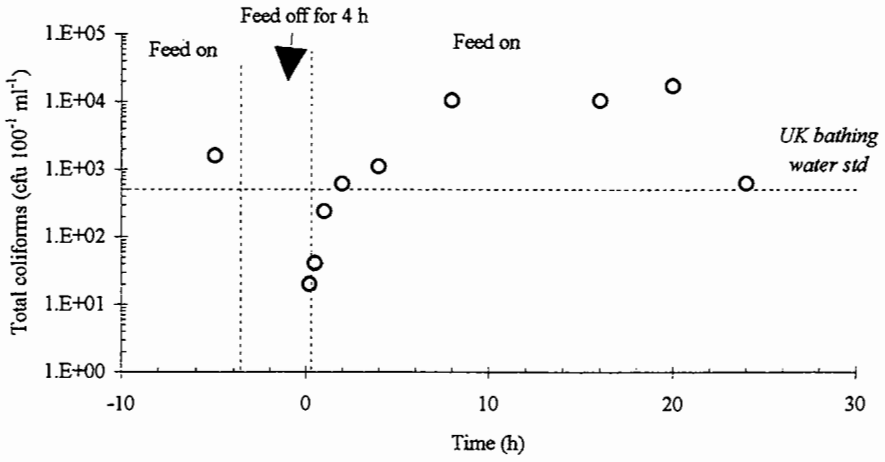


Figure F.25. BAF effluent total coliforms in the 4 h feed-off trial.

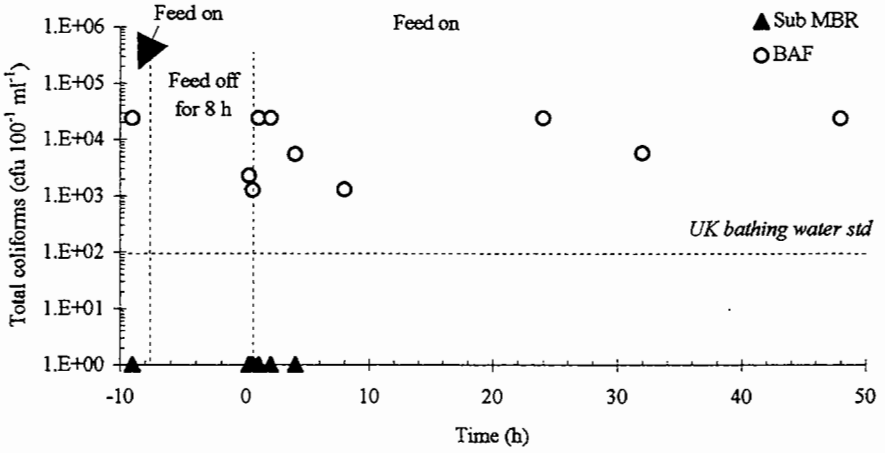


Figure F.26. Submerged MBR and BAF effluent total coliforms in the 8 h feed-off trial.

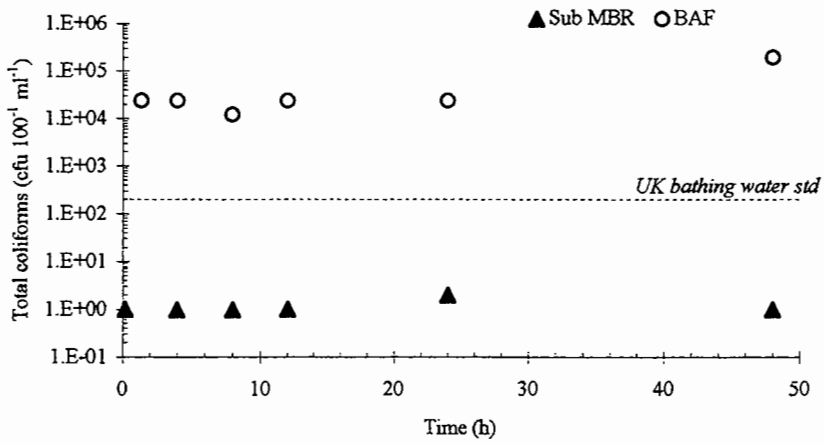


Figure F.27. Submerged MBR and BAF effluent total coliforms in the 3.5-week feed-off trial.

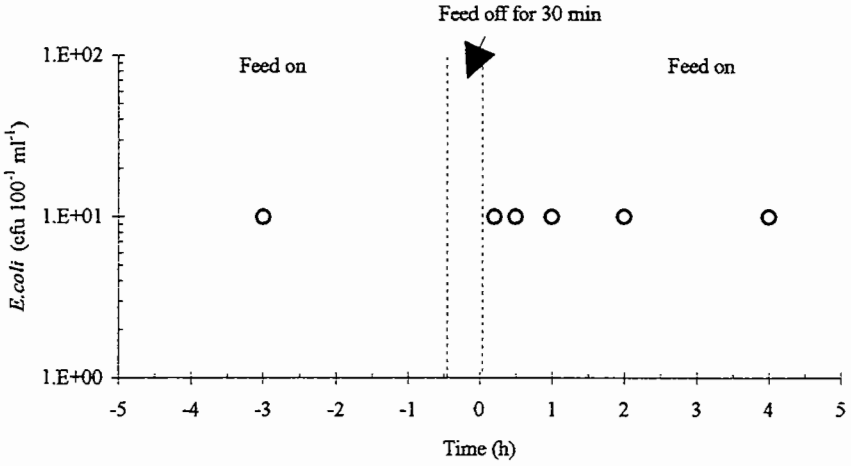


Figure F.28. BAF effluent *E.coli* in the 30 min feed-off trial.

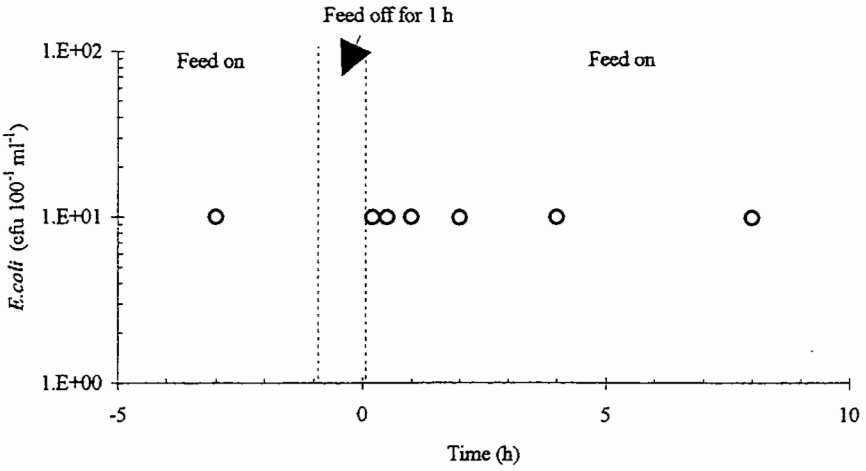


Figure F.29. BAF effluent *E.coli* in the 1 h feed-off trial.

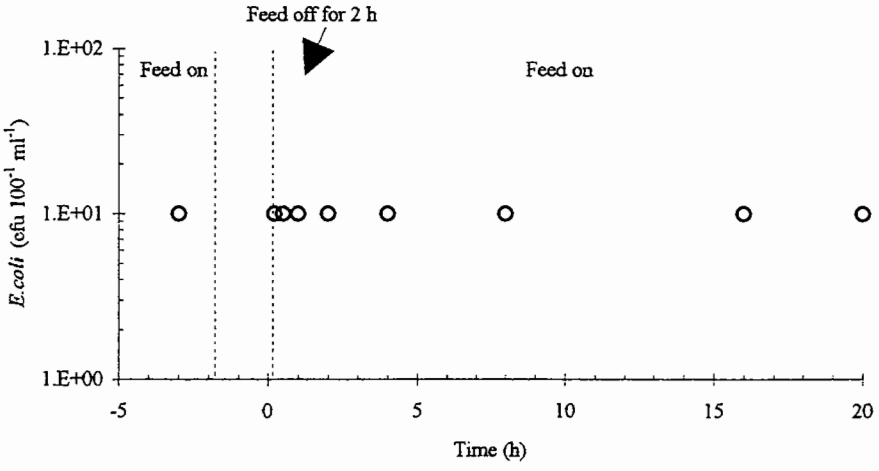


Figure F.30. BAF effluent *E.coli* in the 2 h feed-off trial.

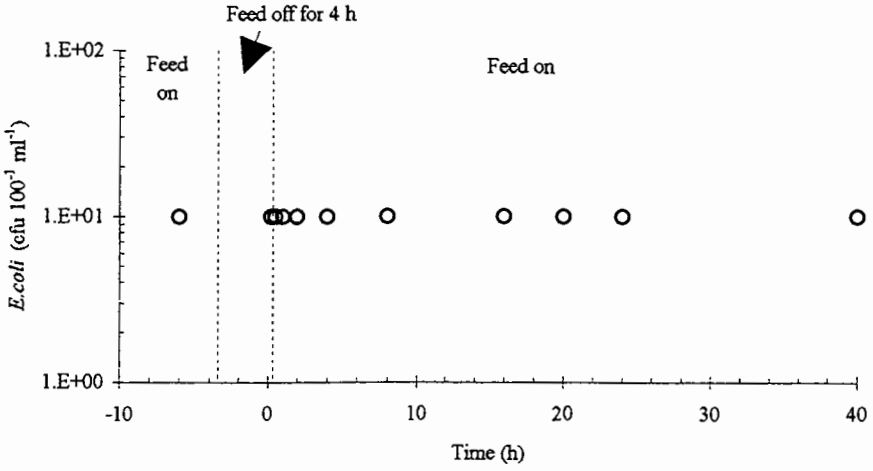


Figure F.31. BAF effluent *E.coli* in the 4 h feed-off trial.

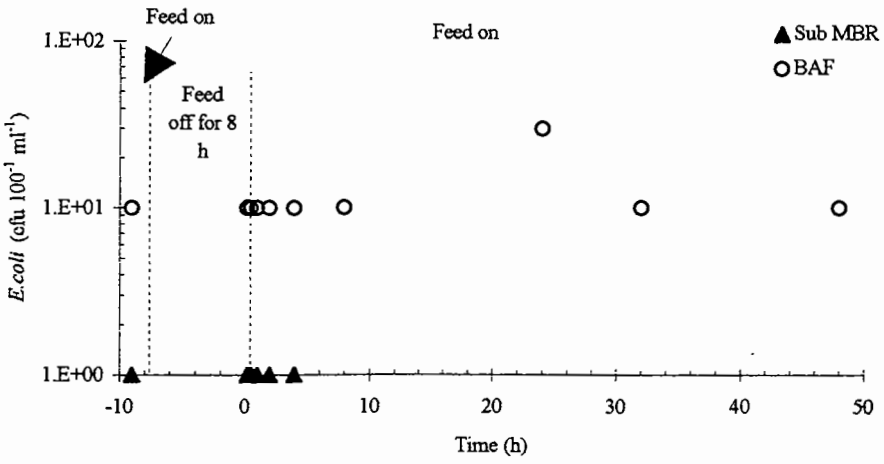


Figure F.32. Submerged MBR and BAF effluent *E.coli* in the 8 h feed-off trial.

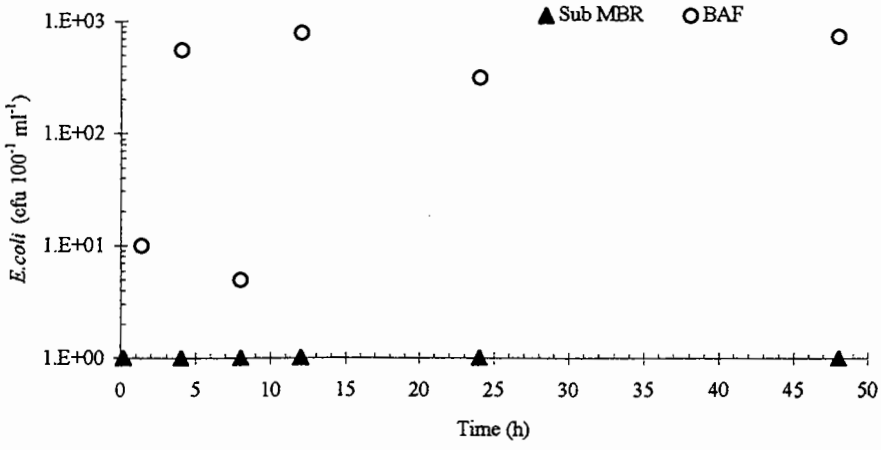


Figure F.33. Submerged MBR and BAF effluent *E.coli* in the 3.5-week feed-off trial.

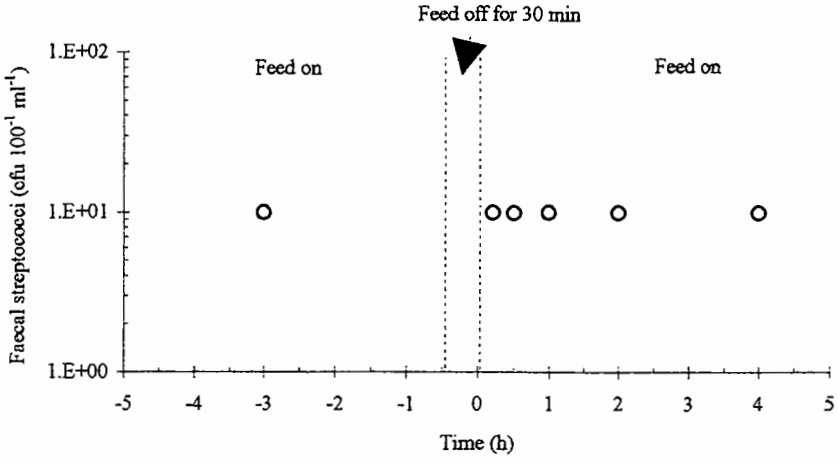


Figure F.34. BAF effluent faecal streptococci in the 30 min feed-off trial.

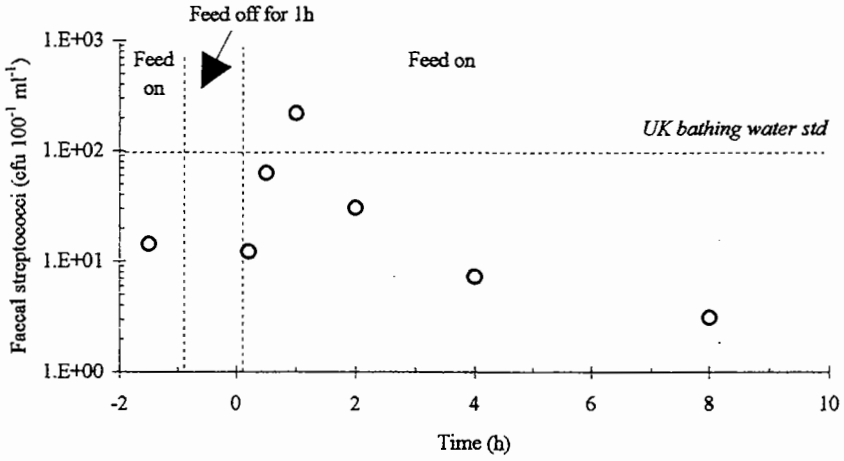


Figure F.35. BAF effluent faecal streptococci in the 1 h feed-off trial.

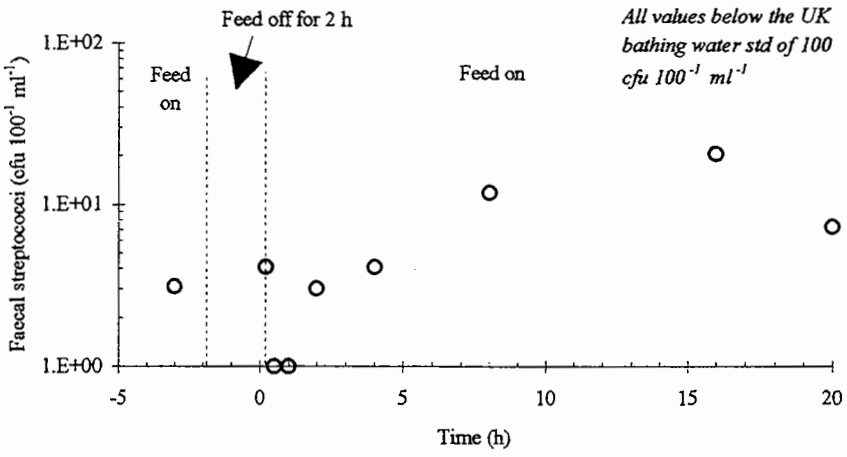
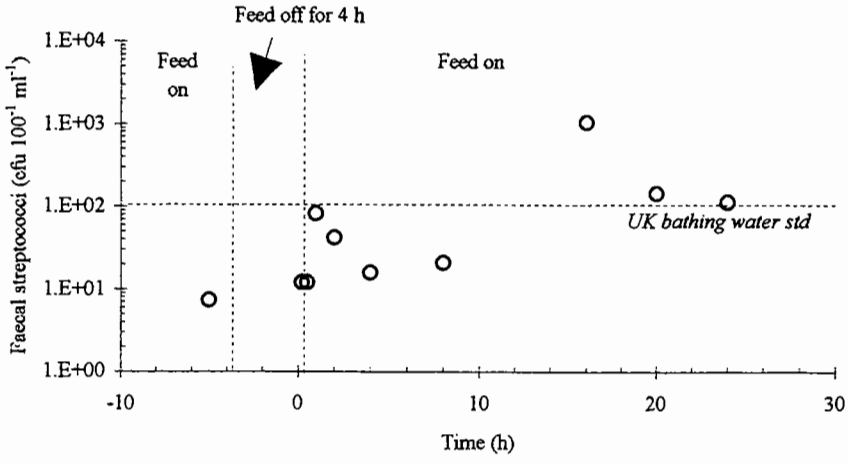
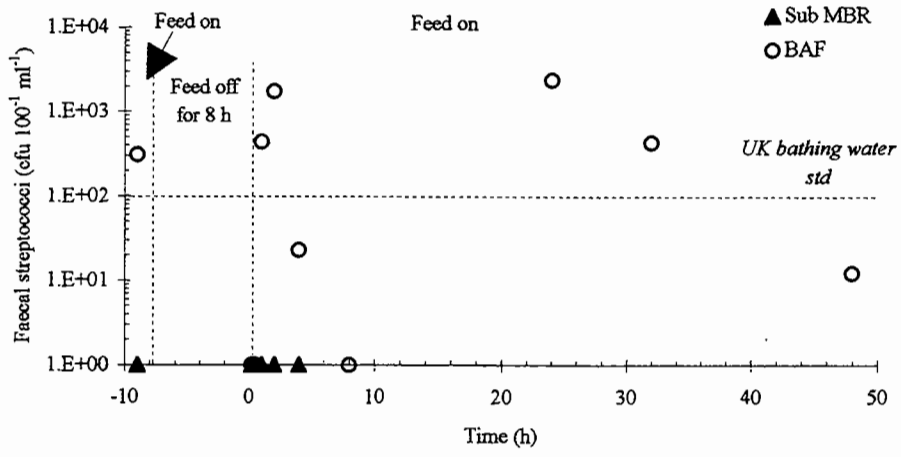


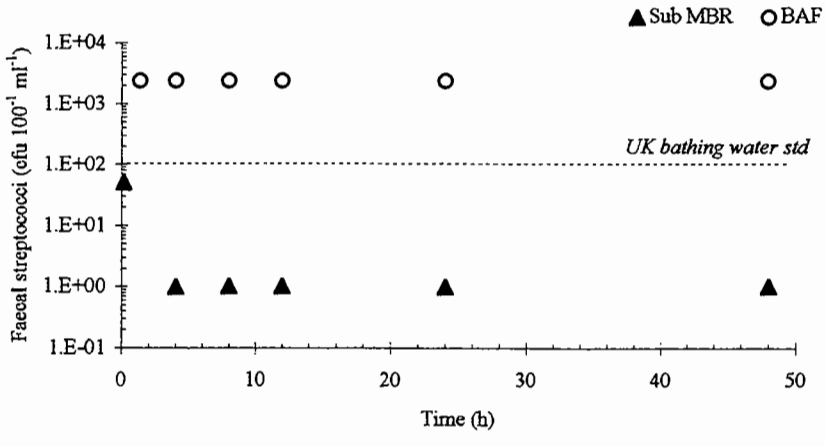
Figure F.36. BAF effluent faecal streptococci in the 2 h feed-off trial.



igure F.37. BAF effluent faecal streptococci in the 4 h feed-off trial.



igure F.38. Submerged MBR and BAF effluent faecal streptococci in the 8 h feed-off trial.



igure F.39. Submerged MBR and BAF effluent faecal streptococci in the 3.5-week feed-off trial.

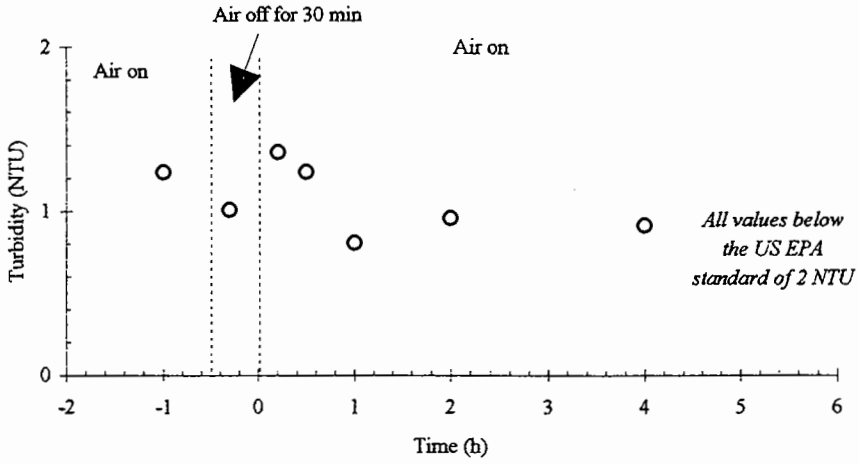


Figure F.40. BAF effluent turbidity in the 30 min air-off trial.

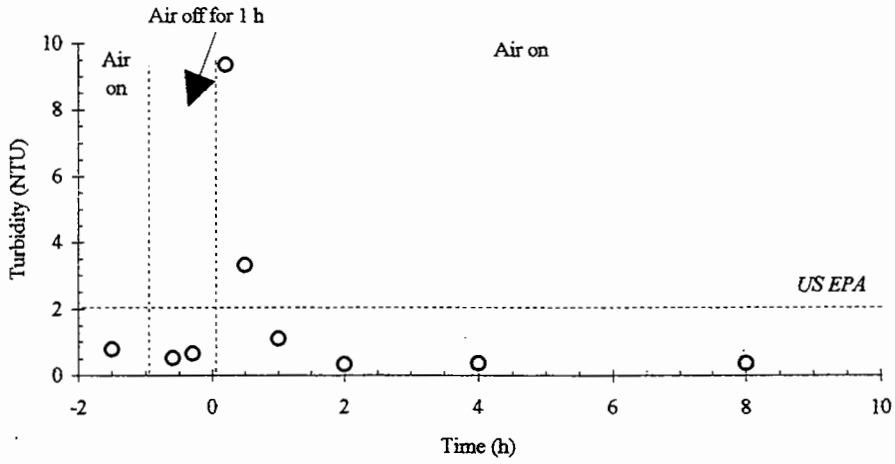


Figure F.41. BAF effluent turbidity in the 1 h air-off trial.

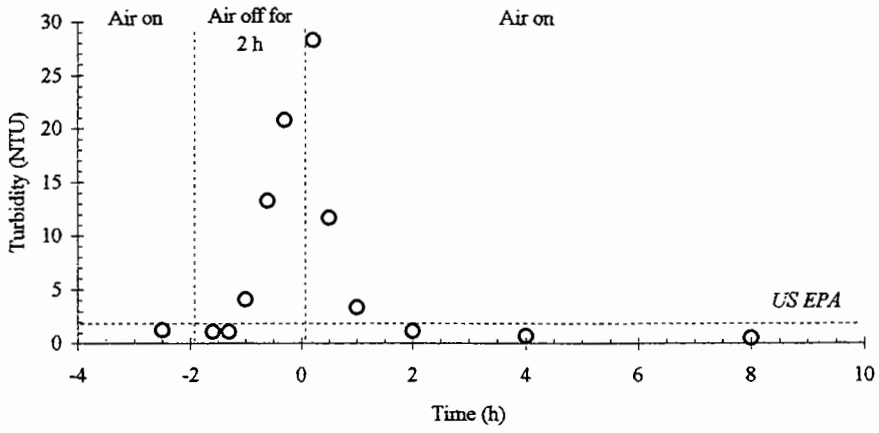


Figure F.42. BAF effluent turbidity in the 2 h air-off trial.

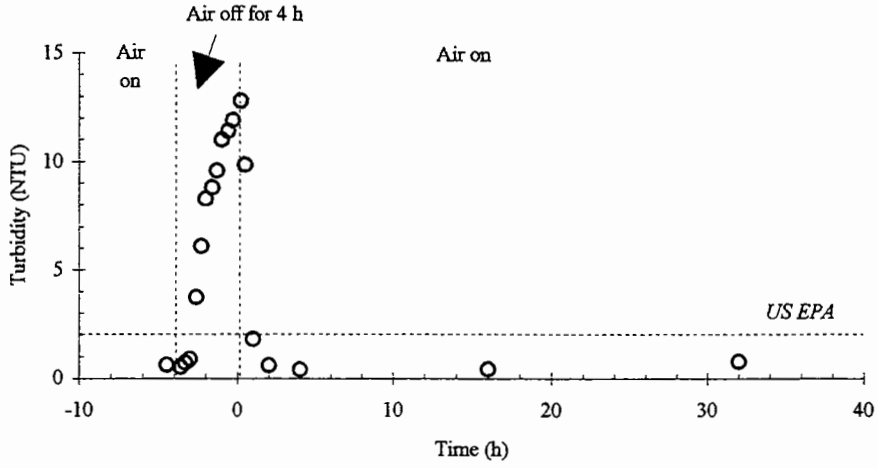


Figure F.43. BAF effluent turbidity in the 4 h air-off trial.

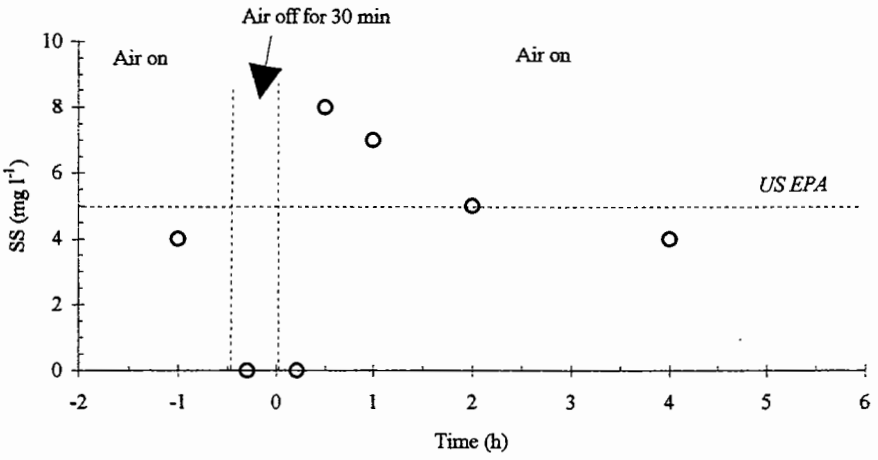


Figure F.44. BAF effluent solids in the 30 min air-off trial.

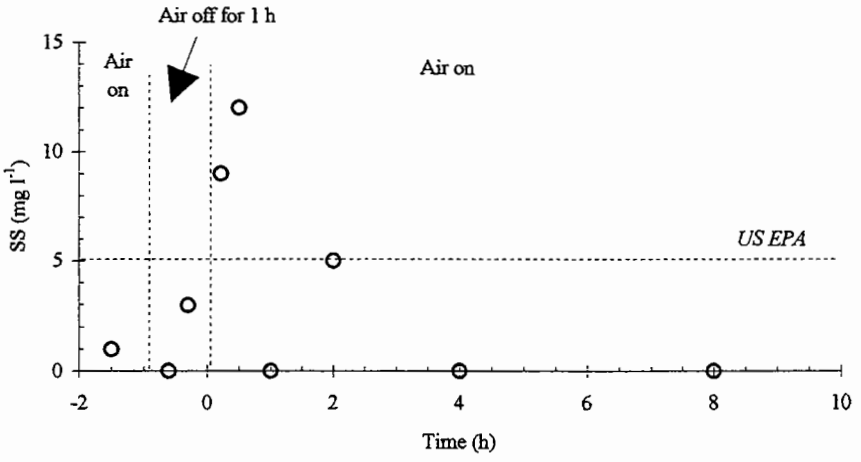


Figure F.45. BAF effluent solids in the 1 h air-off trial.

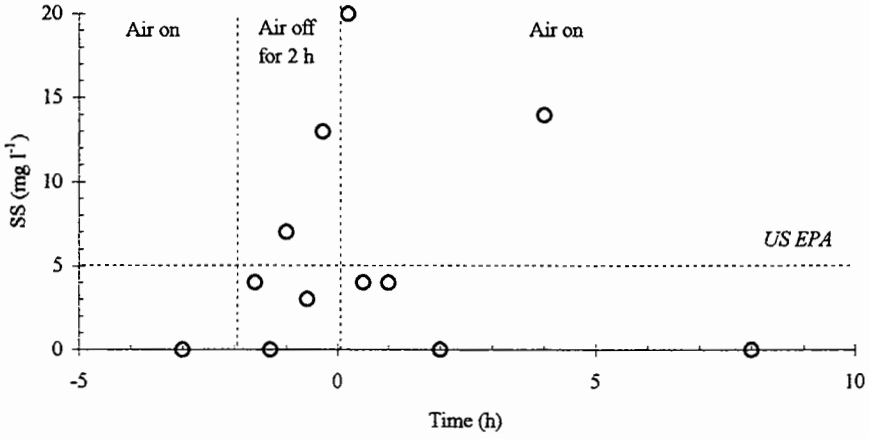


Figure F.46. BAF effluent solids in the 2 h air-off trial.

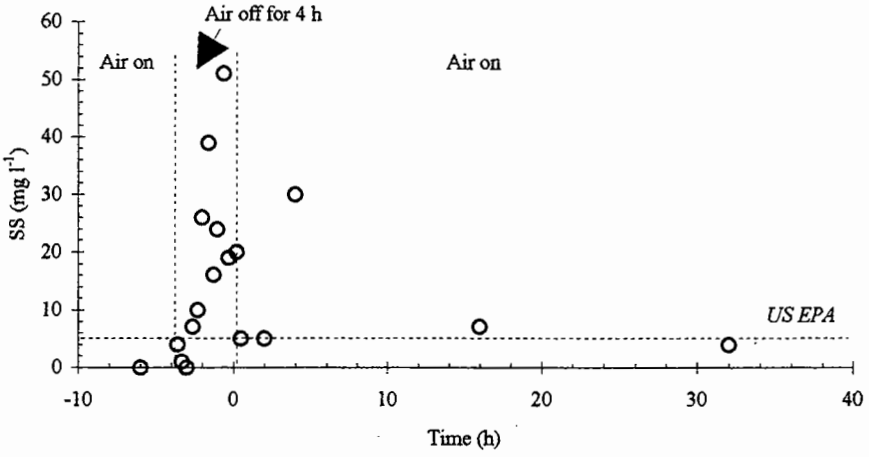


Figure F.47. BAF effluent solids in the 4 h air-off trial.

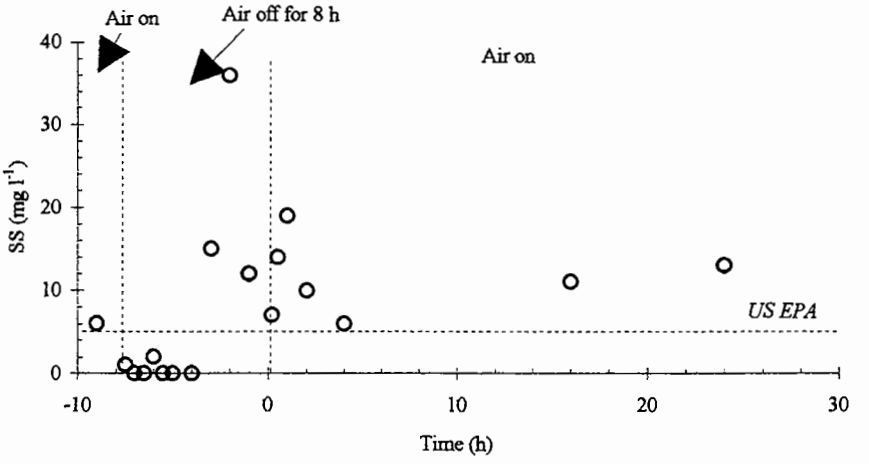


Figure F.48. BAF effluent solids in the 8 h air-off trial.

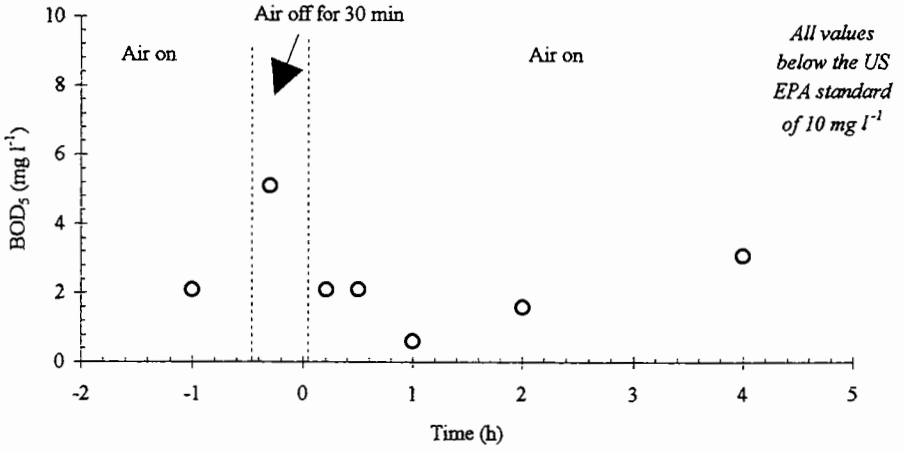


Figure F.49. BAF effluent BOD in the 30 min air-off trial.

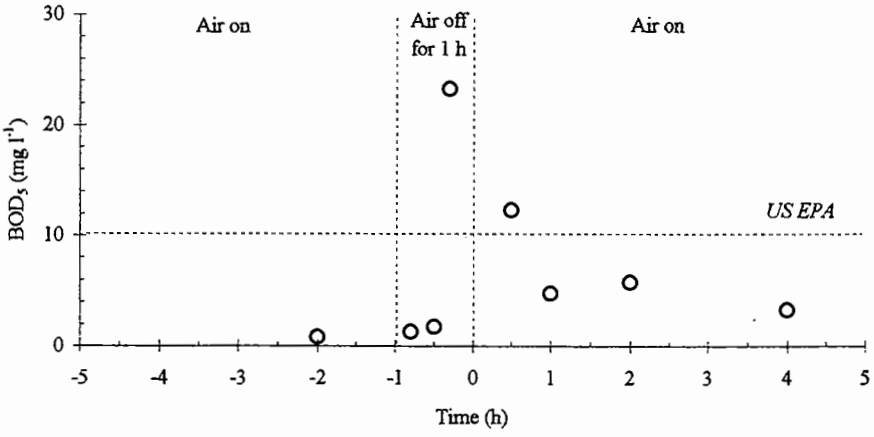


Figure F.50. BAF effluent BOD in the 1 h air-off trial.

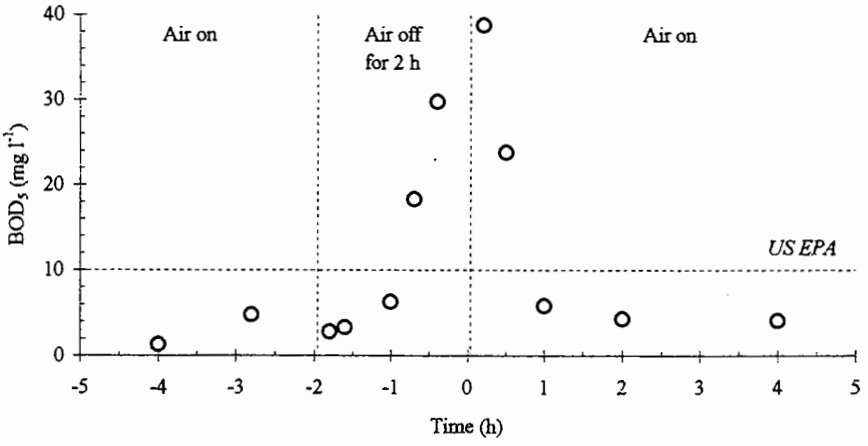
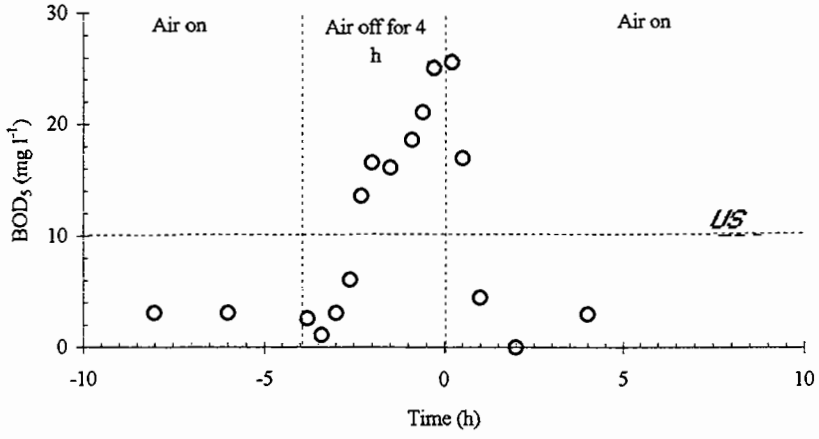
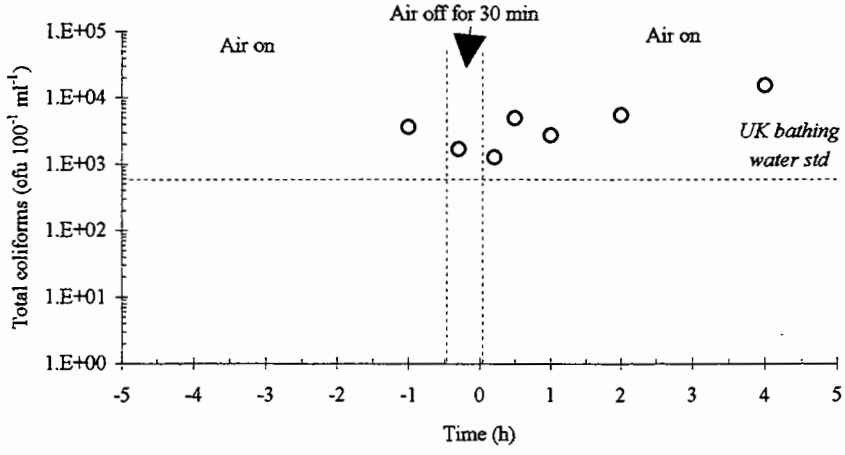


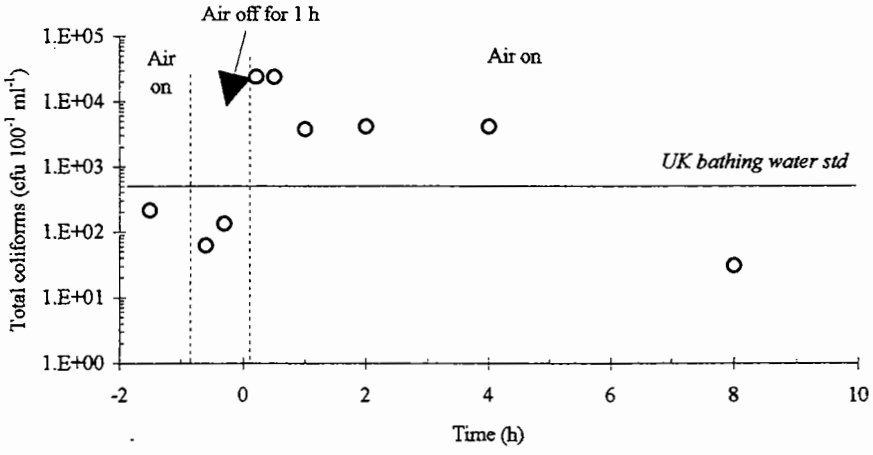
Figure F.51. BAF effluent BOD in the 2 h air-off trial.



gure F.52. BAF effluent BOD in the 4 h air-off trial.



gure F.53. BAF effluent total coliforms in the 30 min air-off trial.



gure F.54. BAF effluent total coliforms in the 1 h air-off trial.

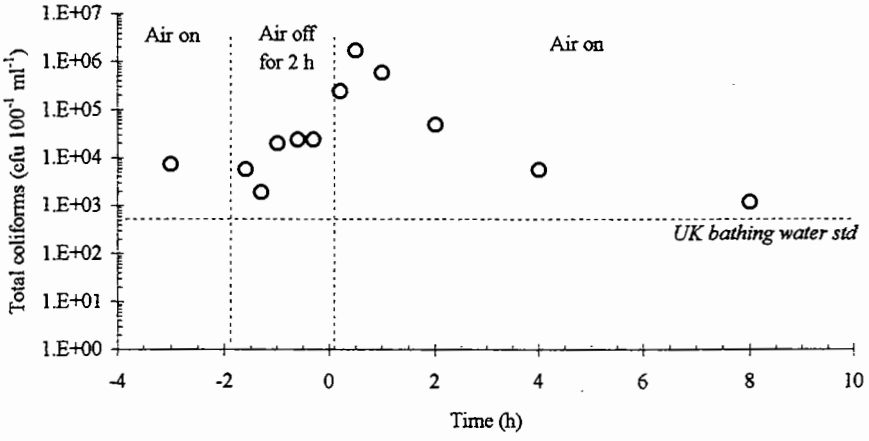


Figure F.55. BAF effluent total coliforms in the 2 h air-off trial.

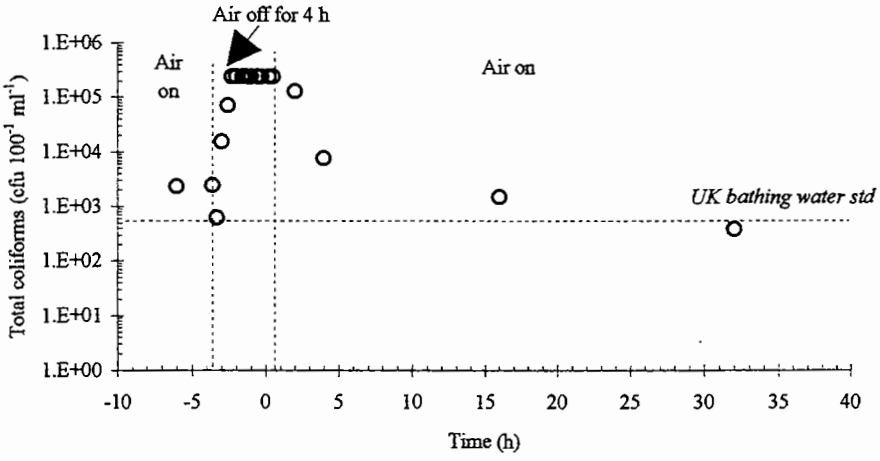


Figure F.56. BAF effluent total coliforms in the 4 h air-off trial.

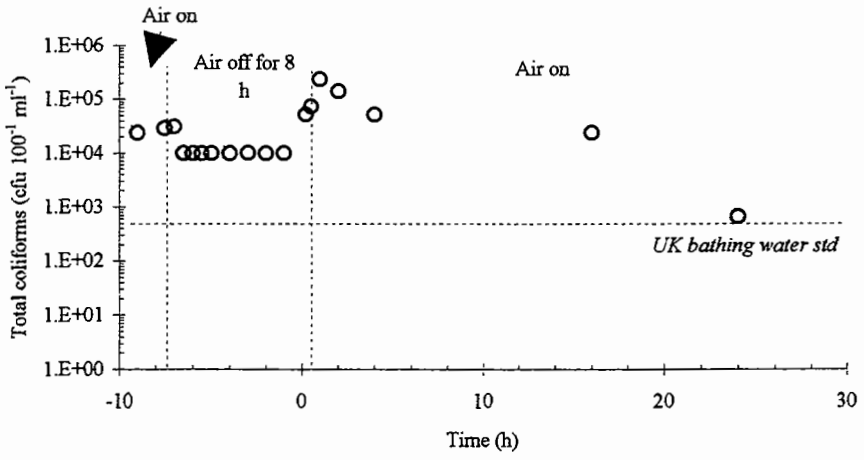


Figure F.57. BAF effluent total coliforms in the 8 h air-off trial.

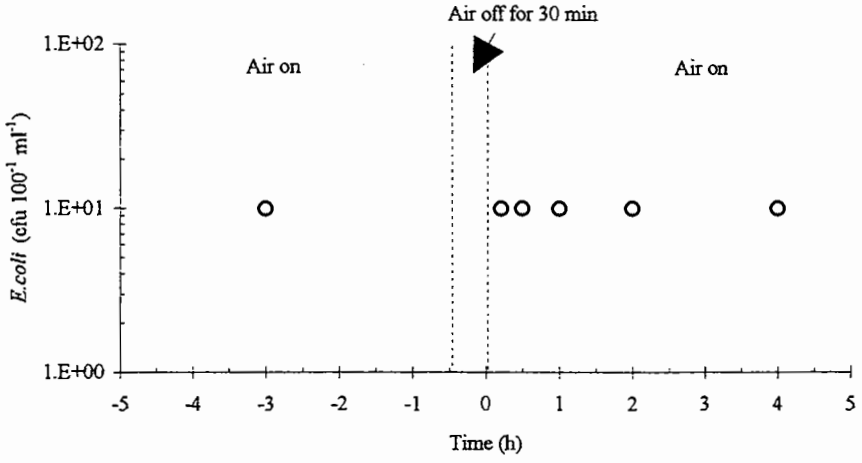


Figure F.58. BAF effluent *E.coli* in the 30 min air-off trial.

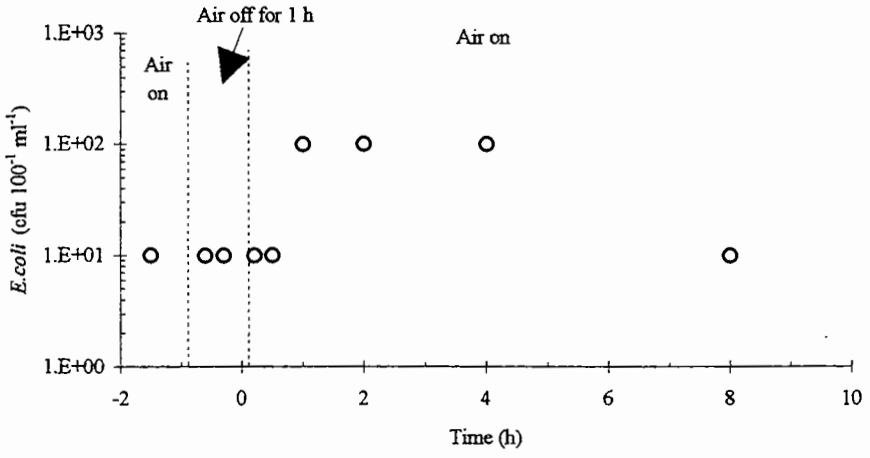


Figure F.59. BAF effluent *E.coli* in the 1 h air-off trial.

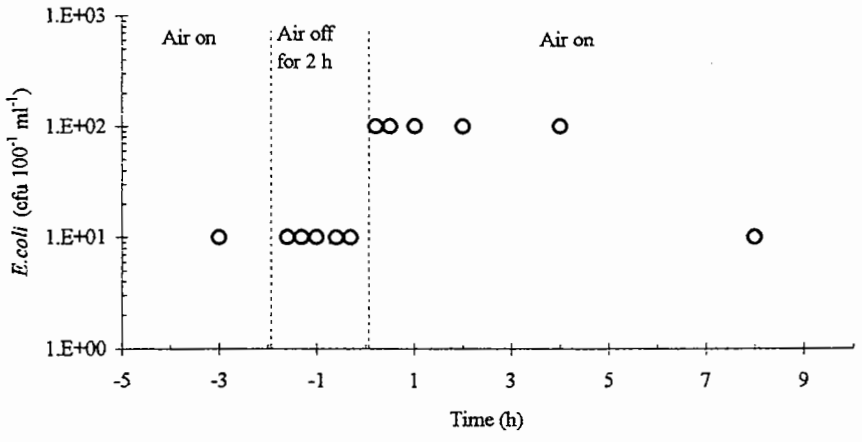


Figure F.60. BAF effluent *E.coli* in the 2 h air-off trial.

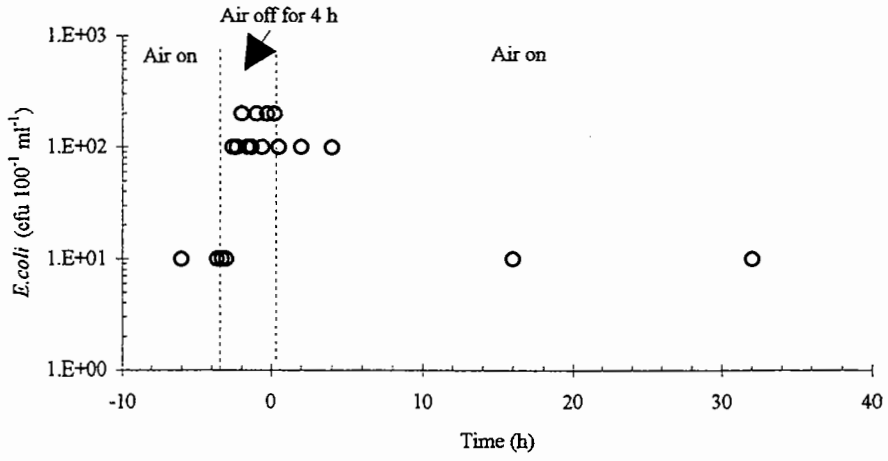


Figure F.61. BAF effluent *E.coli* in the 4 h air-off trial.

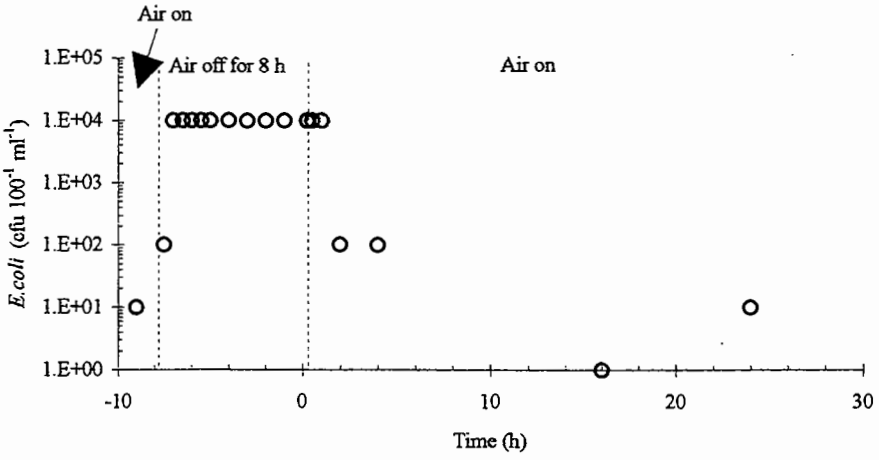


Figure F.62. BAF effluent *E.coli* in the 8 h air-off trial.

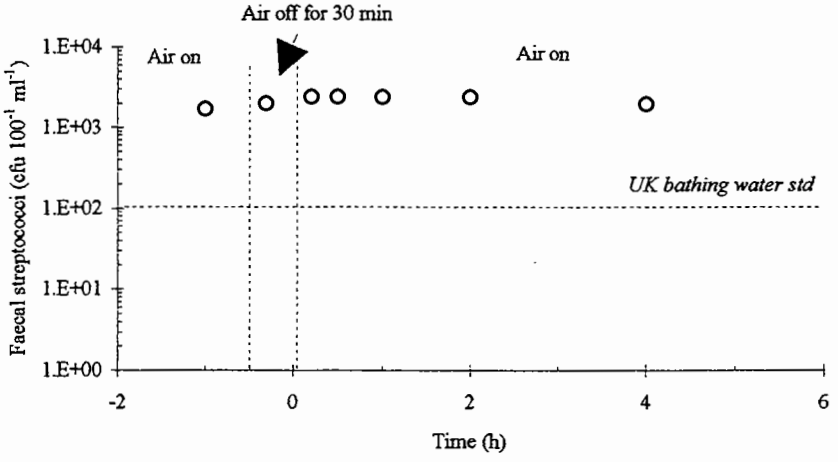


Figure F.63. BAF effluent faecal streptococci in the 30 min air-off trial.

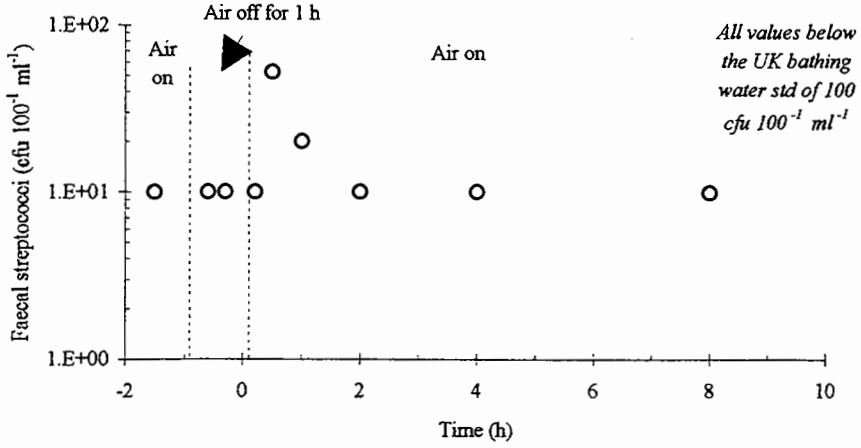


Figure F.64. BAF effluent faecal streptococci in the 1 h air-off trial.

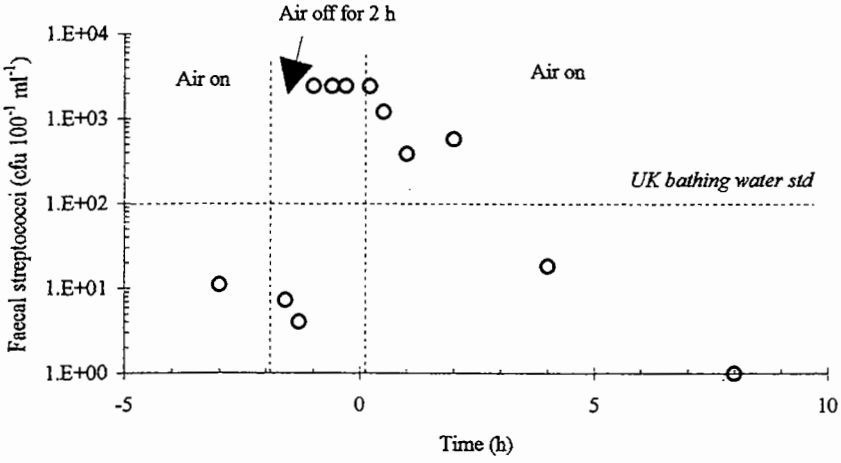


Figure F.65. BAF effluent faecal streptococci in the 2 h air-off trial.

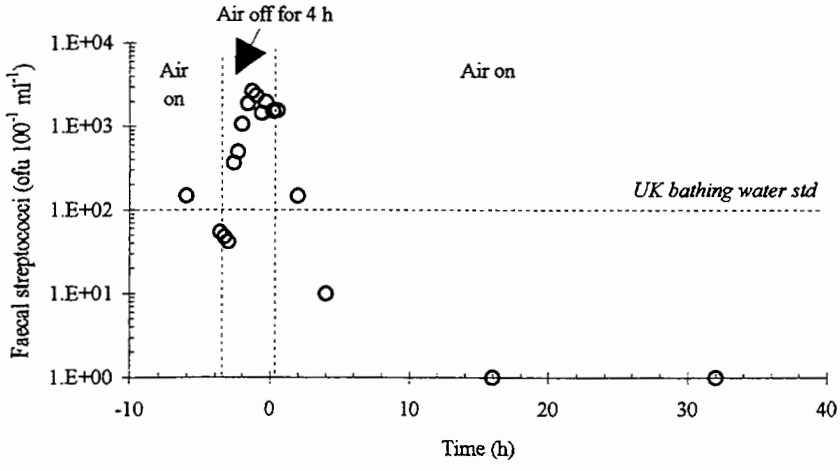


Figure F.66. BAF effluent faecal streptococci in the 4 h air-off trial.

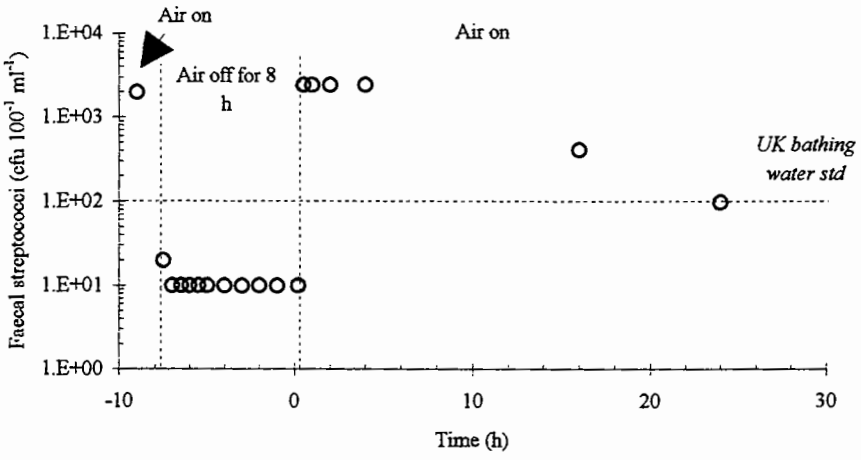


Figure F.67. BAF effluent faecal streptococci in the 8 h air-off trial.

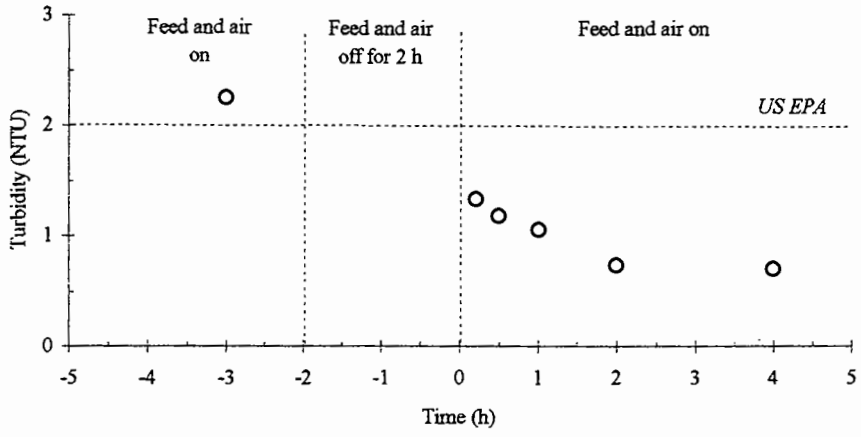


Figure F.68. BAF effluent turbidity in the 2 h feed and air-off trial.

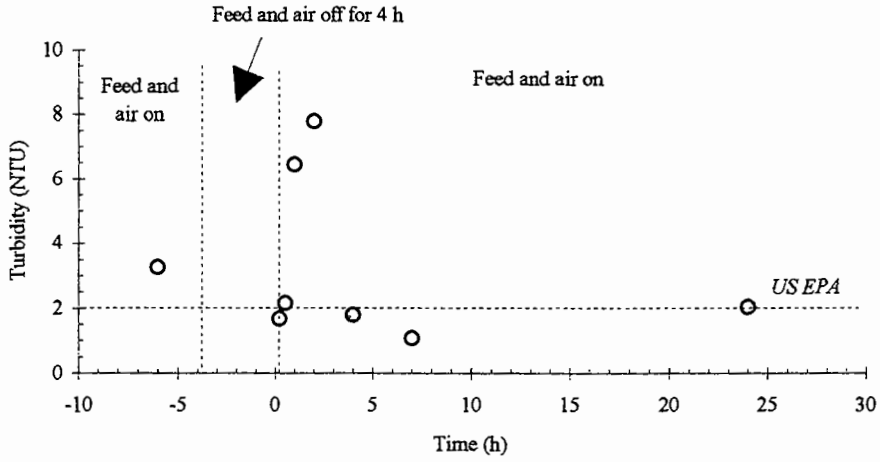


Figure F.69. BAF effluent turbidity in the 4 h feed- and air-off trial.

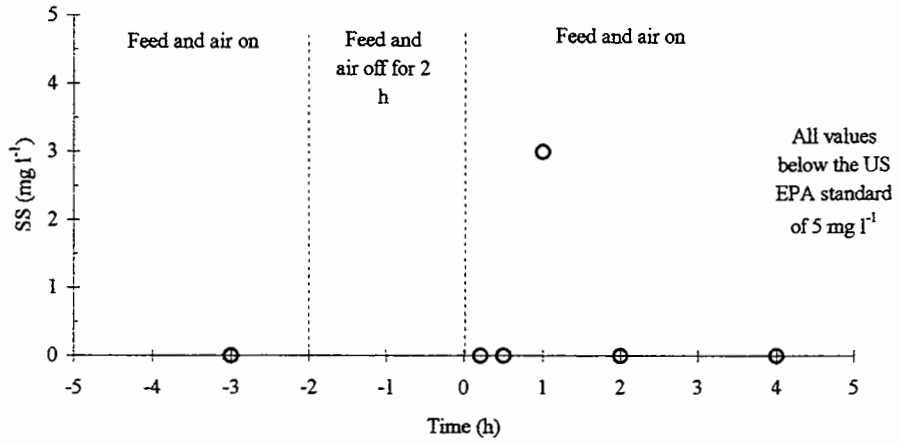


Figure F.70. BAF effluent solids in the 2 h feed- and air-off trial.

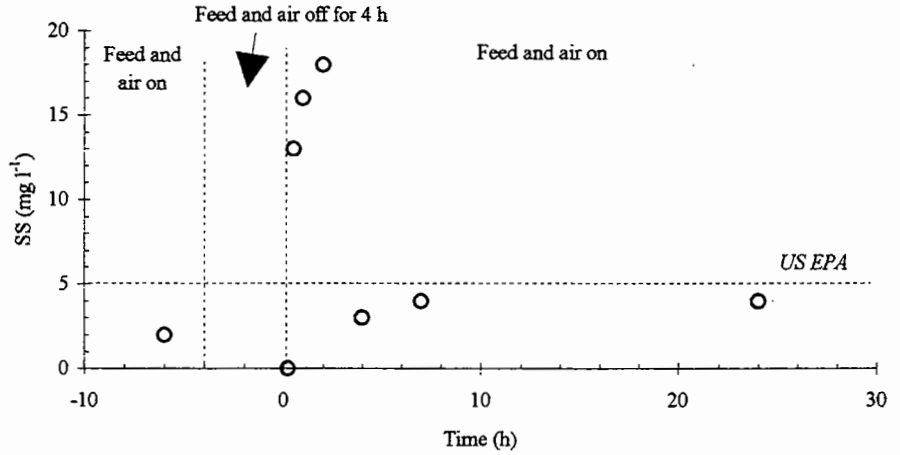


Figure F.71. BAF effluent solids in the 4 h feed- and air-off trial.

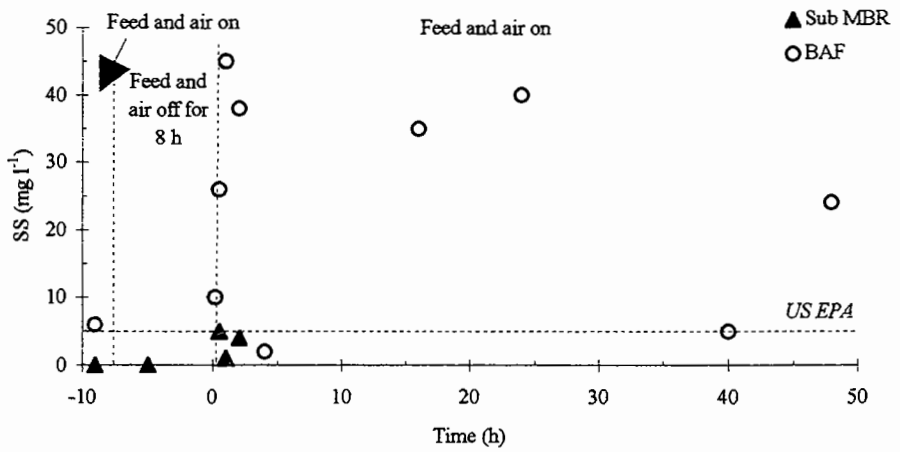
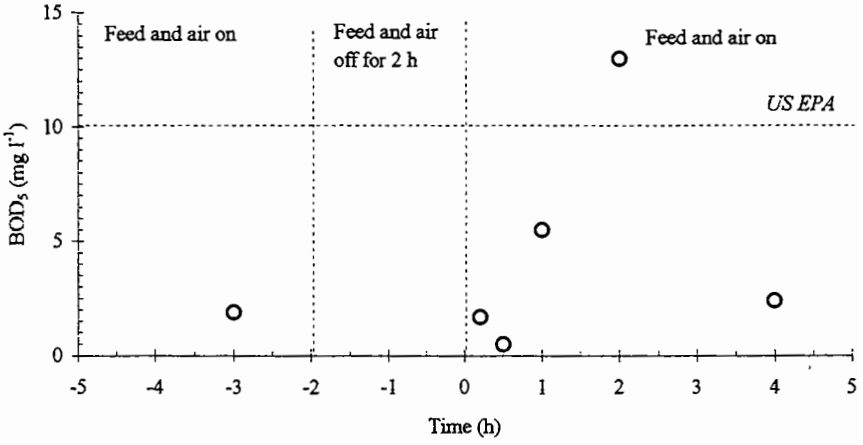
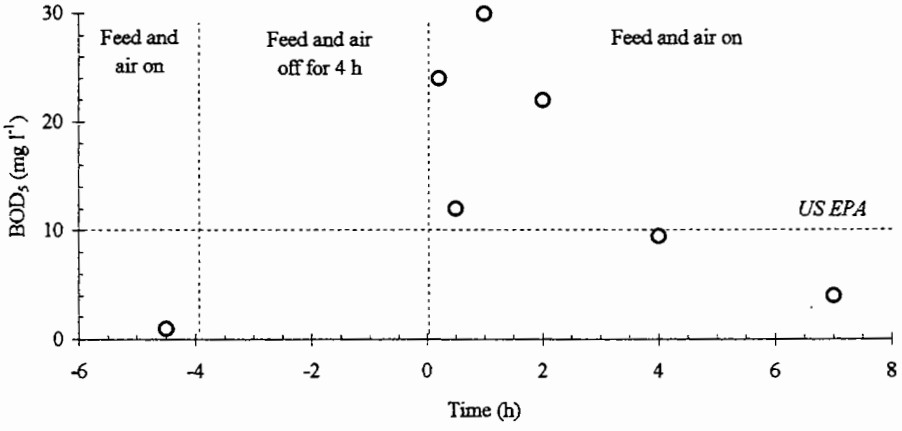


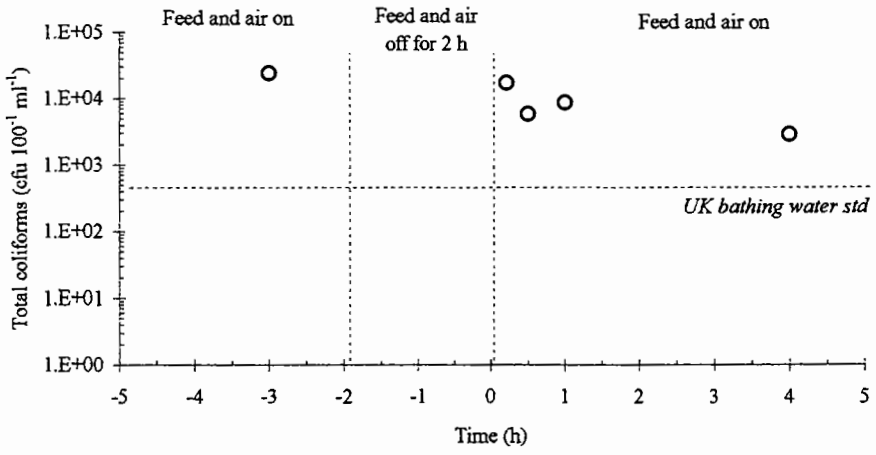
Figure F.72. Submerged MBR and BAF effluent solids in the 8 h feed- and air-off trial.



re F.73. BAF effluent BOD in the 2 h feed- and air-off trial.



e F.74. BAF effluent BOD in the 4 h feed- and air-off trial.



e F.75. BAF effluent total coliforms in the 2 h feed- and air-off trial.

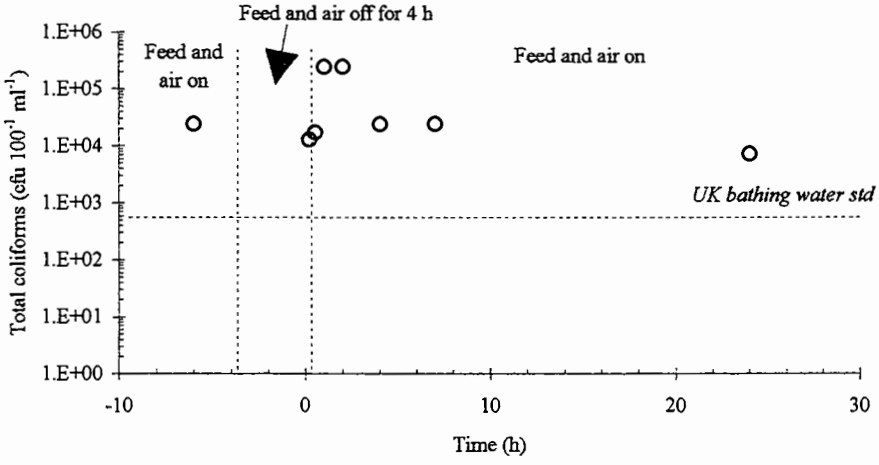


Figure F.76. BAF effluent total coliforms in the 4 h feed- and air-off trial.

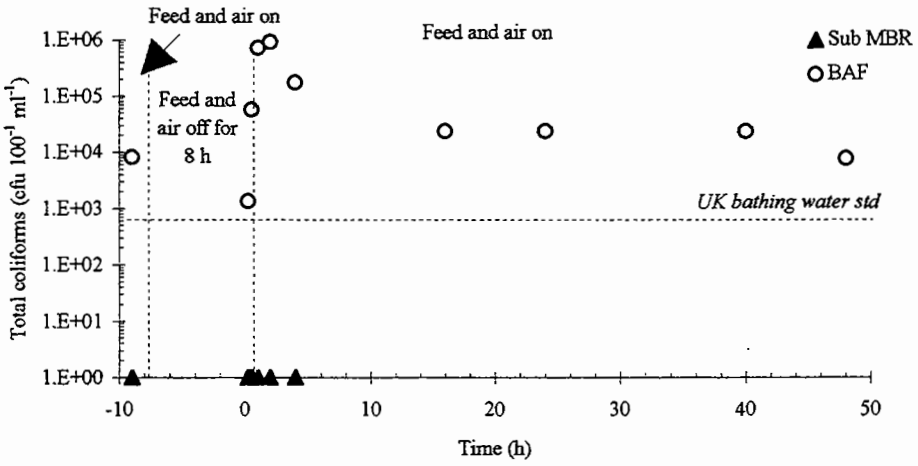


Figure F.77. Submerged MBR and BAF effluent total coliforms in the 8 h feed- and air-off trial.

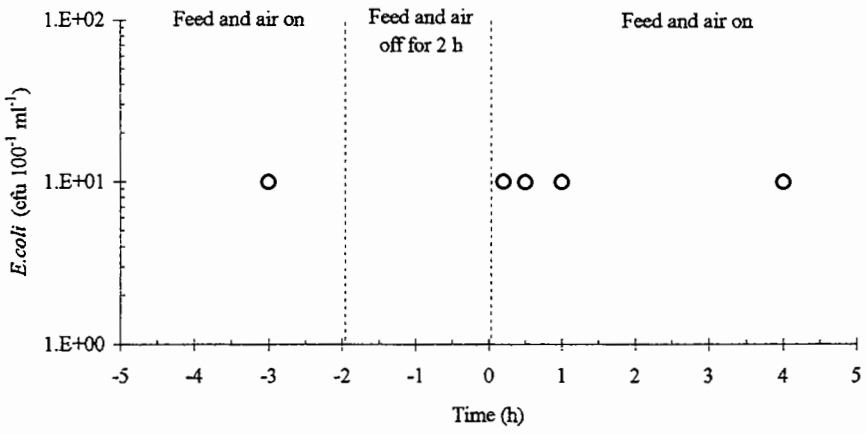
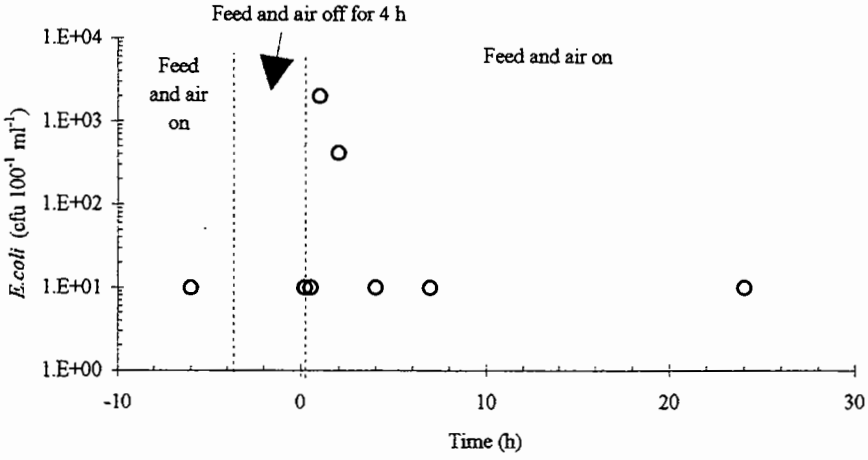
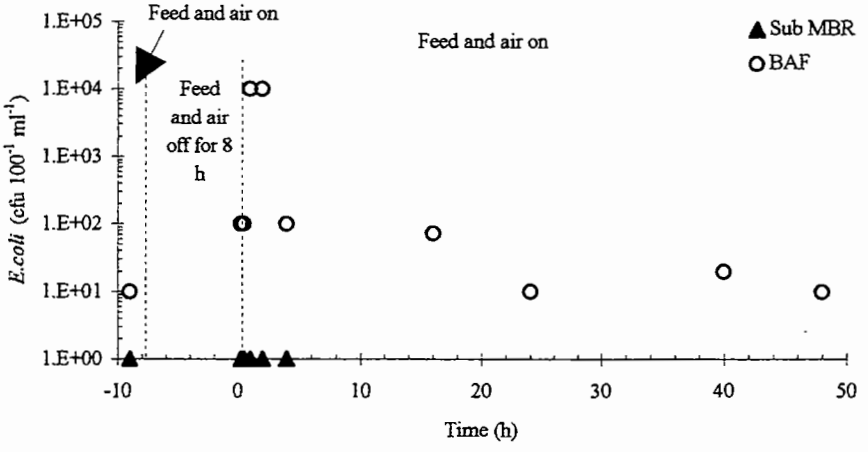


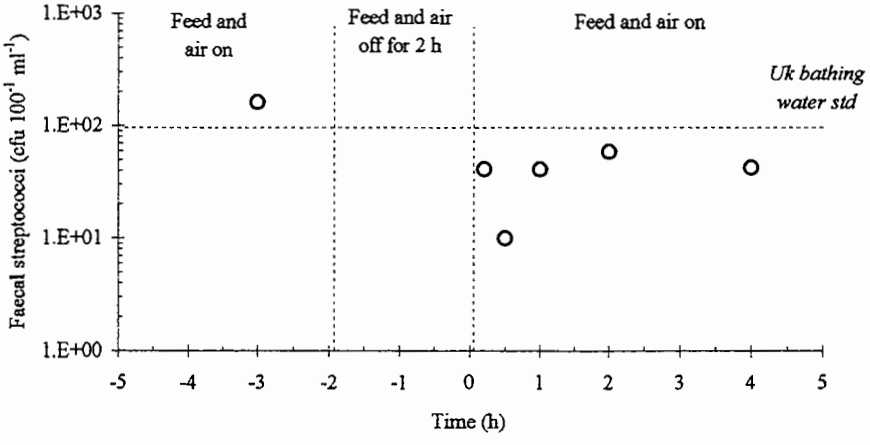
Figure F.78. BAF effluent *E.coli* in the 2 h feed- and air-off trial.



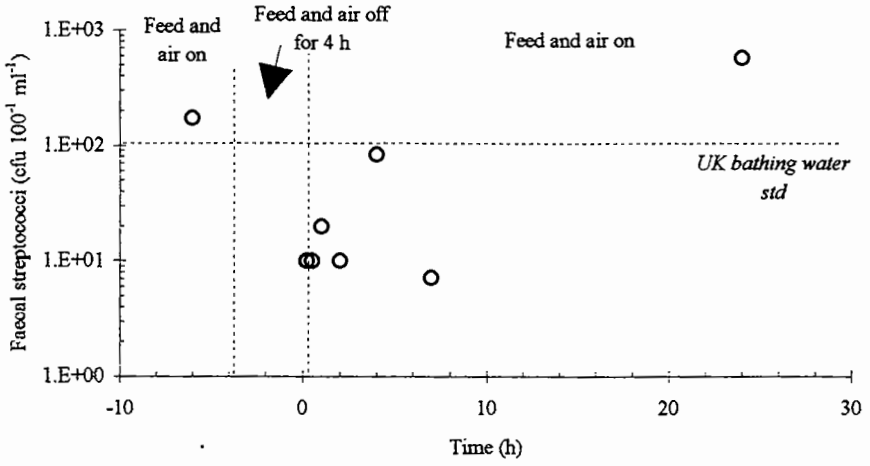
re F.79. BAF effluent *E.coli* in the 4 h feed- and air-off trial.



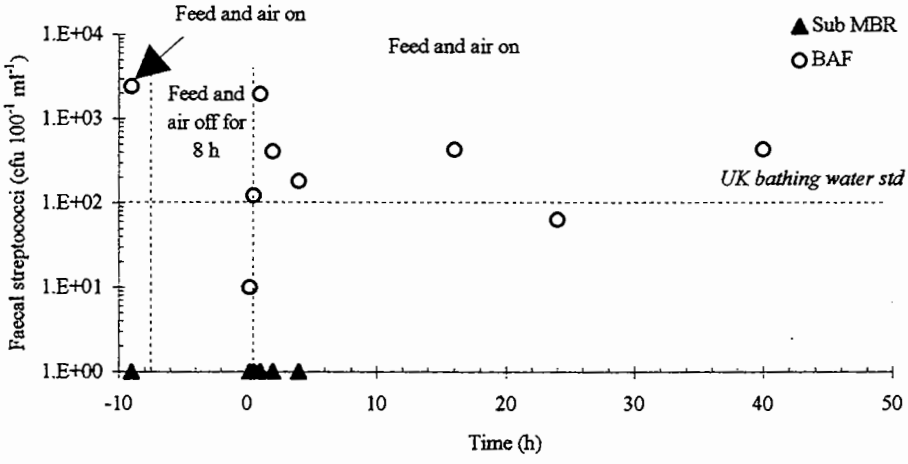
re F.80. Submerged MBR and BAF effluent *E.coli* in the 8 h feed- and air-off trial.



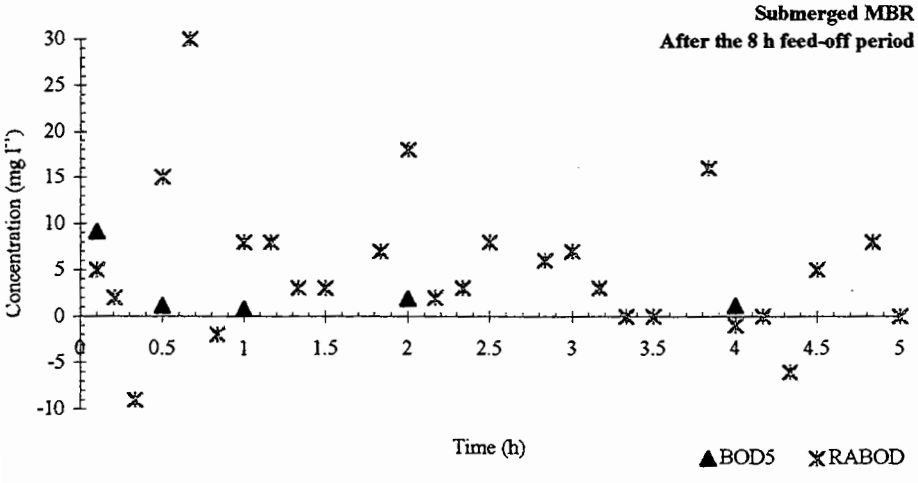
re F.81. BAF effluent faecal streptococci in the 2 h feed- and air-off trial.



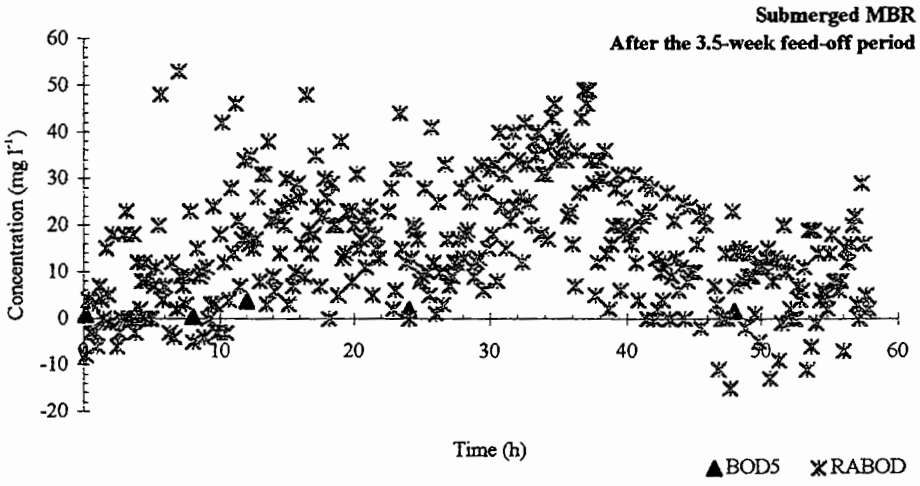
ure F.82. BAF effluent faecal streptococci in the 4 h feed- and air-off trial.



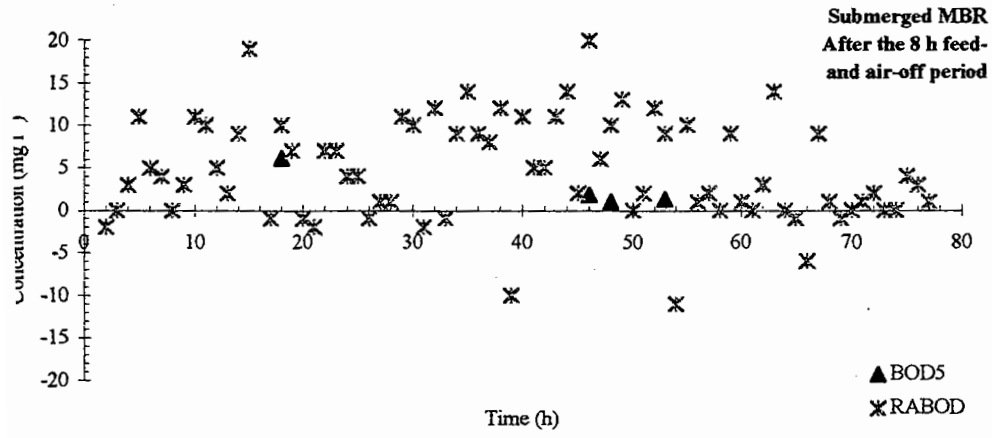
ıre F.83. Submerged MBR and BAF effluent faecal streptococci in the 8 h feed- and air-off trial.



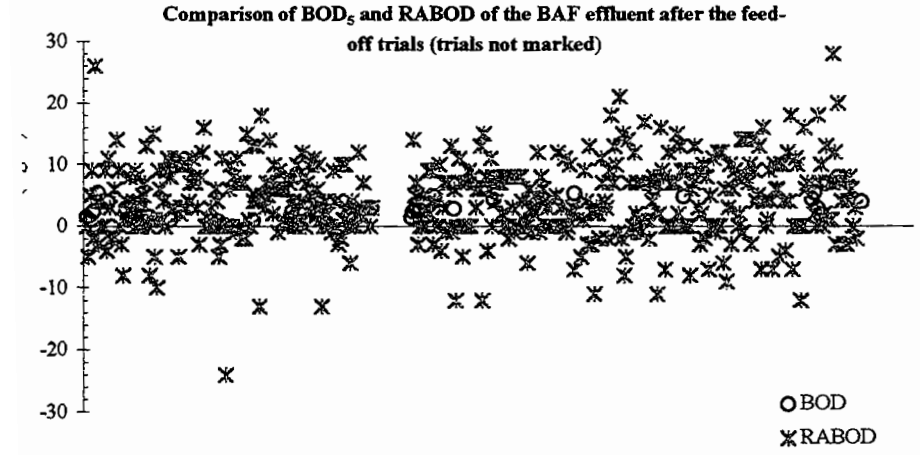
re F.84. Comparison of BOD_5 and RABOD of the submerged MBR effluent after the 8 h feed-off ıd.



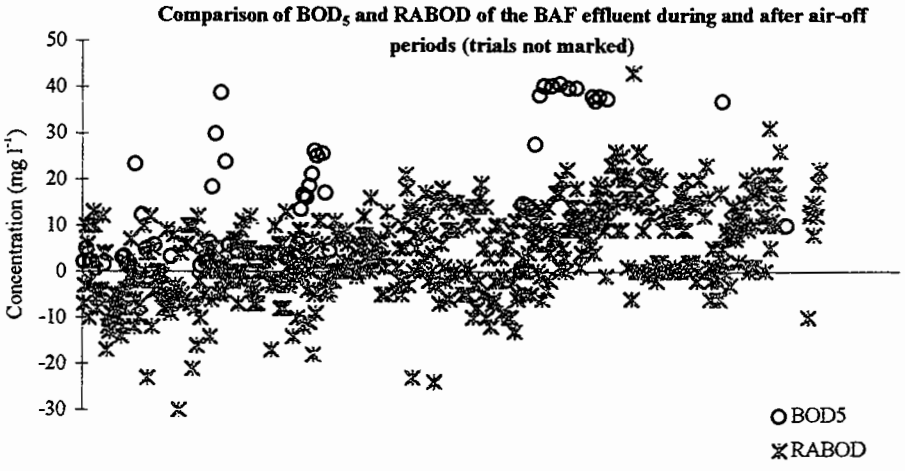
re F.85. Comparison of BOD_5 and RABOD of the submerged MBR effluent after the 3.5-week ·off period.



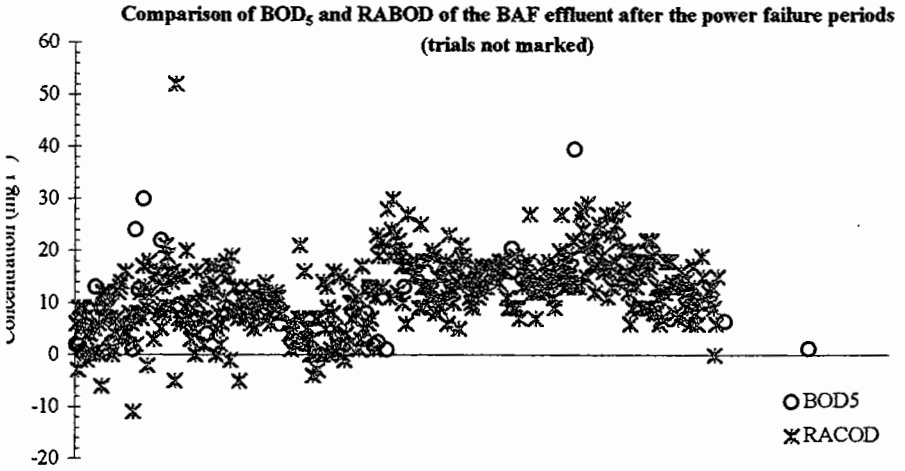
·e F.86. Comparison of BOD_5 and RABOD of the submerged MBR effluent after the 8 h power ɜ period.



ɜ **F.87.** Comparison of BOD_5 and RABOD of the BAF effluent after the feed-off periods.



re F.88. Comparison of BOD_5 and RABOD of the BAF effluent after and during the air-off ods.



re F.89. Comparison of BOD_5 and RABOD of the BAF effluent after the power failure periods.

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19 May 1999

Dear [Sir/Madam]

Thank you for taking the time to answer this questionnaire. Research has been conducted over the last two years examining the potential for reusing water within a home or office. This has focussed on the recycling of bath, shower and hand basin water (grey water) for toilet flushing and irrigation purposes. The benefits of this can be up to a 30% saving in drinking water use, reducing the overall demand on supplies and costs to customers. The results of this study will provide important information required for the research into the efficacy of domestic grey water recycling.

One of the problems in assessing the suitability of treatment systems for grey water is the effect of certain 'out of the ordinary' contaminants on the system performance. However, little is known of the type and frequency of substances that are drained away with grey water. The aim of this questionnaire is to help improve understanding of the types of substances that may be involved. This will enable us to test the technologies under investigation to see what effects, if any, these substances may have.

Thank you again for your time, if you have any queries about the questionnaire itself or water recycling in general please do not hesitate to call.

Yours sincerely

[Dr Bruce Jefferson]

Table G.1. Questionnaire on consumers' behavioural patterns.

Sex	Male ☐	Female ☐
Age	<25 ☐	26-40 ☐ >40 ☐

Please tick the appropriate box showing the frequency with which the substances below have been drained away with **bath, hand basin and shower water** in the home accidentally or intentionally)

	Frequency					
Substance	Once a week	Once a month	Once a year	Occasional	Once	Never
Bleach	☐	☐	☐	☐	☐	☐
Oil (body/food)	☐	☐	☐	☐	☐	☐
White spirits	☐	☐	☐	☐	☐	☐
Hair dye	☐	☐	☐	☐	☐	☐
Alcohol	☐	☐	☐	☐	☐	☐
Perfume	☐	☐	☐	☐	☐	☐
Body fluids e.g. blood, urine	☐	☐	☐	☐	☐	☐
Mud	☐	☐	☐	☐	☐	☐
Oil (car/bike)	☐	☐	☐	☐	☐	☐
Pet care products e.g. shampoo	☐	☐	☐	☐	☐	☐
Detergents e.g. carpet cleaner	☐	☐	☐	☐	☐	☐
Food	☐	☐	☐	☐	☐	☐
Bathroom cleaners	☐	☐	☐	☐	☐	☐
Washing powder	☐	☐	☐	☐	☐	☐
Make up remover	☐	☐	☐	☐	☐	☐
Caustic soda	☐	☐	☐	☐	☐	☐

Other (please specify)

From the above list, please list the three substances which you perceive to be the most harmful to the environment

Table G.2. Test substances used in the survey respirometry work.

Test substance	Brand	Ingredients
Bleach	Tesco's bleach 1314C	Amongst other ingredients less than 5% of sodium hypochlorite
Food oil	Tesco's sunflower oil Sainsbury's pure vegetable oil	Sunflower oil Vegetable oil
White spirit	Great Mills' white spirit	n/a
Hair dye	L'Oréal Excellence Creme hair colourant	Colourant: aqua, cetearyl alcohol, propylene glycol, deceth-3 laureth-12, ammonium hydroxide, oleth-30, lauric acid, hexadimethrine chlorine, glycol distearate, polyquaternium-22, silica dimethyl silylate, ethanolamine, pentasodium pentetate, sodium metabisulfite, resorcinol, carbomer, p-phyenylenediamine, t-butyl hyroquinone, m-aminophenol, 2,3-diaminophonoxoxyethanol HCl, hydroxybenzomorpholine, parfum (C3418/2) Developer: aqua, hydrogen peroxide, cetearyl alcohol, tridiceth-2 carboxamide mea, ceteareth-30, glycerin, pentasodium pentetate, sodium stannate, tetrasodium pyrophosphate (C11174/1)
Alcohol	Ethanol	Ethanol (diluted to 40%) AnalaR, BDH Merck, Poole, UK
Perfume	Joe Bloggs Juice Female eau de toilette spray	Alcohol denat, parfym, propylene glycol
Body fluids	Urea	14.8 mg l^{-1} urea (based on body fluid analogue references from literature in which urea is the biggest single component) AnalaR, BDH Merck, Poole, UK
Car oil	Castrol super T high quality mineral based 2 stroke motorcycle engine oil	n/a
Pet shampoo	Bob Martin flea shampoo for dogs and puppies under 12 weeks' of age	Pyrethrins equivalent to 0.0475% w/w as Pyrethrum extract B.P. (Vet), piperonyl butoxide B.P (Vet) 0.49% w/w
Carpet cleaner	Vanish shampoo - carpet and upholstery stain remover	Amongst other ingredients less than 5% of polycarboxylates, anionic surfactants and non-ionic surfactants
Bathroom cleaner	Tesco bathroom cleaner 1252C	Amongst other ingredients preservative and less than 5% of amophoteric surfactants and phosphonate
Washing powder	Ariel washing liquid (automatic heavy-duty liquid detergent with biological action)	Less than 5% of phosphonates and cationic surfactants, 5-15% of nonionic surfactants and soap, 15-30% of anionic surfactants, and enzymes and optic brighteners
Make-up remover	L'Oréal Plénitude gentle make-up remover	Aqua, hexylene glycol, disodium cocoamphodiacetat, panthenol sodium laureth-8 sulfate, sodium laureth sulfate, triethanoamine, allantoin, quartenium-15, sodium oleth sulfate, magnesium laureth sulfate, magnesium laureth-8 sulfate, magnesium oleth sulfate, methylparaben, sodium benzoate, chlorohexidine dihydrochloride, disodium EDTA, parfym

Caustic soda	dp washing soda crystals	n/a
Food	Tesco's fresh tomato and basil soup	Vegetable stock (water, salt, dextrose, onion powder, flavourings, hydrogenated vegetable oil, yeast extract, sugar, mushroom extract, spices, leek extract, citric acid, herb, herb extract), tomato purée, tomatoes (18%), onion, carrot, single cream (3%), sugar, vegetable oil, modified maize starch, red wine vinegar, orange juice concentrate, salt, basil, garlic purée, black pepper, parsley

Table G.3. Perceived harmfulnes of the substances to the environment shown as the percentage of each group in relation to the entire survey population (the group itself).

Substance	All females	Females <25	Females 26-40	Females >41	All males	Males <25	Males 26-40	Males >41
Bleach	52.9	16.5 (63.6)	17.1 (42.6)	19.4 (56.9)	46.2	11.1 (57.6)	20.5 (43.8)	14.6 (43.1)
Body/food oil	1.8	0.0 (0.0)	0.0 (0.0)	1.8 (5.2)	0.6	0.0 (0.0)	0.0 (0.0)	0.6 (1.7)
White spirit	31.8	7.1 (27.3)	11.2 (27.9)	13.5 (39.7)	27.5	8.2 (42.4)	9.9 (21.3)	9.4 (27.6)
Hair dye	11.3	2.9 (11.4)	4.1 (10.3)	4.7 (13.8)	6.4	1.2 (6.1)	4.1 (8.8)	1.2 (3.4)
Alcohol	0.6	0.0 (0.0)	0.6 (1.5)	0.0 (0.0)	1.8	0.6 (3.0)	0.6 (1.3)	0.6 (1.7)
Perfume	0.0	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	2.3	0.6 (3.0)	1.8 (3.8)	0.0 (0.0)
Body fluids	2.4	0.6 (2.3)	1.2 (2.9)	0.6 (1.7)	1.2	1.2 (6.1)	0.0 (0.0)	0.0 (0.0)
Mud	0.6	0.6 (2.3)	0.0 (0.0)	0.0 (0.0)	0.0	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
Car oil	62.9	18.2 (70.5)	24.1 (60.3)	20.6 (60.3)	63.7	9.4 (48.5)	28.1 (60.0)	26.3 (77.6)
Pet care products	1.8	0.6 (2.3)	0.6 (1.5)	0.6 (1.7)	3.5	0.0 (0.0)	2.3 (5.0)	1.2 (3.4)
Detergents	35.3	11.2 (43.2)	18.2 (45.6)	5.9 (17.2)	37.4	4.7 (24.2)	18.7 (40.0)	14.0 (41.4)
Food	0.8	0.6 (2.3)	1.2 (2.9)	0.0 (0.0)	1.2	0.0 (0.0)	0.6 (1.3)	0.6 (1.7)
Cleaners	14.7	5.9 (22.7)	5.9 (14.7)	2.9 (8.6)	15.2	2.3 (12.1)	7.6 (16.3)	5.3 (15.5)
Washing powder	3.5	1.2 (4.5)	0.6 (1.5)	1.8 (5.2)	8.8	1.8 (9.1)	4.1 (8.8)	2.9 (8.6)
Make-up remover	0.6	0.0 (0.0)	0.6 (1.5)	0.0 (0.0)	2.3	0.6 (3.0)	0.6 (1.3)	1.2 (3.4)
Caustic soda	38.2	8.8 (34.1)	14.7 (36.8)	14.7 (43.1)	21.1	4.7 (24.2)	8.8 (18.8)	7.6 (22.4)
Other	14.1	0.0	0.0	0.0	16.4	0.0	0.0	0.0

Table G.4. Discharge frequency (%) of the substances listed in the survey.

	Bleach	Body/ food oil	White spirit	Hair dye	Alcohol	Perfume	Body fluids	Mud	Oil	Pet care products	Detergents	Food	Cleaners	Washing powder	Make-up remover	Caustic soda
All F																
Once a week	35.3	33.5	0.0	1.8	5.3	4.7	7.1	12.4	1.2	0.6	7.6	24.1	76.5	32.4	16.5	1.2
Once a month	15.9	11.2	3.5	8.8	8.2	2.4	11.2	12.9	1.8	0.6	11.2	2.4	13.5	12.9	4.1	1.8
Once a year	2.4	1.2	5.9	5.3	1.8	1.2	2.4	3.5	0.6	0.6	4.7	1.8	0.0	2.4	0.6	3.5
Occas-ionally	24.7	24.1	25.9	28.2	24.7	18.2	36.5	47.1	14.7	11.8	31.2	23.5	7.6	24.7	21.2	13.5
Once	0.6	2.4	3.5	4.1	2.9	1.8	3.5	3.5	1.8	0.0	2.9	0.6	0.0	3.5	0.6	2.9
Never	21.2	27.6	61.2	51.8	57.1	71.8	39.4	20.6	79.4	86.5	42.4	47.6	2.4	24.1	57.1	77.1
F <25																
Once a week	27.3	38.6	0.0	2.3	11.4	6.8	11.4	13.6	2.3	2.3	6.8	38.6	65.9	27.3	18.2	0.0
Once a month	25.0	13.6	6.8	6.8	13.6	0.0	9.1	11.4	2.3	2.3	11.4	0.0	20.5	15.9	4.5	2.3
Once a year	0.0	2.3	6.8	9.1	0.0	2.3	2.3	4.5	0.0	0.0	4.5	2.3	0.0	2.3	0.0	4.5
Occas-ionally	22.7	18.2	27.3	31.8	25.0	13.6	29.5	50.0	18.2	13.6	34.1	22.7	9.1	25.0	29.5	18.2
Once	0.0	2.3	2.3	6.8	0.0	0.0	2.3	6.8	4.5	0.0	2.3	0.0	0.0	2.3	0.0	4.5
Never	25.0	25.0	56.8	43.2	50.	77.3	45.5	13.6	72.7	81.8	40.9	36.4	4.5	27.3	47.7	70.5

F = females

Table G.4. continued

	Bleach	Body/ food oil	White spirit	Hair dye	Alcohol	Perfume	Body fluids	Mud	Oil	Pet care products	Detergents	Food	Cleaners	Washing powder	Make-up remover	Caustic soda
F 26-40																
Once a week	27.9	32.4	0.0	2.9	2.9	4.4	5.9	10.3	1.5	0.0	2.9	20.6	75.0	25.0	16.2	0.0
Once a month	14.7	8.8	2.9	8.8	10.3	1.5	11.8	14.7	1.5	0.0	11.8	5.9	16.2	16.2	7.4	0.0
Once a year	2.9	0.0	4.4	4.4	2.9	1.5	1.5	2.9	1.5	0.0	4.4	1.5	0.0	2.9	0.0	4.4
Occas-ionally	29.4	32.4	27.9	27.9	27.9	14.7	41.2	50.0	16.2	10.	27.9	29.4	7.4	30.9	20.6	8.8
Once	1.5	2.9	2.9	2.9	1.5	4.4	4.4	2.9	1.5	0.0	2.9	1.5	0.0	4.4	0.0	0.0
Never	23.5	23.5	61.8	52.9	54.4	73.5	35.3	19.1	76.5	89.7	50.0	41.2	1.5	20.6	55.9	86.8
F >41																
Once a week	50.0	31.0	0.0	0.0	3.4	3.4	5.2	13.8	0.0	0.0	13.8	17.2	86.2	44.8	15.5	3.4
Once a month	10.3	12.1	1.7	10.3	1.7	5.2	12.1	12.1	1.7	0.0	10.3	0.0	5.2	6.9	0.0	3.4
Once a year	3.4	1.7	6.9	3.4	1.7	0.0	3.4	3.4	0.0	1.7	5.2	1.7	0.0	1.7	1.7	1.7
Occas-ionally	20.7	19.0	22.4	25.9	20.7	25.9	36.2	41.4	10.3	12.1	32.8	17.2	6.9	17.2	15.5	15.5
Once	0.0	1.7	5.2	3.4	6.9	0.0	3.4	1.7	0.0	0.0	3.4	0.0	0.0	3.4	1.7	5.2
Never	15.5	34.5	63.8	56.9	65.5	65.5	39.7	27.6	87.9	86.2	34.5	63.8	1.7	25.9	65.5	70.7

F = females

Table G.4. continued

	Bleach	Body/ food oil	White spirit	Hair dye	Alcohol	Perfume	Body fluids	Mud	Oil	Pet care products	Detergents	Food	Cleaners	Washing powder	Make-up remover	Caustic soda
All M																
Once a week	28.7	31.0	0.0	0.0	6.4	3.5	4.7	15.8	1.2	2.3	8.8	22.8	56.7	22.2	8.8	0.6
Once a month	19.3	6.4	3.5	4.7	6.4	1.2	6.4	21.6	4.7	1.8	8.2	1.8	24.0	12.9	1.2	2.3
Once a year	3.5	2.3	11.7	1.2	4.1	0.6	2.3	2.3	1.8	1.8	5.3	1.2	0.0	2.9	0.6	7.6
Occas-ionally	21.6	24.0	19.9	14.0	25.1	9.4	45.6	39.8	17.5	6.4	27.5	24.6	14.6	25.1	14.0	15.8
Once	4.1	2.9	5.3	5.3	5.3	1.2	2.9	2.3	1.8	2.3	4.1	0.6	0.0	2.9	0.0	2.3
Never	22.8	33.3	59.6	74.9	52.6	84.2	38.0	18.1	73.1	85.4	46.2	49.1	4.7	33.3	75.4	71.3
M <25																
Once a week	33.3	39.4	0.0	0.0	9.1	3.0	9.1	30.3	3.0	9.1	9.1	27.3	63.6	21.2	15.2	0.0
Once a month	24.2	3.0	9.1	6.1	6.1	3.0	0.0	6.1	3.0	0.0	15.2	0.0	9.1	12.1	0.0	3.0
Once a year	6.1	3.0	0.0	3.0	6.1	0.0	0.0	3.0	3.0	0.0	3.0	0.0	0.0	0.0	0.0	6.1
Occas-ionally	21.2	24.2	15.2	24.2	27.3	6.1	60.6	39.4	27.3	12.1	21.2	27.3	18.2	21.2	9.1	15.2
Once	0.0	0.0	9.1	6.1	9.1	0.0	0.0	3.0	3.0	3.0	0.0	0.0	0.0	3.0	0.0	3.0
Never	15.2	30.3	66.7	60.6	42.4	87.9	30.3	18.2	60.6	75.8	51.5	45.5	9.1	42.4	75.8	72.7

M = males

Table G.4. continued

	Bleach	Body/ food oil	White spirit	Hair dye	Alcohol	Perfume	Body fluids	Mud	Oil	Pet care products	Detergents	Food	Cleaners	Washing powder	Make-up remover	Caustic soda
M 26-40																
Once a week	23.8	32.5	0.0	0.0	7.5	2.5	6.3	11.3	1.3	0.0	8.8	25.0	51.3	23.8	8.8	1.3
Once a month	26.3	8.8	1.3	1.3	10.0	1.3	8.8	33.8	6.3	3.8	7.5	3.8	33.8	17.5	2.5	1.3
Once a year	3.8	2.5	17.5	1.3	5.0	1.3	5.0	1.3	2.5	2.5	7.5	2.5	0.0	5.0	1.3	8.8
Occas-ionally	18.8	26.3	26.3	10.0	26.3	10.0	38.8	35.0	15.0	6.3	31.3	26.3	12.5	22.5	11.3	16.3
Once	7.5	5.0	6.3	6.3	5.0	1.3	3.8	3.8	2.5	2.5	6.3	1.3	0.0	5.0	0.0	1.3
Never	20.0	25.0	48.8	81.3	46.3	83.8	37.5	15.0	72.5	85.0	38.8	41.3	2.5	26.3	76.3	71.3
M >41																
Once a week	35.2	25.9	0.0	0.0	3.7	5.6	0.0	14.8	0.0	1.9	9.3	18.5	64.8	22.2	5.6	0.0
Once a month	7.4	5.6	3.7	9.3	1.9	0.0	7.4	14.8	3.7	0.0	5.6	0.0	20.4	7.4	0.0	3.7
Once a year	1.9	1.9	11.1	0.0	1.9	0.0	0.0	3.7	0.0	1.9	3.7	0.0	0.0	1.9	0.0	7.4
Occas-ionally	27.8	22.2	14.8	14.8	24.1	11.1	50.0	50.0	16.7	3.7	27.8	22.2	16.7	33.3	22.2	16.7
Once	1.9	1.9	1.9	3.7	3.7	1.9	3.7	0.0	0.0	1.9	3.7	0.0	0.0	0.0	0.0	3.7
Never	33.3	50.0	75.9	79.6	72.2	88.9	46.3	24.1	87.0	98.1	57.4	66.7	5.6	40.7	79.5	75.9

M = males

ıble G.5. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of alcohol and throom cleaner.

icohol				**Bathroom cleaner**			
ıl l^{-1})	pH (-)	Oxygen uptake	COD removal	c (ml l^{-1})	pH (-)	Oxygen uptake	COD removal
	7.6	0.03	-0.35	10 [a]	7.8	0.03	-0.36
	7.6	0.02	-0.75	20 [b]	7.9	0.03	-0.17
	7.6	0.02	0.47	25 [c]	7.9	0.03	0.11
	7.6	0.03	0.92	30 [b]	8.1	0.02	-0.18
	7.5	0.03	2.30	35 [d]	8.2	0.03	0.04
	7.5	0.02	-3.01	40 [e]	8.2	0.02	-0.04
0	7.3	0.02	5.08	50 [f]	8.1	0.01	-0.38
0	7.1	0.01	7.69	60 [f]	8.3	0.09	-0.59
0	7.1	0.01	23.08	70 [f]	8.6	0.01	-0.71
ntrol	7.3	0.02	0.00	80 [g]	8.5	0.01	-1.28
				90 [a]	8.6	0.01	-0.81
				[a] control	7.6	0.02	0.00
				[b] control	7.6	0.02	0.00
				[c] control	7.2	0.01	0.00
				[d] control	7.2	0.01	0.01
				[e] control	7.1	0.01	0.01
				[f] control	7.0	0.02	0.00

ıble G.6. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of urea.

rea			
ml l^{-1})	pH (-)	Oxygen uptake	COD removal
25	7.6	0.03	-0.02
25	7.9	0.02	-0.02
5	7.9	0.03	-0.02
	8.0	0.02	-0.04
	8.1	0.02	-0.03
	8.1	0.02	-0.03
ntrol	7.2	0.01	0.01

ıble G.7. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of bleach and dy/food oil.

each				**Body/food oil**			
ıl l^{-1})	pH (-)	Oxygen uptake	COD removal	c (ml l^{-1})	pH (-)	Oxygen uptake	COD removal
5 [a]	7.7	0.02	-0.02	10	7.4	0.01	-0.01
	7.9	0.02	-0.01	20	7.4	0.02	0.00
	8.0	0.01	-0.05	30	7.4	0.01	-0.01
	8.3	0.01	-0.08	40	7.4	0.00	0.00
	8.5	-0.07	-0.09	50	7.4	0.00	-0.00
	8.7	0.00	-0.07	60	7.4	0.00	0.01
	8.9	0.00	-0.56	70	7.4	0.00	-0.01
ontrol	7.4	0.02	0.00	80	7.3	0.00	0.00
ontrol	7.3	0.02	0.01	90	7.4	0.00	0.02
ontrol	7.2	0.02	0.08	control	7.4	0.02	0.00

able G.8. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of car oil and rpet cleaner.

ar oil				**Carpet cleaner**			
nl l^{-1})	pH (-)	Oxygen uptake	COD removal	c (ml l^{-1})	pH (-)	Oxygen uptake	COD removal
[a]	7.3	0.02	-0.01	5 [a]	7.4	0.03	0.09
)[a]	7.4	0.01	-0.01	10 [a]	7.3	0.04	0.16
)[a]	7.4	0.01	-0.00	15 [a]	7.2	0.05	0.54
)[a]	7.2	0.01	0.00	20 [a]	7.2	0.03	0.19
)[b]	7.2	0.01	0.03	25 [a]	7.1	0.02	0.31
)[b]	7.2	0.00	0.01	30 [a]	7.3	0.02	0.26
)[c]	7.7	0.00	0.00	35 [a]	7.3	0.01	0.21
)0[b]	7.7	0.00	-0.02	40 [a]	7.3	0.01	0.17
20[d]	7.7	0.00	0.023	45 [a]	7.3	0.01	0.08
30[e]	7.7	0.00	-0.01	125 [b]	7.3	0.00	-0.61
40[e]	7.7	0.00	-0.03	150 [b]	7.4	0.00	-0.71
50[e]	7.7	0.00	-0.02	175 [b]	7.3	0.00	-0.61
control	7.1	0.01	0.00	200 [b]	7.3	0.00	0.60
control	7.3	0.01	0.01	[a] control	7.5	0.01	0.01
control	7.6	0.01	0.00	[b] control	7.6	0.01	0.00
control	7.6	0.01	0.02				
control	7.7	0.02	0.03				

able G.9. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of food and hair ye.

ood				**Hair dye**			
nl l^{-1})	pH (-)	Oxygen uptake	COD removal	c (ml l^{-1})	pH (-)	Oxygen uptake	COD removal
7	6.9	0.03	-0.07	4.2	7.6	0.02	0.01
2	6.5	0.02	-0.09	8.3	8.2	0.04	0.01
3	6.2	0.01	-0.24	12.5	8.6	0.04	0.02
25	5.7	0.00	0.27	16.7	8.8	0.03	-0.01
67	5.5	0.01	-0.23	20.8	9.0	0.02	-0.04
33	4.9	0.00	-0.11	25	9.0	0.01	-0.07
17	4.8	0.00	-0.39	29.2	9.1	0.01	0.06
00	4.7	0.00	-0.31	33.3	9.1	0.01	-0.09
ontrol	7.4	0.01	0.00	37.5	9.2	0.02	0.01
				control	7.65	0.01	0.00

able G.10. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of caustic soda ıd washing powder.

'austic soda				Washing powder			
nl l^{-1})	pH (-)	Oxygen uptake	COD removal	c (ml l^{-1})	pH (-)	Oxygen uptake	COD removal
[a]	9.7	0.02	-0.08	1.5 [a]	8.0	0.05	-1.19
[a]	10.2	0.03	-0.03	2 [a]	8.1	0.05	0.13
[a]	10.4	0.02	-0.06	2.5 [a]	8.1	0.05	-0.18
[a]	10.5	0.01	-0.10	3 [a]	8.2	0.05	0.19
[a]	10.6	0.01	-0.03	3.5 [a]	8.1	0.05	0.31
[a]	10.7	0.01	-0.15	4 [a]	8.3	0.05	0.67
[a]	10.8	0.01	-0.15	4.5 [a]	8.2	0.05	0.98
0 [a]	10.8	0.00	0.32	5 [a]	8.2	0.04	0.45
0 [b]	11	0.00	-0.06	7.5 [b]	8.5	0.04	0.58
0 [b]	11.1	0.00	-0.18	10 [b]	8.5	0.03	2.25
0 [b]	11.2	0.00	-0.20	20 [c]	8.6	0.02	0.71
0 [b]	11.2	0.00	-0.01	40 [c]	8.5	0.00	0.00
50 [b]	11.4	0.00	0.05	80 [c]	8.5	0.00	-4.72
50 [b]	11.4	0.00	-0.05	160 [c]	8.5	0.00	0.00
control	7.4	0.02	0.00	[a] control	7.5	0.02	-0.01
control	7.7	0.01	0.00	[b] control	7.7	0.01	0.17
				[c] control	7.7	0.01	0.00

'able G.11. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of make-up :mover and perfume.

lake-up remover				**Perfume**			
nl l^{-1})	pH (-)	Oxygen uptake	COD removal	c (ml l^{-1})	pH (-)	Oxygen uptake	COD removal
0	7.4	0.04	0.03	1	7.4	0.03	-0.48
0	7.5	0.05	-0.94	5	7.5	0.03	-45.64
00	7.5	0.06	-1.63	10	7.4	0.03	-92.46
50	7.5	0.05	-0.84	20	7.4	0.01	-196.92
00	7.5	0.04	-0.21	40	7.4	0.01	-358.51
00	7.6	0.02	-0.09	80	7.4	0.01	29.20
00	7.6	0.01	0.15	160	7.3	0.00	72.53
00	7.6	0.02	-0.81	320	7.3	0.00	-26.50
00	7.6	0.01	0.58	control	7.3	0.02	0.01
ontrol	7.5	0.01	0.00				

able G.12. Oxygen uptake (kgO2/kgMLSS/d) and COD removal (kgCOD/kgMLSS/d) of white spirit id pet shampoo.

'hite spririt				Pet shampoo			
ıl l^{-1})	pH (-)	Oxygen uptake	COD removal	c (ml l^{-1})	pH (-)	Oxygen uptake	COD removal
5	7.6	0.03	-0.01	0.1 [a]	7.6	0.01	0.01
ı	7.5	0.02	-0.01	1 [a]	7.6	0.03	0.02
)[a]	7.6	0.02	-0.00	2.5 [a]	7.5	0.05	-0.06
;[a]	7.5	0.03	-0.05	5 [a]	7.5	0.05	0.01
)[a]	7.5	0.01	-0.02	7.5 [a]	7.5	0.05	0.79
;[a]	7.6	0.02	-0.01	10 [a]	7.5	0.04	0.01
)[a]	7.7	0.02	-0.01	20 [a]	7.5	0.04	-0.57
;[a]	7.6	0.02	0.14	40 [a]	7.4	0.01	0.01
)[b]	7.6	0.02	-0.11	80 [a]	7.3	0.01	0.65
)[b]	7.1	0.02	-0.04	90 [b]	7.3	0.01	-0.92
)[b]	7.2	0.02	0.00	100 [b]	7.3	0.01	0.07
)[c]	7.5	0.02	-0.23	110 [b]	7.2	0.00	1.48
)[c]	7.7	0.01	-0.21	120 [b]	7.2	0.06	1.05
)[c]	7.6	0.02	-0.18	130 [b]	7.2	0.01	4.53
)0[d]	7.4	0.02	-0.06	140 [b]	7.2	0.01	0.58
10[d]	7.4	0.02	-0.92	150 [b]	7.1	0.00	1.60
20[d]	7.4	0.01	-0.65	160 [b]	7.1	0.01	2.63
50[e]	7.4	0.01	-	170 [b]	7.1	0.01	-
92[e]	7.4	0.01	-	[a] control	7.6	0.01	0.00
33[e]	7.5	0.01	-	[b] control	7.6	0.01	0.00
50[e]	7.4	0.01	-				
17[e]	7.5	0.01	-				
58[e]	7.5	0.01	-				
)0[e]	7.5	0.01	-				
control	7.6	0.02	0.00				
control	7.1	0.01	0.00				
control	7.4	0.01	0.01				
control	7.3	0.02	0.04				
control	7.2	0.02	-				

Table H.1. Summary of the parameters measured during the transition periods to grey/blackwater and **blackwater**. Range.

Parameter	Influent	Submerged MBR	BAF
OLR		0.05-0.21	0.32-0.90
(kgBOD m^{-3} d^{-1})		**0.13-0.25**	**0.63-1.14**
Flow	-	0.038-0.053	0.302-0.346
(m^3 d^{-1})	-	**0.046-0.053**	**0.331-0.382**
BOD_5	48-141	1-2	2-19
(mg l^{-1})	**91-171**	**1-3**	**5-42**
SS	73-136	0-6	4-47
(mg l^{-1})	**94-204**	**0-13**	**1-66**
Turbidity	-	0.2-0.8	4.3-21.9
(NTU)	-	**0.1-0.3**	**1.3-43.3**
pH	6.6-7.1	6.2-7.3	7.0-7.4
(-)	**7.4-8.0**	**6.3-7.6**	**6.5-7.6**
Total coliforms	6-7	0-1	4-7
(log)	**6-7**	**0-1**	**3-7**
E.coli	3-6	0-0	2-6
(log)	**4-6**	**0-0**	**2-6**