An experimental investigation into filter ripening: contact filtration of lowland reservoir water.

by

Michael John Chipps

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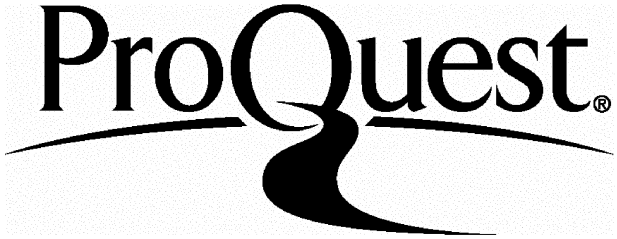
Department of Civil and Environmental Engineering
University College London
Gower Street
London WC1E 6BT



ABSTRACT

Pilot rapid gravity filters, treating lowland reservoir water, were operated to examine chemical, physical and biological influences on filter ripening. Ripening occurred if, after backwashing, filtrate turbidities showed an initial peak, then improvement to a minimum value. Backwashing using collapse pulsing prevented biological fouling of media, and ensured constant starting head losses for three years.

Ripening was examined using sand roughing filtration and dual media contact filtration with pre-ozonation and iron (III) sulphate coagulation. Turbidity removal data from sand filters showed that ripening behaviour varied with seasonal changes in applied loading. Biological maturation of filter media over several months caused improved turbidity removals. Subsequently, ripened turbidity removals varied proportionally with water temperature. Within a filter run ripened turbidity removal was independent of load at constant flow rate, but varied inversely with flow rate.

Dual media filters showed ripening behaviour in roughing and contact filtration modes. Iron dosing produced superior turbidity removals. A novel coagulation control method was devised, based on the shape of the filtrate turbidity curves. Ripening time varied inversely with coagulant dose, but higher doses caused turbidity breakthrough. A temporary initial overdose could control ripening. Pre-ozonation enhanced roughing and contact filtration, but did not change the time taken to achieve maximum removals.

With contact filtration ripened removals were a fixed proportion of the applied turbidity in winter, but filtrate turbidity was independent of load in summer, achieving <0.2 NTU. Flow rates of 12, 17 and 22 $m.h^{-1}$ produced identical ripening curves, indicating a complex relationship between flow rate and particle capture. This was confirmed with data showing inconsistent relationships between turbidity, particle loading and ripening, and poorer ripening behaviour on longer filter runs. Further detailed particle size data were required. Six definitions of ripening were examined but none was satisfactory as ripening behaviour changed with chemical treatment and seasonal factors.

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DEDICATION

For Tracey and Mum, who saw the whole thing through, Ben and Eleanor who joined us, and Dad and Grandma (will we have to call you Sir?), who couldn't stay for the end.

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CHAPTER 1. INTRODUCTION

The issue of filter ripening in water treatment has come into prominence in recent years. Although experimental evidence for ripening goes back over 60 years, its significance has only recently been recognised due to concerns over the penetration of water-borne pathogens such as *Giardia* cysts and *Cryptosporidium* oocysts through conventional water treatment plants. It may also be due to the adoption of less expensive, but potentially more risky, contact or direct filtration methods, instead of two-stage separation and filtration processes.

The term ripening was applied in this thesis to the improvement in filtrate quality from a fixed bed, rapid gravity, granular media filter, measured in the initial stages of an individual filter run. Often this meant after a backwash, although some laboratory studies have used fresh media for each filter run. The improvement in filtrate quality which takes place over several days in slow sand filtration due to the development of the biological community has also be termed ripening (Collins *et al.*, 1992), but this was outside the scope of this investigation. Long term biological, physical or chemical changes to rapid gravity filter media which improve the filter performance over weeks or months were classed as maturation effects in this thesis.

Fixed beds operating at flow rates from around 5 $m.h^{-1}$ with periodic backwashes every few hours or days, typically 24 h, are termed rapid gravity filters (RGFs) when open to atmosphere, and pressure filters if enclosed. RGFs are designed to operate in either upflow or downflow modes (Diaper, 1963). Downflow filtration flow rates may range from 5 to 25 $m.h^{-1}$ (Cleasby, 1990) although pilot scale investigations have resulted in a design rate up to 33 $m.h^{-1}$ at Los Angeles (McBride and Stolarik, 1986). Brief introductions to pressure filtration have been written by Cleasby (1990) and Letterman (1991). The literature search did not find information on ripening in upflow or pressure filters, and as these were not the subject of the experimental work they were not discussed further. Experiments with downflow RGFs employing glass spheres, sand, anthracite and granular activated carbon (GAC) as filter media were reviewed in this thesis, and experimental trials conducted with sand and anthracite were reported.

Chapter 2 presents a literature review covering ripening in rapid gravity filtration during potable water treatment. The types of particles which must be removed from raw waters were described. In particular, the public health problems arising from poor filtration of *Giardia* cysts and *Cryptosporidium* oocysts were discussed. Some authors have reported the passage of these pathogens through filters during the ripening phase. The principles behind the design and operation of rapid gravity filters were summarised, showing it to be a dynamic process in which the particles in the water and the filter media interact to produce a filter cycle of up to seven discernible phases. The ripening period covers four of these phases. Although ripening is a simple concept, no systematic means of quantifying this phenomenon have been proposed. A number of possible ripening points were suggested.

The literature review examined whether ripening was universally observed, or whether it was associated with a particular set of plant or operating conditions. Many common measures of particles in water were examined for evidence of ripening. The use of turbidity as an indicator of water quality was discussed. Some limited evidence that head loss development curves could indicate the termination of ripening was presented. Investigations into some possible causes of filter ripening were discussed, together with ways of controlling the ripening period.

Chapter 3 describes the design, operation, and monitoring of the experimental pilot plant. The objective was to build pilot filters that realistically simulated full scale design and operation. The pilot plant was designed for continuous unattended operation and monitoring, and operated successfully for four years.

The project was funded by Thames Water and used 6 pilot RGFs located at Ashford Common water treatment works (WTW), near Staines, Middlesex. The study examined the filtration of reservoir stored River Thames water. It was of high hardness, low in colour and turbidity, but eutrophic, hence subject to periodic algal blooms. Stored water quality data are presented in chapter 3.

At the start of the research treatment at Ashford Common consisted of microstraining, followed by slow sand filtration and chlorination. The uprating of the works to produce higher outputs of better quality water required, in part, a new rapid gravity primary filter plant, and these trials were established to provide the process design. A range of filter media and flow rates were examined. The details are summarised in the appendix. The investigation sought to maximise particle removal by filtration combined with pre-ozonation and iron(III) sulphate coagulation. Chapter 3 includes calibration data from the ozone generators and the iron(III) sulphate dosing pumps. An assessment of the turbidimeters was included.

In chapter 4 the subject of filter backwashing is examined. It was important that the filters should be operated for many years without problems, but experience in Thames Water with biological fouling of primary filter media had shown that existing backwash procedures were unsatisfactory. In order to assess the ripening behaviour of different filter media and flow rates throughout a range of seasonal water quality variations it was necessary to avoid the complication of biological changes to the filter media. It was not practical to pre-chlorinate water ahead of slow sand filters, or steep the media in chlorinated water, so a satisfactory backwash procedure had to be developed. The objective of this research was to produce a backwashing strategy that limited the attachment of biofilm on the media and avoided the formation of cracks or mudballs. It also needed to be a practical design for a full-scale plant.

Chapter 4 examines the literature for the best way to clean single and dual media filters, and produces experimental evidence showing that the use of a backwashing procedure termed collapse-pulsing was the solution. This was demonstrated by data from media samples and from evidence that flow and temperature normalised starting filter head loss did not change over many months of operation.

Once baseline filter media condition could be maintained it was possible to examine in detail the chemical, physical and biological influences on filter ripening. Chapters 5 and 6 explore the

phenomenon of ripening in rapid gravity filters by taking an empirical long-term monitoring approach, using pilot filters operating without pre-treatment and in contact filtration mode.

In chapter 5 the impact of chemical conditions on ripening is examined. Background data from laboratory and plant-scale investigations were discussed. Reference was made to the considerable body of evidence showing the importance of coagulants and oxidants in optimising particle removal, and influencing filter ripening. The mechanisms of coagulation were summarised together with possible ways in which ozone benefits filtration. The water quality conditions when pre-ozonation was likely to be beneficial were considered, along with the conditions for which contact or direct filtration would be a satisfactory treatment process. Practical measures of determining the correct dose of coagulant were discussed.

The experimental investigation into the significance of iron(III) sulphate and ozone dosing is described in chapter 5 and the results were contrasted with filtration without chemical dosing. The objective of the research was to examine filter ripening under practical conditions, where realistic ozone and iron doses were used. The difficulties with conventional coagulant dose control were considered and a novel method of coagulation control was presented. The conventional wisdom that there was one suitable coagulant dose for any given situation was challenged.

Having established in chapter 5 that undosed filter behaviour could only be interpreted by using turbidity removals, and developed a methodology for optimising coagulant dosing, the effect of physical variables on filter ripening was examined. Some fundamental considerations of what takes place at the microscopic level on filter media were discussed, and the results of laboratory investigations into the impact of media size and type, flow rate, influent suspension concentration and particle size distribution were examined. The pilot plant data were examined to determine whether any of the observations in the literature were supported or refuted under treatment plant conditions, away from the idealised conditions used in fundamental laboratory studies.

Chapter 7 examines the effect of biology on the filtration of particles in the water and on the filter media itself. Some thought was given in chapter 8 to why the use of a systematic definition of ripening was not feasible, and what further studies should be undertaken. A summary is presented in chapter 9 and whether ripening was an issue in these studies was considered. Problems with the operation of the pilot plant, the methodology and the data processing were discussed. Suggestions for futher work are presented in chapter 10. The experimental programme is detailed in the appendix, and a list of references follows.

CHAPTER 2. RIPENING IN RAPID GRAVITY FILTRATION OF POTABLE WATER

2.1 INTRODUCTION

In the treatment of water for public supply, where the water originates from a surface source containing appreciable numbers of particles, filtration through fixed beds of granular media is used throughout the developed world and in many developing countries. Arguably the most commonly employed technology is the deep bed granular media rapid gravity filter (RGF). A history of water treatment has been provided by Baker (1949).

2.1.1 WATER TREATMENT OBJECTIVES

Legislation requires water suppliers to produce water meeting limits on many chemical and biological contaminants. British water suppliers must meet the standards laid down in the Water Supply (Water Quality) Regulations (1989). Water is treated to remove:

pathogenic particles which cause water-borne disease - generally biological organisms such as viruses, bacteria and protozoa - Tate and Arnold (1990) listed over 30 disease-causing organisms;

other contaminants of significance to public health - mainly inorganic or organic chemicals, with toxic, neurotoxic, carcinogenic, mutagenic or teratogenic effects (Tate and Arnold, 1990);

contaminants which affect the aesthetic appeal of the water, either directly, for instance animals, colour, taste and odour, particles, cloudiness (turbidity), iron and manganese, or which promote the deterioration of water quality in the distribution system e.g. algae, particulate organic carbon and actinomycetes (Ives, 1955, Watson, 1990, Janssens and Buekens, 1993).

For some of these parameters filtration is necessary, often in combination with chemical treatment. Soluble contaminants, such as colour, have to be chemically "bound up" into a hydroxide floc if they are to be removed by granular media filtration. There has, however, been considerable debate

in the USA about the need to filter all surface-derived water. The surface water treatment rule in the 1986 amendments to the Safe Drinking Water Act required all surface waters to be disinfected to achieve the elimination or inactivation of 99.9% of *Giardia* cysts (3 log removal) and 99.99% (4 log) of viruses (O'Melia, 1991).

Some States required all surface water to be filtered. In others, where there are surface waters which are low in turbidity, colour, and bacterial counts, water may be treated by chlorination alone, provided effective watershed catchment control and management could be demonstrated (Cotruvo and Vogt, 1990, Geldreich *et al.*, 1990). However the inability of chlorine to disinfect waters containing *Giardia* cysts or *Cryptosporidium* oocysts at the doses used in water treatment to protect consumers from bacterial and viral pathogens (Smith *et al.*, 1990) is the main reason for concern in un-filtered water.

Camp (1969) has provided a summary of the dissolved, colloidal and particulate material found in natural waters, together with information on chemical treatment and water works design. Some of the particles monitored in water filtration experiments are listed with their dimensions in table 2.1.

Table 2.1. *Sizes of some natural and artificial particles in water for filtration, in μm. (After Ainsworth, 1990, Ives, 1970, Montgomery, 1985, Purchas, 1981).*

Particle	**Size range (μm)**
Viruses	0.015 -0.3
Bacteria	0.4-2.0
Clay	0.1-1.0
PVC microsphere	1.3
Cryptosporidium oocysts	4-7
Giardia cysts	7-10 wide 8-14 long
Silica particle	20
Asterionella (diatom)	30
Sand	50 and up
Bacterial flocs	0.5- 1000
Algae	1-200
$Fe(OH)_3$ or $Al(OH)_3$ floc	0.1 - 1000

2.1.2 CAUSES OF WATER-BORNE DISEASE

Failures of filtration plants which have resulted in outbreaks of water-borne disease are rare but have caused great concern. In 1990 a committee under Sir John Badenoch reported on water-borne cryptosporidiosis following the Swindon outbreak in 1989.

In reviewing water-borne disease in the UK from 1937-1986, Galbraith *et al.* (1987) found that 21 of the 34 cases were due to public water supplies. Of the 21, 10 were due to contamination occurring in the distribution system, whilst 11 were due to source water contamination. In 8 of the 11 cases water was either unchlorinated or not chlorinated properly. In 2 of the final 3 cases the pathogen was *Cryptosporidium*. At the time Galbraith *et al.* (1987) were writing the two *Cryptosporidium* outbreaks (in Cobham, Surrey) had not been shown causally to be linked to the water supply, however there was a geographical link. These were the only cases of disease reported where the filtration and chlorination system were not implicated in being defective at the same time as the water source was contaminated.

Craun (1988) reviewed causes of water-borne outbreaks of disease in the USA from 1971-1985 and found that inadequate control of pre-treatment or filtration accounted for between 1-8% of water-borne disease outbreaks. Giardiasis was the most commonly identified cause of water-borne disease from 1971-85, accounting for just under 20% of outbreaks and cases of illness. The difference between giardiasis and other water-borne diseases was that 71% of outbreaks of giardiasis were due to inadequate treatment of surface water, compared to 14% for other diseases.

Badenoch (1990) reviewed a number of outbreaks of cryptosporidiosis and showed that it was not easy to prove a water-borne origin for the organisms in all cases. However there were cases where it appeared that inadequate treatment of surface waters was to blame, either through lack of suitable plant, e.g. at Loch Lomond where only microstraining was available, or through operational practises which caused the filter plant to fail e.g. Carrollton, Georgia; Swindon/Oxfordshire (Farmoor/Worsham WTW); North Humberside (Barmby-on-the-Marsh WTW).

During an outbreak of cryptosporidiosis in 1993 at Milwaukee, Wisconsin, The Milwaukee Journal featured several articles on water and health from 19-26 September 1993. They reported EPA data from the USA between 1981 and 1990 showing 117 outbreaks of water-borne disease (table 2.2).

Carrington and Miller (1993) reported that 13,000 people were believed to have been affected by an outbreak of cryptosporidiosis in 1987 at Carrollton, Georgia. In Swindon 500 cases were confirmed, but up to 5,000 people may have been affected. The cryptosporidiosis outbreak in 1993 at Milwaukee, Wisconsin, affected more than 370,000 citizens (Bolden and Farrell, 1994). In the first quarter of 1994 78 confirmed cases of water-borne cryptosporidiosis were reported in Las Vegas, 61 of these were HIV-infected adults. By 30 June 1994 32 of these adults had died and at least 20 of these had cryptosporidiosis listed on their death certificates (Goldstein *et al.*, 1996).

Table 2.2. *Outbreaks of water-borne disease in USA 1981-90 (source: The Milwaukee Journal 20/9/93).*

Causative organism	Number of outbreaks 1981-90
Giardia	71
Shigella	22
Hepatitis A	11
Campylobacter	10
Cryptosporidium	3

2.1.3 FAILURE OF FILTRATION PLANTS TO PROTECT PUBLIC HEALTH

It has been shown earlier that public health may be at risk if filtration or disinfection processes fail, but it has also been found that treatment plants which were thought to be correctly operated had in hindsight used practices which made the filters vulnerable to the passage of protozoan oocysts. Since *Cryptosporidium* oocysts are resistant to chlorine doses that effectively inactivate bacteria and viruses (Ives, 1990), a treatment plant has to remove these cysts reliably by well-managed solids-liquid separation processes.

The numbers required for infection are not known but thought to be very low. Average concentrations of 63 oocysts.l^{-1} were found in samples taken during the Carrollton outbreak (Smith

and Rose, 1990), so the presence of these in any quantity is undesirable. This was a high concentration for oocysts, but was a low figure compared with the counts of particles found between 5 and 10 μm in a low turbidity treated water. This puts emphasis on obtaining the best quality filtrates at all times.

At Carrollton the outbreak of cryptosporidiosis was linked partly to inadequate flocculation, but primarily to filter operating practice (Logsdon *et al.*, 1988). Due to plant design, changes in water demand were met by taking operational filters out of service. They were re-started without backwashing and this caused high filtrate turbidities. It was this practice which was thought to permit the penetration of *Cryptosporidium* oocysts. Values of filtrate turbidity ranging from 1.6 - 5.0 NTU were measured from re-started dirty filters, ten times higher than values from four clean filters (0.07 - 0.59 NTU) treating the same influent. A normally backwashed filter was examined and turbidity was found to fall from 0.5 NTU to 0.2 NTU in 20 minutes of ripening time (Logsdon *et al.*, 1988).

As evidence for this Logsdon *et al.* (1988) recounted a direct filtration experiment carried out by Logsdon *et al.* (1981) where *Giardia* cysts were dosed into a pilot filter for 24 hours and eventually the supply of cysts was exhausted but the filter had not shown breakthrough so the run was continued. After a further 2 hours breakthrough occurred and the filtrate was sampled. Large numbers of cysts were found which was taken to show that the filter could accumulate cysts during normal steady state operation but that a release of cysts was possible "during a period of non-equilibrium or disturbance".

Failures of rapid mixing, filter rate control valves, and in particular, the use of polymers without alum were linked to a cryptosporidiosis outbreak in Oregon by Leland *et al.* (1993). The use of alum improved filtrate turbidities from 0.4 to 0.15 NTU and pre-chlorination improved coagulation. The practice at Oxford's treatment works of recycling wash water sludge tank supernatant liquors was implicated in concentrating numbers of *Cryptosporidium* oocysts in the water applied to the filters (Badenoch, 1990). Over the time period which included the cryptosporidiosis outbreak the

works were found not to have produced water which showed adverse changes in turbidity, chlorine demand, bacteriological quality or residual coagulant.

This was significant because it showed that a conventional chemical coagulation, sedimentation and filtration works, with chlorine disinfection, which was thought to be correctly operated, could be vulnerable to penetration by pathogens, causing public health problems.

The Milwaukee Journal of 19/9/93 said "insufficient amounts of a substitute chemical early in the water treatment process likely allowed heavy concentrations of Cryptosporidia to reach the water plant's sand filters, which could not remove all of the contaminants". Three days later they blamed the epidemic of cryptosporidiosis on "a gamble to substitute one treatment chemical with another allowing a much larger than normal volume of particles, including Cryptosporidia and other microbes, to pass through the filters and into the distribution system".

With hindsight the Farmoor plant was concentrating the *Cryptosporidium* load applied to the filters. The question is whether a well-operated plant is actually vulnerable to penetration by *Giardia* and *Cryptosporidium*? Evidence that it may be comes from the water-borne outbreak of cryptosporidiosis in 1994 from a "state of the art" direct filtration treatment plant at Las Vegas. Reports by Goldstein *et al.* (1996) and Roefer *et al.* (1996) said that this was the first outbreak where the water treatment plant was properly operated, with no treatment deficiencies or breakdowns. The reservoir stored water was very low in particle numbers, *Cryptosporidium* counts and turbidity, as was the filtrate. However these authors failed to consider that, like Farmoor, this plant recycled settled backwash water.

It may implied from Roefer *et al.* (1996) that standard filter operation is no longer considered satisfactory, since steps taken to improve the Las Vegas plant include adding cationic polymer to the backwash water and planning to install filter to waste in 1997 and pre-ozonation by 2000.

It is not yet clear whether there will be any impact from the Las Vegas experience on water utilities. Data are still being gathered in the USA under the Information Collection Rule to allow decisions to be made about regulations for treating *Cryptosporidium* from the year 2000 under the proposed Enhanced Surface Water Treatment Rule (Solo-Gabriele and Neumeister, 1996).

Clearly there is currently a need for vigilance in raw water monitoring and plant operation. Leland *et al.* (1993) thought that filtration of surface water supplies could probably not at all times produce water free from *Cryptosporidium* oocysts. They advised that plant operators should act as if oocysts were constantly present, since it was impossible to predict when they might show up in the raw water. Carrington and Miller (1993) showed that *Cryptosporidium* oocysts were widely present in the environment in the UK. It is therefore essential that the operation of filter plants is optimised to remove pathogenic oocysts, and the behaviour of these particles during filtration must be understood to enable this.

2.2 RAPID GRAVITY FILTERS IN WATER TREATMENT

RGFs may be used in water treatment in three distinct configurations:

i) the first way is as a polishing stage in "conventional" water treatment after pre-oxidation, usually with chlorine, coagulation, flocculation and sedimentation or flotation;

ii) the second is termed "direct" or "contact" filtration, with chemical treatment but no prior separation stage. Direct filtration implies a separate flocculation stage before filtration, whereas contact or "in-line direct" filtration does not use flocculation tanks (Cleasby, 1990);

iii) thirdly, the same filter technology, generally without chemical treatment, has been used as a pre-treatment stage, to reduce particle loads onto slow sand filters (Ridley, 1967, Toms, 1987, Rachwal *et al.*, 1988).

In each case the objective is to achieve depth filtration. When rapid filtration is the final solids/liquid separation stage it must meet filtrate quality targets suitable for water that is going into supply. Generally, small fluctuations in individual filter performance are tolerated as the total plant

output is considered. In some cases this means no action is taken to separate the filtrate from the ripening period out of the overall treated water stream. This may be a complacent approach if this allows even small numbers of pathogens of a low infective dose to reach the public.

A filter should be designed with a media grain size and bed depth that will remove particles from the water so that filtrate quality targets are achieved, together with run length, flow rate and head loss development goals. The usual quality targets have been turbidity, colour, residual coagulant, and ions such as iron and manganese, when present in the water. Recently, there has been concern over the removal of particles in the same size range as *Giardia* cysts and *Cryptosporidium* oocysts. In Georgia, USA, State law requires particle removal to be monitored (Hiac/Royco, 1995).

Economic objectives, such as capital costs for filter construction and revenue costs for filter operation, must be considered along with quality objectives by the filter designer. Reducing the capital cost requires high flow rates, low backwash flow rates and low cost media. The revenue costs dictate long filter runs, low chemical usage, an acceptable filtrate quality, and effective backwashing using minimum washwater quantities. These are conflicting goals, for example, effective solids removal shortens filter run lengths, so a compromise has to be reached. One criterion given by Ives (1982) was that an optimum filter design should enable the maximum solids holding capacity of the filter and the maximum acceptable head loss to be reached simultaneously.

RGFs typically use sand ranging from 0.5 - 1.0 mm effective size (E.S.), although media grains may be larger. The E.S. is the sieve opening upon which 90% of the media by weight is retained (Cleasby, 1990). To avoid surface clogging, deep coarse media filters have been developed using sand or anthracite coal (Cleasby, 1990). For the same reason dual media filters, comprising a layer of sand underneath a layer of coarser, lower density anthracite have been used to produce a size graded bed. A third, finer, layer of dense garnet or ilmenite may be placed below the sand to produce triple or mixed media filters. Dual and triple media filters promote filtration through progressively finer media layers, preventing coarser particles from rapidly blinding fine media.

There is one filtration medium in widespread use that is different from the usual materials that are simply quarried or mined and size graded. Granular activated carbon (GAC) is manufactured from natural materials such as coal, wood, peat, or coconut shells, which are pulverised then bound together to form granules or extruded pellets, and heat treated to develop the active sites. GAC is sometimes used as an alternative filter medium to sand in addition to its specialist role in removing some organic compounds from water (Graese *et al.*, 1987, Hyde *et al.*, 1987).

The literature however presents very little information on ripening in terms of particle removal by GAC filter-adsorbers. GAC is too expensive a medium to be used simply for its particle removal properties, so studies concentrating on this would be unlikely to use GAC. There is however good reason for being concerned with ripening with GAC as filter media because of the possible release of bacteria mixed with carbon fines and protected from chlorine disinfection (Camper *et al.*, 1987).

Cleasby (1990) stated that chemical treatment was essential to depth filtration. Rapid filtration of low turbidity water may meet turbidity targets but not achieve 2 log *Giardia* removal without chemical coagulation (Craun, 1988, Al-Ani *et al.*, 1986, Saterdal *et al.*, 1988). Chemical treatment acts by flocculating colloid-sized particles to a larger size enhancing transport forces, and directly influences attachment forces. Many authors have written on rapid gravity filter operation and design. Useful reviews have been provided by Cleasby (1990), Kawamura (1975), Montgomery Consulting Engineers (1985) and Monk (1987), however filter ripening is seldom discussed in engineering text books.

In plants where primary filters provide preliminary treatment ahead of slow sand filters there may be intermediate water treatment goals. The need to remove particulate organic carbon (POC) to prolong slow sand filter (SSF) run lengths (reducing cleaning costs and down-time) can be appreciated by considering Toms and Bayley's (1988) cumulative POC load limit of 100 g $C.m^{-2}.run^{-1}$. For a filter to run for 30 days at 0.25 $m.h^{-1}$ (180 $m^3.m^{-2}.run^{-1}$) before reaching 100 g $C.m^{-2}$ an average applied loading of 555 μg $POC.l^{-1}$ would be needed. A useful target for primary filtration might therefore be 500 μg $C.l^{-1}$ or lower. Woodward *et al.* (1991) said that London's

reservoirs had a typical winter background POC of 200 - 300 $\mu g.l^{-1}$ when no chlorophyll *a* was measurable. Typically algae added 30 - 50 $\mu g.l^{-1}$ of POC for every 1 $\mu g.l^{-1}$ of chlorophyll *a*. Chlorophyll a peaks in excess of 10 $\mu g.l^{-1}$ were reported.

Cleasby *et al.* (1984) stated that a raw water suitable for slow sand filtration should contain less than 5 $\mu g.l^{-1}$ of chlorophyll *a* and a turbidity $<= 5$ NTU. It would be reasonable to apply these standards to primary filtered water. Woodward *et al.* (1991) gave primary filtrate chlorophyll *a* targets of < 5 $\mu g.l^{-1}$ average and < 15 $\mu g.l^{-1}$ peak for the operation of SSFs at > 0.3 $m.h^{-1}$. Janssens *et al.* (1982) set a turbidity limit of 0.2 JTU for direct filtration of low turbidity reservoir water ahead of SSFs operated at high rates of 0.40 $m.h^{-1}$.

SSFs have removed bacteria, viruses and much of the other particulate matter passing primary filters in London, reducing the requirement for careful control of primary filtrate quality. There is evidence that SSFs remove *Cryptosporidium* oocysts successfully (Timms *et al.*, 1995) so possibly targets for *Giardia* cyst and *Cryptosporidium* oocyst removal by rapid filtration can be relaxed where SSFs follow RGFs. Because of the difficulties of counting *Cryptosporidium* oocysts it might be prudent to treat the Timms *et al.* (1995) data with caution: Ives (1990) said that SSFs could not be considered an absolute barrier since Toms and Bayley (1988) had reported the passage of unicellular algae of similar size to oocysts through SSFs.

2.2.1 PRINCIPLES OF RAPID GRAVITY FILTRATION

The essence of deep bed filtration is the removal of small particles from water as the water passes through a bed of filter media. Ives (1970) presented an illustration showing that the particles which are to be removed from potable water are generally much smaller than the diameter of the filter media grains and also smaller than the interstices between the media grains.

Ives (1970) said that rapid filtration was applicable to dilute suspensions (less than 500 $mg.l^{-1}$) of particles from 0.1 to 50 μm. He said that larger particles should be strained or sedimented, smaller

particles flocculated and higher concentrations treated by cake filtration. Numbers of particles in water have been found to fall as size increases following power law functions (Montgomery, 1985). The design of a water treatment plant and the operating criteria, such as flow rates and chemical doses must be matched to the composition of the raw water, both in its chemistry and in its particle characteristics. Figure 2.1 shows that this mix of raw water and treatment processes determines the particulate loading applied to a filter. It is widely accepted that both particle transport and attachment are influenced by particle size, the chemistry of the water, additional chemical treatment, as well as the physical filtration conditions, such as filter design and operation. Filtration is thus a dynamic process influenced by a number of variables.

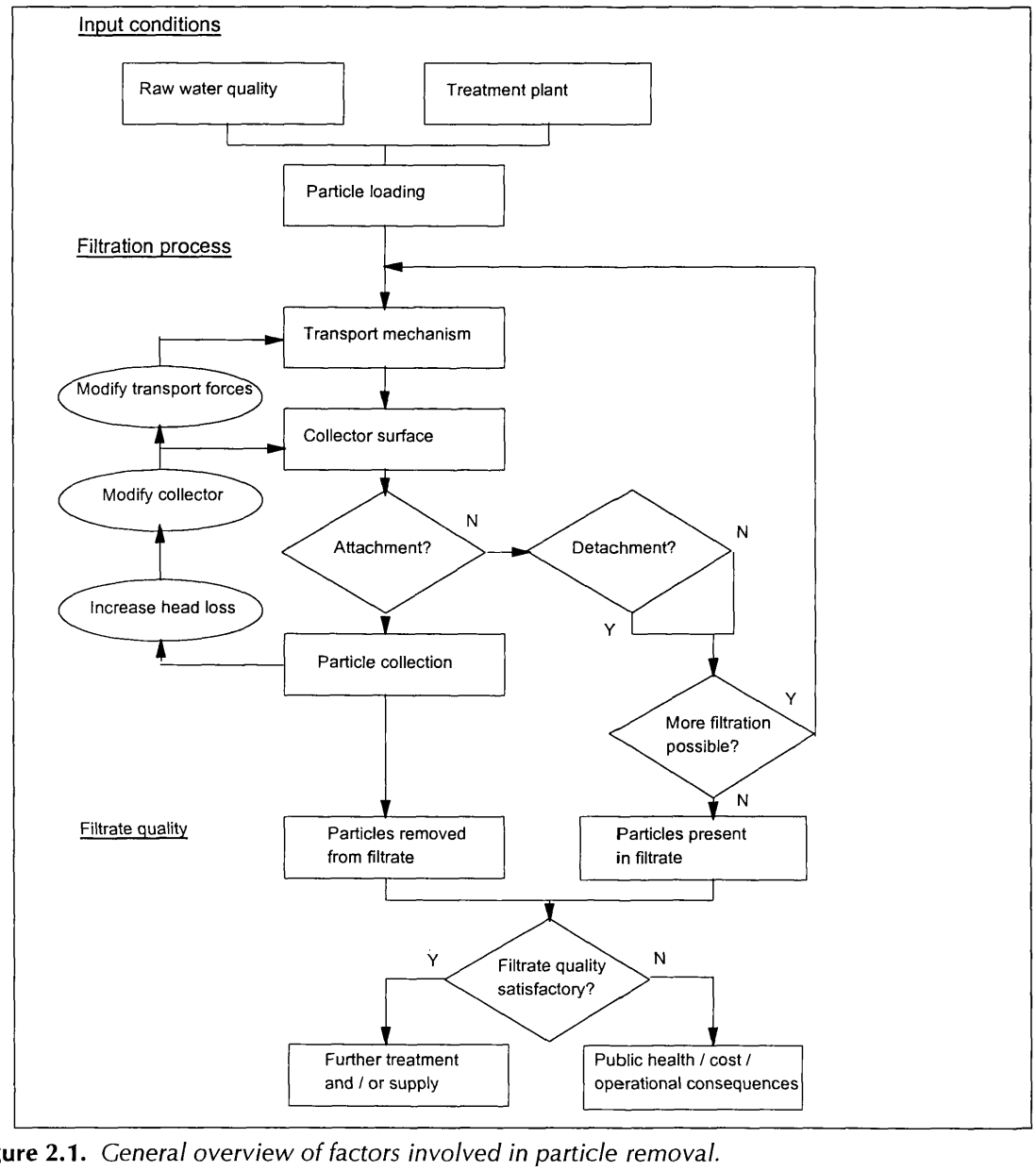


Figure 2.1. *General overview of factors involved in particle removal.*

Figure 1.1 shows that if collection is achieved the deposit modifies the collector structure which then influences transport and attachment forces, potentially to the benefit and even subsequent detriment of filtration. It constricts the pores in the media, and causes increased head loss. If attachment does not take place a particle will pass deeper into the bed and have further opportunities for removal. If there is no further filter available it will be present in the filtrate. This is termed breakthrough.

A deep bed filter is designed to store small particles on the surface of relatively large filter media grains. As filtration proceeds these deposits protrude into the pore spaces between the collecting medium. Pores become partially or fully clogged, and the resulting increased friction consumes the driving head across the bed (Ives, 1980). Therefore, as well as transport and attachment forces acting on the initial deposition, increasing fluid shear forces may cause some degree of modification to the structure of the deposit and even detachment during filtration. In order not to sustain excessively rapid filter clogging and loss of head it is essential that RGF design enables filtration to take place in the depth of the bed, as opposed to surface straining (Ives, 1970). Straining does take place occasionally on RGFs when high concentrations of particles such as filamentous algae are present in water (Ives, 1970).

A typical RGF achieving depth filtration will have a linear head loss curve (assuming constant flow rate). This is due to laminar flow conditions throughout the depth of the filter (Cleasby and Baumann, 1962a). If surface straining occurs in an RGF head loss development will have a curve where rate of change starts off slowly and gradually accelerates (Ives, 1980). Head loss may be described by the Kozeny equation (Cleasby, 1990).

If a filter continues to operate when no further driving head is available the filter may reach unacceptably low flow rates. High head loss, low flow rate or poor effluent quality are conditions which require the filter to be cleaned. A filter must be backwashed to remove accumulated deposits so that it can re-start at a low head loss. This is achieved by backwashing with clean water, often assisted by air scour.

2.2.2 MECHANISMS OF RAPID GRAVITY FILTRATION

Efficient filtration with porous media involves the capture of particles through the depth of the filter bed. For depth filtration to take place a **transport mechanism** is required to bring the particle into contact with the collector and an **attachment process** is required to ensure it is held there and remains there until backwashed to waste. Useful reviews of filtration for drinking water treatment, including filter modelling, have been prepared by Mints (1966), Ives (1970, 1980, 1982), O'Melia (1985), Amirtharajah (1988), Gimbel (1989), Tien (1989), Cleasby (1990) and Letterman (1991).

Small particles which are suspended in water are carried into the filter in the fluid streamlines. Ives (1980) has said that the flow in a filter is laminar, meaning that head loss is proportional to flow rate, and the streamlines do not cross, even when the pores are significantly clogged with deposits. Friction drag (viscosity) causes flow rates to be lower close to the media. Camp (1964) described the fluid streamlines converging at contractions in the pore channels.

In order to be filtered the particles must come out of the streamlines and come to rest on the media. If the particles are uniformly distributed in the flow not all of them will be in streamlines that bring them directly into contact with the media surface permitting capture (Ives, 1970). Furthermore, capture may not be fully achieved by every particle - collector collision: the adhesion probability is not necessarily unity. Therefore, if the filter is considered as consisting of serial layers not all the particles will adhere to, or even come into contact with, the first layer. The aim of depth filtration is to provide sufficient layers for adequate particle removal.

A number of physical, chemical and biological variables influence filtration but evidence is lacking on whether they influence ripening. Ives and Sholji (1965) reviewed a number of empirical studies and concluded that filter efficiency was inversely proportional to grain size, filtration rate, and the square of water viscosity. The effect these have on the drag forces and torques acting on the particle have been considered by the trajectory models of filtration, discussed in depth by Tien (1989).

The approach velocity of the water is conventionally quoted for flow rate, but as water passes into the filter its absolute velocity increases as it passes into the spaces (pore structures) between the grains. A typical sand filter has a porosity of around 0.4 (Stevenson, 1995), so interstitial velocities are on average 2½ times greater than approach velocities, although the range of pore geometries in a clogged filter may result in varied interstitial flow rates (Baumann and Ives, 1987).

Since both filter media and natural particles in water are negatively charged chemicals have been used for assisting particle removal in water treatment for about 100 years. Two principal effects, coagulation, and flocculation were described by Gregory (1989). Coagulation is the reduction in electrical charge on the particle (destabilisation) by the application of counterions which reduces the repulsion effect of the electrical double layer interaction. Flocculation of destabilised particles results in larger aggregations which may be more easily filtered. Chemical use must be closely controlled: particles which have adsorbed excessive polymer may have enhanced stability, termed steric stabilisation (Gregory, 1989).

Little consideration has been paid to biological processes in filter ripening. Tien (1989) ruled out any consideration of biological effects on filtration (in a footnote on page 7 of his book), stating "The presence of bacterial growth and its effect are not considered in this book".

2.2.2.1 TRANSPORT MECHANISMS

Camp (1964), describing Stein's (1940) work, said that removal of floc was principally due to the convergence of streamlines and the surface of the collectors or floc already deposited on the collector, where they contract to pass through the pore spaces. Where a suspension of many particle sizes is filtered different mechanisms will affect the different sizes. Yao *et al.* (1971) demonstrated that transport efficiencies are minimum for particles around 1 μm in diameter, as they are too large for diffusion, but too small for the effects of gravity (Ives, 1980). For this reason Ives (1980) said that bacteria have to be flocculated to be removed effectively on a RGF. The transport forces assisting particle filtration have been summarised by many authorities. They are presented in

table 2.3. Ives (1970) also mentioned that Camp (1964) had suggested that filter pore flocculation would assist filtration, although it is not a transport mechanism as such.

Ives (1980) has said that the effect of gravity falls as flow increases whilst inertial effects increase. As many studies have demonstrated that filter efficiency falls as flow rate increases Ives (1980) said that inertia as a filter mechanism was not applicable to water filtration.

Table 2.3. *Summary of possible transport forces in filtration. (After Ives and Gregory, 1967, Ives, 1970, 1975, 1980, Amirtharajah, 1988, Gimbel, 1989, Tien, 1989, and Letterman, 1991).*

Particle diameter	Mechanism	Action
< 1 μm	Diffusion	The diffusion mechanism operates on sub-micrometre diameter ("sub-micron") particles; they move across the streamlines according to Brownian motion causing them to encounter media grains.
> 1 μm	Gravity	Ives (1970) gave the example of the sedimentation of larger particles e.g. Kaolinite 2-10 μm onto the collector, and Ives (1980) said that even low density particles, with a low settling velocity, may be acted upon by gravity, due to the very low velocity of the water flowing near to the filter grain surface. Ison and Ives (1969) observed gravity effects even with up-flow filtration.
> 1 μm	Interception	The particle is in a streamline which flows so close to the collector, known as the limiting trajectory, that the particle touches the collector. This mechanism arguably is the final condition which all the others reach.
> 1 μm	Inertia	The particle is unable to change direction as the streamline bends around the collector
> 1 μm	Hydrodynamic shear	The different flow rates of the streamlines act on non-spherical shapes to cause an apparently random motion which cause a particle to move across the streamlines and bring it into contact with the collector.
> 1 μm	Hydrodynamic retardation	Due to the approaching particle displacing the volume of water between it and the collector. The motion of this fluid slows the approach of the particle and causes it to rotate. It also uses up the kinetic energy of the particle which may present an energy barrier preventing adhesion.
> 1 μm	Straining	This may take place in the depth of the filter where bridging of the pore constrictions is caused by the simultaneous arrival of several particles. There might also be "pinch" points created by the random packing of angular faces of adjacent collectors.

2.2.2.2 ATTACHMENT MECHANISMS

Once the particle has made contact with the filter grain attachment forces must hold it there. These are described in table 2.4. Gregory (1975) and Ives (1980) have illustrated diagrammatically how

the electrical repulsion and van der Waals attraction forces sum to produce a total interaction curve for the attachment of particles. A review of colloidal interactions may be found in Gregory (1989).

Table 2.4. *Summary of possible attachment mechanisms in filtration. (After Ives, 1970, 1980, Ives and Gregory, 1967, Gregory, 1989, and Letterman, 1991).*

Mechanism	Action
Electrical double layer interaction	Electrostatic forces may be favourable, for instance, where the electrical double layers on filter media and colloidal particles are oppositely charged, they will attract each other. If they are the same there will be a tendency to repulsion. In a lowland river Ives (1970) said that these forces only act over short distances, around 10 nm.
London - van der Waals forces	Attraction between atoms and molecules, due to fluctuating dipoles set up by the movement of electrons (Gimbel, 1989), termed London-van der Waals forces, operate up to a range of 50 nm (Ives, 1970).
Mutual adsorption	Ives (1970) described attachment mechanisms due to polymer bridging between particles, and also where an intermediary ion of opposite charge (Ca^{2+} or H^{+}) linked an anionic polymer and a negatively charged particle.
Electrostatic patch	Where a cationic polymer only covers parts of negatively charged particles an electrostatic patch model for attraction has been proposed (Gregory, 1989).
Hydration	Gregory (1989) thought hydrogen bonding might indirectly induce hydrophobic interactions between particles, however Ives and Gregory (1967) and Ives (1970) thought it was not a mechanism in filtration of colloidal solids,
Biological factors	In a review of slow sand filtration Ellis (1985) described that removals of bacteria could be achieved by predation, and that extra-cellular polysaccharides exuded by bacteria and fungi could also assist filter attachment. In the absence of pre-chlorination it is possible that these mechanisms could contribute to rapid gravity filtration.

2.2.2.3 PORE FLOCCULATION

Filter pore flocculation is a possible mechanism for ripening a) if it actually takes place at all, and b) if shear conditions in the filter change following initial deposition to create larger, more filterable flocs. According to Ives (1970), Camp (1964) proposed that particle aggregation could take place by orthokinetic flocculation in filter pores, although Camp (1964) said sedimentation and pore coagulation (taken to mean pore flocculation) were of minor significance. Ives (1975a) said it was a well known phenomenon.

There has been, however, some debate as to whether filter pore flocculation does come about and the evidence for pore flocculation is doubtful, or its significance in filtration is only minor. Graham

(1988) thought that pore flocculation as a filtration mechanism was third in significance behind changing particle-collector attachment efficiency with chemical destabilisation and polymer bridging, and preferential deposition on already attached particles.

Tanaka and Pirbazari (1986) showed the growth of particles through the filter and explained their origin from either contact flocculation (i.e. on the media surface) or within pore flocculation. For it to be contact flocculation re-entrainment in to the filtrate must occur for the larger particles to be measured.

Adin and Rajagopalan (1989) thought that flocculation in the filter pores was not a mechanism occurring in filtration, and that the growth of particles occurred in contact with deposited particles. Contact flocculation may help increase the attachment probability, and has been inferred by Darby and Lawler (1990) and Moran *et al.* (1993a) from large particles in the effluent and assuming that detachment of large aggregates had taken place. The hypothetical formation of dendrites in the O'Melia and Ali (1978) ripening model was possibly an example of contact flocculation.

Averill *et al. (1990)* presented particle size distribution data which they claimed showed pore-space flocculation was a significant mechanism in changing particle size distribution with a filter. They preferred this to re-suspension since particle size growth increased then reduced as flow rates continued to rise. They thought that detachment would continue as flow rate increased. Whether at high shear stresses detached flocs actually broke up into smaller components was not considered. A further reason was that the pore flocculation effect was said to seen early in the filter run, presumably before detachment would be taking place, but Averill *et al.* (1990) did not show the data upon which they based this conclusion.

The fluid shear conditions within a filter were modelled by Hunt *et al.* (1993). Calculated clean bed and clogged bed fluid shear rates were higher than those used in flocculation basins (table 2.5).

Table 2.5. *Fluid shear rates through two sizes of clean and clogged sand (G s^{-1}) given by Hunt et al. (1993).*

Approach velocity ($m.h^{-1}$)	0.54 mm sand G (s^{-1}) clean	0.54 mm sand G (s^{-1}) clogged	1.29 mm sand G (s^{-1}) clean	1.29 mm sa G (s^{-1}) clogged
5	110	790	58	220
10	200	1430	110	390
20	220		820	

2.2.2.4 THE DYNAMIC PARTICLE ADHESION PROCESS

Gimbel (1989a) explained attachment probability by describing a "dynamic particle adhesion process" where the particle, after it has reached the collector, is subject to both attachment and separation forces. Tobiason and O'Melia (1988) summarised these forces as the attraction due to London-van der Waals forces and gravity, the repulsive force of electrical double layer interactions and a lateral force due to fluid drag. According to Gimbel (1989a) the particles will not reach static adhesion until their kinetic energy has been used up, partly by drag forces and torques and partly by conversion to potential energy against the van der Waals attraction.

Gimbel (1989a) observed with a microscope and high-speed camera that after contact had been made between particles (glass beads 30-35 μm diameter) and collector (quartz plate) the particles continued to move along the surface of the collector, but not necessarily at the same distance to the collector. It appeared they were "jumping up and down" as they travelled. Adding $CaCl_2$ to the solution, or cationic polymer to the solution or the collector, reduced the time taken for the particles to finally come to rest. It was thought that the polymer reduced repulsive electrostatic forces and established polymer bridges between the particles and collectors.

2.2.2.5 DETACHMENT

One of the important questions in filtration has been whether the particles can be re-entrained into the flow, the so-called Mints-Ives controversy of the 1960s (Anon., 1976, Averill *et al.*, 1990, Ginn and Amirtharajah, 1990, Ginn *et al.*, 1992). This may be by collision with other particles as they arrive at the grain surface, or by shear forces increasing as flow rate rises to compensate for the pore spaces becoming clogged. More recently Ives has moved from opposing to actually observing detachment, so the debate is perhaps closed (Ives and Clough, 1985, Ives, 1989, 1990a). It is

generally agreed that some particles may become detached after initial successful filtration. The subject has been examined in detail by Moran *et al.* (1993a).

Clearly part of the case for filter pore flocculation hinges on whether detachment of previously retained deposits takes place in filtration. If it does this could account for the growth of particles within the bed; if not then particle size growth would be due to within-pore flocculation.

The effect of detachment in ripening is interesting. Can the production of large particles facilitate the removal of other particles? Does detachment contribute to ripening lower layers of filter media? There is no mention of re-entrainment of deposits during ripening in the endoscopic observations of Ives (1989, 1990a). Detachment was only seen to take place once considerable amounts of deposit had been collected, which were then dislodged by the arrival of further particles. Ison and Ives (1969) said that there was little tendency for detachment in the early stages of filtration. Camp (1964) described the behaviour of flocs in Stein's (1940) two dimensional model filter. A floc observed arriving at a point on a collector did not always adhere firmly. If it did not adhere it would roll along the collector surface and adhere at another point, or it could pass back into the flow.

In ripening the presence of particles on the media probably aids the transport mechanism and attachment efficiency probably increases due to lower rates of detachment of particles brought to the collector surface. The particle is intercepted by other particles on the media surface, or retained by a complex polymer structure, rather than allowed to roll off the collector under the influence of its own kinetic energy. This is analogous to bouncing a ball on soft dry sand compared to bouncing it on a hard surface.

2.2.3 THE FILTER CYCLE

The net effect of transport and attachment processes in filtration is to gradually increase the volume of deposit collected within the filter bed. The consequences of this gradual increase in specific deposit (σ) (volume of deposited particles per unit filter volume) have been examined by many workers by measuring changes in head loss and filtrate quality, or filter coefficient (λ) (collection efficiency), as the filter run proceeds. A number of consecutive operating phases, based on filtrate

quality, may be discerned, which together with the filter backwash define the filter cycle of a granular media deep bed filter.

The relationship between λ and σ has formed the basis of a number of mathematical models of filtration described as empirical or phenomenological models, summarised by Tien (1989). Ives (1975a) summarised in graphical form models of the relationship between filtration coefficient and specific deposit proposed by several workers. A modified version of this diagram is shown in figure 2.2. Where filter coefficient initially increases the model shows ripening.

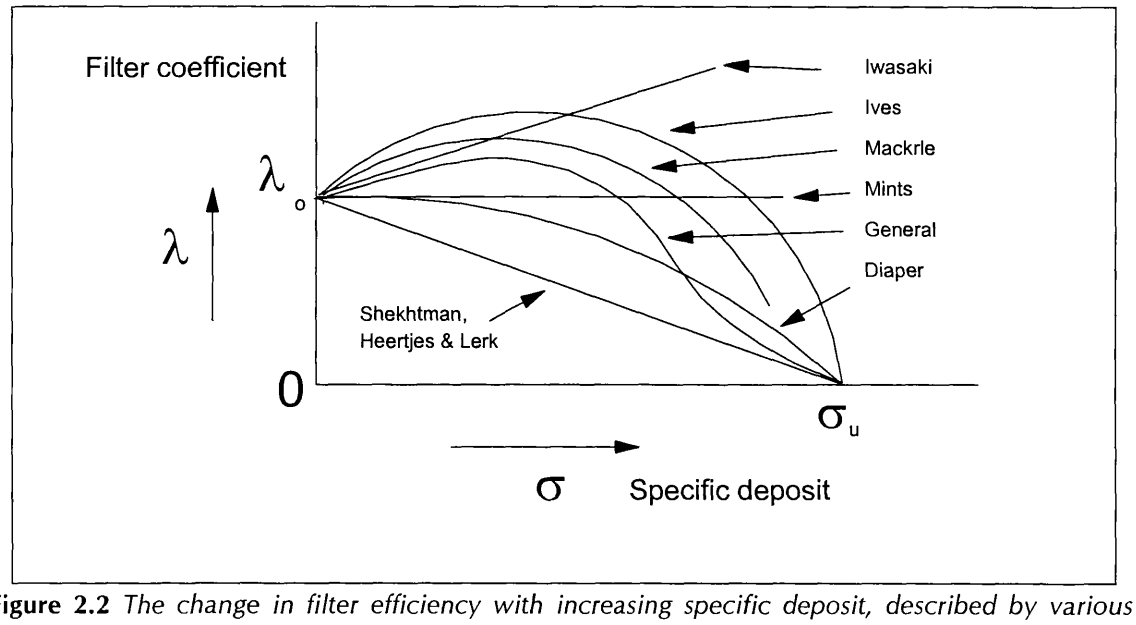


Figure 2.2 *The change in filter efficiency with increasing specific deposit, described by various authors. (After Ives, 1975a, 1993 pers. comm.).*

2.2.4 PHASES OF A FILTER CYCLE

As a consequence of ripening filtrate quality usually reaches an optimum value, and in an ideal situation this is maintained for a large proportion of the remaining filter run. In some instances, however, ripening leads shortly into breakthrough, in other examples good filtrate quality may continue to the end of the filter run. Several studies have identified different phases in the filter cycle but have used slightly different terminology. Figure 2.3 shows an idealised representation of the filter cycle, incorporating seven phases described in the literature. The filter cycle is started at the end of a backwash, and terminated with a backwash.

Not all phases have been observed in any one study, and their duration and filtrate quality may be variable. The lag and deteriorating phases, given as 1-10 minutes by Amirtharajah and Wetstein (1980), are rarely mentioned in the literature. Where they are, measuring conditions, i.e. sampling intervals or instrument responsiveness, may mean these phases, or some of the detail, are missed. In figure 2.3 the numbers in circles correspond to the numbers in the following text:

1) the lag phase (Amirtharajah and Wetstein, 1980, Cranston and Amirtharajah, 1987), where clean backwash water passes out of the filter from the filter underdrains (up to time T_u);

2 and 3) two deteriorating or pre-ripening phases, where effluent quality becomes poorer, exhibiting two peaks (Amirtharajah and Wetstein, 1980, Cranston and Amirtharajah, 1987), caused firstly by dirty backwash water remnants within the bed (up to T_m), then secondly by backwash water above the media because of the particles it contains, and the dilution effect of re-stabilising the influent suspension (up to T_{a+i});

4) the ripening period, where filtrate quality improves (up to T_{r1} or T_{r2}, depending on the definition of ripening);

5) a ripened, effective filtration (Ginn *et al.*, 1992), optimum filtration (Cranston and Amirtharajah, 1987) or working (Janssens *et al.*, 1982, Vigneswaran and Chang, 1989) stage (from T_r to T_b);

6) a period of deteriorating filtrate quality, known as the breakthrough phase (recognised in all the references cited) (from T_b);

7) finally a period of no further deterioration where breakthrough has reached a maximum value (Vigneswaran and Chang, 1989), and wormhole flow (Baumann and Ives, 1987) may be occurring (after T_w).

This thesis will include the pre-ripening or deteriorating phase in the ripening period, it will thus examine the entire period following backwashing up to attainment of an "acceptable" filtrate quality.

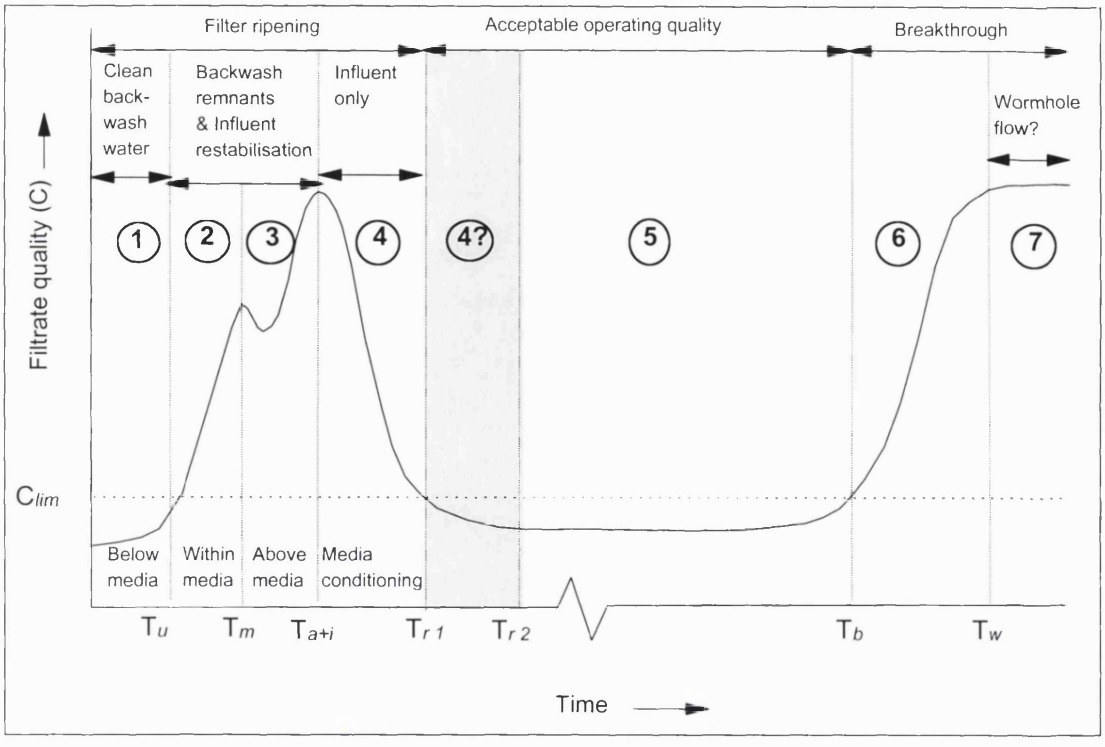


Figure 2.3. *An illustration of a seven phase filter operating cycle showing the pre-ripening phase, the ripening period, a best operating stage and the breakthrough phase. (After Amirtharajah and Wetstein, 1980, Janssens et al. 1982, Cranston and Amirtharajah, 1987, Vigneswaran and Chang, 1989, and Ginn et al., 1992).*

2.3 THE DEFINITION OF RIPENING

In the literature very little is said of what constitutes the end of the ripening period. Is it achieving a quality target or a complete end to the improvement process? Ripening is usually poorly defined, generally with no clear end point. Moran *et al.* (1993) simply defined it as the "improvement in effluent quality early in a filter run". For Tobiason *et al.* (1993) the ripening period was defined as the "period of declining effluent concentration to nearly complete removal". In this review of the literature, where graphs have been examined, the second definition has been taken to signify the end of ripening, except where stated.

The literature shows there is not one universal ripening curve. Typical ripening curves show a rate of improvement that starts high and gradually declines. Ripening generally implies that a ripened stage is reached. It is however not necessarily true that the best phase of filtration is an absolutely constant quality; there are inevitable fluctuations due to "random", or not well understood,

breakthrough events, or breakthrough precipitated by flow variations, or even small variations due to measurement sensitivity. So at what point can ripening be said to be complete?

The end of the ripening period could be defined in a number of ways:

when all filtrate quality parameters meet acceptable targets;

when a useful and easily measured parameter (i.e. well correlated with other water quality parameters) meets a target, i.e. C_{lim} in figure 2.3 which might be a low turbidity value;

when the filter achieves a performance target suggested as being likely to indicate satisfactory performance (e.g. log. removal or percentage removal value);

when no further improvement is noted (difficult to observe except after the event);

when no further significant improvement is observed (i.e. improvements becoming extremely small between readings, especially if close to the reliable sensitivity of the instruments used);

when a fixed proportion of the overall best removal has been reached (e.g. if the filter removed at best 90% of the load, then 90% of this value might be used, i.e. when 81% removal was reached;

when a time limit had expired (likely to be used in filter to waste, but not necessarily directly indicative of the completion of ripening);

when the filtrate reaches a value which can be mixed with other filtrates and not compromise overall treatment plant performance (difficult to assess on a single filter pilot plant).

For primary filters the aim was to assist the main slow sand filtration treatment process so the first two approaches are not appropriate to the filters in this study. The last approach allows consideration of filter runs under the whole range of input water qualities and chemical treatments, and where the filtrate did not reach any recognised absolute target.

In laboratory studies influent water quality may be kept constant. In contrast, in actual water treatment practice the influent quality may change at any time. The variation in influent quality may

lead to difficulties interpreting filter ripening unless the filtrate quality remains constant or the removal is unaffected.

Of the few authors who have attempted to define the end of the ripening period Barnett *et al.* (1992) had a target of 0.5 NTU, which was reached after a filter to waste period averaging roughly 0.5% of the total filter run time. Janssens *et al.* (1982) and François and Van Haute (1985) suggested that ripening was achieved when the improving phase reached 0.2 JTU. The same value was also the point at which breakthrough was defined and filter runs were terminated. Watson (1990) used the EC guide level 0.4 NTU.

The value of 0.2 JTU was not the best quality that Janssens *et al.* (1982) reported their filter could achieve, since values later in the filter run approached 0.1 JTU. This meant that their ripening point was not the point at which quality improvement stopped, so ripening was continuing. It is necessary to distinguish what is meant by ripening in this instance - reaching a target, or reaching the point of no further improvement. Can improvement in filtrate quality beyond the ripening point be considered as anything other than ripening? Their work presented a practical approach to ripening by considering the point at which an acceptable water quality was produced. A plant operator would not desire to run a protracted filter to waste period whilst filtered water quality gradually reached a minimum, if much of the wasted water met filtered water quality targets. An area has been marked 4? and shaded on figure 2.3 to illustrate this point; C_{lim} is the target filtrate quality, but it is not the minimum filtrate quality achieved.

Baumann (1989) reviewed the history of filtration from a health perspective and noted that the American Water Works Association had suggested a goal of 0.1 NTU for potable water leaving a works. Ripening would be completed when the quality of filtrate from a recently backwashed filter could be mixed with the water from other filters in operation without causing elevated final water turbidities. The actual value of the ripening point would depend on the dilution. Plant operation would need to be controlled to avoid affecting overall plant turbidity; breakthrough, which could

result from poor flow control or running filters too long, should be prevented to avoid an impact on the dilution water.

Al-Ani *et al.* (1986) could be considered to have given a definition of successful filtration applicable to their studies on low turbidity waters. They said that if "raw water turbidity is 0.5 NTU and if removal of turbidity is 70 percent or greater, the probability is 0.85 that the removal of *Giardia* cysts will exceed 99 percent."

2.4 RESEARCH INTO FILTER RIPENING

Information in the literature on filter ripening may be categorised into two basic sources: the first is where research has specifically investigated ripening and the second is filtration research which yields information on ripening, even if that was not the principal focus of the work reported.

Three principal research methodologies may be described:

i) empirical data may be gathered, such as head loss measurement and filtrate quality, where general understanding can be inferred;

ii) real-time observations have been made within a filter using endoscopes (Ives and Clough, 1985), and a number of papers have presented details of visual observations from scaled-up model filters (e.g. Camp, 1964, Ison and Ives, 1969, Payatakes *et al.*, 1981);

iii) building a mathematical model and testing it against experimental data.

The first case covers the empirical study of the behaviour of a filter over a period of many filter runs to see what happens, usually at pilot or operational scale. This may include several different measures of filter performance. It includes studies to determine the influence of the influent water quality on ripening, and studies of how backwashing influences ripening. Managing filter operation and water treatment chemistry to control ripening is covered as well as questioning whether ripening is an issue. The significance of ripening for optimising plant operation or safeguarding public health may need to be reviewed on a case by case basis since the usual measured

parameters, turbidity and particle counts, give only indirect evidence of the suitability of water for human consumption.

The second and third categories include studies that have been conducted in the laboratory under controlled conditions to assist with the evaluation of conceptual or mathematical models of filtration and filter ripening. These studies have used uni-sized and multi-sized suspensions of particles. The interactions between particles and the media as filtration proceeds are considered to see how ripening might be explained. Detailed analysis of filtrate quality has provided the data with which insights into filtration have been made.

The mathematical modelling approach attempts to deduce what is happening in filtration, then checks model predictions against experimental results to test the success of the model. It is generally the objective of these models to predict filtrate quality and growth in head loss. Much of this work has its origins in clean water filtration, but some information has been imported from waste water filtration and air filtration.

Filtration modelling has followed two distinct approaches: the empirical method involves producing equations specific to a given set of conditions; the alternative approach involves determining the fundamental forces governing the transport of a particle to a collector and controlling the attachment step. These mathematical models of filtration, based on predicting the path of a particle in a filter, have been developed principally by Tien and co-workers, and are known as trajectory theories (see Tien, 1989).

Tien (1989) and Gimbel (1989) have provided in-depth reviews of the mathematical treatment of transportation, deposition and attachment theories in deep bed granular media filtration. Their major drawback is the need to make simplifying assumptions. For example, the shape of particles and collectors are often spherical (Letterman, 1991), and initially only clean bed conditions were modelled (e.g. Rajagopalan and Tien, 1976). Whilst these may be tested in the laboratory, the results offer only limited insight into what actually takes place in water treatment plants. Later

workers have attempted to model the entire filter cycle. To achieve this a change in deposit morphology was proposed to take place after ripening, which initiated breakthrough, from a smooth coating (where λ increases with σ) to a pore blocking morphology (where λ decreases with σ) (Tien and Payatakes, 1979, Tobiason and Vigneswaran, 1994).

2.5 OBSERVATIONS OF FILTER RIPENING

Many filtration studies have ignored filter ripening. This may be due to deliberate choice, for example where an attempt to model clean bed filter behaviour has been the objective. Historically, the significance of ripening was underestimated and it was dismissed. Ripening can be missed if the choice of sampling times in a filtration study is too late or too infrequent. Engineering text books have generally paid little attention to filter ripening and filter design may reflect this, giving a filter operator little control over the ripening period. Thus, the ripening phase is not always observed or recognised, and is often not reported in investigations of filter performance.

2.5.1 THE HISTORY OF RIPENING OBSERVATIONS

Amirtharajah and Wetstein (1980) cited four studies where initial degradation of filtrate quality had been shown but was not investigated in depth; the references were Hudson (1935), Ling (1955), Cleasby and Baumann (1962) and Harris (1970). Ives (pers. comm.) stated that they missed Stein (1940). Ripening appears to have been ignored as the studies discussed other aspects of filtration, but it would be interesting to know why ripening was not an issue to earlier workers.

Ling (1955) demonstrated on a conventional plant a deteriorating phase and a ripening phase following backwashing. He termed this the "lag period". He noted that the time to reach the peak (poorest) turbidity, and the height of the peak, were related to flow rate, the poorest quality being attained in the fastest time at the highest rate. Since the lag periods lasted less than an hour, which he considered relatively short compared to the overall filter run time, and the peaks were considered to be low turbidity values, Ling (1955) concluded that the "effect of the lag period to the over-all plant removal (was) not as important as it was thought before." This referred to an earlier source (Davis, 1942).

Ling (1955) found that the peak turbidity corresponded to the detention time in the filter for the post-backwash influent water to reach the sample point. He regarded the particles in the influent water to be responsible for the peak. A further interesting finding was that the turbidity removal per inch of filter media started high in the top inch but reduced over time. In subsequent layers the removal built to a peak (ripened) but then deteriorated in a time sequence corresponding to a filtration front moving progressively down through the filter.

Similarly, Camp (1964) demonstrated a progressive ripening, showing from the data of Eliassen (1935, 1941), how rate of removal of iron varied according to depth of sampling within the filter and filter run time. The surface of the filter started with a high removal rate which fell almost immediately. A ripening period of 10 h was reported for the sample point at 0.1 ft (30 mm). At 0.2 ft (60 mm) into the bed a clear ripening period lasting 20 h was almost immediately followed by deteriorating performance. Further down the bed the ripening period was longer, and removal performance remained at the ripened value. Ripening at 0.5 ft (150 mm) lasted 40 h.

The literature reveals that ripening has been observed under many different experimental conditions and using many different water quality parameters. It is therefore likely to be a real effect, potentially of significance. The following sections describe observations of ripening using several different measurements.

2.5.2 FILTRATE IRON CONCENTRATION

Ripening has been demonstrated from measurements of filtrate iron concentration, where iron was the coagulant, by Cleasby and Baumann (1962, 1962a), and Camp (1964) using the results of Eliassen (1935, 1941). Amirtharajah and Wetstein (1980) demonstrated that several phases comprised the ripening behaviour of filters using filtrate iron concentration. It should be noted that the iron concentrations reported by Cleasby and Baumann (1962a) during ripening were greatly in excess of the 0.2 $mg.l^{-1}$ standard applicable to treated water in Europe in the 1990s.

2.5.3 RIPENING OF SUSPENDED SOLIDS

Vigneswaran and Chang (1986) showed a ripening phase for suspended solids of a suspension of 5 μm Kaolin particles lasting up to 2 hours for the lowest concentration and less time at higher solids concentrations At the highest loading the filter showed no ripening, instead breakthrough was immediate. Darby *et al.* (1991) and Clark *et al.* (1992) also demonstrated ripening with suspended solids measurements.

2.5.4 RIPENING MEASURED BY FILTRATE TURBIDITY

Turbidity is perhaps the parameter most commonly used to show filter ripening, starting with Ling (1955), including Cranston and Amirtharajah (1987), Darby *et al.* (1991) and Clark *et al.* (1992) amongst many others. Vigneswaran and Chang's (1989) data of turbidity removal showed the duration of ripening with 12 μm Kaolin lasted between 0.5 and 1.5 hours. The duration of the ripening phase did not show a consistent relationship with flow rate. The degree of turbidity removal was clearly associated with the flow rate with filtrate quality better the lower the flow rate. Since it is discussed widely in the thesis it is not given special attention here.

2.5.5 RIPENING MEASURED BY PARTICLE COUNTERS

A number of techniques are available to count particles >0.5 μm equivalent sphere diameter in water. Studies such as those by Borrill and McKean (1993), Hillis and Colton (1995) and Hargesheimer and Lewis (1995) have demonstrated ripening of particle counts on full scale filters, using a variety of batch and on-line instruments.

At laboratory scale studies have demonstrated ripening in particle counts with a number of different particles (Kaolin, latex spheres, pollen, $CaCO_3$), sources of water, and depths and media grain diameters of filter beds. Coulter counter data have been reported by Clark *et al.* (1992), Darby and Lawler (1990), Darby *et al.* (1991), Darby *et al.* (1992), Kau and Lawler (1995), Mackie *et al.* (1987), Mackie (1989), Mackie and Bai (1992,1993), Moran *et al.* (1993), Tobiason *et al.* (1993), Veerapaneni and Wiesner (1993) and Vigneswaran and Ben Aïm (1985).

2.5.6 RIPENING OF BACTERIA

Bucklin *et al.* (1988) examined water samples over the filter ripening period for total coliform bacteria and heterotrophic plate counts. They found that the former showed trends which related to the turbidity trends and the initial peaks seen with turbidity were reported for coliforms, probably for the same reasons. There were considerable variations in the bacteria data and turbidity was not a good surrogate for bacterial enumeration.

The use of chlorinated backwash water was a significant factor in Bucklin *et al.*'s (1988) results since it caused injury to coliforms which meant it was necessary to use a culture method which measured injured coliforms. They warned against drawing erroneous conclusions from tests which measured only viable bacteria since these might show that bacteria were absent in the initial ripening period, falsely suggesting that a filter to waste period was unnecessary. There was the possibility of injured bacteria recovering later in the distribution system or else providing a nutrient source for other distribution system organisms. High heterotrophic plate counts were most commonly associated with the first appearance of the influent through the filter, when the chlorine in the backwash water had passed through (Bucklin *et al.*, 1988).

Cleasby *et al.* (1984) presented data showing that the ripening of counts of coliform bacteria in direct in-line filtration took up to 6 hours, very similar to their turbidity data. Denny and Pitchers (1993) in contrast said that ripening of thermotolerant coliforms to pre-backwash counts was slower than ripening of turbidity. Turbidity values returned to pre-backwash values within 30-60 minutes, whereas coliform counts were still higher after 4 hours. There was no difference between a GAC and a sand filter in this respect. It must be noted that the coliform numbers were very low (< 2 per 100 ml), although 500 ml samples were taken. This, and the unusual way of judging ripening, might mean their interpretation of slower ripening was rather severe. This method of judging ripening is interesting, but it relies on the filter still being in its optimum phase prior to backwash and not in a breakthrough phase.

2.5.7 RIPENING OF ALGAE

Few studies have demonstrated ripening for algae but some evidence has been presented by Maroudas (1966) for ripening of *Chlorella*, and by Haarhoff and Cleasby (1989) in contact filtration trials with a *Chlorella* monoculture. The latter demonstrated filter ripening by turbidity measurement in their trials. They found high correlations between turbidity and three measures of algal counts (spectrophotometry at 680 nm, $r=0.91$, suspended solids, $r=0.96$, and particle volume, $r=0.91$), giving good circumstantial evidence for the ripening of algae.

To investigate the ripening of a *Giardia* surrogate Bucklin *et al.* (1988) measured concentrations of "*Giardia*-sized" diatoms in the filtrate over the ripening sequence. No *Giardia* cysts were found in the influent but there was a wide range of particles present, and, in the opinion of Bucklin *et al.* (1988) there were sufficient numbers of "*Giardia*-sized" particles present to make the test worthwhile. The filtrate was sampled from 0 - 7.5 minutes after backwash, then 8.5 - 60 minutes and finally 60 - 120 minutes. The first sample contained a "moderate amount" of diatoms, in the second diatoms were present, but "rare", and the third sample was "significantly cleaner" than the previous two. Turbidity peaked at 0.6 NTU falling to 0.4 NTU after 2 h.

2.5.8 RIPENING OF PARTICULATE ORGANIC CARBON

Watson (1990) presented mean POC results averaged from many filter runs on four different media types. Single, dual and triple media filters were examined and each showed the same trends, although the actual filtrate quality was superior from the triple media filter.

During May and June the filtrate POC quality continued to improve for the first 24 hours of filter operation. The triple media (anthracite, sand and garnet) ran for 48 hours continuing to improve, whereas the anthracite and sand dual media filter appeared to reach a ripening point at around 24 hours operation. GAC filters reached maximum head loss before quality improvement had finished. Results from August to November showed that ripening occurred on all the filters, and was achieved by 8 hours in each case. It is clear from Watson's (1990) work that the abundance of algae in the water varied, as did the species composition, and the effect of different algae on flocculation and filtration was variable.

Watson (1990) also showed typical filtrate turbidity curves with a deteriorating phase, a rapid ripening phase then a slow ripening phase and finally a steady optimum phase. The data for two of the filter runs showed remarkable agreement between turbidity and POC.

2.5.9 RIPENING OF *GIARDIA* CYSTS AND *CRYPTOSPORIDIUM* OOCYSTS

Since ripening is a period of sub-optimal filter performance, even in a well-operated plant, there is a potential threat to human health if water treatment plants fail to remove *Giardia* cysts and *Cryptosporidium* oocysts during the ripening period since these are not inactivated by conventional chlorine disinfection. Ozone appeared to be a potential agent for *Cryptosporidium* inactivation at practicable doses and contact times (Ives, 1990), but Ives (pers. comm., 1993) said that this was in doubt following a recent WRc Foundation report.

A number of studies have indicated that during ripening filters allow the passage of *Giardia* cysts and *Cryptosporidium* oocysts. Goldgrabe *et al.* (1993) stated that LeChevallier *et al.* (1991) had shown that the filter ripening period was the time in the filter cycle when the most *Giardia* cysts could pass through the filter. Work by Logsdon *et al.* (1981) with a low turbidity water coagulated with alum or alum and polymer showed that during a ripening phase of 0.5 h the penetration of *Giardia muris* cysts was 10 to 25 times greater than in the ripened filter. Remarkably there was less than a 0.1 NTU turbidity improvement over the period from ripening to the filter operating at its best. *Giardia muris* was used as a model for *G. lamblia* which is the human pathogen. Ainsworth (1990) reported that 99% *Giardia* cyst removal could be achieved by coagulation and filtration, but that filtrates often contained higher cyst numbers for up to 30 minutes following backwashing. Further evidence of oocyst penetration during ripening in U.S. treatment works was collected in an OSTEMS report (Benton *et al.*, 1991) mentioned by Ives (1995).

2.6 THE CAUSES OF FILTER RIPENING

Three major causes have been identified for filter ripening. In order of occurrence in a filter cycle, the first is due to the remnants of the dirt from a previous run being present in the pores of the

media and in the residual backwash water above the bed (Amirtharajah and Wetstein, 1980, Amirtharajah, 1985, Cranston and Amirtharajah, 1987), the second was the mixing of the influent with the residual backwash water (Cranston and Amirtharajah, 1987), and the third was due to the clean filter media being less efficient than media bearing retained deposit, possibly by the formation of dendrites (O'Melia and Ali, 1978) or the blockage of pore spaces (François and Van Haute, 1985).

A further cause of poor filtrate quality following backwashing was suggested by Cranston and Amirtharajah (1987) and Bucklin *et al.* (1988) as being due to the general disturbance to particles which had previously settled in the pipework due to the movement of valves during or following the backwash.

The net effect of these mechanisms is to produce the complex ripening curve presented in figure 2.3. In this diagram phases 1, 2 and 3 may be termed the pre-ripening phase, whilst phase 4 is ripening proper, although the overall period is referred to generally as the ripening phase.

2.6.1 THE INFLUENCE OF BACKWASHING ON FILTER RIPENING

Amirtharajah and Wetstein (1980) showed that the peak iron concentration in the filtrate during the first peak, and also the total iron in the effluent in the 15 minutes following backwashing were inversely proportional to the time taken by the backwash valve to close. They said that this was not due to scouring of deposits from the media. François and Van Haute (1985) found no evidence of the rate of closure of the backwash valve influencing filtrate quality but suggested that the rate of closure affected the packing of the filter bed. The rapid closure of the backwash valve might not allow the bed to settle in a stratified manner, and this would mean the bed grains could pack together more tightly, reducing pore dimensions and increasing the likelihood of pore blockage and thus influence ripening.

In water treatment plants backwashing is not a perfect process but its effectiveness will influence ripening. Water treatment plant engineers have to achieve backwashing in a limited time period

without the use of excessive quantities of water. The amount of deposit still in the filter bed or top water after washing was an influence on ripening (Amirtharajah, 1985).

Backwash practice will also govern maturation by influencing the amount of chemical deposit and biological growth that is able to develop on the media over the course of time. The influence of biological and chemical maturation on ripening requires investigation.

Bucklin *et al.* (1988) produced data which gave conceptual support to François and Van Haute's (1985) finding that 5% of the initial poor filtrate quality was due to the backwash remnants and 95% due to the influent water. But whereas François and Van Haute (1985) thought the influent water passed through the filter before an adequate floc structure had started to block the pores and "dead spaces", Bucklin *et al.* (1988) thought that the mixture of influent and backwash water remnants interfered with the destabilisation of the charges on the influent particles and resulted in a peak in filtered water turbidity. Ripening followed this peak.

Laboratory studies, under controlled conditions, show that ripening is not solely a factor of backwash remnants. Tobiason *et al.* (1993) and others found ripening occurs with new, totally clean, or chemically pre-treated media, where the filter run was not preceded with a backwash.

Although backwash remnants may only account for 5% of the material passing the filter during ripening it could include concentrated collections of particles from throughout the previous filter run.

2.6.2 BACKWASH REMNANTS AND THE PRE-RIPENING PHASE

Despite the ripening of filters having been examined in detail with well-controlled experiments by many authors e.g. O'Melia and Ali (1978), Mackie *et al.* (1987), Darby and Lawler (1990), Darby *et al.* (1991, 1992), Clark *et al.* (1992), and Veerapaneni and Wiesner (1993), few have presented data from a pre-ripening phase. This first phase was explained by Chang and Vigneswaran (1990) and Tobiason *et al.* (1993) as a dilution effect due to the displacement of particle free water from

the bed by the suspension under examination, i.e. a question of timing in defining the start of the experiment. The peak at the end of the deteriorating phase was taken to be the clean bed removal. Work detailing the pre-ripening phase at laboratory scale has been presented by Amirtharajah and Wetstein (1980) and Cranston and Amirtharajah (1987), and at full scale by Bucklin *et al.* (1988). It was noted at pilot scale by Goldgrabe *et al.* (1993).

TWIN PEAKS OF BACKWASH REMNANTS.

According to Amirtharajah and Wetstein's (1980) description, firstly clean backwash water remaining under the filter floor and in the common pipework appeared in the filtrate. The quality then deteriorated, displaying two peaks. The first peak was due to dirty backwash water remaining within the pores of the media and originating from the resettlement of the expanded media (although Amirtharajah (1985) stated that the settlement of the bed as fluidisation stopped would not dislodge particles from the grains provided the wash had been effective). The second peak originated from dirty backwash water remaining between the top of the media and the wash-out weir. Amirtharajah and Wetstein (1980) thought the height of the second peak was dependent on the rate of filtration (from Cleasby and Baumann's (1962) data). This was also observed by Bucklin *et al.* (1988).

The existence (or at least the magnitude and duration) of this initial degradation period is probably strongly linked to the type of backwash employed, its duration, the size, shape, density and numbers of particles in the water being treated, and whether chemical conditions are optimum for filtration. Backwashing never usually washes a filter totally clean; the attempt is to remove accumulated deposits and return the filter to service with a low head loss with a minimum use of backwash water. There is an economic and functional balance to be sought. It may be that retaining some dirt in the water assists ripening, although there is the risk of concentrated "filtrate residue" passing through the filter by this strategy.

Overall, Bucklin *et al.* (1988) concluded that their observations on turbidity showed that the initial degradation period had a magnitude and duration that a) varied seasonally, b) varied from plant to plant, c) was dependent on flow rate and d) to a lesser extent, depended on raw water turbidity.

2.6.3 EXPERIMENTS WITHOUT BACKWASHING

Several laboratory experiments have reported ripening during contact filtration with fresh clean media each filter run, so that backwashing was not a cause of ripening, and backwash remnants were not an influence on ripening. In some trials charge neutralisation of particles and filter media was attempted using a cationic polymer or a Calcium salt but conditions of water chemistry and background particle matrix were not reported in some studies. In all but one of the cases examined here spherical glass filter media was used.

Early investigations of ripening, such as Habibian and O'Melia (1975) and O'Melia and Ali (1978), concentrated on filtration of suspensions of a single size of particles (monodisperse suspensions). More recent work has covered monodisperse suspensions as well as combinations of two or three particle size classes and broad ranges of particle sizes (termed heterodisperse or polydisperse suspensions), for example Clark *et al.* (1992), Darby and Lawler (1990), Darby *et al.* (1991, 1992), Kau and Lawler (1995), Mackie *et al.* (1987), Mackie (1989), Mackie and Bai (1992, 1993), Moran *et al.* (1993), Tobiason and O'Melia (1988), Tobiason *et al.* (1993), Veerapaneni and Wiesner (1993), Vigneswaran and Aïm (1985) and Vigneswaran *et al.* (1990).

In considering laboratory experiments there may be significant differences from the way operational filters behave. Filter media such as sand and anthracite and are not perfectly spherical, and the tendency of media to change with time due to abrasion (Fulton, 1988), organic and inorganic material deposition (Galvin, 1992) and biofilm development must be borne in mind. Therefore what constitutes "clean" media may be different in the laboratory from in the field. Also, economic criteria mean that at practical scale municipal water filtration plants will never be likely to use media such as glass spheres. An operational filter would also be backwashed, and would not be pre-conditioned with a particle-free solution containing coagulant.

Because of doubts concerning the practical significance of experiments with glass spheres, Kau and Lawler (1995) repeated the experiments of Moran *et al.* (1993, 1993a) with sand as the filter media

and found that ripening still occurred, but since clean bed filtration was better with the sand, ripening was less marked.

2.6.4 THE INFLUENCE OF THE INFLUENT WATER ON RIPENING

François and Van Haute (1985) reported a strong relationship which showed an increase in aluminium sulphate or coagulant dose corresponded to a decrease in maximum peak turbidity ($r=-0.68$). There was also a strong positive correlation between the influent turbidity and the height of the filtrate peak ($r=0.91$). Janssens *et al.* (1982) found that peak turbidity was statistically significantly ($r=0.86$) related to the raw water turbidity, with secondary and third influences from alum and polymer doses, respectively.

François and Van Haute (1985) showed more evidence of the effect of influent quality in a experiment where, having changed the influent from tap water to flocculated river water after 75 minutes, filtrate deterioration and ripening occurred. Comparing the two peaks which resulted, 5% of the poor water quality was in the first peak and due to the backwashing and the presence of old material above and within the bed, whereas the fresh material in the river water accounted for 95% of material passing the filter in ripening. Further evidence of the relatively small significance of old material breakthrough during ripening was shown by the presence of only 10% of material with the low zeta potentials associated with old floc. In a similar experiment Janssens *et al.* (1982) found the backwash remnants produced a C_{max} of 0.07 JTU during downflow of clean water. When normal influent water was restarted a C_{max} of 0.53 JTU was reached. A high initial turbidity and long ripening period occurred even after all backwash water remnants had been washed out of the filter by tap water.

2.6.5 FILTER RIPENING SEQUENCE INCORPORATING BACKWASH CONDITIONS AND INFLUENT WATER

Cranston and Amirtharajah (1987) modified the twin peak ripening model of Amirtharajah and Wetstein (1980) in view of the importance of the incoming water quality (François and Van Haute, 1985) and the influence of deposits on ripening shown by O'Melia and Ali (1978) and Payatakes *et al.* (1981).

The revised model had three stages, the first of which, the "remnant" stage, included the three mechanisms of the deteriorating phase in Amirtharajah and Wetstein's (1980) model. The quality of the water within and above the bed was very dependent on the efficiency of the backwash. Cranston and Amirtharajah (1987) showed that the zeta potential of the previously retained particles became more negative after mixing with the clean backwash water, making those that remained after backwashing less easy to filter (table 2.6).

Cranston and Amirtharajah (1987) termed the second stage "influent mixing and particle stabilisation". They showed mixing the dosed influent water with coagulant free backwash water resulted in an increased charge on previously destabilised particles which adversely affected filtration performance. Filtrate turbidity was at its highest value coinciding with the most negative zeta potentials.

Table 2.6. *Changes in mean turbidity and particle zeta potential each minute during backwash. (Adapted from Cranston and Amirtharajah, 1987).*

Time into backwash (mins)	Wash water turbidity (NTU)	Particle zeta potential (mV)
0-1	39	-11.5
1-2	40	-10.3
2-3	4	-11.3
3-4	3	-13.2
4-5	2	-12.9

The third phase was termed "filter media conditioning" by Cranston and Amirtharajah (1987) and consisted of the deposits on the filter grains enhancing filtration performance. Cranston and Amirtharajah (1987) thought that both the dendrite growth and pore clogging mechanisms (François and Van Haute, 1985) could act synergistically, and that the influence of one or other mechanism would vary during ripening and vary according to whether polyelectrolytes or metal salts were used for coagulation.

It is interesting to speculate whether the end of backwashing should be the point where the zeta potential of the particles in the water left in the filter is the same as the incoming water. This is

presumably easiest achieved by dosing the backwash water. It could be that there should be sufficient dirt left in the water to pre-coat the filter media to accelerate ripening.

2.7 OPTIMUM RIPENING TIME

In defining optimum filtration conditions Janssens *et al.* (1982) suggested a compromise between three conflicting drives was required: they said that the filtrate should be the best possible, with the lowest possible rate of head loss development, and the time between filter backwashes maximised. They defined the **optimum ripening time** as that where the ratio of ripening time (t_{r1}) to overall useful productive time of the filter (t_b) was minimised. The optimum coagulant dose was one that gave a rapid ripening but not at the expense of early breakthrough, i.e. before maximum head loss was reached. Trials showed a higher dose improved ripening time but shortened the overall run length by promoting breakthrough.

2.8 THE SIGNIFICANCE OF FILTER RIPENING

Ripening is characterised as being the only phase where a marked improvement of filtrate quality takes place. Ripening therefore implies a period of filter operation which is sub-optimal. For this to have meaning there must be an optimal phase of filter performance when nothing acting on filter performance has been deliberately changed: the filter has simply achieved its ripened state. This implies that the factors impacting on filter performance which are filter design, filter operation, the influent water quality and any chemical pre-treatment are optimised for overall filter effectiveness, and they remain constant throughout the ripening and optimum performance phases. Because of this it is necessary to understand what contributes to optimum filter performance so that artefacts of inadequate filter operation may be eliminated from the examination of ripening.

The ripening period is poorly defined, and the lack of reference to it in many filter studies, and even the conscious elimination of ripening data from reports of filter performance, leads to the conclusion that it is often not perceived as being of significance. However, if it is assumed that a filter is being operated optimally, in the ripened condition, to a set of water quality standards by an

operator working under the economic constraint of "just good enough", it must be questioned whether having a preceding phase of sub-optimal quality can be acceptable either to public health or customer appreciation of the final product or alternatively, the economic operation of subsequent treatment stages.

Therefore, since RGFs are utilised to remove a diverse range of particles from water intended for human consumption, there is a chance that the water will not meet quality standards and that particles that may impact on human health may be present in the filtrate.

2.9 THE SIGNIFICANCE OF TURBIDITY

In this thesis the experimental investigation was limited to continuous monitoring of turbidity. This represents examination of the "cloudiness" of the water by measuring the amount of scattering of light through 90° by the particles in the water. It has been demonstrated by Gregory (1994) for equal mass suspensions, that nephelometric turbidity measurement is most sensitive to sub-micrometre diameter particles, with a peak in sensitivity to latex spheres at 0.25 μm diameter. Low turbidity values do not necessarily mean low numbers of particles in the water of larger sizes (Baumann, 1989) like algae or *Giardia* cysts and *Cryptosporidium* oocysts.

Studies by Logsdon *et al.* (1981) reported by Cleasby (1990) concerning changes in filtrate quality when flow rate was stepped up from 11 to 27 $m.h^{-1}$, showed that breakthrough of *Giardia* cysts increased 25 times, whereas turbidity increased by 4 times. Clearly turbidity was not a good indicator of this other quality parameter. Is this true during ripening? If so, what is accompanying turbidity breakthrough during the ripening phase. It would be expected that the mechanics of ripening and breakthrough would be different, since breakthrough could include shedding of previously retained deposits.

Watson (1990) found that filtrate turbidity showed a strong correlation with filtrate iron concentration and filtered water POC, both were significant at the 1 in 2000 level of chance. This indicated the usefulness of turbidity as a water quality parameter.

Mosher and Hendricks (1988) concluded that turbidity could be used as a surrogate indicator of a filter's ability to remove *Giardia* cysts under the conditions which they quoted from Al-Ani *et al.* (1985) for treating mountain waters of less than 1 NTU who said " ..if turbidity removal exceeds 70% and if filtered water turbidity is lower than 0.10 NTU, the probability is 0.85 that removals of *Giardia* cysts would equal or exceed 99%." Mosher and Hendricks (1988) achieved 100% *Giardia* removals with turbidity removals not less than 87%. Al-Ani *et al.* (1986) repeated the earlier quote in a slightly modified form: "If, for example, raw water turbidity is 0.5 NTU and if removal of turbidity is 70% or greater, the probability is 0.85 that removal of *Giardia* cysts will exceed 99%." It is not clear if this only referred to mountain waters.

Gregory (1994) regarded turbidity as a useful surrogate for general particulate matter and "micro-organisms", excepting *Giardia* cysts and *Cryptosporidium* oocysts, in contrast to Al-Ani *et al.* (1986). LeChevallier and Norton (1992) concluded that removal of turbidity, and particles $< 5\ \mu m$ were useful predictors of *Giardia* and *Cryptosporidium* removal, but said that this contradicted an earlier study by LeChevallier *et al.* (1991). They thought that the relationship between cysts and oocysts and turbidity or total particles would vary with different source waters, and that this relationship would have to be determined locally at individual treatment works.

With secondary sewage effluent Darby *et al.* (1991) observed ripening of suspended solids, turbidity and total particle number. Ripening for suspended solids and turbidity was not large, due perhaps to the absence of coagulant. The improvement in removal efficiency was more marked with particle data.

Clark *et al.* (1992) presented data from the filtration of lake water which, like Darby *et al.* (1991), showed ripening for suspended solids and turbidity. Two rates of ripening were observed. Ripening measured by suspended solids took place rapidly for the first 1 - 2 h and then continued at a lower rate of improvement for the remainder of the 12 h run. Turbidity ripening proceeded at a rapid rate for the first 2 h, and at a slower rate thereafter. Turbidity removals ripened from 70% to 80% and suspended solids removals improved from 63% to 93% over about 300 minutes. In the same

experiment, increased removal of total particle volume from 75% to 93%, measured at 165 mm depth, showed ripening was completed in the first 2 h.

Moran *et al.* (1993) reported better removal of suspended solids compared to turbidity, initially and during the ripening phase. This reflects the sensitivity of these measures to particle size and supports the detailed particle removal work of Mackie and Bai (1992, 1993).

2.10 HEAD LOSS AS INDICATOR OF RIPENING

The change in filtration efficiency that marks the end of the ripening period is not easily defined. A change in the rate of head loss development might provide a simple means of determining, or defining, the completion of the ripening phase. Head loss gradient would increase if interstitial filtration rate changed due to the constriction or blockage of pore spaces, even if approach velocity was constant, or if attachment efficiency had increased with a constant influent quality.

The literature was examined to see whether head loss gradient could be used as an indicator of the completion of ripening. Unfortunately contradictory results were found.

There was the suggestion from some studies that head loss development started at a low rate and changed to a steeper gradient as a consequence of ripening. Wiesner *et al.* (1987) used the O'Melia and Ali (1978) ripening model to predict head loss development in GAC filters. They plotted a theoretical head loss line which showed a flat start over first 2 h, corresponding to theoretical ripening time, before the head loss line started to rise. Actual data however showed no indication of a slower ripening rate of head loss. The model of Tobiason and Vigneswaran (1994) for 0.27 μm particles showed an exponential head loss increase, but experimental data showed a clear change from a shallow to a steep gradient at around 70 minutes. This corresponded with the end of the improving phase for particle removal.

Vigneswaran and Chang's (1989) data showed a steepening of the rate of head loss development when ripening was achieved. Clark *et al.* (1992) showed a distinct increase in the rate of head loss

development about 30 minutes after maximum particle removal efficiency was achieved. Vigneswaran and Chang's (1986) head loss curves showed a steeper gradient about 1-2 hours after ripening was achieved.

In Tare and Venkobachar's (1985) presentation of Habibian's (1971) data a change in head loss gradient corresponding with the end of ripening was evident with 0.1 μm particles but not 1.0 μm particles. Tobiason *et al.* (1993) found that head loss development was linear over time throughout the runs for polymer destabilised monodisperse suspensions of 1.32 and 10.0 μm particles. However the 0.27 μm particles showed a head loss graph with two distinct linear phases; an initial lag phase lasting 70 minutes was followed by a steep rise in head loss gradient. This coincided with the completion of ripening.

In contrast to the findings of Tobiason *et al.* (1993) Veerapaneni and Wiesner's (1993) data showed head loss development rate over time was not linear, the rate accelerated with time, after an initial lag phase of about 60 minutes. Also contrasting was the finding that the rate of head loss development for polydisperse suspensions did not fall between the two monodisperse components but was greater than either of them.

At plant scale there was a hint in the data presented by Cleasby *et al.* (1992) that the end of the improving phase corresponded with a slight increase in the rate of head loss development.

Mackie (1989) and Mackie and Bai (1992, 1993) have shown that the development of head loss is dependent on a complex interaction of particle size and location of deposition within the filter. This determines the degree of clogging of the pore spaces and the porosity of the deposits themselves and thus the loss of head.

2.11 CONTROLLING FILTER RIPENING

Cranston and Amirtharajah (1987), commenting on findings of Logsdon *et al.* (1985) that increased numbers of potentially pathogenic micro-organisms passed through filters in the ripening phase,

and that the duration of this phase could exceed 2 hours, said there was a need to reduce the filter ripening time for the benefit public health and, where filter to waste was practised, to reduce the loss of treated water. It has already been stated that ripening is seen when filters are behaving optimally during the majority of the filter run, so these control measures are in addition to the selection of the correct chemical and operating conditions for normal filter operation.

Amirtharajah and Wetstein (1980) suggested that reviving an earlier practice of rapid dumping of initial filtrate to waste at highest possible rates would allow the filter to return to service the most quickly. Whilst this would reduce one of the causes of poor water quality during the ripening phase, it would not eliminate ripening altogether.

Amirtharajah and Wetstein (1980) and Bucklin *et al.* (1988) have suggested physical and chemical ways of mitigating, but not eliminating, poor initial quality, optimising filter start up as an alternative to filter to waste. These are summarised in table 2.7.

Table 2.7. *Physical and chemical means of controlling filter ripening. (After Amirtharajah and Wetstein, 1980, and Bucklin et al., 1988).*

Physical methods	
	Improve backwashing with combined air and water to reduce the dirt left on the filter grains.
	Finish the backwash with a sub-fluidising water wash to reduce the effect of grain collisions as the bed packed down.
	Discharge the above bed water after the backwash.
	Wash with a low freeboard reducing the remnant volume.
	Control flow rate, avoiding sudden flow rate changes.
	Use slow starts, i.e. a gradual filter restart.
Chemical methods	
	Ensure proper chemical treatment of the raw water.
	Use polymer as a filter aid.
	Use a higher coagulant dose in the early part of the filtration cycle.
	Add alum, alone or with polymer, during the water only stage of backwashing, ceasing to dose for the last few minutes, to eliminate an extra spike caused by the alum.
	Add polymer to the backwash water towards the end of the wash.
	Utilise chlorinated backwash water.

2.11.1 CHEMICAL METHODS OF CONTROLLING RIPENING

INITIAL HIGH DOSE IN INFLUENT

Janssens *et al.* (1982) and François and Van Haute (1985) demonstrated that an excess coagulant dose at the start of the filter run could reduce the initial turbidity peak and shorten the ripening period. François and Van Haute (1985) considered that the higher coagulant dose in the influent helped by maintaining particles in the destabilised state, and by increasing the concentration of particles. When the overdose was too great early breakthrough resulted. When it was applied for too long the turbidity value rose at the end of dosing and despite a further ripening period the filtrate quality from this experiment to be inferior to that of a control that did not receive any overdosing. Janssens *et al.* (1982) and François and Van Haute (1985) explained this finding by suggesting that the overdose produced a floc which was weak and susceptible to break up due to fluid shear.

ADDING CHEMICALS TO THE BACKWASH WATER

Harris (1970), Yapijakis (1982) and François and Van Haute (1985) showed on pilot plants that adding a polymer to the backwash water when alum was the primary coagulant reduced the initial turbidity peak and ripening time. Pilot plant studies by Cranston and Amirtharajah (1987) investigated several combinations of alum and/or polymer as primary coagulant and added to the backwash water. The details of the ripening sequence varied with the precise combination of chemicals but reductions in the duration and peak turbidity (magnitude) of the ripening sequence were reported. It was possible to overdose the coagulant in the backwash water, reducing or eliminating the reported benefits.

The use of polymers in the backwash water was thought to reduce chemical repulsion forces and create a three dimensional filter "matrix", generating polymer bridges (Gimbel, 1989a) between the media or dendrites in the streamlines, which could be analogous to increasing the roughness of the media. Backwash dosing of aluminium sulphate assisted filtration by preventing the re-stabilisation of particles in the mixture of backwash water and influent, the destabilised particles being readily retained by the filter media. Additional alum would strengthen the flocs, and produce extra flocs, facilitating particle capture (Cranston and Amirtharajah, 1987). By adding alum to the backwash

water Cranston and Amirtharajah (1987) achieved very small zeta potentials in the mixture of incoming water and backwash remnant (-5 to -7 mV). Despite adding extra load to the filter, Cranston and Amirtharajah (1987) felt that its short duration would not affect overall filter run lengths.

One drawback with adding coagulant to the backwash water was an earlier peak, lasting from 2-5 minutes into the filter run, because the overdose formed stable flocs which passed through the filter during the remnant phase. Whilst this could increase the first peak, the overall filtrate quality was still better than the conventionally dosed filter. Cranston and Amirtharajah (1987) suggested that the early turbidity peak caused by aluminium sulphate as a backwash water coagulant could be removed by a very short filter to waste period or by terminating coagulant injection 1-2 minutes before backwashing ceased.

Following these laboratory studies on artificial waters Bucklin *et al.* (1988) produced very similar ripening behaviours at full scale. Cleasby *et al.* (1992) also demonstrated on a full scale filter the benefit of dosing a non-ionic polymer into the backwash water for the final 3 minutes of washing, in combination with a wide range of primary coagulants.

VARYING BACKWASH COAGULANT TIME

Cranston and Amirtharajah's (1987) experiments showed that backwash water only had to be dosed with coagulant for long enough to displace all the water within and above the bed to ensure even mixing within the backwash remnant water, which in their trials appeared to be the last 1.5 minutes of a 5 minute wash. This conflicted with their suggestion (repeated by Bucklin *et al.*, 1988) that it was necessary to cease dosing aluminium sulphate into the backwash water 1-2 minutes before the end of the wash to avoid an initial aluminium hydroxide spike.

2.11.2 PHYSICAL METHODS OF CONTROLLING RIPENING

VARYING THE REMNANT VOLUME ABOVE THE FILTER BED

Cranston and Amirtharajah (1987) showed that varying the depth of backwash water above the filter (the remnant volume) affected the timings of the events in the filter ripening sequence. The length of

the remnant stage (which included the first peak) was directly proportional to the detention time of this volume of water. A larger volume resulted in a small reduction in the peak but a greater duration. It might be concluded that this had the effect of diluting the remnant phase, but that the same total number of particles had to pass through the filter before this stage was completed. The magnitude of the filtrate turbidity peak was closely linked to the turbidity of the remnant water. It would be an important part of filter design and operation to use sufficient volume and rate of water to give the desired initial post-backwash filtrate quality.

With aluminium sulphate added to the wash water Cranston and Amirtharajah (1987) showed a benefit from greater depths of water above the bed. They suggested the greater depth allowed better mixing, and gave more time for forming stronger flocs, or better destabilisation of particles.

VARYING THE TOTAL VOLUME OF BACKWASH WATER

Assuming the wash rate to be constant, varying the volume is the same as varying the wash time. Cranston and Amirtharajah (1987) presented data on backwash time which showed the importance of backwashing to a low remnant water turbidity, as when this remnant was highly turbid it significantly increased the magnitude and duration of the post backwash spike. Cranston and Amirtharajah (1987) suggested that backwashing should stop once remnant water turbidity was reduced by 95%.

SLOW STARTS

Until recently there appears to have been very little in the literature covering slow starts, despite the fact that it has been widely practised in the UK (Monk, 1987). Monk (1987) furnished no references to slow starts. Keay (1995), Hillis and Colton (1995, 1995a) and Borrill and McKean (1993) presented evidence from on-line light obscuration particle counters that showed that a slow start did not satisfactorily address filter ripening. Keay (1995) produced anecdotal evidence that the slow start did not reduce the total number of particles passing the filter compared to a full rate start, because the slow start prolonged the ripening period. Borrill and McKean (1993) showed similar initial peaks of 1-5 μm particle numbers from their fast start and slow start filters. The slow start simply delayed the peak.

Hillis and Colton (1995) showed that nearly 50% of the 2-5 μm particles penetrating the filter during a 48 hour run did so during the ripening period. The effect of the slow start was to prolong the ripening period from about 25 minutes to 45 minutes. The slow start showed a two peak ripening period in contrast to the single peak when no slow start was used. Measuring 2-5 μm diameter particles, the single peak reached 30 000 particles.ml^{-1}, whereas the two peaks were each around 22 000 particles.ml^{-1}. After ripening the filtrate contained around 50 particles.ml^{-1}. The cumulative number of particles passed in a given filtered water volume showed a clear benefit of the slow start, as it reduced treated water particle counts by about 30%. However the authors were aware that the counts during ripening were still high. Hall and Pressdee (1995) reported a similar reduction (average 30%) in particle counts comparing a slow start with a full rate start. Once again the ripening period was not eliminated as the filtrate during the remainder of the filter run contained much lower particle numbers.

Cleasby *et al.* (1992) found that the initial post backwash peak was exacerbated by an operating strategy that caused the filter flow rate to be in excess of the desired flow at the end of the filter to waste period. Trials with a "gradual" manual opening of the filtrate valve from zero flow to the correct flow showed that the peak was eliminated or reduced, though there was no change in overall ripening time. There was no detail of what flow rates or timings constituted a "gradual" valve opening. Tobiason and O'Melia (1988) warned against causing hydrodynamic shocks to the filter bed when changing over from a filter to waste period to putting the filter back into production. In reviewing data from several studies comparing constant rate and declining rate filtration Cleasby (1993) argued that one advantage of declining rate filtration was that there are no sudden surges in flow rate during the filter cycle. The major disadvantage with this mode of filtration is that the filter is returned to service at its highest possible flow rate. A comparison between a declining rate filter and a constant rate filter on a conventional full scale works, reported by Cornwell *et al.* (1991), revealed no difference in peak turbidity, ripened turbidity or ripening time.

FILTER TO WASTE

Cleasby (1990) said that the initial degradation and improvement in turbidity following backwashing amounted to only a small proportion of the turbidity passed over an entire run and

that filtering to waste had been "largely abandoned". However he thought that this might not apply where giardiasis was an issue, due to the low numbers required for infection and its resistance to chlorine. Cleasby's (1990) view contrasts markedly with the experimental findings of Hillis and Colton (1995) with particle data.

Leland *et al.* (1993) followed up a cryptosporidiosis outbreak with particle counting studies on raw water and clear-well water. The test revealed higher 2 - 5 μm diameter particle counts for 15 minutes following backwashing. This resulted in a 30 minute filter to waste period being adopted. Turbidity measurements were not sensitive to the quality changes shown by the particle counter.

Mosher and Hendricks (1988) and Gertig *et al.* (1988) said that a filter waste period following backwash was not necessary with a low turbidity water. Gertig *et al.* (1988) reported that trials with spikes of *Giardia* cysts during and after ripening showed no difference in the removal of *Giardia* cysts, coliforms or turbidity with correct chemical dosing. However their criterion for optimising chemical dose was that the "proper" chemical dose should give a filtrate turbidity "near or less than 0.1 NTU during the first few minutes of the filter run", so there is a circularity in their argument. Trials without a coagulant achieved erratic cyst removals ranging from 89% to over 4-log (99.99%). All cyst and oocyst data should be treated with some caution due to methodological difficulties with recovery and counting.

In contrast, Cleasby *et al.* (1984) stated that, for their direct in-line filtration experiments initial turbidity often exceeded 1 NTU. Ripening often lasted several hours, although filtrate was poorest in the first hour. They concluded that in waters where *Giardia* cysts were a concern a filter to waste period would be appropriate. Goldgrabe *et al.* (1993) suggested that particle size distributions and counts could be used to optimise the filter to waste time for controlling particles similar in size to *Giardia* cysts.

Table 2.8 shows Bucklin *et al.*'s (1988) suggestions for when to incorporate a filter to waste period. The conclusion was that filter to waste would only be suitable where a very poor quality of filtrate

of short duration was experienced, and where the objective of a filter to waste period was clearly understood.

Bucklin *et al.* (1988) concluded that the need for a filter to waste period should be judged on a plant by plant basis. It was an expensive and difficult retro-fit option. At Bozeman, it was found to have an effect in disrupting filtrate quality by causing a surge in filter flow rate. They thought that the quality of filtrate at the plants studied would be better controlled by manipulation of the operating variables rather than having a filter to waste period.

Table 2.8. *Filter to waste selection criteria. (After Bucklin et al., 1988).*

Peak filtrate turbidity range	Duration of ripening period	Recommendation for filter to waste
0.3 - 0.6 NTU	1 - 6 hours	Not justified
1 - 2 NTU and above	0.5 hours	Recommended
0.6 - 1 NTU	0.5 - 1 hour	Depends on other plant criteria such as flow rate, number of filters, microbiological quality and other alternatives

2.11.3 BIOLOGICAL CONTROL OF FILTER RIPENING

MANAGING "MATURE" MEDIA

The maturation which takes place on SSFs suggests that managing backwashing to leave an active biofilm might assist with filtration and possibly ripening. A long term improvement in turbidity removals was ascribed to media maturation by Chipps *et al.* (1995). However, Goldgrabe *et al.* (1993) demonstrated no benefit for ripened filter performance deriving from the presence of biomass. Whilst ripening data were presented from their pre-chlorinated and backwash chlorinated filters, ripening data were not presented from their biological filter, so further studies are required. Recently, Urfer *et al.* (1997) stated that Huck *et al.* (1997) had found higher initial peak turbidities in biologically active filters than conventional pre-chlorinated filters. Thereafter ripened removals were no different.

CHAPTER 3. EXPERIMENTAL CONDITIONS

3.1 INTRODUCTION

An investigation into ripening in filtration of lowland reservoir stored water took place from February 1989 to March 1993 at Thames Water's Ashford Common water treatment works. A pilot plant was designed and constructed with six filters and the option of pre-ozonation and iron(III) sulphate coagulation. The plant ran automatically with flow, head loss and turbidity data being logged every 15 minutes. Several thousand filter runs were recorded. The phenomenon of ripening in RGFs was examined using turbidity data from filters operating without chemical pre-treatment and in contact filtration mode.

The experimental work investigated the effect on filter performance of changing chemical coagulant dose, pre-ozonation, backwash method, filter media and flow rate, as well as the seasonal changes in stored water conditions such as temperature and algal taxa and concentrations.

The filters were developed to pre-filter reservoir water prior to slow sand filtration. Pre-filtration has been used in treating London's river-derived water supply since the late 1920s (Chevalier, 1953). It was required to prolong slow sand filter (SSF) run lengths and permit operation at flow rates above 0.1 $m.h^{-1}$. Rachwal *et al.* (1988) stated that reservoir management and improvements in primary filtration would be necessary to run SSFs at relatively high flow rates averaging 0.3 $m.h^{-1}$ with peak flows of 0.5 $m.h^{-1}$.

Thames Water and its predecessor, the Metropolitan Water Board, had explored chemical coagulation in the treatment of reservoir water in studies at Kempton Park of conventional clarification and filtration (Chevalier, 1953, Ives, 1955), and contact filtration (Quaye, 1976), and in contact and direct filtration at Coppermills works in 1985-8 (Rachwal *et al.*, 1988). The latter study did not combine coagulation with pre-ozonation, they were seen as alternatives. Jar tests were used for setting coagulant dose, but there were problems with coagulant dose selection, resulting in premature clogging of a downstream slow sand filter in one case. Work with a coiled pipe

hydraulic flocculator had shown that contact filtration was no different to direct filtration (Bauer pers. comm., 1997). This was confirmed by Chipps *et al.* (1995a).

3.2 WATER SOURCES AND MANAGEMENT

The water tested was reservoir stored lowland River Thames water. The reservoirs supplying Ashford Common were predominantly Queen Mary and Wraysbury, with the Queen Mother (Datchet) used occasionally (figure 3.1). When water quality in the Queen Mary reservoir deteriorated, threatening to limit the output from Ashford Common, the inlet water would either be blended, or be switched totally to Wraysbury or Queen Mother water, whichever was determined to be suitable for treatment at Ashford Common. As a result water treated was usually low in inorganic and organic particles, and water containing algae that presented treatment difficulties was utilised as little as possible.

Stored water was pumped to the pilot plant by two submersible centrifugal Flygt B2066 pumps, located in a well feeding the main works. The pumps were mounted on a steel 2" manifold, feeding a 3" ABS main underground to the pilot filters. Initially the pumps were set up on a duty/standby basis. When higher volumes of water were called for later in the project both pumps were operated together.

The stored water contained a diverse algal community typical of a eutrophic lowland reservoir, together with a moderate loading of turbidity (0.5 to 10 NTU). Ridley (1967) briefly summarised the names and numbers of algal taxa in these reservoirs.

The Queen Mary reservoir was the usual choice, but being relatively shallow was more prone than the deeper, managed, Wraysbury and Queen Mother reservoirs to algal blooms. Artificial mixing of the water through jetted inlets prevented thermal stratification and limited primary productivity (Bauer *et al.*, 1997).

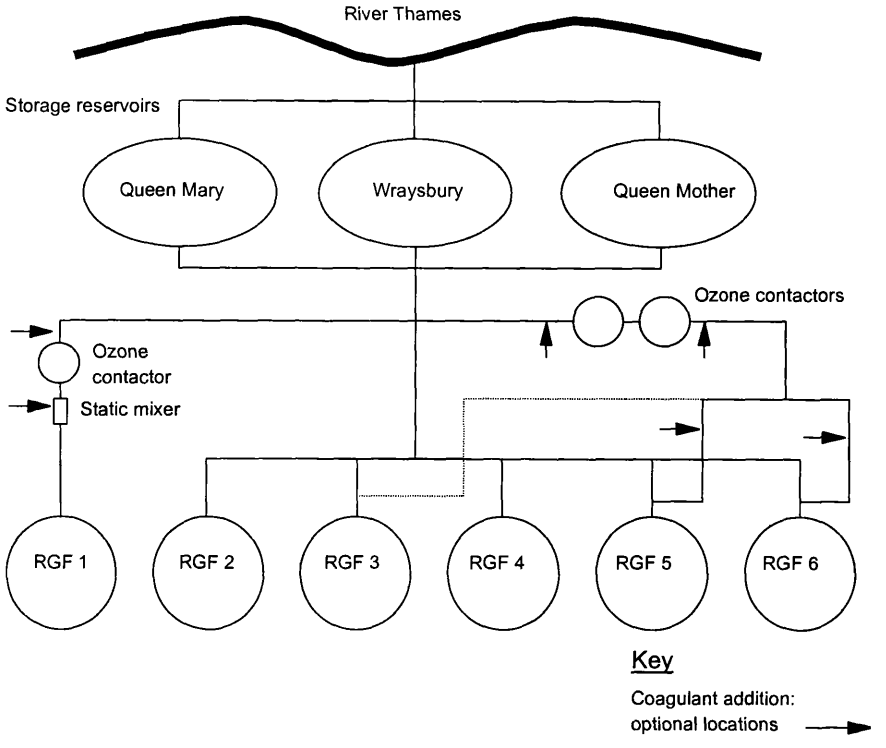


Figure 3.1. *Schematic of the pilot plant layout, showing stored water sources and outline of chemical pre-treatment options.*

3.3 FILTER COLUMNS

Six filter columns were installed in two adjacent, purpose-built, frameworks. All pipework and electrical services were mounted directly onto the frameworks. A schematic of a filter is shown in figure 3.2 and photographs of the pilot plant in figures 3.3 and 3.4. The retention time of the water through the filters and associated pipework is shown in table 3.1.

The roof was in-filled with steel checker plate, allowing access to the top of the columns and the sides were clad with plywood for insulation. Eight 1kW electrical tube heaters provided heating for frost protection. All dimensions have been quoted in metric units where they had a bearing on calculations . Pipework was in 2" PVC-U except the low rate wash water in 1" PVC-U, and the head loss measurement and air supply in brass compression fittings and nylon hose (up to 12 mm).

N.B. In constructing the pilot plant it was found that Imperial units of measurement are still frequently used for plastic pipe fittings. It has been decided that where measurements have a bearing on filter calculations they shall be quoted as the nearest metric equivalent. Fittings which do not require conversion will be referred to as supplied.

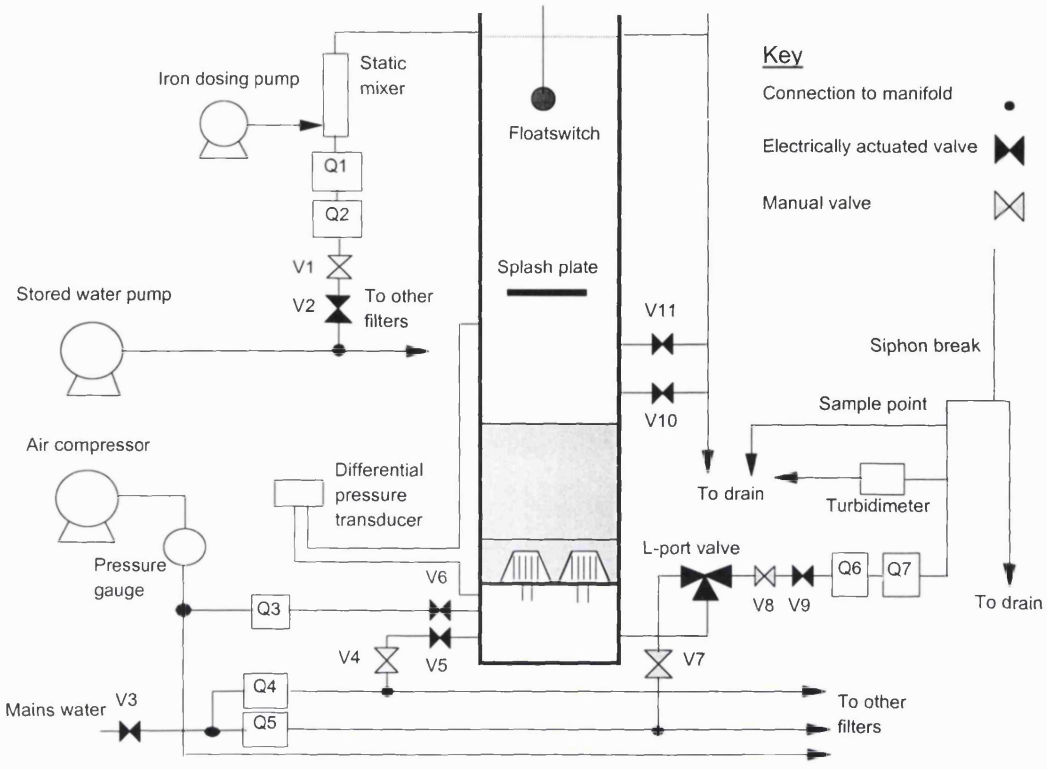
The columns used were of two designs, both 584 mm internal diameter, and made up of flanged tube sections, bolted together. Columns 1,2 and 3 were of one design (type A), 4,5 and 6 another, earlier design (type B). Each had a plenum chamber 600 mm deep with a base plate underneath and topped by a nozzle plate making a false floor for the filter. Connections to the plenum were via two flanges, one 2", to which an electrically actuated L-port valve was fitted, providing an exit for filtrate and an entrance for high rate backwash water, and one 8" (A) or 6" (B) blank flange into which low rate washwater, air scour supply, pressure transducer and manometer connections were made.

The nozzle plates were 25 mm thick and held 14 type HS10 (Cadar Ltd.) 0.2 mm slit nozzles with slotted (35 mm x 1.5 mm) 100 mm long air tail pipes with 2 mm diameter air bleed hole, arranged in a rhomboid pattern at a density of 50 m^{-2}.

The main body sections of the filters were made of PVC-U flanged tubes 1.2 m or 2.1 m long. 2" flanged spigots were fitted to the sides of the columns to permit stored water to enter and backwash water to exit.

In the type A column construction the larger (2.1 m) section housed the media and was placed immediately above the nozzle plate. Also included was a clear PVC window, enabling observation of the surface of the media during normal operation and during backwashing measurement of bed expansion and observations of the effect on the bed of the air scour. The 1.2 m section was mounted above this to allow water levels to rise sufficiently to provide driving head once the filters started to become clogged.

The columns were GRP reinforced and could be drilled and tapped to enable head loss measurements to be made using probes screwed through the side of the filter wall. The 1.2 m section housed the media (making for easy access to the media when the columns were dismantled), with the 2.1 m section above.



KEY

Valves		Flow meters	
V1	Manual inlet flow control	Q1	Inlet flow gap meter
V2	Inlet open/close and flow control	Q2	Inlet paddle wheel flow sensor (pwfs)
V3	Mains water pressure reducing valve	Q3	Air flow meter
V4	Low rate backwash flow control	Q4	Low rate backwash pwfs
V5page 77	Low rate backwash open/close	Q5	High rate backwash pwfs
V6	Air scour open/close	Q6	Outlet pwfs
V7	High rate backwash open/close	Q7	Outlet flow integrator
V8	Manual outlet flow control		
V9	Outlet open/close and flow control		
V10	Backwash drain down		
V11	Backwash water outlet		

Figure 3.2. *Process schematic of an individual pilot plant filter, showing connections to common manifolds.*

A further column section was added to each column in November 1989 to allow still higher head losses to be developed. The type A columns used an additional 1.2 m section, the type B sections were 0.77 m long. In February 1990 the type A columns were reinforced to take tapping points, 100 mm apart. In November 1990 the 0.77 m sections were replaced with new 1.2 m sections containing a window, placed immediately above the section housing the media. The relative

positions of the column sections were rearranged when deep beds of media were investigated to ensure that pipework connections gave sufficient freeboard during backwashing and that backwash observations could be made.

The outlet pipework was orientated horizontally out of the filter, then, on the side of the framework adjacent to column 1, it rose up 1.1 m before turning over and down to a common 4" drain pipe. This was to prevent the filters draining dry in the event of a water supply interruption. A siphon break was provided. Sample valves and tappings for the on-line turbidimeters were taken off this rising section.

Splash plates were fitted in November 1989, when the columns were operating in rising mode to prevent incoming raw water displacing the media surface at time of low water level (low head loss or immediately after a backwash in later operation). The filters were independent, in that backwashing one filter did not affect the flows through the others. Only one filter could wash at a time. Stored water, backwash water and air were provided from common manifolds.

Table 3.1. *Water residence time in the filter and associated pipework. The filters were 4.35 m from nozzle plate to overflow. The minimum media depth was 0.6 m and the maximum 2.0 m.*

Filter section	Filter section dimensions	Residence time at 5 $m.h^{-1}$ (mins)	Residence time at 10 $m.h^{-1}$ (mins)	Residence time at 15 $m.h^{-1}$ (mins)
Water above media	3.65 m	44	22	15
	3.25 m	39	20	13
	2.25 m	27	14	9
Filter media depth (includes 0.1 m gravel)	0.7 m	3	2	1
	1.1 m	5	3	2
	2.1 m (assume porosity = 0.4)	10	5	3
Plenum	0.6 m deep, outlet at 0.3 m (assume dead space below)	4	2	1
Outlet pipe	2" nominal bore; up to 6 m long (worst case)	0.5	0.25	0.16
Sample hose	8 mm nominal bore, 1 m long, assume 500 $ml.min^{-1}$,	0.1	0.1	0.1
Turbidity cell volume	2 l, sampled at 250 - 750 $ml.min^{-1}$, assume 500 $ml.min^{-1}$ and plug flow	4	4	4
TOTAL (1.1 m media)		53	29	20

Original in colour

Figure 3.3. *The pilot plant under construction in 1989.*

Figure 3.4. *The pilot plant in 1992, with deeper filters and ozone contactors enclosed for weather protection.*

3.3.1 CONTROL STRATEGY AND INSTRUMENTATION

The pilot plant was designed to be controlled either manually or by computer. Control of flows and maximum head loss was manual over the first few months then by computer when the system had proven itself reliable and accurate. Backwashing was always under computer control, with occasional extra washes initiated manually. Considerable flexibility was built into the both the hardware and the software of the entire system to enable development of the pilot plant.

The computer used programs written in-house (Davy, 1989) to control electrically actuated process valves via relays for flow control and backwash sequencing and timing. The computer also logged on-line flow, head loss and turbidity data, storing the values with date and time to disc. Each backwash was counted and logged. The software read the instruments at a high frequency, storing the results to a total and counting the number of values taken. Every few seconds these values were used to calculate an average which was temporarily stored in an array. The current values were used to refresh the screen display and were logged every 15 minutes to create daily data files. Full details were given by Davy (1989).

Rapid gravity filters can be operated in three ways, declining rate, fixed rate with varying top water level, and fixed rate, fixed level with control of the outlet flow (Tebbutt, 1971). The pilot filters were operated initially in a rising mode, where flow rate into the filter was controlled and the outflow was fixed. The water level in the column compensated for pressure loss across the filter media and pipework, rising as head loss increased to maintain flow rate. Eventually the level would rise to a point where a float switch would be displaced, triggering a backwash.

As confidence was gained in the control and monitoring system, the backwash was initiated when 1.2 m head loss was measured across the filter bed and floor by a differential pressure transducer. To enable high rate studies to be performed and terminal head losses in excess of 1.2 m, an extra section was added to each filter in November 1989 and the backwash system was set to wash each column at 1.8 m. Backwashing was triggered at 2.5 m on columns 5 and 6 from 28.1.92 and column 4 from 9.6.92.

Initially flow control was achieved by controlling a ball valve on the inlet side of the filter in conjunction with a paddle-wheel flow sensor. From November 1990 flow control on filters 2 to 6 was switched to a ball valve and flow meter on the outlet side of the filter to enable the filters to operate full. This was important to try to mimic a realistic residence time in the water above the filter media in case of floc growth in the iron dosed filters. A general arrangement of the filter process equipment is shown in figure 3.2.

The outlet flow valve opened fully after the backwash. When the filters were changed to outlet flow control there was the potential for an initial flow surge. The flow control loop operated swiftly enough to prevent this distorting ripening turbidity data. From June 1992 the valve timings were reset to just nudge open these valves after the backwash.

3.3.2 OZONE GENERATORS AND CONTACTORS

Pre-ozonation of water for column 1 was supplied by a Trailigaz Labo ozoniser nominally rated at 9 g $O_3.h^{-1}$. Compressed air was fed by a Gast oil-free compressor through an air drier to a spark discharge ozoniser tube and the ozonised air passed into a contact column through a sintered stainless steel dome, providing fine bubble diffusion. All lines carrying ozonised air were in PTFE hose and 316 stainless steel.

The contactor was designed by the Water Research centre (WRc) to run water counter to the ozone bubble flow, and had a nominal retention time of 5 minutes for a column flowing at 10 $m.h^{-1}$, based on a volume of approximately 0.228 m^3. This ozoniser and contactor were installed in July 1989.

Following installation of additional ozone equipment in November 1990, columns 5 and 6 were supplied by a Trailigaz Ozobloc ozoniser rated at 36 g $O_3.h^{-1}$. Compressed air was fed by a Hydrovane compressor through an oil trap and air drier and to two ozoniser tubes and into a two stage contact column through two sintered stainless steel diffuser plates. The contactor was designed by Ozotech Ltd. The first column provided counter current diffusion and was followed by

a co-current stage. Each stage was approximately 0.51 m^3 capacity, providing ozonised water after a maximum of 10 minutes contact time to two columns each running at 10 $m.h^{-1}$, and a minimum of 5 minutes for two filters each treating 20 $m.h^{-1}$.

Each contactor had a free discharge over a weir providing a hydraulic outlet de-gassing stage. The inlet pipework to filters 1, 5 and 6 was re-routed to pass through the ozone contactors. Columns 5 and 6 could be fed by either the raw water manifold or the ozonised raw water pipework.

The ozone was operated to achieve a target ozone residual of 0.25-0.5 $mg.l^{-1}$. Ozone residuals were measured colorimetrically by two methods; firstly a Wallace and Tiernan DPD dropper kit and secondly by Hach Indigo ampoules (0-0.25 $mg.l^{-1}$ low range or 0-0.75 $mg.l^{-1}$ mid range ampoules).

3.3.3 IRON DOSING EQUIPMENT

Iron(III) sulphate ("iron" in this thesis) was dosed into the raw water feed to certain filter columns using Prominent type 1000, 1201 or 2001 diaphragm pumps. The chemical was obtained as an 11.6 % solution by weight as Fe^{3+}, specific gravity 1.55 $kg.l^{-1}$, i.e. 180 g $Fe^{3+}.l^{-1}$. The iron pumps were calibrated *in situ* by using a burette as the supply reservoir.

The dosing system was initially set up with a pump mounted on the top of a 30 l drum containing the chemical. When it was found that the dosing frequently failed due to gas bubbles becoming trapped in the pump head, the pump was positioned below the storage tank to provide a positive head on the suction side of the pump. The pumping was also found unreliable when low stroke volumes were used. This was unfortunate as high stroke volumes required low stroke frequencies to provide the correct dose and meant pulsed injection of chemical when near-continuous would have been preferred to ensure good mixing. It was decided that the iron solution would be diluted to 25% in order to keep the pump stroke and frequency as high as possible. Storage tests showed that a 25% solution was stable for over three months.

The mixing of iron was assisted by in-line static mixers or orifice plates or weirs, and changes in pipe direction, as well as movement of water above the bed in the filter column itself.

The static mixers used were a Sulzer 300 mm x 25 mm unit on column 5 in 1989, and Kenics 6 element 600 mm x 40 mm unit for columns 1, 2 and 5 during 1990, and for column 2 late in 1991. The Kenics unit was chosen as it had a lower head loss than the Sulzer device, by virtue of a larger diameter. For columns 2 and 5 they were mounted in the pipework where the flow rose vertically from the raw water manifold, before a 90° elbow and into the column. In the case of column 1 the mixer was originally located horizontally in the pipework between the ozone contactor and the filter, but was subsequently removed to allow higher flow rates into the filter. The Kenics mixer supplier quoted 0.07 bar head loss at flows equating to filtration rates of 7.5 - 15 $m.h^{-1}$. A rule of thumb mixing energy of 600 mm head loss is considered adequate for rapid mixing.

Further mixing was achieved when the flow entered the column, either hitting a splash plate when the water level was low, or hitting the surface of the water in the column. This was the situation when the columns had only two sections above the nozzle plate. After some overflow modifications to the type B columns in Summer 1989 column 5 operated with a flooded inlet towards the end of each filter run. Once the extra column section had been added, the raw water entered the column below the top water level during the extended high head loss phase. Part of the reason for changing filters 2-6 to outlet flow control, in late 1990, was to stop the raw water falling down into the columns by maintaining flooded inlets. This was then achieved at all times except immediately following a backwash.

The trials with iron early in 1991 used the Kenics static mixer for column 1 and hydraulic mixing across an orifice plate for columns 5 and 6. Subsequently, for 5 and 6 iron application was moved to the de-gassing outlet of the ozone contactor to permit polyelectrolyte injection just prior to the orifice plate. Further changes to columns 1, 5 and 6 moved the iron injection point into the pipework feeding stored water into the ozone contactors. For column 1 this was followed by the

removal of the mixer from the pipework between the contactor and the filter, enabling higher flow rates to be tested: previously the contactor had overflowed at rates in excess of 50 $l.min^{-1}$.

Iron was briefly dripped into the top of the first ozone contactor of the column 5 and 6 pair. This caused ozone leaks and coincided with poor filtration results and was discontinued.

In the middle of 1991 pipework changes were made so that column 2 received water via either column 5 or 6's feed pipework, so that it was iron and ozone dosed. When column 5 was not receiving water via the contactor a Kenics mixer on the original pipework ensured iron could continue to be dosed. A mixer was not available for the column 6 pipework, so only raw water was received when the ozone contactor was by-passed. The inlet for column 6 was at the top of the second column section. The inlet to column 5 was at the top of the third section.

It was considered that this type of dosing was defined as in-line or contact filtration, which implies there was only rapid mixing and no flocculation tank. Whether the ozoniser counted as a flocculator is not clear, but iron dosing before or after the contactor did not appear to show a difference in filtrate quality or run length. No controlled experiment was carried out to check this.

Calculations of mixing energy G (units sec^{-1}) and Gt, mixing x time (no units) were not attempted for the following reasons:

the iron dosing was applied at very low pump frequencies which was not ideal for mixing;

in the trials without outlet flow control any mixing in the static mixers and feed pipework, forming flocs, would have been disrupted by the violent entry of the water into the column hitting the splash plate - column 1 operated with only inlet flow control for the duration of the project;

dosing iron before the ozone contactor did not use a static mixer but relied on bends in the pipework; as the water left the contactor the hydraulic de-gassing fall would have damaged flocs; bubble contactors do not act as flocculators (Ives, 1981);

dosing iron into the hydraulic de-gassing stage was followed by complex pipe geometry into filters 5 and 6;

the residence time in the pipework and the water above the filters varied with flow rate;

the doses of iron used were too low to form flocs in a jar test.

The samples of iron dosed water were taken from the filter column immediately above the media. They were measured using Palintest low range test tablets (range 0-1 $mg.l^{-1}$) and a bench colorimeter. The dosed values plotted in the results section are those taken from actual spot samples, not from pump curve calibrations. When the dose was >1 $mg.l^{-1}$ the samples were diluted with tap water or distilled water because the high range test tablets were found not to work. The manufacturers were unable to provide an explanation for this.

Iron samples were taken from raw water and treated tap water as a control. Generally these values were read on the colorimeter as <0.01 $mg.l^{-1}$. Filtrate samples were taken only from those columns where iron was dosed.

The targets set out for iron residuals in the filtrate were the same as if these filters were on a conventional coagulation, settlement and filtration plant, i.e. 0.02 $mg.l^{-1}$ average, 0.05 $mg.l^{-1}$ 95%-ile maximum, and 0.20 $mg.l^{-1}$ maximum (Council of European Communities, 1980). This was decided upon as the best way of ensuring that the iron residual would not be deleterious to subsequent slow sand or GAC filtration units downstream of the RGFs.

Development work might look at how accurate the Palintest low range iron tablets were against laboratory methods for iron determination, but the costs ruled this out as a routine procedure.

3.3.4 TURBIDIMETERS

During the course of the project on-line Hach 1720C turbidimeters were installed on each filtrate stream and the raw water. Flow was controlled by means of a valve to the manufacturer's specification of 250 - 750 $ml.min^{-1}$. The flow rate to each meter was checked weekly and adjusted

as required. The meters were set to produce a 4-20 mA output to the computer data logger across a suitable range of turbidities; generally, 0-3 NTU for column 1, 0-5 NTU for columns 2-6 and 0-10 NTU for raw water. Logging of the turbidimeters by the computer commenced: Columns 1,2, 5 and 6 - 26/4/90, Stored water - 10/12/91, Columns 3 and 4 - 2/1/92.

The daily measurement of turbidities by a sample read on a Hach B2100 bench turbidimeter were used to check the accuracy of the on-line instruments. If the readings were widely different, say 25% out, particularly if the on-line instrument read higher than the bench sample, then the meter would be checked for flow rate and solids contamination, for example iron residues, zooplankton or general dirt within the flow cell, and cleaned if necessary.

According to the manufacturer (Hach, 1988) the 1720C digital low range turbidimeter had the following performance: Accuracy $\pm 2\%$ from 0 - 30 NTU; Resolution 0.0001 NTU; Repeatability $\pm 1.0\%$ or 0.002 NTU, whichever is the greater; Response time for a full scale step change 90% response in 5 minutes at 500 $ml.min^{-1}$. They warned that the amount of light scattered depended on the size, shape, composition and refractive index of particles in a suspension so that, whilst there was a relationship between the suspended solids concentration of a suspension and its turbidity, it varied between suspensions and was not quantifiable.

3.3.5 DIFFERENTIAL PRESSURE TRANSDUCERS AND MANOMETERS

The manometer and pressure transducer tappings above each filter bed and out of each plenum chamber were simple openings, tapped with brass compression adaptor fittings. Nylon hoses connected the pressure transducer to the column. The transducer and the manometers were used to measure total pressure drop across the filter media and nozzles.

Computer logged pressure measurement was by a Tekflo TF2 variable reluctance transducer and TR8 pressure transmitter, calibrated to give 4-20 mA output for 0-2 m pressure difference. A three way valve assembly was used to assist calibration of the transducer against the water manometer.

Readings were displayed on the computer data logger and checked during the daily readings with the water manometers.

These units were found to be very reliable, and re-calibration was not attempted if the on-line and manual pressure readings differed by less than 50 mm, and re-calibration was only performed if the unit consistently varied from the manometer reading. The meters were found to give a reliable output up to 3.5 m differential pressure.

Intermittent problems with pressure reading for column 2 were experienced. Eventually small wiring problems were found to have been the cause of these problems. The first was a loose connection in the terminal box for column 2 which only caused problems in the winter, and the second fault which developed later was a broken solder joint in the transducer plug.

3.3.6 FLOW MEASUREMENT

Flow metering was achieved with three distinct units. To allow complete manual control during the first plant operational phase a gap meter was used in conjunction with a diaphragm valve to set manually the desired flow into the filter. An electrically actuated 2" ball valve was controlled by the computer to stop the flow during a backwash. The flow was measured electronically using a GF paddle-wheel flow sensor and signal conditioning unit, and logged by the computer. The electronic flow meters were calibrated by performing water rise rate tests in each column with all outlets closed.

Once confidence had been gained in the system, flow control was achieved by the computer program and the diaphragm valve and gap meter were used to set a flow level in excess of the desired amount, forcing the ball valve to operate around its midpoint. The computer controlled the valve by energising an opening or closing relay for a brief period, calculating an average flow, and comparing this with the set point to determine further action.

On the outlet side of the filter a standard 50 mm Kent PSM water meter was used to integrate total flow treated. As there are accuracy standards to which, by law, these meters must adhere these meters were used to cross-check the electronic units, and calculate average daily flow rate. Calibration of the Kent PSM meters by water level drop tests was not carried out.

When target flow rates to columns 2 to 6 were increased to 75 $l.min^{-1}$ (15 $m.h^{-1}$) during the rising mode phase of operation it was found that the pressure drop across the PSM units was too high for the pilot plant since the columns would be full before 1.8 m head loss had been developed across the bed. This meter worked in a way known as semi-positive displacement which used a rotary piston mechanism to measure the flow. After a short period of operating without a flow integrator the meters were replaced with Kent Helix 3000 units, which, by using a small rotating paddle in the flow, consumed far less head.

These meters were installed for columns 2 to 6 in November 1990, at the same time as the electronic units and flow control electrical ball valves were moved to the outlet side to permit outlet flow control. The paddle wheel flow sensors were re-calibrated in the 1" housings against the Kent Helix meters. Additionally a butterfly valve was provided to enable the ball valve to operate near its mid-range. The manufacturers' recommendations of upstream and downstream straight pipe lengths were observed. These were 15" before and 5" after for the electronic units and 20" and 10" for the Helix meters. The PSM meters did not require this.

Because the flow to column 1 was limited by the hydraulics of the ozone contactor, no attempt was made to increase the flow to this column beyond 15 $m.h^{-1}$ and so the pipework was left in its original state.

3.3.7 FILTER MEDIA

Table 3.2 lists the filter media used. The type, dates and depths of media in each trial are presented in tables in the appendix. An excess of each media layer was put into each column and the media then given a fluidising and stratifying backwash. The column was then drained and the excess

material, approximately 50 mm deep, consisting mostly of fines was removed using a vacuum cleaner. A 100 mm depth of support gravels was used to cover the nozzles and assist with the dispersal of air and water during backwash. The sand had a density of 2.65 $g.ml^{-1}$, the anthracite 1.4 $g.ml^{-1}$ and the GAC 0.43 $g.ml^{-1}$.

Table 3.2. *Media types used in the pilot plant as quoted by suppliers.*

Media	Size range (mm)	Effective size (mm)	Uniformity coefficient
Chemviron F400 GAC US Mesh 12 x 40 BSS 10-36	0.42 - 1.7	0.6 - 0.7	2.0
BSS 6-14 sand	1.18 - 2.8	1.25 - 1.7	< 1.7
BSS 10-18 sand	0.85 - 1.7	0.9 - 1.18	< 1.4
BSS 14-25 sand	0.6 - 1.18	0.63 - 0.85	< 1.4
BSS 16-30 sand	0.5 - 1.0	0.54 - 0.71	< 1.4
Grade 2 Anthracite	1.2 - 2.5	1.3	< 1.5
Grade 2 Anthracite Coarse Cut [1]	1.7 - 2.5	1.8	1.1
Grade 3 Anthracite	2.5 - 4.0	2.6	< 1.5
Gravel	2 - 6		

BSS = British standard sieve

Effective size (ES) is calculated from the sieve size grading curve where 10% by weight of media (d_{10}) is smaller (Cleasby, 1990)

Uniformity coefficient (UC) is calculated from the size grading by d_{60}/d_{10} (Cleasby, 1990)

[1] Referred to as 2CC in this thesis

3.3.8 DATA COLLECTION

Each weekday a daily sheet was filled in to provide results of filtration performance, to check flow rates and chemical dosing, to cross-check the calibration of on-line instruments and to ensure that any faults which might have developed would be spotted quickly and remedied. Tables 3.3 and 3.4 describe the daily and weekly routines respectively. A stick type meter was used to measure pH.

Weekly samples were taken to measure the concentrations of the predominantly biological particles in the stored water and filtrates. Chlorophyll *a* was measured by spectrophotometry or fluorimetry

after filtration of 2 l samples on a glass fibre pad and digestion in boiling methanol. A second sample was filtered and POC was measured by titration following digestion in chromic acid (method in appendix of Watson, 1990). Particle size analysis was carried out using a Coulter counter with a 200 μm diameter orifice. Weekly checks were carried out upon instruments and the filters themselves.

Table 3.3. *Daily monitoring and sampling routine.*

Parameter	Monitoring	Action
Flow	Note gap flow meter Note computer flow values Note electronic flow meter raw values Note flow integrator	Adjust diaphragm or butterfly valve
Backwash flow	Note last backwash flow to ensure valves functioned correctly	
Head loss	Note computer dp values Note manometer tubes	
Turbidity	Note computer values Note on-line meter values Perform bench measurement	Take samples Clean flow cell if required
Iron dosing	Note pump settings Measure dosed water values Measure filtrate values	Take samples Bleed pump diaphragm if air-locked
Ozone dosing	Note ozoniser power, flow and pressure settings Measure ozone residuals after contactor using DPD and ampoule methods	Take samples
pH	Measure pH of water samples	Take samples
Temperature	Measure raw water temperature	Take sample
Computer	Check computer is reading sensors correctly	Change floppy disc when necessary

Table 3.4. *Weekly routine activities.*

Parameter	Monitoring	Action
POC	Laboratory analysis	Take sample
Chlorophyll *a*	Laboratory analysis	Take sample
Particle size analysis	Laboratory analysis	Take sample
Turbidity	Check that flow rate is within limits and clean flow cell Clean bench cells if required	
Iron	Measure value for iron in tap water	
Iron dosing	Refill dosing tank	
Media levels	Check distance from top of column to top of media to ensure media has not been lost over the course of successive backwashes	
Backwash flow	Ensure high and low rate flows are correct, that wash occurs correctly and that media expansion is sufficient	Adjust flow rates if necessary

3.4 PILOT PLANT OPERATIONAL DATA

3.4.1 INFLUENT WATER QUALITY

Several water quality parameters form the environmental background to the experiments. Figures 3.5 and 3.6 show respectively the results of daily measurements of reservoir stored water temperature and turbidity, indicating the seasonal ranges.

Weekly samples of the incoming water were analysed for particle numbers and size class between 4 and 80 μm equivalent sphere diameter using a Coulter counter with a 200 μm orifice. These data are presented in figure 3.7 as total particle number (ml^{-1}) and the counts converted to total particle volume (ppm) are presented in figure 3.8.

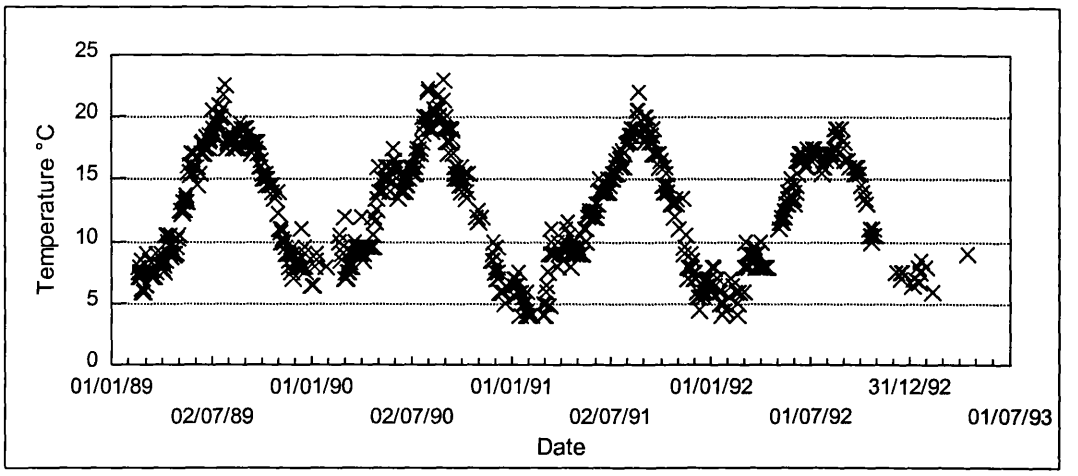


Figure 3.5. *Seasonal variation in stored water temperature measured at the inlet to the pilot plant.*

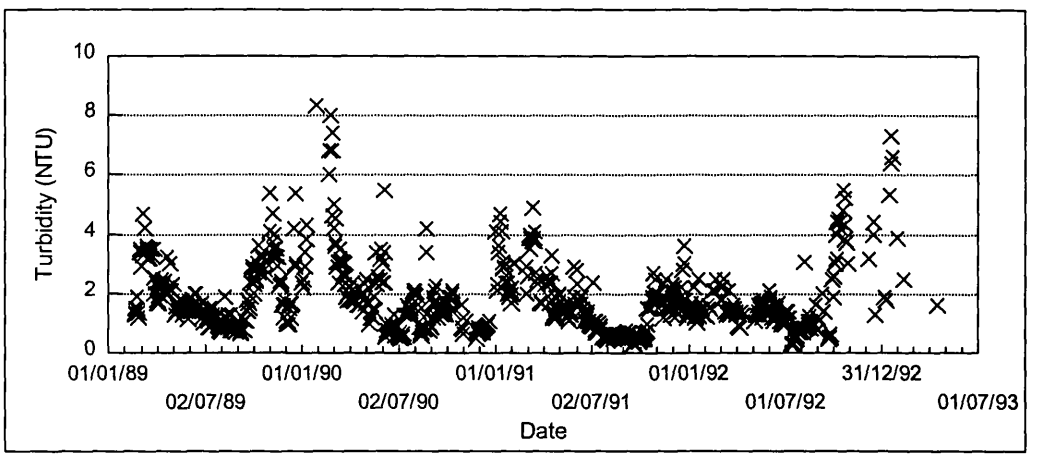


Figure 3.6. *Daily stored water turbidity results from the inlet to the pilot plant.*

Figure 3.9 shows an indication of the algal concentration in the influent measured by chlorophyll a concentration ($\mu g.l^{-1}$) and the concentration of particulate organic carbon (POC $\mu g.l^{-1}$) is plotted in figure 3.10. POC indicates the living material and detritus to be removed by the filters. A comparison of techniques for measuring algal concentrations, and the relationships between these parameters, was provided by Watson (1990).

The variation in particle loading was considerable. There was around one order of magnitude variation in total particle number and volume. Chlorophyll a concentrations showed seasonal increases regularly up to 20 times higher than the lowest winter concentrations, and POC varied typically by a factor of 5 between winter and summer. The impact these changes had on the effectiveness of filtration and the location of the deposition within the filter could be considerable,

so these data must be borne in mind when considering filter ripening in detail in later chapters of the thesis.

Stored water pH measured at the pilot plant 1989-93 averaged 8.37 with a maximum of 8.83 and a minimum of 7.50. The modal pH range was 8.5 - 8.6. It should be noted that these daily samples were not evenly distributed over the experimental period, see figure 3.11. The data suggest that the effect of reservoir storage was to elevate the pH, which is consistent with the photosynthetic utilisation of carbon dioxide by algae, although there might have been some diurnal variation.

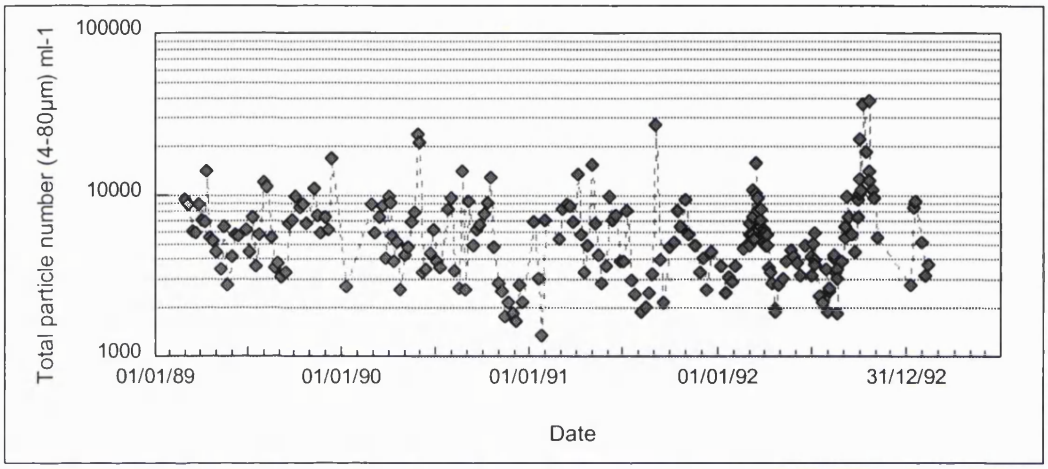


Figure 3.7. *Weekly stored water total particle numbers measured by Coulter counter.*

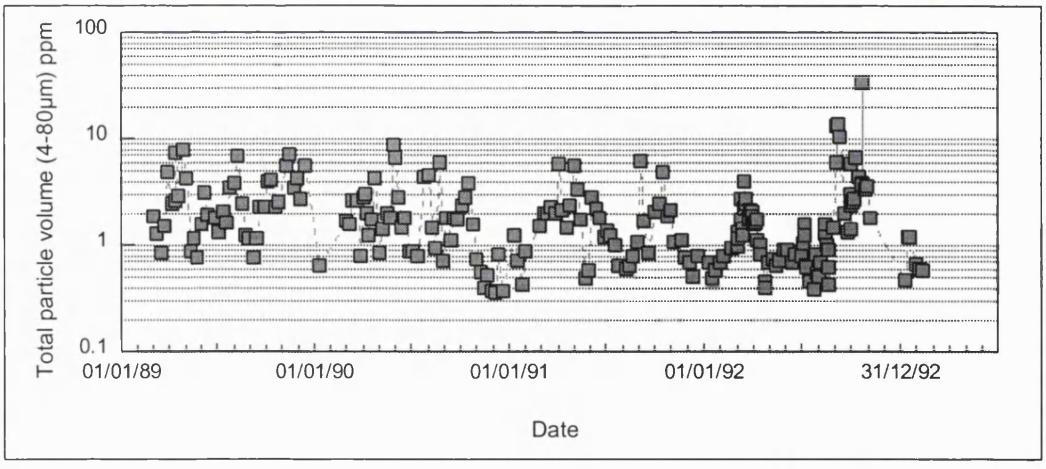


Figure 3.8. *Weekly stored water total particle volume measured by Coulter counter.*

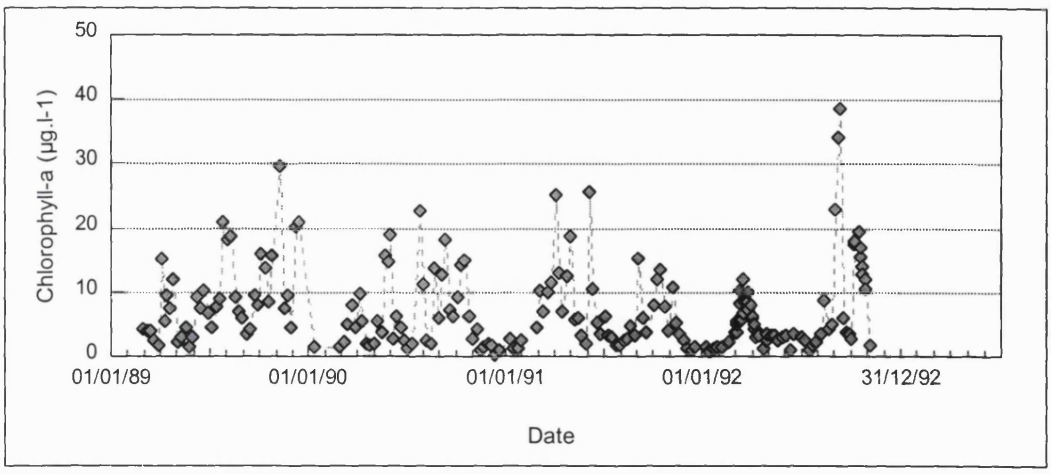


Figure 3.9. *Weekly stored water chlorophyll a concentration.*

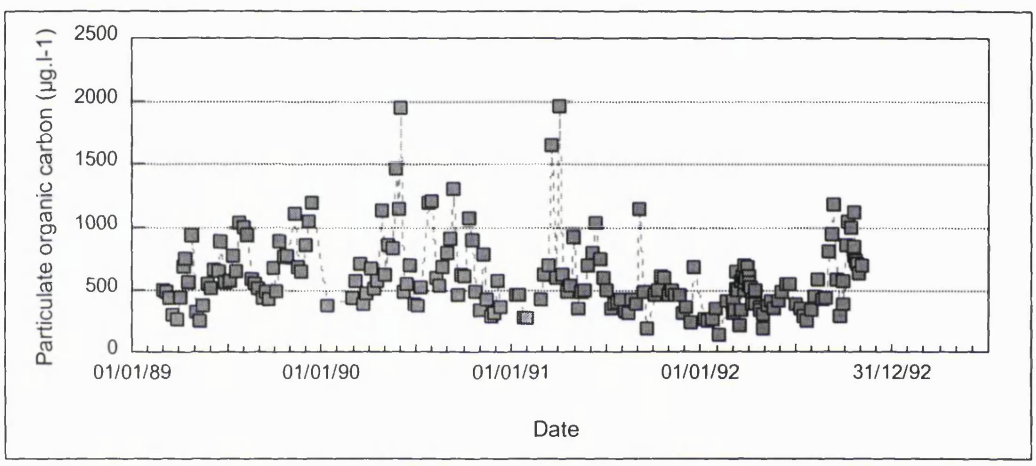


Figure 3.10. Weekly *stored water particulate organic carbon concentration.*

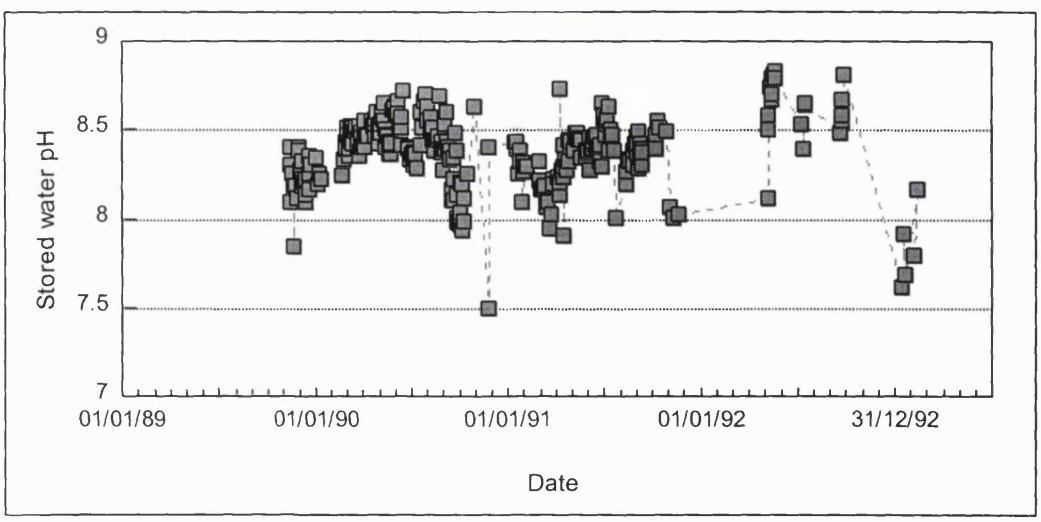


Figure 3.11. *Daily stored water pH values.*

3.4.2 RIVER THAMES WATER CHEMISTRY

In order to understand the impact of pre-ozonation and the conditions for coagulant dosing during the pilot plant trials it is necessary to determine certain inorganic and organic water quality parameters. Table 3.5 presents data from Arnac and Burke (1994) and Graham *et al.* (1996) on some water quality parameters measured for River Thames water in the lower Thames valley, upstream of the tidal limit.

The data show that the water was typical for a lowland river with low turbidity, low colour and low metals concentrations, but high in dissolved solids, hardness and conductivity, and moderately high total organic carbon concentration. It is acknowledged that particle loads change dramatically during reservoir storage, principally due to algal blooms. The pH and turbidity of the water would be expected to change in the reservoirs, but the general chemistry of the water was unlikely to change.

Table 3.5. *Physical and chemical water quality data from the River Thames 1989 - 1994 (years 1989-1992 after Arnac and Burke, 1994; years 1992-1994 after Graham et al., 1996).*

Determinand	1989 mean	1990 mean	1991 mean	1992 mean	1992-4 mean	1989-94 maximum	1989-94 minimum
Colour (Hazen)	13	13	13	20	-	56	6
Turbidity (FTU)	19.5	17.3	11.8	21.5	-	207	1.9
Conductivity ($\mu S.cm^{-1}$)	651	650	680	652	641	980	391
pH	8.02	8.17	8.16	8.22	7.9	9.1	7.1
Nitrate as N (mg $N.l^{-1}$)	7.6	7.6	8.3	8.6	-	15.5	4.6
Carbonate hardness (mg $CaCO_3.l^{-1}$)	220	223	217	205	-	258	134
Non carbonate hardness (mg $CaCO_3.l^{-1}$)	70	63	78	91	-	-	-
Total hardness (mg $CaCO_3.l^{-1}$)	290	286	295	296	283	385	134
Alkalinity (mg $CaCO_3.l^{-1}$)					198	257	162
Total dissolved solids ($mg.l^{-1}$)	468	470	481	-		565	366
Aluminium ($mg.l^{-1}$)	0.06	0.04	0.05	0.06	-	0.40	<0.01
Iron ($mg.l^{-1}$)	0.5	0.48	0.36	0.43	-	2.6	0.06
Manganese ($mg.l^{-1}$)	0.02	0.02	0.02	0.03	-	0.06	<0.01
Total organic carbon ($mg.l^{-1}$)	-	-	-	6.2	5.4	13.7	3.5
Chemical oxygen demand ($mg.l^{-1}$)	20	16	-	-	19.1	30.7	10.6
Suspended solids ($mg.l^{-1}$)	-	-	-	-	12.9	48	2.8
UV absorbance 254 nm (m^{-1})	11.4	10.6	10.8	15	-	40	7.6

3.4.3 TURBIDITY MEASUREMENT

Above 0.2 NTU the bench and on-line instruments showed good agreement. On-line turbidimeter readings below 0.2 NTU were usually lower than the equivalent bench sample reading. Because the on-line units gave results largely in accord with the bench measurements no attempt was made to reset their factory calibrations.

A slight reduction in the turbidity value recorded between the on-line instrument display and the computer display was noted on site for two instruments. This was of significance at the very low values recorded with ozone and iron(III) sulphate dosed filters. Column 1 data were lower by 0.025 NTU, and Column 5 by 0.04 NTU. Column 6 values were unchanged. Confirmation of this, and further details not recognised on site, came from processing the daily sample records (table 3.6).

Figure 3.12 presents the results from 520 daily samples of turbidity out of column 5 measured by the on-line instrument (x-axis) compared with the bench instrument value (y-axis) and the computer reading (2nd y-axis). The dashed line indicated x = y. Logarithmic axes have been used to clarify the point. Table 3.6 further illustrates this point for the other turbidimeters, by showing the mean difference between the bench (B) and the on-line instrument (I) reading, and the mean difference between the on-line and logged computer (C) values, together with their standard deviations, and number of readings.

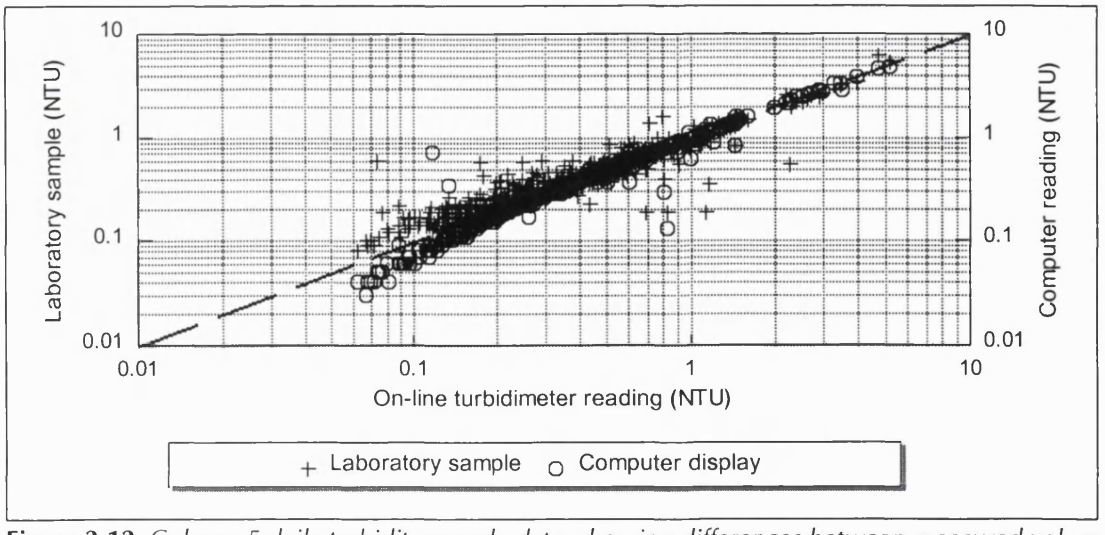


Figure 3.12. *Column 5 daily turbidity sample data, showing differences between measured values.*

Table 3.6. *Means and standard deviations of differences between the three daily recorded turbidity sample values, bench (B), on-line instrument (I) and computer reading of on-line instrument (C).*

Column	Mean (B-I)	Standard deviation (B-I)	Number of samples (B-I)	Mean (I-C)	Standard deviation (I-C)	Number of samples (I-C)
1	0.038	0.18	527	0.026	0.09	527
2	-0.059	0.21	406	0.041	0.09	406
3	-0.062	0.20	73	0.058	0.26	73
4	-0.007	0 09	171	0.039	0.02	165
5	0.008	0.17	520	0.036	0.08	520
6	0.047	0.12	499	-0.007	0.06	499

3.4.4 OZONE PRODUCTION MEASUREMENT

The production of ozone was calculated by bubbling ozonised air through a potassium iodide solution and titrating with sodium thiosulphate using the method of Ozotech Ltd. (1990). Two ozone generators were used, with maximum production capability rated by their manufacturer at 9 $g.h^{-1}$, and 36 $g.h^{-1}$. The smaller unit supplied ozone to the water fed to column 1, and the other supplied columns 5 and 6. The larger generator's output was measured 43 times at approximately two month intervals between April 1991 and February 1992. The smaller unit was measured 21 times. The physical parameters needed for calculating the ozone concentration were generator power (kW on the larger unit, A on the small), and air flow rate ($m^3.h^{-1}$) and pressure (bar) through the generator. The calibration of the larger unit was calculated as follows. From the measured ozone production ($g.h^{-1}$), shown as squares in figure 3.13, a multiple regression was calculated. This took the form:

$$y = 68.9x + 3.7a - 26.5b + 10.1$$

where: y = ozone production ($g.h^{-1}$)
x = generator power (kW)
a = air flow rate ($m^3.h^{-1}$)
b = air pressure (bar)

The values calculated from this regression are shown as diamonds in figure 3.13. The value of r^2 for this regression was 0.9048. A simplified regression basing production purely on power setting returned a value of $r^2 = 0.8965$ for the equation:

$$y = 68.1x - 5.3$$

This equation is shown as the line in figure 3.13. Because the r^2 values show that the ozone produced was strongly related to the power setting, the simpler equation has been used to calculate ozone dose.

The calibration of the smaller unit was based on the data plotted in figure 3.14. A linear regression took the form $y = 8.39x - 1.43$, $r^2 = 0.87$, where x = generator current (A).

The ozone generator power readings, the water flow rate and the ozone residual were recorded daily. This allowed approximate ozone dose to be calculated for the filter runs taking place at that time, and extrapolated to subsequent runs if there had been no interruptions to plant operation by the time of the next set of daily readings.

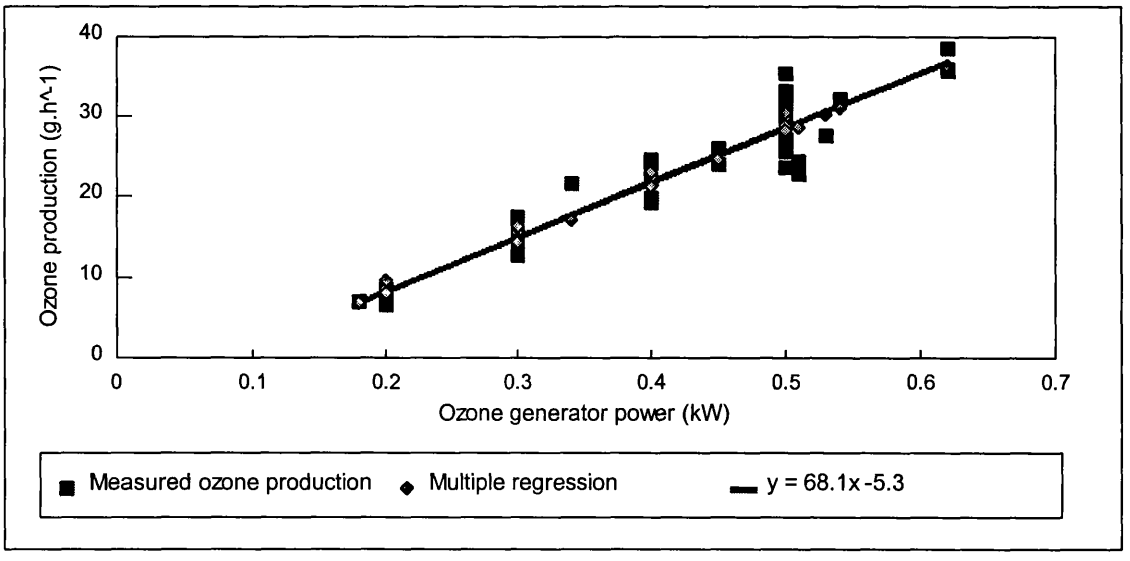


Figure 3.13. *Ozone production calibration curve for the generator supplying filters 5 and 6.*

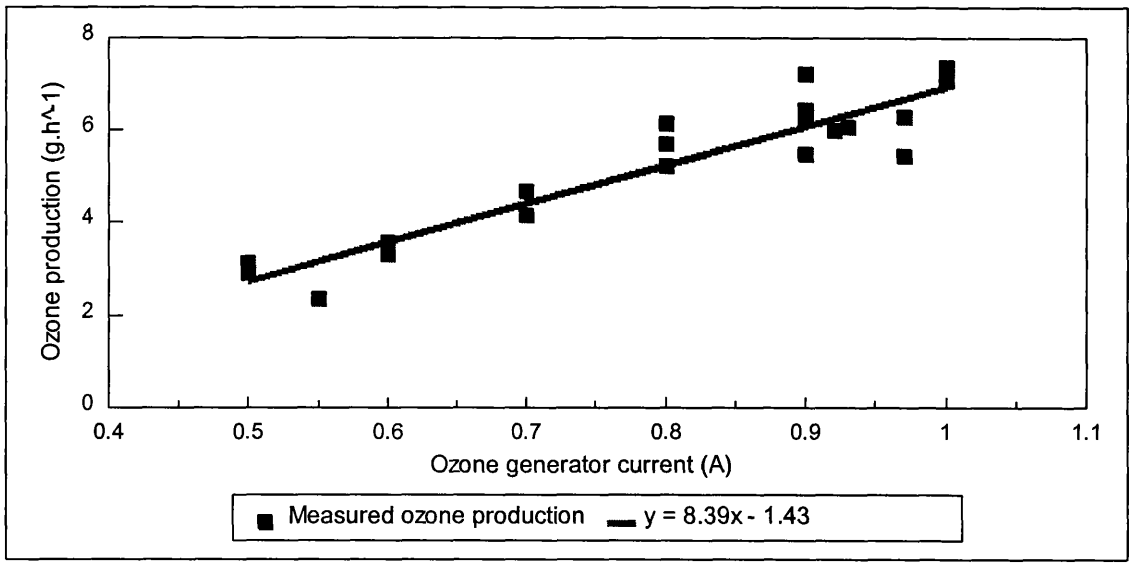


Figure 3.14. *Ozone production calibration curve for the generator supplying filter 1.*

3.4.5 CALIBRATION OF IRON DOSING EQUIPMENT

The measurement of the concentration of iron dosed into the filters requires discussion. Two sizes of pump were used: a class 1000pp pump supplied column 1 and a 2001pp pump supplied the common feed to columns 5 and 6. The dosing equipment was optimised by trial and error to avoid problems experienced early on in the trials. Each was situated below a reservoir containing 25 l of iron(III) sulphate. The full pump stroke was always used, and dose was varied by changing pump stroke frequency. This arrangement was necessary to avoid air locks in the pumps.

The coagulant was a commercial grade, diluted to 25% of the supplied concentration because of very low pump frequencies used when low coagulant doses were required. The supplier's data stated a typical product specific gravity of 1.55 $kg.l^{-1}$, and iron(III) content of 11.5% w/w. The 25% solution therefore contained approximately 45 mg $Fe^{3+}.ml^{-1}$.

The pumps were calibrated in situ, pumping dilute iron(III) sulphate from a burette. Three type 1000pp pumps were calibrated. The calibration curves are shown in figures 3.15 and 3.16. The regression $y = 0.049x$ was calculated with the data from all three pumps. The r^2 value was 0.999. It appears the pumps were very similar in performance. The approximate concentration of iron per pump stroke was $0.049 * 45 = 2.2$ mg Fe^{3+}. One type 2001pp was calibrated with water. This

produced a regression of y = 0.227 x with an r^2 value of 0.999. Since the coagulant solution was denser than water it is not safe to assume the pump would deliver the same volume per stroke. However it may be assumed that each pump stroke could not deliver more than an estimated maximum of 0.227 * 45 = 10.2 mg Fe^{3+}. The pump outputs were linear.

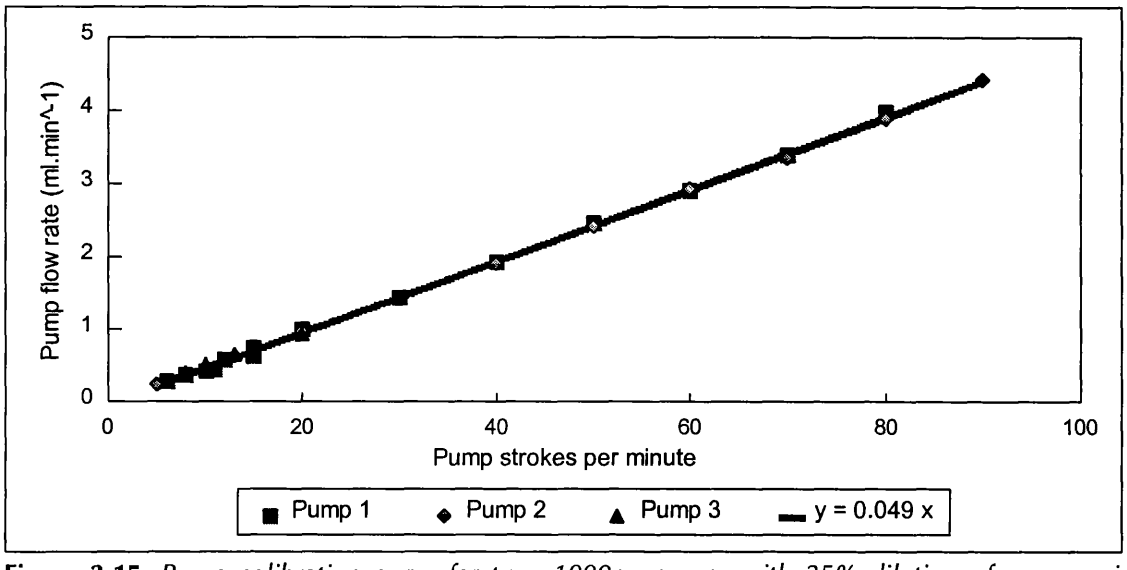


Figure 3.15. *Pump calibration curve for type 1000pp pumps with 25% dilution of commercial iron(III) sulphate solution.*

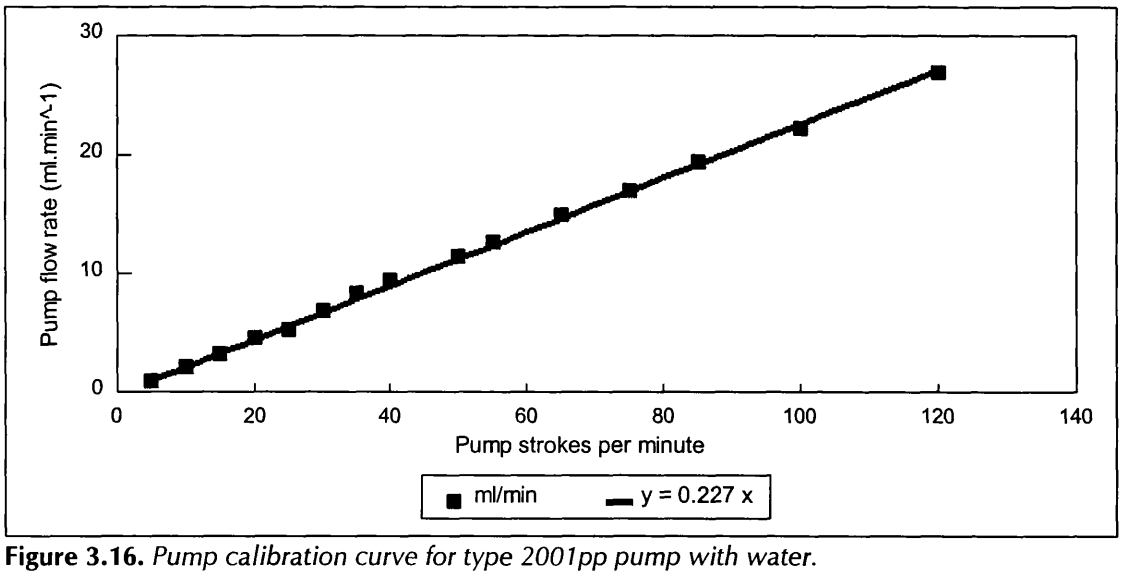


Figure 3.16. *Pump calibration curve for type 2001pp pump with water.*

The iron dose was recorded in two ways. The pump stroke frequency was recorded on daily readings sheets, and whenever the pump setting was altered it was timed and noted in the site log book. Samples of dosed water were taken from the filter just above the media and measured by the Palintest colorimetric method on site. Using the measured concentration of iron in the water, the

flow rate of water into which the iron was dosed and the pump stroke frequency the weight of iron dosed per stroke was estimated.

Figure 3.17 shows the results for the weight of iron delivered per pump stroke for two 1000pp pumps, calculated from the measured water sample, water flow rate and pump setting. The graph has been divided into four regions. Prior to 22.4.91 the iron was dosed ahead of a static mixer after the ozone contactor. The mean value was 2.7 $mg.stroke^{-1}$. In the three following sections the iron was dosed ahead of the ozone contactor, and the static mixer was removed. The middle line (15.12.91) was the day that a "yellow sludge" was removed from the iron injection point. Both middle sections had an average delivery value of 2.2 $mg.stroke^{-1}$. This value was the same as the value determined by the pump calibration. The data measured after 22.4.92 were from a different pump of the same model. The average delivery of this pump was 1.7 $mg.stroke^{-1}$.

Figure 3.18 shows the estimates of iron delivered per pump stroke for a 2001pp pump, calculated as above. The inflow to columns 5 and 6 was recorded daily from a electro-magnetic flow meter. The graph has been divided into three regions. In the left hand and right hand regions the iron was dosed into the feed to the ozone contactor immediately upstream of a 90° elbow in 3" PVC pipe at ground level. In the centre period, from 13.1.92 to 3.6.92, the delivery pipe and injection nozzle were suspended about 6 m off the ground, discharging into the ozone contactor outlet. The water passed over a weir and dropped approximately 0.5 m. The mean dose per stroke value appeared to fall steadily during each of these periods. For the time when stored water turbidity was logged (10.12.91 onwards) the mean stroke delivery was 6.7 $mg.stroke^{-1}$ in the first period, 3.6 $mg.stroke^{-1}$ in the second and 4.9 $mg.stroke^{-1}$ in the third.

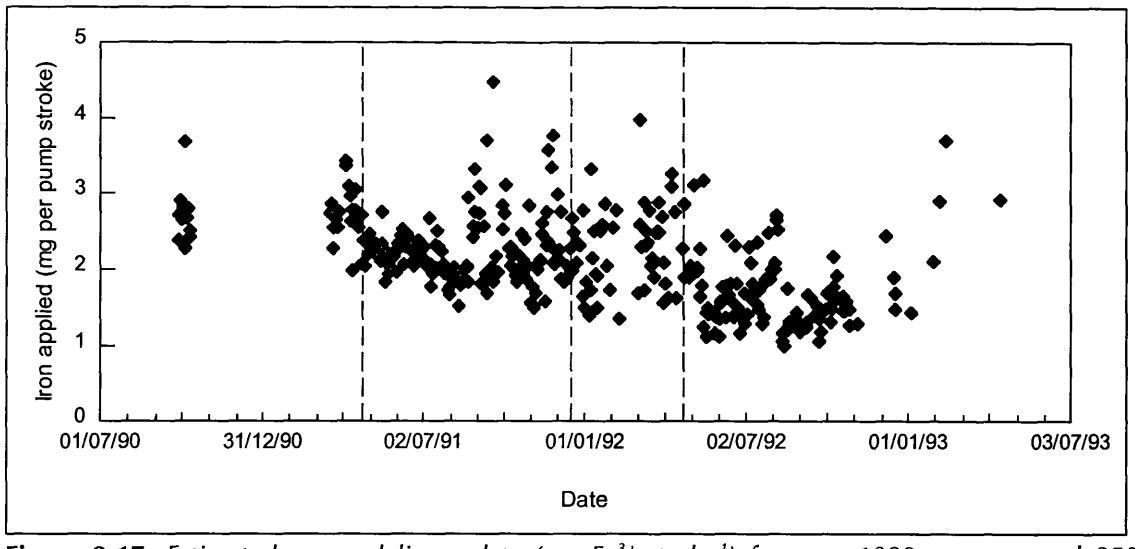


Figure 3.17. *Estimated pump delivery data (mg Fe^{3+}.stroke^{-1}) for type 1000pp pump and 25% iron(III) sulphate, based on the measured dose of iron in the water above the filter, the water flow rate and the pump stroke frequency.*

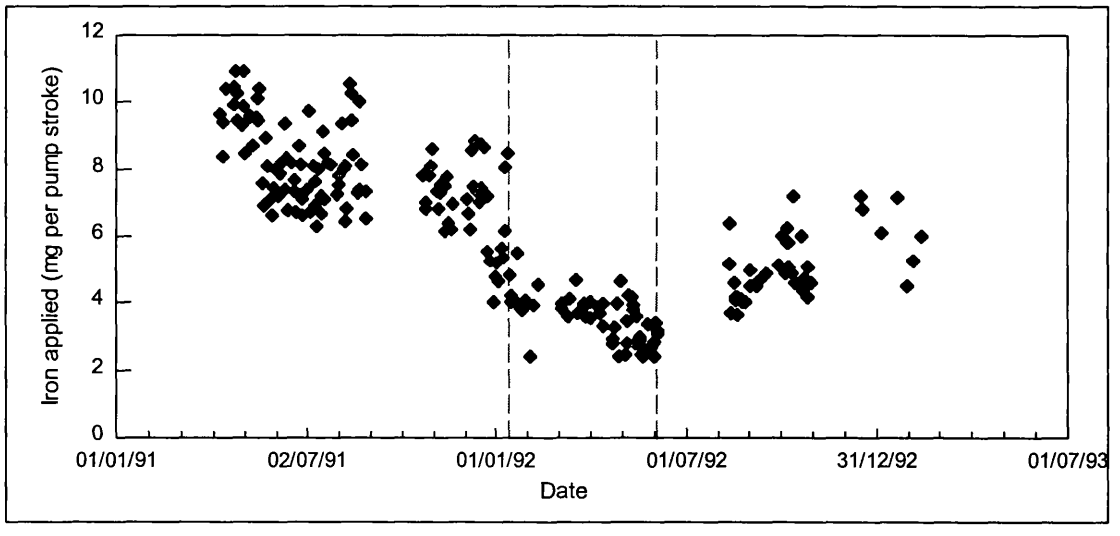


Figure 3.18. *Estimated pump delivery data (mg Fe^{3+}.stroke^{-1}) for type 2001pp pump and 25% iron(III) sulphate, based on the measured dose of iron in the water above the filter, the water flow rate and the pump stroke frequency.*

3.5 DISCUSSION

3.5.1 TURBIDITY MEASUREMENT

The on-line and bench turbidity instruments provided useful cross checks for one another, although, as figure 3.12 showed, at low turbidity values the bench method read higher than the on-line instrument and the on-line value was under-registered by the computer. There were several causes for the differences in turbidity readings: for example experimenter error, accumulated dirt on the on-line sensor, scratches or scale on the glass sample tubes, gradual deterioration in secondary

standards, and the decision not to re-calibrate the on-line instruments. The glass tubes needed regular acid cleaning. The bench standards deteriorated slightly over time and required re-calibration against a primary standard. The logged data showed step changes at low values due to the computer rounding the on-line turbidity reading from 3 to 2 decimal places.

Turbidity measurements with Isoton particle free water and distilled water revealed that the bench turbidimeter could read as low as 0.025 NTU, although scratched tubes could account for an additional 0.02 NTU. Care was required to avoid condensation with cold water samples. The low on-line turbidity values measured with the ozone and iron treated sample may have measured higher in the bench sample due to pin-head sized bubbles of oxygen coming out of solution under the heat of the turbidimeter lamp, since the ozonated water would be super-saturated with oxygen.

With the on-line meter being a flow-through device, and the bench unit requiring sampling into a glass cell, the two turbidity meters were not measuring light scatter in quite the same way. Neither method was wrong, but with very low target values the choice of instrument would have some bearing on the chance of meeting these targets. Since an analysis of monitoring techniques was not part of this project, the fact that both results were largely in quite close agreement was taken to show that both methods were satisfactory, and each acted as a control for the other.

For these reasons it has been decided that the logged data would not be any more accurately presented later in the thesis if an adjustment was made on the basis of the results in table 3.6. There was no reason to regard the bench data as more meaningful than the on-line values - the methods simply read the turbidity of the water in a different way and returned a slightly different value. However a degree of caution should be applied to the absolute accuracy of any turbidity values quoted later, especially at very low values.

3.5.2 OPERATION OF OZONE EQUIPMENT

In hindsight the management of the ozonation system was not ideal. Time was required to keep the system going safely. Access problems meant mass balances could not be performed. The objective

with the ozonation was to achieve a residual with an excess of ozone. No attempt was made to destroy the residual. Several reasons were found for inconsistent ozone dosing or variable ozone transfer efficiency: there were occasional leaks in the ozone pipework; problems with air drying resulted in nitric acid fouling the generators; deterioration in the contactor diffuser bubble patterns showed they required periodic cleaning. As a result the calculated ozone dose may not have been fully transferred to the water, and values quoted in the thesis should be treated as nominal.

3.5.3 OPERATION OF IRON DOSING EQUIPMENT

There are several possible reasons why the pump output appeared to vary over the course of the experiments. The change in dosing point resulted in different delivery heads for the pump. Unfortunately no loading valve was used with the pumps and so pump output could have varied with back pressure when the dosing point was moved. The deposition of sludge at the injection point may also have been accompanied by fouling of the dosing hoses and extra back pressure. The pump performance may have deteriorated with use, i.e. component wear. It is not clear why a replacement pump produced a lower delivery when the calibration suggested these types of pump were capable of nearly identical performance.

To avoid the iron dosing pump from getting air-locked it was necessary to pump with the largest stroke volume and low stroke frequencies. The iron dose was very low in the winter, and even using a 25% dilution of the manufacturer's product did not prevent stroke rates as low as 2 or 3 per minute. The mixing of chemical into the water was probably not optimum at these low frequencies.

The measurement of iron clearly showed much scatter. The degree of chemical mixing, particularly at low pump frequencies, the accuracy of sampling, the fate of the iron in terms of precipitation reactions or aggregation with particles, and the repeatability and accuracy of the measuring technique were all potential causes of variation in the assessment of pump performance.

The examination of dosing pump performance was carried out to provide a simple calibration to enable the pump settings to be converted to an approximate coagulant dose. This was added to the

database record of filter performance. This method eliminated the large variations that the measurements gave, and allowed the relative performance of different pump settings to be considered. It cannot be considered that coagulant dose was accurately recorded, but subsequent work carried out by Thames Water using a calibrated sight glass on the suction side of the pump confirmed that doses of < 1 mg $Fe^{3+}.l^{-1}$ were optimum for contact filtration (Chipps *et al.*, 1995a).

3.5.4 DATA LOGGING

Fifteen minute data logging intervals were chosen because more frequent logging would probably have resulted in the collection of anomalous data during the ripening period. The first reason for this was the backwash sequence ended by fully opening the outlet valve up to 17.7.92. After this date it was only partially opened. In each case the computer then took two minutes to establish the correct flow rate from the programmed value by gradually adjusting the valve position.

Secondly, in some trials there was the potential for the build up of chemicals in the inlet pipework during backwashing. This was flushed out of the filter with the tail end of the backwash water, but its effect on the initial filtration conditions, although minimal, could not be quantified. To overcome this problem columns 5 and 6 were fed with water from a common manifold with common iron and ozone dosing from 1991 onwards.

Thirdly, the turbidimeter was designed with a 2 l bubble trap. With a sample rate of around 500 $ml.min^{-1}$ this meant that any rapid transients in filtrate turbidity were likely to be missed. In addition the timing of the end of a backwash was not logged, so the first sample could be taken from 1 to 15 minutes after the wash ended. As a result the precise details of the pre-ripening phases described by Cranston and Amirtharajah (1987) could not be confirmed.

3.5.5 DATA PROCESSING

The problems with plant reliability meant that significant time and effort were required to modify the database to ensure only representative data were examined. Entire filter run records were deleted from the database where it was known that the backwash had failed, where the filter run was terminated early or otherwise interfered with, or when the turbidimeters were dirty, or when

the logger stopped functioning. Individual records were deleted where the daily log books showed the turbidimeters were being calibrated.

The lack of instrumentation to measure the concentrations of iron and ozone in the water, made it necessary to combine daily readings and plant log book notes into the logged database, using the pump calibrations and daily samples to provide an estimate of the applied iron and ozone doses. Modifying the logged database to deal with operational problems and the lack of on-line iron and ozone dose measurement was extremely tedious. Runs where the supply of either chemical started or stopped part way through were then identified and not included in the summary statistics. Since on-line measurement of iron and ozone is expensive and difficult, it would have been desirable to record generator and pump electrical status.

It would also have been desirable to log a backwash status indicator to eliminate runs where the wash had not been achieved successfully, and a run time counter would have been very useful. It was regrettable that logging of on-line turbidimeters for the stored water and columns 3 and 4 was provided late in the trials.

3.5.6 PLANT OPERATION

The operation of the plant was reasonably representative of full-scale filters, although more attention to ozone transfer efficiency and coagulant mixing would be required for optimising a large plant. The build up of ozone and coagulant in the column 1 feed pipework could have been avoided if a bypass had been provided during backwashing. This extra load was allowed to pass out of the filter with the backwash water at the end of the wash, but this was possibly not completely satisfactory, given the sensitivity of the system to chemical doses. The installation of common supply pipework to columns 5 and 6 overcame this problem.

The pair of 2" outlet ports did allow free discharge of dirty backwash water at fluidising water rates. During the backwash the water level was higher than these outlets, so some dirty water had to pass out of the filter after the high rate water stopped and after the inlet water flow had restarted.

The control program was modified from 17.7.92 to stop the opening of the outflow valve fully after the wash. The software did not allow for slow starts, and attempts to manually slow start the filter would have be thwarted by the computer attempting to adjust the flow to the set point. Future pilot plants should consider allowing slow starts. In addition, most laboratory and pilot plant studies are conducted at fixed flow rates. In reality many filter plants operate with varying throughputs. Some further work is required to examine the effect this has on filtrate quality, and what rate of flow rate change is suitable for slow starts and for flow rate change in ripened filters.

CHAPTER 4: FILTER BACKWASHING.

4.1 INTRODUCTION

When filtering water from a source containing a diverse biological community it is reasonable to assume that biological growths could become established on filter media. Scanning electron micrographs have confirmed the presence of bacteria, fungi and algae on dirty filter sand (Bayley, pers. comm., 1993). The presence of a matrix of biofilm on filtration has not been properly examined as an influence in rapid gravity filtration, and is not well understood for slow sand filters (Letterman, 1991). Following on from O'Melia and Ali's (1978) view of ripening, the presence of biofilm is likely to affect ripening. Replacing the media for each filter run has been undertaken in laboratory studies (Tobiason *et al.*, 1993) but is extremely hard to achieve with a pilot plant, and does not simulate full scale conditions.

The literature suggests two potential means of backwashing filters: separate air scour and fluidising water rinse, and the use of air combined with water (Amirtharajah, 1978). Both of these methods have been developed principally to wash chemical floc out of filters, often where the water treated contained a chlorine residual. The backwashing of filter media bearing biomass has not been reported. The impact of biofilms on rapid filtration has received limited attention e.g. Goldgrabe *et al.* (1993). Many filtration studies omit details of backwashing.

This chapter examines what long term changes can take place to filter media, and examines the methods of backwashing to determine how they influence media condition in the long term. It was intended that the filter media should be given sufficient backwashing to enable the study of filter ripening to draw conclusions that were not distorted by artefacts of media condition. Since the use of separate air and water was simpler and cheaper to engineer than combined air scour and backwash water (CASBW) systems (Brignal, pers. comm., 1997) it was decided to investigate the efficacy of the separate wash system in limiting long term biological and chemical changes taking place on the filter media in order to limit any change the filtration conditions. If this method was unable to maintain media in good condition the CASBW was to be used.

4.2 LITERATURE REVIEW

The backwash is an integral part of the operation of a rapid gravity filter and has an important influence on the performance of the filter during the ripening period (Amirtharajah and Wetstein, 1980). Cleasby (1990) believed that the "backwashing system is the most frequent cause of filter failure." The filter design and backwash sequence must facilitate removal of dirty backwash water but prevent media loss.

In the short term, the deposits which are not removed by the backwash may add extra load to the filter. Whilst this extra load might help ripen the filter there have been no reported studies investigating this. Amirtharajah and Wetstein (1980) have shown that particles remaining after the backwash can pass through the filter.

In the long term, chemical (Galvin, 1992) and biological (Bayley, pers. comm., 1993) deposits can accumulate on the media surface if not removed by the backwash. It is important that the backwash efficiently removes deposits to avoid the development of "mudballs" which are aggregates of dirt, media and coagulant. An excessive film of biological material and/or inorganic matter must not be allowed to develop on the media. This can bind the grains together causing the washed bed to return to service with high head losses. It may ultimately lead to the development of cracks through the bed, or dead spots in the bed leading to high localised flow velocities and "jetting" during backwashing (Cleasby, 1990). Kawamura (1975) illustrated how cracks up to 10 mm wide appeared in a filter containing "many" mud balls. Jetting may disrupt the bed structure as gravels are pushed up into the sand (Cleasby, 1990). Depending on the filter floor arrangement, sand can then pass into the under drains or block filter nozzles. Fulton (1988) discussed problems with caking of media grains due to inadequate backwashing.

Kawamura (1975) stated that backwashing a filter may be initiated by one of four triggers: filtrate turbidity (or other measure of quality if appropriate), head loss, time, or manual intervention. With constant rate filtration a filter requires backwashing when the head loss limits the flow rate. In

declining rate filtration washing is initiated on flow rate or water level (Cleasby, 1993). Ideally head loss and filtrate quality limits should be reached simultaneously to make optimum use of the filter (Ives, 1982, Tien and Payatakes, 1979). If scheduling is required in filter blocks containing many units in order to even out plant output, wash water demand or dirty wash water treatment plant utilisation, washing may take place on a pre-set time interval. Washing on time may also be necessary if backwashing is not optimal, resulting in excessive long term dirt deposition. In this case some improvement in media cleanliness can be gained by washing filters frequently (Bayley, pers. comm., 1993).

The objectives of backwashing are:

to remove superficial deposits from the media and to transport these out of the filter so as to recover the available voids for particle deposition and obtain a satisfactory back-to-service ("starting") head loss;

to remove or prevent the growth of "permanent" undesirable biological or chemical deposits on the filter media, whilst allowing some degree of biological or chemical "maturation";

to prepare the filter for the subsequent filter run; this could mean restoring or maintaining media stratification in multi-layer filters, or dosing extra chemicals to prepare the media or the influent water (Cranston and Amirtharajah, 1987).

Information on the hydraulics of backwashing and design of dirty backwash water outlet troughs has been presented by Kawamura (1975b). Further design information was presented by Norman and Gould (1984) and Cleasby *et al.* (1975, 1977).

4.2.1 BACKWASHING METHODS

In an introduction to filter backwashing Cleasby (1990) described four methods: i) upwash with full fluidisation, on its own and ii) supplemented by surface washing; and backwashing assisted by air scour, with the air either iii) preceding a water wash or iv) combined with it. Water alone was the

least effective cleaning method and combined air scour and sub-fluidising water washing was the most effective. Methods ii) and iii) were of similar, intermediate, effectiveness (Cleasby, 1990). The assistance provided by surface washing was demonstrated by Kawamura (1975a). Cleasby *et al.* (1975) showed the benefit of preliminary air scour over water-only washing, but said that this method could not prevent long term accumulation of material coating the sand grains. For dual media filters Cleasby *et al.* (1975) suggested that further auxiliary washing should be provided at the media interface. The danger of media loss when air scour is used, and ways it may be avoided, were discussed by Cleasby (1990). One effect of air scour was to break up large flocs and allow their removal by lower water rates (Adin and Hatukai, 1991)

a. BACKWASHING WITH WATER ALONE

In the US backwashing has traditionally used a fluidising upwash, with bed expansion of 15-30% (Cleasby, 1990), often assisted by a surface water scour or mechanical rakes. Amirtharajah (1978) and Quaye (1976), reviewing the history of backwashing, said that Hulbert and Herring (1929) had found 50% bed expansion was necessary for effective backwashing, whilst Hudson (1935) and Baylis (1937, 1959) required additional surface washing to eliminate mud-ball formation. Quaye (1976) did not regard 50% bed expansion for optimum backwashing as theoretically justified. Amirtharajah (1978) and Quaye (1976) said it was necessary to set up conditions for maximum hydrodynamic shear to optimise removal of deposits. These conditions were met at expanded bed porosities of 0.65-0.70 (Amirtharajah, 1978) or 0.75 and 0.78 for UK anthracite and sand respectively (Quaye, 1976). Amirtharajah (1978) said that his porosity values corresponded to the old "rule of thumb" bed expansions of 40-50%. Cleasby (1990) recommended that backwashing with full bed fluidisation should be achieved with a water rate of 1.3 V_{mf} (velocity of minimum fluidisation), based on a V_{mf} calculation using the d_{90} sieve size, therefore allowing the coarsest media grains to be mobile.

Ives (1982) was concerned that Amirtharajah's (1978) work indicated that 70-80% bed expansions were required, and that these were "too high for practical use." He did not state why, but reasons

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The V_{mf} is the point at which the pressure drop measured as water flows up through a bed of filter media stops increasing, being equal to the buoyant weight of the media, and the bed starts to expand (Cleasby, 1990).

could have included the volume of wash water required, problems of filter floor design, backwash pump sizing, clean wash water pipework diameter and holding tank volume, and dirty wash water treatment and disposal, with their consequent impact on construction and operating costs.

b. BACKWASHING WITH AIR ASSISTANCE

Amirtharajah (1978) noted that European tradition had been to utilise air scour, either followed by a water velocity sufficient to fluidise the media marginally (British practice), or simultaneously with water (mainland Europe), but concluded that the most effective way of washing a filter might involve a simultaneous air scour and sub-fluidising water wash.

Cleasby (1990) summarised typical backwash air and water flow rates. In Britain for 0.5 mm ES sand he said air flows ranged from 18 - 36 $m.h^{-1}$, and water from 12 - 20 $m.h^{-1}$. In the United States for dual media with 1.0 mm ES anthracite and 0.5 mm ES sand air rates of 55 - 91 $m.h^{-1}$ and fluidising water rates of 37 - 49 $m.h^{-1}$ were typical. For CASBW with 1.0 mm ES sand he reported air rates of 27 - 73 $m.h^{-1}$ and water at 15 $m.h^{-1}$.

The development of theory and practice of combined air and water backwashing have been described by Amirtharajah (1978, 1984, 1993), Fitzpatrick (1990) and Amirtharajah *et al.* (1991a). Combining air scour with sub-fluidising rates of upflowing backwash water produced the conditions, descriptively termed "collapse-pulsing", which resulted in the optimum removal of deposits from filter media. The efficacy of collapse-pulsing backwashing has been demonstrated by Amirtharajah (1984, 1993), Regan and Amirtharajah (1984), Amirtharajah *et al.* (1991a), Addicks (1991) and Fitzpatrick (1990, 1993).

Linear regressions of water flow rate as a percentage of V_{mf} against air flow rate under conditions of collapse-pulsing with sand and sand/anthracite filters were produced by Amirtharajah *et al.* (1991a). Lower water rates required higher air rates and *vice versa*. It was recommended that air rates should fall in the range 30 to 135 $m.h^{-1}$. At the lower end of the air flow range water rates should be 40 to 60% of V_{mf}. At higher air flow rates water flows should be in the region of 25 to 45% of V_{mf}.

Amirtharajah (1984) presented a general equation for predicting collapse-pulsing conditions, generated from soil mechanics theory, of the form:

$$aQ_a^2 + \left(\frac{V}{V_{mf}}\right) = b$$

where a and b = dimensionless constants for a particular system
Q_a = air flow rate
V = water flow rate
V_{mf} = velocity of minimum fluidisation

Amirtharajah (1984) presented experimental data that produced a linear regression close to his prediction with a high correlation coefficient (r = 0.88).

Visual observations, at laboratory scale, by Fitzpatrick (1990, 1993) showed optimum cleaning of filters dosed with kaolin suspensions occurred under conditions of collapse-pulsing. Practical confirmation of the effectiveness of backwashing using collapse-pulsing has been reported by Amirtharajah *et al.* (1991a) and Amirtharajah (1993) at pilot plant scale and full scale where collapse-pulsing conditions produced optimum cleaning in sand, dual media and GAC filters.

4.2.2 THE MECHANISM OF BACKWASHING

In the 1970s many workers sought to understand the conditions for optimum backwashing with water alone, and concluded that fluid shear was a poor mechanism, but inter-particle collisions caused removal of deposits: this is why surface washing or air scour were effective (Kawamura, 1975a, Cleasby *et al.*, 1975, Valencia and Cleasby, 1979). In complete contrast, Amirtharajah (1978) said that particle collisions were of negligible importance in cleaning media grains, particularly as the energy in the water was used to fluidise the media and this kept the grains apart. Since an optimum porosity, between 0.6 and 0.8, was required for effective cleaning of media with fluidising water, corresponding to bed expansions in excess of 50% (Amirtharajah, 1978, Quaye, 1976, 1987, Valencia and Cleasby, 1979) Amirtharajah (1978) and Fitzpatrick (1993) attributed the majority of filter media cleaning to hydrodynamic shear forces. Fitzpatrick (1993) observed media grain collisions during collapse-pulsing.

4.2.3 BACKWASH WATER VOLUMES

Ives (1981a) stated that backwash water volumes used were normally 1% of filter production, 3% was high and 5% was considered excessive. This was based on UK experience with low backwash water rates of 36 $m.h^{-1}$, a rate sufficient to fluidise fine media, supplemented by air scour. Backwash water usage may also be quoted as empty bed volumes, where 2 - 3 is typical, i.e. for a 1 m depth of media 2 to 3 $m^3.m^{-2}$ would be used.

The Montgomery Engineers (1985) regarded 8 $m^3.m^{-2}$ as a typical volume of water for a filter backwash. They suggested that a production efficiency of over 95% was desirable, so that less than 5% of the water produced should be used for backwashing, and stated a target productivity of 200 $m^3.m^{-2}.run^{-1}$. They also said that as much as 50% of a plant's capital cost would be spent in the provision of backwashing and washwater treatment facilities.

Carns and Parker (1985) reported savings in backwash water by switching from direct filtration with alum to direct filtration with clay plus polymer for a low turbidity (<3.5 NTU) water. Wash water usage fell from 4.3 - 6.3% to 2.4 - 3.0% across three plants. The wash method was not described, but the reason for the wash water saving was that the change in coagulant allowed longer filter runs and higher filtration rates.

Because of the shorter runs caused by the higher filtration rates generally employed in direct filtration, Janssens *et al.* (1989) stated that this treatment method used up to 6 - 8% of production in backwash water, compared with 2 - 4% for conventional filters. This must be offset against the extra water volumes lost in conventional treatment due to settlement tank de-sludging or flotation tank de-scums.

Amirtharajah (1985) showed that backwashing with water alone produced a constant wash water quality, i.e. no further dirt removal from the media surface, after 6.0 $m^3.m^{-2}$. He stated that the water velocity used and time could be varied yet reach the same cleanliness as long as the product of velocity and time was the same. This ignored the possibility that dirt particles might have different

sizes and densities and only be fluidised by high rates of water. 6.0 $m^3.m^{-2}$ is 6 empty bed volumes for a 1 m deep bed.

A volume of 6.0 $m^3.m^{-2}$ when backwash water is flowing at 48 $m.h^{-1}$ gives a high rate water backwash time of 7.5 minutes. The Ashford Common pilot trials used wash water at similar rates and times as it was found by visual observation through the filter windows that backwash water appeared to be clear by this time.

4.3 EXPERIMENTAL METHODS

The backwash was developed using the traditional British method as a start, and modifying it in the light of international experience, to provide a solution which would be practicable at full scale. Air and water rates and volumes were utilised which could be justified on the basis of literature references. Three methods were used for assessing backwash effectiveness:

the condition of the filter media in terms of attached organic and inorganic matter (POC and solids) on samples of backwashed filter sand; this was measured in the supernatant water after vigorously shaking a sample of filter sand in water;

the head loss at the start of each filter run, normalised by flow rate and temperature to take into account the different flow rates examined and the seasonal variation in water temperature;

turbidity data were used to investigate whether filter "maturation" took place under these conditions: if filtration performance improved over weeks or months, it would provide circumstantial evidence for biological and/or chemical changes on the media grain surfaces.

Head loss development rates could not be compared between filter runs due to the variations in influent particle loadings and composition.

4.3.1 BACKWASH PROCEDURE AND APPARATUS

Backwash was carried out under computer control from one of two head loss triggers. In the first months of pilot plant trials the level of water in the filter rose during a filter run to accommodate

increasing head loss. When a float-switch was reached the backwash was initiated. This was at head losses of 1.1 - 1.4 m. From October 1989 the backwash commenced at a pre-set differential pressure value. The details are in Tables A2.1, A2.2 and A2.3 in the appendix.

The computer controlled the sequence and duration of valve operations necessary to close the inlet and outlet valves to the filter, drain down the supernatant water, introduce air scour, introduce mains water, drain the dirty water, refill the filter and restart filtration. A time delay of 2 hours between consecutive washes on the same filter was set into the program.

Backwash water was taken from a mains water supply. A pressure reducing valve kept the supply pressure to 2 bar. The water contained a low residual of chlorine (< 0.5 mg.l^{-1}).

Initially, the supernatant was not drained through the bed in case a failure to drain to a low enough level caused media to be washed away. Stored water above the filter was allowed to pass out through the dirty water washout valves (V10 and V11 figure 3.2) prior to commencing the wash. The backwash was modified to drain the above bed water down through bed in columns 5 and 6 from July 1992 onwards.

From 1991 the wash cycle for column 1 included time after the inlet flow restarted to allow incoming stored water to pass straight out of the drain down valves to waste. This was to eliminate iron and ozone that had built up in water standing in the inlet pipework and ozone contactor during the wash.

SEPARATE AIR SCOUR AND WATER WASH

In the first year of trials the wash consisted of:

closing the inflow and outflow valves;

drain down through the backwash waste water outlet valves to approximately 300 mm above the media surface;

3 minutes of air scour at 30 m.h^{-1} ;

5 minutes fluidising wash water to achieve a bed expansion of between 20% and 30%. Filtration commenced immediately inlet and outlet valves opened and the water level in the column gave sufficient hydrostatic head.

COMBINED AIR SCOUR AND WATER WASH

From February 1990 the wash was changed to:

a shorter air only phase (15 seconds);

a combined air scour and low rate water (<10 m.h^{-1}) phase, lasting 5 minutes;

a final high rate fluidising rinse for 7.5 minutes, set to achieve 20 - 30% media expansion;

the inlet valve opened, followed by the closure of the washout valve and the opening of the filtrate valve;

where iron or ozone were dosed, chemicals which accumulated in the inlet pipework were flushed from the filter by leaving the washout valve open for 3 minutes.

To prevent media loss there was no discharge of dirty water out of the filter during the combined phase. Because the filters accommodated different media depths if the expanded bed position was close to the lower wash out valve (V10) this remained closed during the rinse to prevent media loss.

For comparison, column 6 had only 6.5 minutes of high rate water. Figure 4.5 indicates that this does not appear to have affected starting head loss.

AIR SCOUR

Air scour was provided by a compressor with reservoir, pressure reducing valve and flowstat arrangement. Pressure in the reservoir was set at 2 bar, but when the air was flowing during the air scour phase pressure was found to be leaving the reservoir at 1.5 bar. Flow rate was set to 120 l.min^{-1} STP scale (standard temperature and pressure, i.e. 273 K, 760 mm Hg, Kneen *et al.*, 1972). Using the short formula in the flow meter manufacturer's catalogue (this excludes temperature effects)

$$Actual\ flow\ (free\ air) = scale\ reading\ (STP) * \sqrt{1 + P_E}$$

where P_E = air pressure (bar), results in a flow of 120 l.min^{-1} at 1.5 bar becoming 190 l.min^{-1} (43 m.h^{-1}) as free air (47 m.h^{-1} if pressure remained at 2 bar). By Charles's law the volume of air would be 9% larger at 25°C than 0°C (at constant pressure the coefficient of expansion for all gases is 0.003663 per °C).

The rate of air used in these trials was higher than typically used in the UK and lower than used in the US for separate air and water washing (Cleasby, 1990) but within the range suggested by Amirtharajah *et al.* (1991a) for CASBW.

HIGH RATE BACKWASH

High backwash rates were set up on the pilot plant to give a bed expansion of between 20 and 30% of the total depth of media, excluding gravels. The bed expansion varied within the range due to seasonal water temperature variations. Expansions were measured directly on the columns with windows. Where no window was available the bed expansions were estimated by briefly opening the lower (V10) and upper (V11) backwash outlet valves to check for media during fluidisation at various rates.

An electronic paddle-wheel flow meter measured backwash rate. This was calibrated using water rise rate measurements. The high rate backwash water rates used were set as shown in table 4.1. The meter reading x 0.175 gave the backwash rate in m.h^{-1} . The high water flow rate from the last backwash was displayed for each filter on the computer screen; these were available to check whether the wash had been achieved. The data showed there were no problems with the availability of the backwash water.

LOW RATE BACKWASH

DETERMINATION OF V_{mf}

A laboratory fluidisation test apparatus was used to determine the minimum fluidisation velocity (V_{mf}) for each of the filter media types used. The anthracite was soaked in water for 4 h to ensure

wetting. Repeat measurements after 72 h gave the same V_{mf} values. The V_{mf} of 6-14 sand was not measured as the apparatus could not receive sufficient water flow to fluidise this grade of sand.

The V_{mf} selected for the trials was based on the value for the sand at 14°C, since this was the median stored water temperature. In setting the backwash water rate for dual media filters it was decided that keeping the finer sand layer clean was likely to be more important that the coarser anthracite layer in preventing increases to starting head losses.

The low rate water pipe work and valves were installed in February 1990. The flow meter reading x 0.024 gave the water flow rate in $m.h^{-1}$. Flows were set to be in the range 25 - 50% of V_{mf}. This range was reported by Amirtharajah (1984) as promoting collapse-pulsing through the depth of the bed.

4.3.2 POC AND SOLIDS REMAINING ON FILTER MEDIA

Samples of filter sand were taken for analysis of accumulated deposits when the trial with that media configuration terminated, and in some cases during its operating period. The filter was backwashed and drained and then dismantled. The condition of the surface layers was noted, especially the presence or absence of "mudballs" (aggregates of foreign matter). For sand filters 1 kg samples were placed into separate plastic bags from four depths in the filter: the surface 0 - 25 mm, and from filter depths of 100 - 120 mm, 250 - 300 mm and 500 - 600 mm. As the filter was dug out the presence or absence of mudballs continued to be noted. For dual media filters the anthracite layer was removed and its condition described, allowing the sand to be sampled as above. The 500 mm sand sample could not be taken when the sand layer was only 340 mm deep. Filters 4, 5 and 6 were designed to enable direct access to beds of 1.2 m depth or less. Columns 1, 2 and 3 required sampling using an improvised scoop, with the media being removed by vacuum cleaner. The media samples were either taken directly to the laboratory or frozen to reduce changes to the organic material.

The analytical procedure involved measuring a volume of sand into a 250 ml stoppered measuring cylinder, adding water, noting the total volume of sand and water, and shaking it vigorously for 1 minute to liberate attached material from the sand. The supernatant was sampled and filtered to determine dried suspended solids (105°C) and POC. A trapezoidal integration of the concentrations of deposits liberated from the sand from the four depths was used to calculate the mass of material in the filter sand (Bayley, pers. comm., 1989). Every effort was made to ensure as little anthracite as possible was left with the sand samples from the dual media filters, but 100% separation was not possible.

4.3.3 CALCULATION OF STARTING HEAD LOSS

The data for starting head loss were taken from on-line monitoring of filter head loss and filter flow rate. The database comprising logged data from each filter contained approximately 130,000 records, collected at 15 minute intervals[1] for four years. The following method was used to calculate flow and temperature normalised starting head loss for each type of filter media used:

records were deleted from the database for filter runs where it was clear that the logger had failed and filter runs were incomplete or there was uncertainty that backwashes had been carried out properly;

the first four records of positive flow and head loss values (i.e. not during the wash), were averaged from each filter run after the backwash counter increased. Some scatter in early results (up to 5.10.90, week 91) was due to the backwash counter increasing at the start of the backwash. This means that a high head loss value from the end of the previous run could have been included in the average. The software was altered to avoid this after week 91. Thereafter a change in counter indicated a successful backwash;

the average recorded flow value was used to normalise the average head loss value to a standard flow rate of 10 $m.h^{-1}$;

all the flow normalised starting head losses for a given week were averaged. This was to smooth out variations in the starting head loss due to the timing of the sampling events relative to the end of the backwash;

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[1] From 14:46 on 29/9/92 to 17:51 on 4/11/92 (weeks 196-201) data were logged every 5 minutes.

the weekly flow normalised starting head losses were normalised with respect to water temperature by using a weekly mean of daily temperature measurements. The recorded head loss was adjusted using the water viscosity at the measured temperature and normalising to the water viscosity at 15°C;

the mean weekly flow and temperature normalised starting head loss data were plotted against the week number. Week 1 was the start of January 1989.

The flow data used up to 25.5.89 to calculate the normalised head loss were all 8 $m.h^{-1}$, which was the nominal flow rate in all the filters from the time logging commenced till this date, after which the on-line flow meters were used.

Using weekly average temperature values and weekly averages of individual flow normalised starting head losses, the flow normalising calculation took the form:

$$H_n = H_{QT}.\frac{Q_n}{Q}.\frac{\mu_n}{\mu}$$

where H_n = normalised head loss (m)
H_{QT} = measured head loss at flow rate Q ($m.h^{-1}$) and temperature T (°C)
Q_n = flow rate used as standard for normalisation (10 $m.h^{-1}$)
Q = measured flow rate ($m.h^{-1}$)
μ_n = absolute viscosity of water at 15°C (1.145×10^{-3} $kg.m^{-1}.s^{-1}$)
μ = absolute viscosity of water at the mean weekly recorded water temperature ($kg.m^{-1}.s^{-1}$)

4.3.4 FILTER MATURATION

To examine filter maturation daily stored water and filtrate turbidity sample data were condensed into weekly average turbidity removal values to see whether there was a maturation effect taking place i.e. a long term change in filter performance. Each point comprised between 1 and 5 daily samples. To eliminate the effect of chemical dosing on turbidity removals, only data from filters where no coagulant or ozone was dosed have been used. The sample time was effectively random with respect to the filter cycle, and since there were no coagulants used, there is a reasonable probability that these data represented ripened filter performance.

4.4 RESULTS

4.4.1 MEDIA MANAGEMENT

ELIMINATION OF AIR FROM THE MEDIA

In order to remove air from the plenum chamber and filter bed and allow media grains in the dirty water to settle back into the expanded bed, the two dirty water outlet valves were not opened until 90 seconds after the high rate water started. It took one minute to eliminate the air from the bed and plenum chamber. A further 30 seconds was allowed for the anthracite grains to settle, before opening the valves as the water level was higher than the outlet. A full scale installation with wash out launders or a side weir would need only the one minute period before water reached the outlet.

MEDIA STRATIFICATION

A 150 mm internal diameter clear PVC filter was used to observe the behaviour of sand and anthracite layers during the backwash of a dual media filter containing 600 mm depths of 0.6 - 1.18 mm sand and 1.7 - 2.5 mm anthracite. It was found that considerable mixing of sand and anthracite took place during combined air and water washing. A fluidising water rate of 45 - 50 $m.h^{-1}$ was found to be the optimum rate to ensure a rapid and thorough separation of the layers. This was completed within 3 minutes at 7°C.

SELECTION OF FLUIDISING WATER FLOW RATES

Table 4.1 shows the rates of backwash water selected. These ensured a bed expansion ranging between 20 and 30% at the range of stored water temperatures encountered (4 to 23°C). Details of media grades were given in table 3.2. Media depths are given in the appendix. The increase in flow rate for column 4 14-25 sand in 1991 was due to changing from estimating the bed expansion to measuring it following the installation of a new filter column section with a window. The flow rates were checked weekly.

Table 4.1. *Fluidising backwash water flow rates giving 20-30% bed expansion at water temperatures between 4 - 23°C.*

Column	Year	Media	Backwash water flow rate ($m.h^{-1}$)
1	1989 -90	F400 GAC	12-18
1	1990 -93	Dual media: Grade 2CC/ 14-25 sand	42-51
2	1989	10-18 sand	68-70
2	1989 -90	F400 GAC	10-20
2	1990	Deep coarse bed: Grade 2CC	46-47
2	1991 -92	Deep coarse bed: 6-14 sand	83
3	1989	Dual media: Grade 2/ 14-25 sand	38-45
3	1990	Dual media: Grade 2/ 14-25 sand	45-47
3	1991 -92	10-18 sand	65
4	1989 -90	14-25 sand	49-52
4	1990	14-25 sand	46-48
4	1991 -93	14-25 sand	61-64
5	1989 -90	Dual media: Grade 2/ 16-30 sand	33
5	1990	Dual media: Grade 2CC/ 14-25 sand	39-42
5	1991	Dual media: Grade 3/ 10-18 sand	69
5	1991 -93	Dual media: Grade 2CC/ 14-25 sand	47
6	1989 -90	Dual media: Grade 2/ 16-30 sand	33-37
6	1990	Dual media: Grade 2CC/ 14-25 sand	39-44
6	1991	Deep coarse bed: Grade 3	77
6	1991 -93	Dual media: Grade 2/ 14-25 sand	43

LOW RATE BACKWASH

The measurement of V_{mf} produced the data given in table 4.2, some examples are plotted in figure 4.1. Determining the V_{mf} for the angular anthracite was complicated by the continuing increase in head loss beyond the V_{mf} in order to separate interlocking media grains. The data from weekly checks of low rate water flows are shown in table 4.3. Following the publication of design data for collapse-pulsing by Amirtharajah *et al.* (1991) the low flow rates were adjusted in February 1992 to 9.6 $m.h^{-1}$ which was within 40 - 60% of V_{mf} recommended for air at 47 $m.h^{-1}$. The flow rates are expressed as a percentage of V_{mf} for each media in tables 4.3 and 4.4.

Table 4.2. *Experimentally determined values of minimum fluidisation velocity (V_{mf}).*

Media	Water temperature (°C)	V_{mf} (m. h^{-1})
0.5 - 1.0 mm sand (BSS 16-30)	14	16
0.6 - 1.18 mm sand (BSS 14-25)	14	19
0.85 - 1.7 mm sand (BSS 10-18)	14	22
1.2 - 2.5 mm anthracite (Grade 2)	7	13
	14	16
1.7 - 2.5 mm anthracite	7	16
(Grade 2CC coarse cut)	14	23
	23	24
2.5 - 4. 0 mm anthracite (Grade 3)	7	40

The weekly flow rate checks carried out through 1990 to March 1991 had shown consistent flow rates and the measurements were checked infrequently thereafter. A check of flow rates in February 1992 (data in table 4.4) revealed that the rates for column 4 and 6 had changed from their original settings. Column 2 had been reset using an estimated value for coarser media (6-14 sand). Carbonate deposits caused a solenoid valve failure (V9, figure 3.2) on column 1, resulting in only 1.1 $m.h^{-1}$ of low rate water passing through the filter. The valve was serviced on 2/3/92 and a resulting improvement to 6.3 $m.h^{-1}$ flow rate was achieved (33% of V_{mf} sand, 39% of V_{mf} anthracite). This problem was unique to column 1 as the others had an electrically actuated ball valve as V9.

Table 4.3. *Low rate backwash water flow rates 1990-1991. Flow rate as $\%V_{mf}$ refers to the media in the filter or the sand in dual media filters.*

Column	Time period	Media	Flow range ($m.h^{-1}$)	Flow rate as % V_{mf}
1	08/90 - 05/91	Dual media: Grade 2CC/14-25	6.4-7.4	34-39
2	08/90 - 10/90	Deep coarse bed: Grade 2CC	6.7-7.5	42-47
3	08/90 - 02/91	Dual media: Grade 2/14-24	7.0-7.7	37-41
		10-18 sand		32-35
4	08/90 - 09/91	14-25 sand	6.7-7.7	35-41
5	08/90 - 03/91	Dual media: Grade 2CC/14-25	6.7-7.2	35-38
		Dual media: Grade 3/10-18		30-33
6	08/90 - 04/91	Dual media 2CC/14-25	6.7-7.8	35-41
		Deep bed Grade 3		17-20

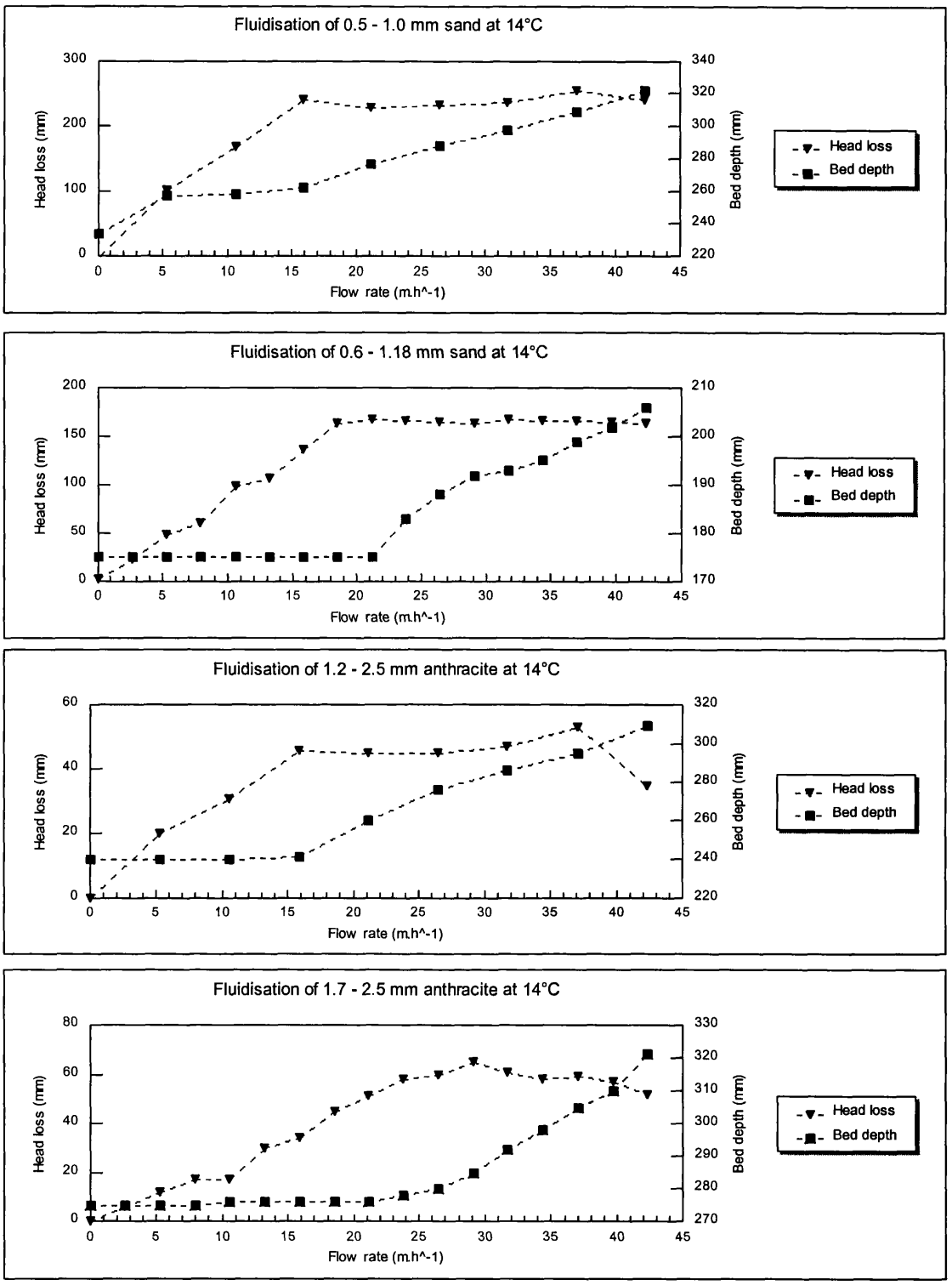

Figure 4.1. *Experimental data showing how head loss and bed depth changed with increasing water flow, from which the V_{mf} was estimated.*

Table 4.4. *Low rate backwash water check, 6th February 1992. After this check all low rate flows were reset to 9.6 m.h^{-1}. Flow rate as %V_{mf} refers to the media in the filter or the sand in dual media filters*

Column	Media	Flow (m.h^{-1})	Flow rate %V_{mf}	Flow (m.h^{-1})	Flow rate %V_{mf}
1	Dual media: 2CC/14-25	1.1 (*)	6	9.6	51
2	6-14 sand	16.8	n/a	9.6	n/a
3	10-18 sand	8.1	36	9.6	44
4	14-25 sand	14.8	78	9.6	51
5	Dual media: 2CC/14-25	7.2	38	9.6	51
6	Dual media: 2/14-25	4.1	22	9.6	51

Note: the value for column 1 marked (*) had the rate of flow passing through the outflow meter deducted, showing the effect of the solenoid valve failing to shut, due to carbonate deposition.

BACKWASH WATER USE AS A PROPORTION OF THROUGHPUT

In order to check that the volumes of water used in backwashing the pilot plant filters would be acceptable to plant designers and operators the volumes of water were considered as a percentage of the daily filter production. The examples in table 4.5 have been calculated for each column assuming productivities based on a filter area of 0.265 m^2 and: one 23.5 hour filter run at 10 m.h^{-1}, with 30 minutes for backwashing (62.3 m^3.day^{-1}), and three 7.5 hour runs at 15 m.h^{-1}, with 1.5 hours out for washing (89.4 m^3.day^{-1}). Note that column 6 had a shorter high rate wash duration. Not all the volume of backwash water that enters the bed is lost during a backwash. Some water is left between the filter floor and the discharge weir. Table 4.5 shows the amount of water that was put into the bed to achieve cleaning, with the volume left in the bed subtracted. The calculation assumes 1.1 m depth of media at a porosity of 0.4, and 0.4 m freeboard. For the pilot filter area of 0.265 m^2 this gave a volume of 0.22 m^3 to be subtracted. The volume of backwash water as a proportion of daily throughput, assuming constant rate filtration, was calculated using the formula:

$$\%_{throughput} = \frac{(V_w - V_l) * n_w}{Q * (24 - (t_w * n_w)) * A} * 100$$

Where:

V_w = Volume of water used in a wash (m^3)
V_l = Volume of wash water left in bed at the end of a wash = 0.22 m^3
n_w = number of washes per day
Q = flow rate m.h^{-1}
t_w = time a filter is out of service whilst washing (0.5 hours)
A = area of filter = 0.265 m^2

Table 4.5. *Backwash water utilisation, as a proportion of throughput, for the filters operated from 1992-3. Examples worked assuming one wash per day for a filter operating at 10 $m.h^{-1}$ and three washes per day for a filter operating at 15 $m.h^{-1}$.*

Column	High rate water flow rate, duration and volume			Low rate water flow rate, duration and volume			Total wash volume	% daily throughput used in washing	
Media	flow rate ($m.h^{-1}$)	duration (mins)	water volume (m^3)	flow rate ($m.h^{-1}$)	duration (mins)	water volume (m^3)	V_w (m^3)	1 wash [10 $m.h^{-1}$]	3 washes [15 $m.h^{-1}$]
1 Dual media	48	7.5	1.6	9.6	5	0.2	1.8	2.5 %	5.3 %
2 6-14 sand	83	7.5	2.8	9.6	5	0.2	3	4.5 %	9.3 %
3 10-18 sand	65	7.5	2.1	9.6	5	0.2	2.3	3.3 %	7.0 %
4 14-25 sand	61	7.5	2.0	9.6	5	0.2	2.2	3.2 %	6.6 %
5 Dual media	47	7.5	1.5	9.6	5	0.2	1.7	2.4 %	5.0 %
6 Dual media	43	6.5	1.2	9.6	5	0.2	1.4	1.9 %	4.0 %

Note: column 1 dual media = 0.66 m of 1.7-2.5 mm anthracite over 0.34 m of 14-25 sand;
column 5 dual media = 0.60 m of 1.7-2.5 mm anthracite over 0.60 m of 14-25 sand;
column 6 dual media = 0.60 m of 1.7-2.5 mm anthracite over 0.60 m of 14-25 sand.

4.4.2 POC AND SUSPENDED SOLIDS ON FILTER MEDIA

Tables 4.6 and 4.7 present results of sand samples analysed for POC and suspended solids. The values presented have been normalised to a sand layer depth of 650 mm to permit comparison with the Thames Water standards of less than 250 $g.m^{-2}$ of POC and 1500 $g.m^{-2}$ of suspended solids derived empirically from full scale backwashing trials (Chipps *et al.*, 1995).

The number of months that the media was in service is recorded in the column marked "Operational period". When the same filter media was sampled more than once tables 4.6 and 4.7 show a change in operating period but the media column is left blank. The value of 29 (17) for the operating period of the 14-25 sand in column 4 represents the media where the bottom 600 mm had been in service for 29 months but the top 300 mm had been added after 1 year, resulting in an operational period of 17 months. Where values for POC or suspended solids are shown in parentheses a sample was missing, either due to a lost sample or the media not being deep enough, and a value from the sample above, or the average of the sample above and below, has been used in the integration calculation of media condition.

SEPARATE AIR AND WATER BACKWASHING

In January 1990 the first set of trials was concluded and the media examined. The results of both tests showed for sand in 1989 that the backwashing had been ineffective. It may be seen in table 4.6 that with separate air and water backwashing all three results for sand-only filters failed the targets. Visual inspection showed that the fluidising wash had kept the finest media grains continually exposed to the dirtiest influent, and inspection of the top 100 mm of the 14-25 media after 10 months in operation showed it to have changed from a golden colour to a dark brown, and was so covered in clay-like material it could be formed into balls by hand pressure.

In the dual media filters the sand under the shallow anthracite layer in column 3 failed the POC test but was within the suspended solids limit. The shallow layer of sand in columns 5 and 6 lying underneath 660 mm of anthracite passed both targets, although only marginally in the case of POC. The deeper anthracite layer in columns 5 and 6 appears partly to have protected the sand but it was getting dirty. The suspended solids values for column 5 and 6 were quite different, whereas the POC values were virtually identical. It should be noted that these values were under-estimates as the missing samples were given values of 0. Two samples were lost due to problems in the laboratory and could not be replaced.

One test was carried out with sand from column 3 where approximately 10% of anthracite was present (judged by observation) and with a sample where an effort was made to remove the anthracite using tweezers. The results shown in table 4.6 suggest that the anthracite did not increase the result by more than 10%.

Table 4.6. *POC and suspended solids residues on filter sand washed with separate air scour and fluidising water rinse. POC data: g.m^{-2}, target < 250 g.m^{-2}. Suspended solids data: g.m^{-2}, target < 1500 g.m^{-2}.*

Column	Media (BSS)	Operational period (months)	POC (g.m^{-2})	Suspended solids (g.m^{-2})
2	10-18	4	427	2286
3	14-25 [1]	11	409 [A]	1267 [A]
			425 [B]	1388 [B]
4	14-25	7	989	3773
		10	877	2371
5	16-30 [2,3]	10	245 [C]	893 [C]
6	16-30 [2]	10	239 [D]	1339 [D]

Notes: [1] 660 mm sand under 150 mm anthracite
[2] 340 mm sand under 660 mm anthracite
[3] Iron dosed
[A] Anthracite removed as far as possible
[B] Approximately 10% anthracite in sand sample
[C] 0-25 mm and 250-300 mm samples only, only 340 mm sand in filter; value integrated to 650 mm using 0 for missing values
[D] 0-25 mm and 100-150 mm samples only, only 340 mm sand in filter; value integrated to 650 mm using 0 for missing values

BACKWASHING INCORPORATING A COMBINED AIR AND WATER WASH

The results in table 4.7 show the POC and suspended solids results from sand sampled out of filters using the enhanced backwash. In several cases the media had been in operation considerably longer than in table 4.6. The filter sands were kept to below the target values in the majority of trials sampled. This was an improvement on the first year's data. Only two samples failed the targets (three if the extrapolated values for column 1 at 19 months were used) and the rest passed. The poor samples were from column 1, taken in April 1993 after 38 months operation, and column 6 dated September 1990. The suspended solids data for column 4 dated May 1990 seemed to be high, but were much lower on two subsequent samples.

Table 4.7. *POC and suspended solids residues on filter sand washed with combined air scour and sub-fluidising water wash with subsequent fluidising water rinse. POC data: g.m^{-2}, target < 250 g.m^{-2}. Suspended Solids data: g.m^{-2}, target < 1500 g.m^{-2}.*

Column	Media (BSS)	Sample date (month/year)	Operational period (months)	POC (g.m^{-2})	Suspended solids (g.m^{-2})
1	14-25 [2,3]	09/91	19	228 [G] (466)	650 [G] (1300)
		04/93	38	651 [G] (1056)	4247 [G] (5877)
2	6-14	05/92	18	75	541
3	14-25 [4]	11/90	8	119	332
3	10-18	05/92	18	143	599
3	14-25 [3,5]	04/93	8	0	113 [G] (218)
4	14-25	05/90	3	42	1076
		09/90	7	106	322
		11/90	8	136	516
4	14-25	04/93	29 (17)	64	333
5	14-25 [3,4]	06/90	4	9 [E] (12)	61 [E] (88)
		10/90	7	88	330
		11/90	8	92	333
5	10-18 [2,3]	08/91	9	13 [G] (25)	56 [G] (129)
5	14-25 [3,6]	04/93	19	131	581
6	14-25 [4]	06/90	4	184	769
		09/90	7	510	1598
		11/90	8	230 [F] (347)	720 [F] (1090)
6	14-25 [3,6]	04/93	19	275	690

Notes: [2] 340 mm sand under 660 mm anthracite [3] Iron dosed [4] 660 mm sand under 340 mm anthracite

[5] 400 mm sand under 400 mm anthracite

[6] 600 mm sand under 600 mm anthracite

[E] 500-600 mm sample missing, calculation assumed these were 0; the values given in parentheses used the value of the 250-300 mm sample for the 500-600 mm sample in the calculation

[F] 250-300 mm sample missing, calculation assumed these were 0; the values given in parentheses used the average of the values from the 100-150 mm sample and the 500-600 mm sample for the 250-300 mm sample in the calculation

[G] 500-600 mm sample missing, calculation assumed these were 0 (only 340 mm sand in filter); the values given in parentheses used the value of the 250-300 mm sample for the 500-600 mm sample in the calculation

4.4.3 MEASUREMENTS OF FILTER STARTING HEAD LOSS

The results of weekly averages of flow and temperature normalised starting head loss are plotted in figures 4.2 to 4.8 below. The x-axes are scaled in weeks from 1.1.89 to relate the scale to the time

of year. The trials started in week 6, and data logging commenced in week 9. These graphs illustrate the differences that backwash method had on normalised head loss at the start of the filter run.

The data from the trials with separate air scour and fluidising water rinse were collected up to week 54. These are plotted in figures 4.2-4.5. These data reveal that the normalised starting head loss increased rapidly over the first 3-4 months of operation to a value some 2-3 times that of new media. It appeared that the normalised starting head loss values reduced (column 1 and column 5), or stopped increasing (column 3 and column 6), after a peak around week 40.

Results from the trials using a combined air and water wash under collapse-pulse conditions between the air scour and fluidising water wash stages are plotted in figures 4.2-4.8. There was some scatter in the data after week 54, particularly up to week 91 when a high head loss value from the end of the previous run was included in some of the calculations. Despite this, the general tendency was that the starting head loss showed a small initial increase then maintained a value slightly higher than that of new media, but not a rising trend. This is consistent with biological and/or chemical maturation of filter media.

The data for 14-25 sand (column 4), shown in figure 4.2, are quite "noisy", explained in part by the changes to media and media depth, and probably an occasional failed backwash if it took place outside working hours and went unrecorded. However there is one period that needs further explanation. A sudden decrease in starting head loss coincided with placing 300 mm of fresh sand onto 600 mm of sand that had been used for 11 months. It had increased around week 133 due to a decision to re-calibrate the differential pressure transmitter. Whether this was the correct decision is an open question but the significant point is that there was consistency in head loss values before and after the re-calibration, suggesting the backwash was performing satisfactorily. The head loss returned to pre-July 1991 values briefly before the media was topped up; the site records do not indicate whether a further calibration took place at this time.

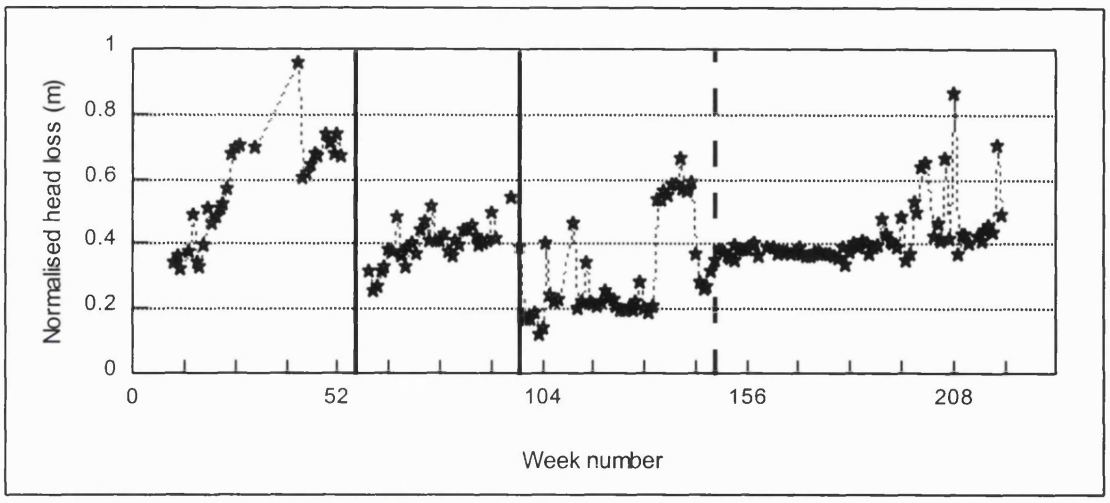

Figure 4.2. *Weekly averaged flow and temperature normalised starting head loss results for BSS 14-25 sand (column 4). Media 1.0 m deep, supplied new in week 4 and week 56. New media added to 0.6 m depth in week 97 and topped up to 0.9 m week 146. Replacement of media is denoted by the solid vertical line, the topping up by the dashed vertical line. Flow rates varied from 5 - 18 m.h^{-1}.*

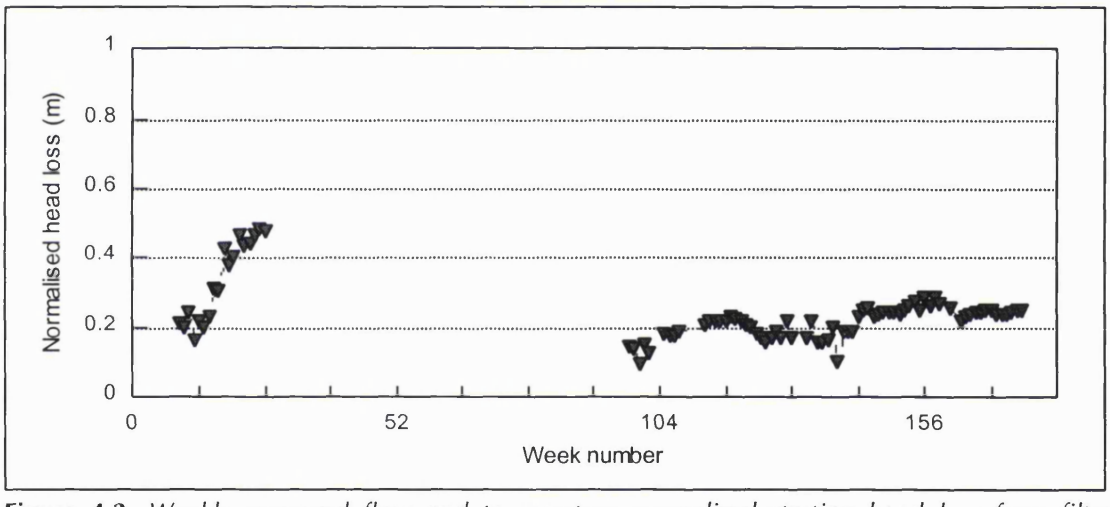

Figure 4.3. *Weekly averaged flow and temperature normalised starting head loss from filters containing BSS 10-18 sand. The data from 1989 were from a 1.0 m depth washed with separate air and water, at filtration rates from 8 - 12 m.h^{-1} (column 2). The results from week 98 onwards were from a 0.9 m deep filter of new sand, treating water flows from 5 - 20 m.h^{-1} (column 3).*

With the 10-18 sand (see figure 4.3) there was a problem with inadequate washing which was spotted when it was realised that column 3 had started to produce better turbidity removals than column 4 (14-25 sand of same depth - 0.9 m) - see figures 4.9 and 4.10. The rise in starting head loss in the weeks at the end of 1991 (up to week 156) was caused by poor backwashing due to the loss of the low rate water through a ball valve which failed to shut during backwashing. The fault was rectified in week 157 and the head loss trend stopped rising.

The results for dual media filters are plotted in figures 4.4 to 4.6. The ineffectiveness of the separate air and water wash is apparent as is the benefit of the additional combined phase. The fault in the low rate water wash on column 1, discovered and corrected in week 162, does not appear to have had a significant impact on normalised starting head loss. It is not clear from the data when this problem started or if it recurred. There was a slight rise and fall in starting head loss between weeks 156 and 169 which might have been due to the failure and correction of the low rate water wash, but further corroboration of this is not available and would be required to state confidently that this was so.

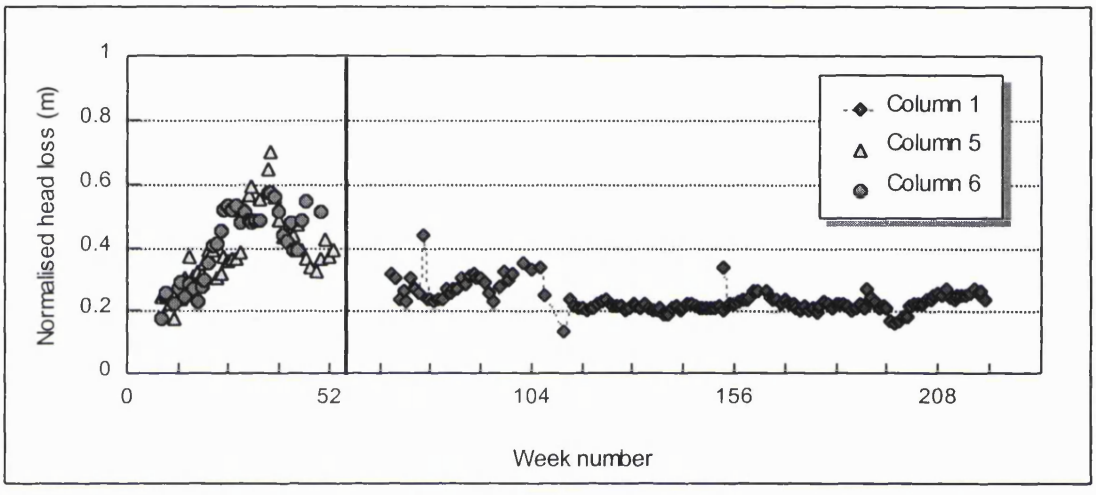

Figure 4.4. *Flow and temperature normalised starting head loss data for two dual media filters comprising 0.34 m sand under 0.66 m anthracite. Columns 5 and 6 used Grade 2 anthracite and BSS 16-30 sand, column 1 used Grade 2CC anthracite and BSS 14-25 sand. Separate air and water backwash used up to week 56. Wash modified to include collapse pulse backwash after week 56.*

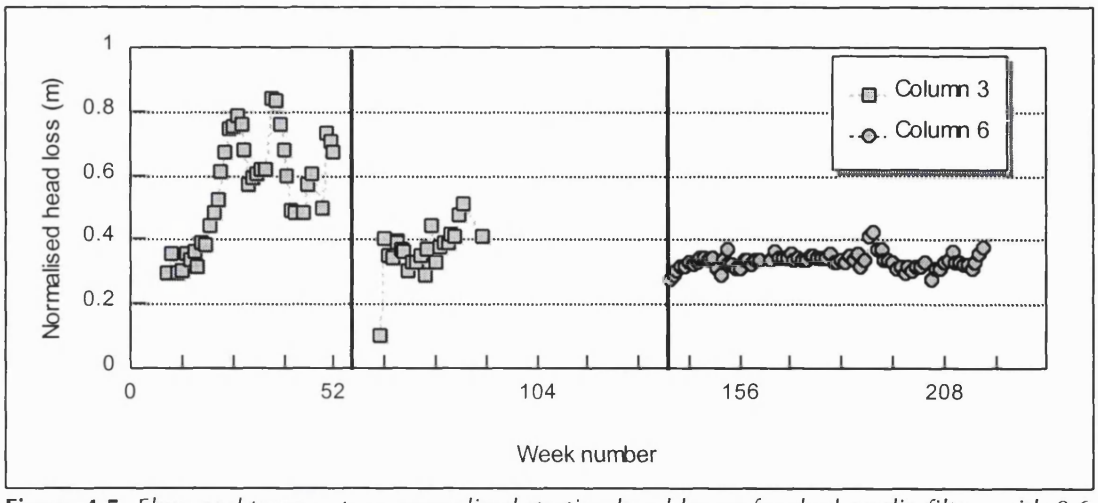

Figure 4.5. *Flow and temperature normalised starting head losses for dual media filters with 0.6 m BSS 14-25 sand. Column 3 had 0.34 m of Grade 2 anthracite. Column 6 had 0.6 m of Grade 2 anthracite from week 140. Changes in media are denoted by the black vertical lines.*

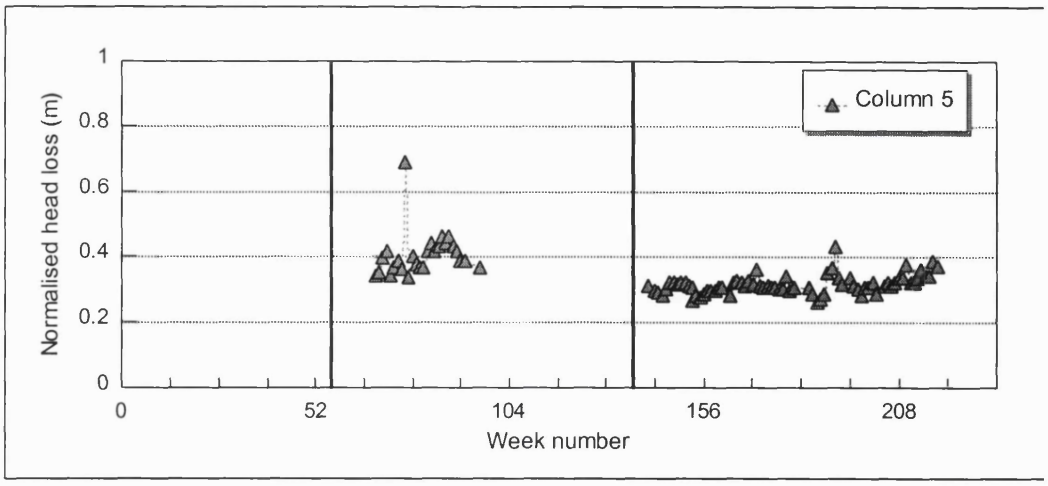

Figure 4.6. *Flow and temperature normalised starting head losses for dual media filters with 0.6 m BSS 14-25 sand. Column 5 had 0.34 m of Grade 2CC anthracite up to week 100 and 0.6 m of Grade 2 CC anthracite from week 140. Changes in media are denoted by the black vertical lines.*

Some trials were carried out with 1.8 - 2.0 m deep beds of coarse single media, and one trial had a coarse dual media 0.66 m grade 3 anthracite over 0.34 m 10-18 sand. Figures 4.7 and 4.8 show that it was possible to maintain a consistent starting head loss with the use of the collapse-pulse wash and a fluidising rinse. In figure 4.7 the scatter in the results for the coarse anthracite was probably due to faults with pressure transducer wiring generating false values, together with the timing problem with the backwash counter increasing before the wash had finished.

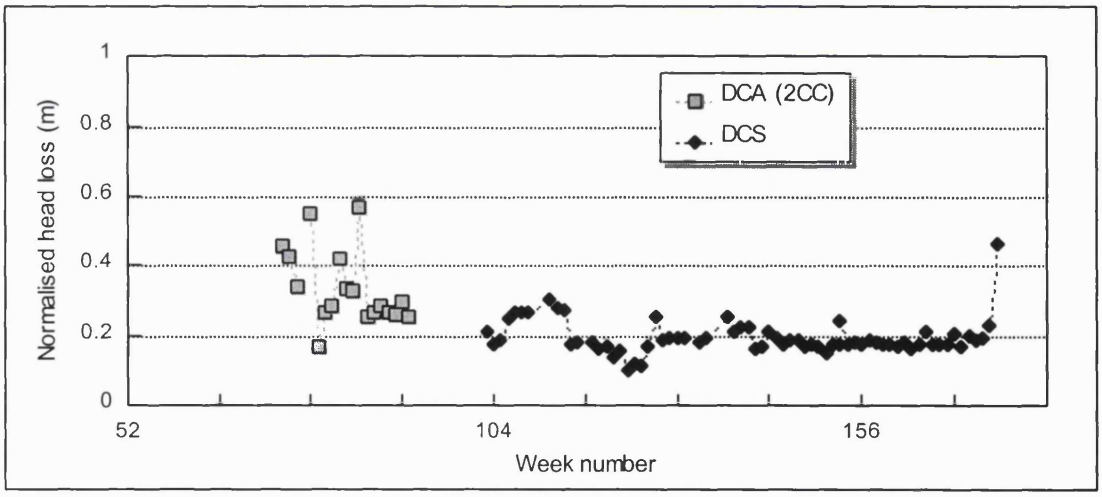

Figure 4.7. *Flow and temperature normalised starting head losses for deep coarse media beds. The data marked DCA (2CC) were from a 2 m deep bed of 1.7-2.5 mm anthracite, and the data labelled DCS from a 1.8 m deep bed of BSS 6-14 sand.*

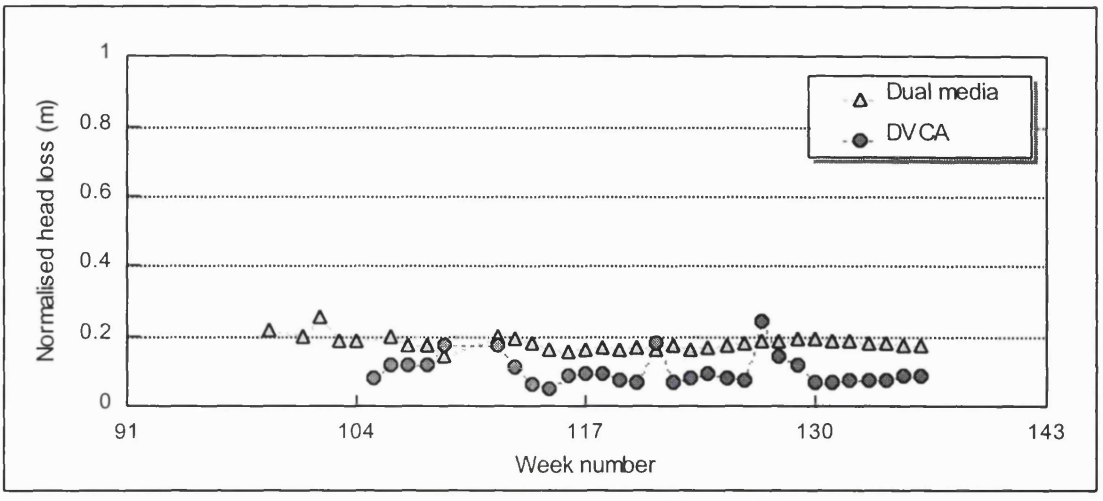

Figure 4.8. *Flow and temperature normalised starting head losses for deep coarse media beds. The data marked dual media were from a 1 m deep bed of 0.66 m 2.5-4.0 mm anthracite over 0.34 m of BSS 10-18 sand, and the data labelled DVCA from a 2 m deep bed of 2.5-4.0 mm anthracite.*

The scatter seen in the normalised starting head loss data may be explained by the following reasons:

logging intervals - generally 15 minutes, average head loss values would tend to be lower for more frequent samples (weeks 196-201) as they would be taken closer to the start of the run;

rate of head loss growth - if head loss increased quickly it would tend to raise the average value of the first four data points;

numbers of filter runs per week - these varied from 1 to over 20 runs depending on the water quality at its impact on clogging the filter: the effect of this is complicated; with many runs the average will not tend to be unduly distorted by one misleading filter run's data unlike where there was only a limited number of filter runs;

timing of data logging moment in relation to the end of the backwash.

4.4.4 FILTER MATURATION

Column 4 with 14-25 sand was operated with two changes of media and one topping-up so should give one example of maturation in a poorly backwashed state and two examples of maturation under good backwashing conditions. These data are shown in figure 4.9 with removal plotted as $1-(C/C_0)$ where C = filtrate turbidity and C_0 = influent turbidity. The data from the filters containing 10-18 sand (columns 2 and 3) are presented in figure 4.10 to show the poor backwashing data

(column 2, up to week 54) and the good backwashing data, plus a period of poor backwashing, caused by a valve failure, within the good backwashing period (column 3, after week 54).

Because the media was often changed at the beginning of the year, it was hypothesised that removals were always low during the winter months. Evidence that this was not the case and that maturation was occurring was provided by comparing the turbidity removals achieved by new sand around week 104 with the filter performance of 1 year old media, around week 156, and 2 year old media around week 208 (see figures 4.9 and 4.10). If no maturation had taken place it would have been expected that the removals of turbidity would have been similar each winter.

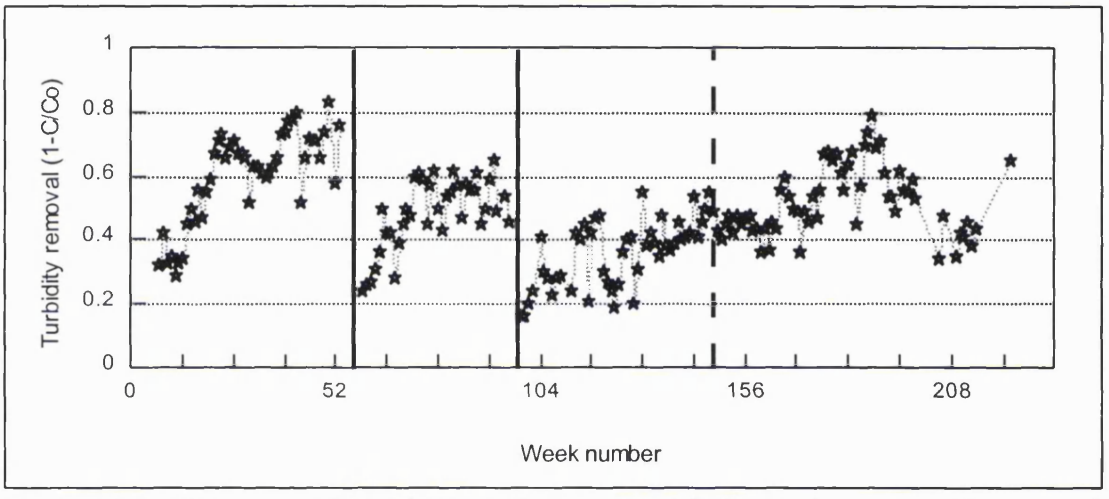

Figure 4.9. *Turbidity removal based on weekly averages of daily stored water and filtrate samples from a BSS 14-25 sand filter. The continuous black lines show when media was replaced, the dashed line indicates when media was topped up from 600 to 900 mm.*

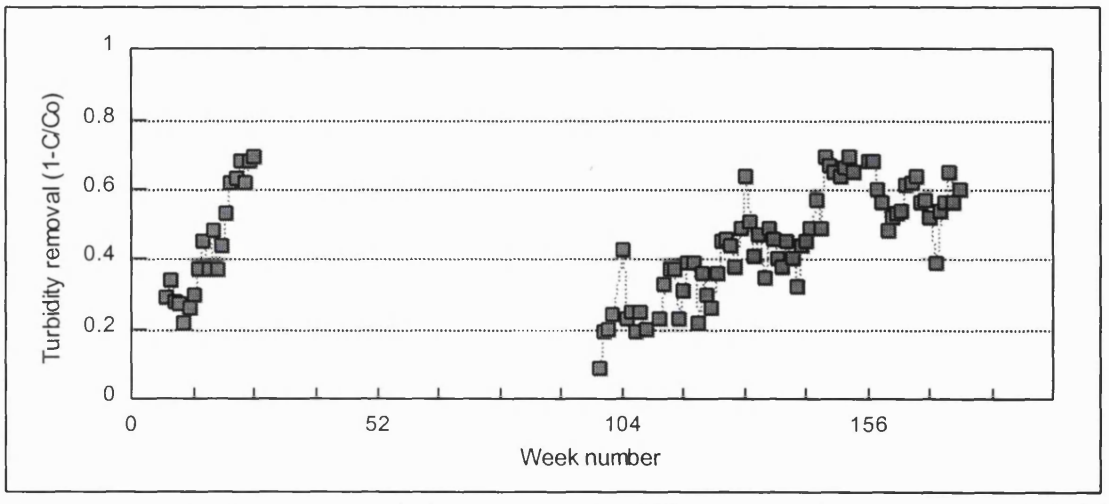

Figure 4.10. *Weekly average turbidity removals by BSS 10-18 sand. Note period of slightly enhanced removals in late 1991 (> 0.6 in weeks 146-156) due to failure of a backwash valve.*

The increase in turbidity removal from < 40% with clean sand to over 50% after a few weeks of operation, shown in figures 4.9 and 4.10, suggested that filter maturation took place, i.e. changes to the filter performance brought about by biological and/or chemical changes on the surface of the media grains.

Turbidity removal was influenced by the method of backwash employed as well as the length of time the filter had been in service, since better removals were achieved earlier with the poorer backwash. Under good backwashing conditions maturation was slower. The improvement in removals in the first period of operation of 10-18 sand was much greater than in the second period i.e. after week 100. The effect of media depth on turbidity removal is shown in figure 4.9. Results from weeks 96-130 with 600 mm sand appeared to show poorer removals than in the previous year (weeks 54-78) after similar maturation periods.

FAILURE OF OUTLET VALVE ON COLUMN 3

Column 3 had a period in late 1991 when the outlet flow control valve failed and so low rate water was also able to by-pass the column. A replacement valve was fitted 6/1/92 (week 157). The turbidity removal data plotted in figure 4.10 showed that column 3 filtration improved from weeks 146 to 156 to values greater than might have been predicted from the rising trend of turbidity removals. Column 3 produced better turbidity removals (>60%, figure 4.10) than the finer sand in column 4 (<50% removals, figure 4.9) at this time. It was this second finding which prompted an investigation into the backwashing of column 3. This found that a valve was failing to close which impeded the combined air and water phase of the backwash. The subsequent reduction in turbidity removals on resuming the correct backwash, and the apparent result of bringing removals back into line with a slowly improving trend suggested that maturation was a long term process that was accelerated by poor backwashing, but was not prevented by the backwash used in these filters.

4.4.5 SURFACE MATTER

An unexpected finding was the presence on the surface of a backwashed and drained filter bed of small deposits, approximately 50 mm^3 in volume. Some were white and powdery, some blue and

soft, and when iron was dosed a bright orange-brown soft material was found. Additionally, many "mud discs" up to 30 mm x 20 mm x 5 mm were observed. These were grey soft lumps which would contain fibrous and mud-like material, sometimes formed around an organic core, e.g. a piece of fish or an aquatic invertebrate. Some incorporated filter media into their matrix. All these objects were found on the CASBW filters.

The mud discs appeared in Summer and disappeared over Winter. They could cover up to an estimated 25% of the filter surface on both sand and dual media filters. Excavations of several filters showed that these discs were only found on the filter surface. There was no evidence of discs having grown into filter mud-balls.

4.5 DISCUSSION

SEPARATE AIR SCOUR AND WATER WASHING

The RGFs at Thames Water's water treatment works in London historically had problems with dirt accumulating on the media causing erratic filtration performance. The build up of material was rapid, taking place in under 15 weeks with separate air scour and backwash water (Toms, 1987). Long term accumulation of dirt caused the sand grains to stick together resulting in large cracks on the surface of filters and "mudballs" deep within the bed. These allowed the water to pass rapidly through the bed through a few large channels, the water thus remaining essentially un-filtered (Bayley, pers. comm., 1993). Accumulated material could also be shed.

At the start of the project it was felt that Thames Water's problems in backwashing full scale RGFs came from following air scour with wash water at a rate that was unable to carry liberated solids up from the bed surface and out of the filter. It was expected that following air scour with a high rate water rinse, sufficient to expand the filter media between 20 and 30%, would remove deposits from the filter grain surfaces and maintain solids in suspension to be washed out of the filter, preventing long-term accumulation of dirt. If successful, this backwash would avoid the engineering complexities of combined air and water washing in the design of full scale plants.

The first year's trials showed from the growth in starting head loss, and the accumulation of POC and suspended solids on media samples, that the backwashing during the first year was not adequate. These data suggested that there were changes taking place on the media grains which were undesirable from an operational viewpoint, and introduced a variable which would create difficulties for interpreting the significance of external factors on filter ripening. Since the objective was to develop a backwash method that offered long term control over filter media condition and could be implemented at full scale, this backwash method did not meet its objective.

The POC data showed that the material on the media was biological in origin, and scanning electron micrographs have confirmed the presence of bacteria and fungi on RGF sand (Bayley, pers. comm., 1993).

Unlike Thames Water filters at Coppermills WTW, where air scour was followed by sub-fluidising water rates of 17 $m.h^{-1}$ (Chipps *et al.*, 1995), the fluidising water rinse had the effect of stratifying the bed and keeping the same media in constant exposure to the influent. This probably had the effect of accelerating the rate of increase in starting head loss.

Studies at Walton WTW in the 1970s, which succeeded in cleaning old filter media by a programme of manual washes with CASBW, produced targets for media cleanliness. These were 250 $g.m^{-2}$ POC and 1500 $g.m^{-2}$ dried suspended solids, based on the 95%-ile value achieved at Walton (Bayley, pers. comm., 1989, Chipps *et al.*, 1995). This study adopted these targets, but only for filter sand, and furthermore, with caution in examining the sand in dual media filters. GAC and anthracite were too friable for suspended solids measurement and, being carbon, the fines could interfere with the POC test.

With both single media and dual media filters the steep rise in starting head loss was noticeable throughout the period with separate air and water washing. It did not appear to depend on the depth of sand or the presence of an anthracite layer, being observed with 1 m deep sand filters (figures 4.2 and 4.3) and dual media filters with 0.34 m or 0.66 m of sand (figures 4.4, 4.5 and 4.6).

COMBINED AIR SCOUR AND WATER WASHING

On the evidence of the first year's trials, a combined air and sub-fluidising water stage was introduced between the separate air and high rate water stages, based on the laboratory work of Regan and Amirtharajah (1984). Amirtharajah *et al.* (1991a) showed that, with air flow rates of 43 $m.h^{-1}$, low rate water flow rates should be 38 - 55 % V_{mf} for sand and dual media filters. This was achieved in this study (tables 4.3 and 4.4).

This study is believed to be the first to confirm, outside of laboratory conditions, and independent of Amirtharajah and colleagues, the benefit of collapse-pulsing backwashing. The results showed that collapse-pulsing backwashing could limit the attachment of organic and inorganic material to the filter media, and could maintain starting head losses on several media sizes and materials similar to those of clean beds of media over a period up to three years.

The pilot plant data do not allow a direct comparison between the two methods of washing since total air and water volumes used in the enhanced wash were greater.

It is not clear why there should have been high POC and suspended solids values in the first trial with 14-25 sand in column 6 in 1990 (table 4.7). It is possible that there had been a solenoid valve problem (as with column 1 in 1992), but the valve had been discarded before the significance of the laboratory data was realised.

A possible explanation for the poor POC and solids results from column 1 after 38 months could have been the recurrence of the solenoid valve problem discovered in February 1992, giving a very low rate of water during the CASBW. Although this was discovered and rectified after about 22 months of operation, the POC and solids on the media measured at 19 months were acceptable. It was likely that there was a gradual reduction in low rate water rate, resulting in a period of time of unknown duration when "collapse-pulsing" was not occurring. It is puzzling that POC and suspended solids results exceeded guidelines in April 1993 because the head loss data did not indicate that the media in column 1 was accumulating deposits up to the termination of data

logging in March 1993. This suggests that this column was being washed adequately, and that the guideline values were not wholly indicative of poor filtration conditions..

Measuring head loss was much simpler than the POC and suspended solids measurement, which was disruptive and labour intensive. This makes it a better tool for operators to judge backwash effectiveness.

Unfortunately no other filters were run with the media undisturbed for the same length of time as column 1, so further work is required to determine whether combined air and water washing was a) effective in the longer term, or b) only kept media clean but could not clean up media that has begun to get dirty, or c) could effectively clean dirty media. The data from the other filters provided considerable evidence that the CASBW method was successful, particularly if the column 4 data for 29 months operation are examined.

MATURATION

The increase in turbidity removal by the filter over time, together with evidence of biological colonisation of the media, suggests a biological maturation process was taking place throughout the operating period of the filter. The combined air and water backwash may have kept down the normalised starting head loss and attached POC and solids concentrations, but maturation was not eliminated.

The changes which occur over time to the surface of filter media have been called "long-term ripening" by Amirtharajah and Wetstein (1980) and ripening by Galvin (1992). These changes involve the "permanent deposition of layers of solids being filtered" (Amirtharajah and Wetstein, 1980). Ives (1955) and Ridley (1967) reported deposition of calcium carbonate and Galvin (1992) found organic materials and the metals Mn, Fe, Al, Ca and Mg. Toms (1987) demonstrated that considerable build up of particulate organic carbon (POC) on sand took place in four months.

Surface irregularities may be increased by the presence of chemical deposits (Galvin, 1992) or biological growths of bacteria and fungi (Bayley, pers. comm., 1993). It is reported that in SSFs a complex community of prokaryotes, fungi, protozoa and metazoa and a matrix of extra-cellular biopolymers play a role in filtration (Ellis, 1985, Ives, 1990), although Letterman (1991) said that very little fundamental research had been conducted into them in slow sand filtration. Some indication of the complex structure of biofilms was given by Coghlan (1996). Unfortunately, biological influences on deposition such as predation, excretion, and the movement of motile organisms have received no attention in RGF literature.

These long-term changes (over periods of time from weeks to months for biological growth and years for chemical changes) are regarded in this thesis as maturation. Ripening refers to the changes which occur during individual filter runs. The degree to which maturation of the media affects ripening merits investigation, especially since maturation may change the surface chemistry and morphology of the filter grains and introduce structures such as fungal hyphae which may penetrate the flow stream lines and may even bridge adjacent media grains. If backwashing was unable to remove them, this would be akin to having dendrites already present on the media at the start of the filter run.

Following the temporary poor backwashing of the 10-18 sand filter for 10 weeks (weeks 146-156), which resulted in improved turbidity removals, it might be asked whether these filters were being backwashed too clean.

It was not the objective of the study to optimise backwash *per se*, although if the filter media was not kept in a consistent condition the study of ripening could have been affected. The starting head loss data provided circumstantial evidence that backwashing with CASBW was satisfactory. Filter maturation did occur, and must be considered when examining ripening. Replacing media each time has been used to overcome the problem of changes to the media grain surfaces in fundamental laboratory studies of filtration (Tobiason *et al.*, 1993), but it is not a practical option at full scale.

It is not possible to say if there is such a thing as a proper or perfect backwash. Operational practice is likely to require a reasonable compromise between lost production time, pore space available for deposition, available head, wash water volume, and pump and pipework sizes. The situation becomes more complex if it is desired that media should be allowed to mature biologically. It is probable that an excess of final rinse washwater was used in this study to make up for problems peculiar to the column design in transporting dirty water out of the columns. However the wash water volume calculations suggest that the water volumes used were not excessive for direct filtration. Backwashing was therefore successful but not necessarily optimised.

CHAPTER 5: CHEMICAL INFLUENCES ON RIPENING

5.1 LITERATURE REVIEW

Certain chemicals may be added to water to enhance particle removal. The importance of chemical treatment in destabilising charges on particles in water to promote effective filtration is widely accepted. The correct use of coagulants and pre-oxidants determines the removal performance of a ripened filter, hence they are critical to the ripening process. It may not be possible to interpret what happens during filtration unless chemical conditioning is understood.

Rapid gravity filters operating before slow sand filters may receive water that has had no chemical pre-treatment. This has been the practice in London. Few water treatment works possess slow sand filters; most have only rapid filters. In these plants the water may be pre-chlorinated or treated with another oxidant such as ozone, as well as coagulated.

Chemical treatment acts in several ways on both the particles and the water in assisting solids/liquid separation. A positively charged coagulant is required. This can be a simple salt, a salt which hydrolyses to a complex structure or a polyelectrolyte (Gregory, 1978, 1978a). Comprehensive reviews of the fundamentals of colloid stability, coagulation and flocculation have been provided by Gregory (1989) and Amirtharajah and O'Melia (1990).

In the first case, chemical conditioning produces flocs for conventional clarification / filtration systems and direct filtration, and promotes "charge neutralisation" (not necessarily to zero zeta potential) for contact filtration.

Secondly, it has been shown for latex spheres (Yao *et al.*, 1971, Tobiason and O'Melia, 1988, and Stenkamp and Benjamin, 1994), DIF/BO powder but not Kaolin (Chang and Vigneswaran, 1990), and *Pseudomonas aeruginosa* (Martin *et al.*, 1992) that the ionic strength of the suspension was important in determining ripening behaviour. This was through changing the electrostatic charges

on the particles and collectors. Yao *et al.* (1971) and Tobiason and O'Melia (1988) showed that the magnitude of this effect on ripening and particle removal was dependent on particle size, in line with trajectory theory. Tobiason and O'Melia (1988) showed negligible filtration took place where repulsion forces were high: in low ionic concentrations no ripening took place, and in some cases the reverse of ripening happened, since filtrate deteriorated as time and deposition proceeded. The deposits were making conditions for filtration less favourable.

Thirdly, the coagulants themselves influenced ripening behaviour. Tanaka and Pirbazari (1986) presented results from which they concluded that the shape and molecular weight of different cationic polyelectrolytes used in contact filtration of kaolin suspensions were influential in producing different shaped ripening curves.

This was also reported by Cleasby *et al.* (1992) whose experiments on full scale filters showed that the filter ripening sequence varied with different chemical treatment conditions. For similar influent turbidities, peak turbidity during ripening was higher with iron(III) chloride and cationic polymer than with bentonite and cationic polymer. In a comparison between alum and iron(III) chloride, both with cationic polymer, the iron was not as effective as alum in removing turbidity, and ripening was only seen with the alum dosed filter. When a non-ionic filter aid polymer was added, both filtrates showed ripening taking between 5 and 10 h, but the iron still produced a poorer filtrate quality. An operating strategy was devised which used bentonite and polymer when raw water turbidity was >7 NTU. When raw water turbidity was below 7 NTU iron(III) chloride and polymer were used, except in winter at low temperature and low pH when the iron was replaced with alum (Cleasby *et al.*, 1992).

Fourthly, it was not necessarily the particles in the water that determined the appropriate chemical dose. O'Melia (1991) pointed out that the natural organic matter (NOM) concentration most often determined the type and concentration of coagulant for conventional, direct and contact filtration. NOM affects the floc structure and sludge quantities and may determine suitability of direct or

contact filtration modes. O'Melia (1991) concluded that NOM was probably more important in water treatment plant design and performance than any other parameter, including turbidity.

Chemical treatment and rapid gravity filtration are two stages in water treatment which go together, and must be optimised together with a view to the final product water quality, and not the "perfection" of either stage. A compromise may have to be made between particle-floc settlement characteristics and filterability (Ives, 1981a). For example Quaye (1976) said that Robeck *et al.* (1964) had shown that having an adequate floc strength in the flocs leaving the sedimentation stage and carried over to high rate coarse filters was actually more important to filtrate quality than floc settleability.

Cleasby (1990) used the data of Robeck *et al.* (1964) to illustrate that, with a strong floc, a ripening then best phase turbidity curve resulted, whereas with weak floc breakthrough occurred at low head loss values. Unfortunately no ripening data were presented for the weak floc condition.

5.1.1 COAGULATION AND FLOCCULATION

The chemicals used in water treatment achieve colour and particle removal by promoting coagulation and flocculation. No universal definition exists for these terms (Gregory, 1989). They are often used as synonyms, whilst they are also used to imply two different stages in one continuum.

In the water industry coagulation is generally defined as a rapid process (from <1 to 7 seconds) which destabilises the charges on colloidal and larger particles, and precipitates colour by chelation. Colloidal particles range from 1 nm to 1 μm in size (Gregory, 1989).

Following coagulation, flocculation occurs if the destabilised particles or hydroxide precipitates are encouraged to agglomerate. This can be by Brownian motion (perikinetic flocculation), low energy mechanical or hydraulic mixing (orthokinetic flocculation), and differential settlement (Gregory,

1989). Flocculation is a slow process taking up to 20 minutes (Purchas, 1981, Rodman, 1982, Amirtharajah and Tambo, 1991).

Coagulation requires a relatively high mixing energy over a short period of time and flocculation requires a prolonged period of low energy mixing. Turbulent rapid mixing is required to produce the small destabilised "pinpoint flocs" suitable for direct filtration (Amirtharajah and Tambo, 1991). Kawamura (1991) stated that good rapid mixing may be achieved by chemical addition across an in-line static mixer, a weir or a hydraulic jump such as a flume, a Venturi or an orifice plate. Approximately 600 mm head loss was required. If head loss could not be afforded mechanical mixers could be used.

Fundamental and practical considerations of mixing have been gathered in a book edited by Amirtharajah *et al. (1991)*. Further information on the definitions of coagulation and flocculation, the electrical double layer model and its relevance to coagulation, calculation of mixing energy, and methods of achieving rapid mixing and flocculation may be found in textbooks by Degremont (1991), Kawamura (1991), Svarovsky (1981), and Shamlou (1993) and a paper by Polasek (1979). It is perhaps surprising, given the amount of literature, that there is little written, and little agreement, on exact conditions for rapid mixing.

When natural particles in water come into close proximity interaction of the diffuse layers generates electrostatic repulsion. To achieve filtration or flocculation the attractive London-van der Waals forces, which depend on the atoms of the particles, their size and distance apart, and are independent of the composition of the solution (Amirtharajah and O'Melia, 1990), must outweigh the electrostatic repulsion forces. The sum of these two forces gives the net interaction energy between two particles. Coagulants reduce the electrostatic repulsion.

The particles found in natural waters typically have zeta potentials from -20 to -40 mV (Amirtharajah and O'Melia, 1990). According to Amirtharajah and O'Melia (1990) the zeta potential depends on the potential at the surface of the particle (Nernst potential) and the thickness

of the electrical double layer. The thickness is governed by the ionic strength of the solution; at low ionic concentrations the diffuse layer extends over a considerably greater distance than in higher ionic strength solutions (Gregory, 1993). In River Thames water the electrical double layer thickness was estimated by Hughes (1981) to be 4 μm (table 5.1). The London-van der Waals force was said by Ives (1981a) to operate at distances between particles of less than 0.1 μm.

Table 5.1. *Estimates of electrical double layer thickness for particles in various ionic strength waters (Hughes, 1981).*

Medium	Electrical double layer thickness (μm)
Distilled water	900
10^{-4} M NaCl	31
10^{-4} M $MgSO_4$	15
River Thames water	4
Sea water	0.4

The most widely used coagulants in water treatment are trivalent iron or aluminium salts which form hydrolysis products in reaction with water (Gregory, 1989). The trivalent cations do not play a role in destabilisation (Gregory, 1989). The hydrolysed chemical species created are dependent on pH and metal ion concentration; this was illustrated in coagulation diagrams, showing charge neutralisation, restabilisation, and floc enmeshment (sweep coagulation) zones, presented by Amirtharajah and O'Melia (1990) for aluminium and iron.

Several mechanisms exist for particle removal by aluminium or iron hydroxides. With colloidal suspensions these are:

destabilisation by charge neutralisation;

reduction of the diameter of the diffuse portion of the electrically charged double layer surrounding the particle;

formation of an insoluble precipitate binding the colloids within metal-hydroxide masses;

the coalescence of these particles into dense "flocs" for settling, or to achieve greater probability of particle removal in a filter.

Gregory (1989) said polyelectrolytes destabilised particles by charge neutralisation and bridged groups of particles to produce flocs. Polyelectrolytes may be used alone, or with a coagulant salt as "coagulant aids". Where the aim is to enhance flocculation, the use of a charged polymer is typical. When used as "filter aids" a non-ionic polymer assists the filtration process.

Natural polymers such as starch or preparations from the seeds of *Moringa oleifera* may be used in coagulation, especially if the presence of polyelectrolyte monomers is undesirable, a biodegradable product is desired, or synthetic chemicals are not available (Ndabigengesere *et al.*, 1995).

5.1.2 SELECTION AND CONTROL OF COAGULANT DOSE

It follows from the preceding sections that effective chemical water treatment requires the application of the correct dose of an appropriate chemical in a mechanically efficient manner. Amirtharajah and O'Melia (1990) described several strategies for this:

the jar test is the most commonly used method;

techniques based on particle charge measurement i.e. zeta potential measurement by zeta meters, and streaming current detectors which produce results which correlate strongly with zeta potential;

control on residual coagulant concentration in the final water (feedback control).

Electrophoretic mobility measurement may also be used as a technique for coagulant control (Letterman *et al.*, 1982), as may measurement of changes in particle size distribution (Yeh and Ghosh, 1981).

Optimum coagulant dose was shown by Habibian and O'Melia (1975) to depend on particle size. Comparison of coagulation by a cationic polyelectrolyte of one suspension of 0.1 μm diameter latex spheres and one of 1.1 μm latex particles, made up to have equivalent surface areas, showed that the suspension of smaller particles required only half the polymer dose needed by the larger particles for effective coagulation to take place.

JAR TESTS

The use of jar tests to determine the type of coagulant, the optimum dose, the optimum pH, mixing energy, residence time, time between chemical additions and any coagulant aid polymer type and dose has been described by Hudson and Wagner (1981) and Ives (1981). The jar test can be performed to assist plant design, for routine dose optimisation, and for changing dose in response to changing raw water quality.

As an alternative, floc size and structure may be visually examined and the optimum dose selected from this, but this method is not likely to be appropriate for flotation or direct/contact filtration. Visible flocs are of the order of 0.1 mm upwards (Ives, 1981). For flotation smaller flocs are desirable (10 - 100 μm diameter, Edzwald and Wingler, 1990). Contact filtration involves filtration of destabilised particles that may be too small to be visible or to settle in a beaker in a short period of time. Visual data from a jar test provide no information if charge neutralisation is the mechanism for direct and contact filtration.

Since clarification and filtration are usually operated together, it makes sense to optimise the two together for overall process and economic efficiency. To this end the jar test may be complemented by a laboratory scale filter to asses the Filterability Number of the suspension (Ives, 1978, Ives, 1981a). Filterabilty tests, or pilot plant filter studies, are likely to be a suitable method for selecting chemical conditions for direct or contact filtration applications. Petrusevski *et al.* (1994) described experiments with a combined jar tester and bench-scale filtration apparatus.

OTHER COAGULANT CONTROL METHODS

For optimising coagulant dose zeta potential, electrophoretic mobility or streaming current may be used. The aim is to obtain a value for particle charge after rapid mixing that indicates destabilisation has been achieved (Amirtharajah and O'Melia, 1990), but which is confirmed as the correct value by practical experience. Yeh and Ghosh (1981) found that minimum residual turbidity in a jar test corresponded to a dose of polymer that produced an electrophoretic mobility of zero. Habibian and O'Melia (1975) found that it was not necessary to achieve complete charge neutralisation, as

determined by electrophoretic mobility, for effective coagulation. Restabilisation was observed to take place before charge reversal had occurred. Letterman and Tanner (1974) reported an optimum zeta potential for direct filtration with polymers of +13 mV, corresponding to maximum turbidity removal, highest rate of head loss development and a short ripening time with no breakthrough. The general view, though, appears to be that a slightly negative zeta potential is acceptable or even preferable.

A streaming current detector (SCD) may be used to give a laboratory or on-line indication of the charge on particles (Dentel, 1994). Amirtharajah and O'Melia (1990) stated that there was a strong correlation between SCD measurements and zeta potential, although SCD values varied with pH. A target SCD reading for a given pH must be determined for each application.

It is possible that the physical measurement of floc size may enable optimum dose selection, although the presence of bound water in the floc and the possibility of flocs shearing in the measurement device may make it difficult to measure flocs accurately. Yeh and Ghosh (1981) reported that a floc size of 20 μm gave the best results with direct filtration of a bentonite suspension, irrespective of the type of cationic polymer used.

MANUAL AND AUTOMATIC COAGULANT DOSE CONTROL

Coagulant dose control procedures based on laboratory trials require an operator to use the data to correct doses on the plant manually. Their advantages are that a wide range of chemical conditions can be investigated simply and reliably. Their disadvantages are that they are labour intensive, there is delay between testing and results - potentially a problem with a rapidly changing water, and the degree of skill required for interpretation (Watts, 1994).

Watts (1994) described several automatic dose control methods including streaming current measurement, proprietary feed forward systems based on influent water quality, an optical monitoring system and a proprietary feedback system, based on filtrate quality. The development of

an advisory system using real-time data and a neural network model, which had potential for closed loop control was reported by Frith (1996).

5.1.3 CHEMICAL TREATMENT OF LOW TURBIDITY WATERS

It has been argued by Geldreich *et al.* (1990) that mountain water of low turbidity could be treated by chemical disinfection alone, provided that there was watershed management, and historical data to back quality assessments.

Saterdal *et al.* (1988) showed it was necessary to use contact filtration to remove completely *Giardia* cysts from low turbidity mountain water (raw water 0.5 NTU). Filtration alone was not sufficient to remove *Giardia* and took out only around 50% of turbidity and other particles from the water. They set a turbidity target of <0.1 NTU to indicate effective chemical treatment and filtration for removal of *Giardia* cysts. In similar trials Gertig *et al.* (1988) and Mosher and Hendricks (1988) found chemical coagulation was essential for *Giardia* removal.

Saterdal *et al.* (1988) noted that in sewage samples numbers of cysts were of the order of tens to hundreds per litre, in contrast to the mountain streams where concentrations were around 10 to 40 per cubic metre. The challenge of *Giardia* to treatment works receiving water subjected to a degree of indirect reuse would be much greater than in these studies.

Mosher and Hendricks (1988) reported that a 10 minute interruption in chemical dosing resulted in *Giardia* cysts in the filtrate, a reduction in coliform removals and a slight fall in turbidity removal from 95% to 88%. This is in contrast to Quaye (1976) who said that he could stop chemical dosing after it had conditioned the filter and by doing this a 24 h filter run could be extended to 36-48 h, with a consistent 0.22 NTU filtrate. In fact Quaye (1976) was treating water with a turbidity below 1 NTU, and only achieving some 60% removal with or without chemical addition.

5.1.4 CONTACT AND DIRECT FILTRATION

Destabilised particles from high quality water sources, i.e. low turbidity, colour, suspended solids or particle concentrations, may be filtered without prior separation processes (Clark *et al.*, 1992).

Filtration without flocculation is termed "direct in-line", "in-line direct" or "contact" filtration (Cleasby, 1990). "Direct filtration", or "floc-filtration" (Janssens and Buekens, 1993), implies flocculation by mixers before the filters.

Contact or direct filtration may be economically attractive by reducing numbers of process units. Additionally, by not requiring large flocs, direct and contact filtration use lower coagulant doses than sedimentation or flotation, even when treating water from the same source (Dolejs, 1994). Watson (1990) cited Culp's (1977) estimates of direct filtration giving capital savings of 30% and chemical cost savings of 10 - 30% over conventional sedimentation and filtration systems. Craig (1985) estimated capital savings of 30-40% and reduced operating costs for what he termed direct filtration (actually contact filtration) compared to conventional treatment. Petrusevski *et al.* (1994) found that direct filtration with pre-oxidation required 4 to 6 times less iron coagulant than conventional sedimentation-filtration (with ozonation after sedimentation), producing comparable or superior filtrate turbidity and particle concentrations.

Janssens and Buekens (1993) warned that contact filtration required more careful monitoring than direct filtration as it was more sensitive to changes in influent water quality. The flocculation stage could accommodate small changes. They said that achieving filtrate turbidities consistently below 0.15 NTU was difficult using contact filtration.

RIPENING IN CONTACT AND DIRECT FILTRATION

Comparing direct filtration of *Chlorella* with contact filtration, using a cationic polymer, Haarhoff and Cleasby (1989) reported differences in ripening. Two contact filtration runs and two direct filtration runs were considered. Ripening appeared to be eliminated by direct filtration, but the first sample was taken after 1 h. Ripening took 4 h on one of the contact filtration runs. Removals of particles $> 4\ \mu m$ were greater with direct filtration than with contact filtration. The rate of head loss development were reduced. In one of the runs breakthrough was delayed by direct filtration, in the other it occurred earlier than with contact filtration.

Edzwald *et al.* (1982) compared contact and direct filtration of low turbidity, coloured waters, treated with cationic polymers. Whilst there appeared to be very little difference in the removal of turbidity they found higher rates of head loss development with contact filtration. Head loss was more concentrated in the upper 200 mm of the bed during contact filtration, and there was better filtration in depth with direct filtration. Edzwald *et al.* (1987) showed a higher post backwash peak with direct filtration than contact filtration.

These are the only studies to have compared contact and direct filtration for ripening. From the work of Darby and Lawler (1990), it is likely that the flocculation used in direct filtration created larger particles causing a lower rate of head loss development. Janssens and Buekens (1993) raised the question whether, as plant size increased from laboratory to full scale, it was likely that the distinction between contact and direct filtration would become less, since there would be some degree of flocculation taking place as water passed along channels from the rapid mixer to the filters, and in the water above the filter bed.

SUITABILITY OF WATER FOR DIRECT OR CONTACT FILTRATION

Gray (1991) was concerned that direct filtration of low turbidity water, with an optimised coagulant dose, could result in ripening times of several hours, prior to achieving an acceptable filtrate. For this reason there might be a lower limit to the quality of water suitable for direct filtration. Furthermore, natural organic matter (NOM) exerted more of a coagulant demand than latex turbidity in model waters, and in natural waters the variation in organic species made coagulation control difficult. He said that some low turbidity waters could be unsuitable for direct filtration because of the high coagulant demand of the NOM. This would be true of some highly coloured upland lake waters. Dolejs (1994) said that NOM was some ten times more important in influencing coagulant dose than turbidity. Janssens and Buekens (1993) presented findings that if the specific UV absorbance of the water was less than 3, the coagulant demand of dissolved organic carbon (DOC) was small, whereas at values of 4 to 5 it was DOC, not turbidity, that determined coagulant dose.

Selection of treatment processes on the basis of raw water particle types and concentrations were discussed by Janssens and Buekens (1993) and by O'Melia (1985), Wiesner and Mazounie (1989) and Wiesner *et al.* (1988). The choice between sedimentation and filtration, flotation and filtration, direct, or contact filtration, should be made according to the loading of turbidity and algae / chlorophyll *a* (Janssens and Buekens, 1993) or the average particle diameter and the number concentration of particles (O'Melia, 1985, Wiesner and Mazounie, 1989, Wiesner *et al.*, 1988).

Contact and direct filtration were said, by Janssens and Buekens (1993), to be suitable for chlorophyll *a* concentrations up to 10 $\mu g.l^{-1}$, although data from the Wahnbachtalsperreverband in Germany indicated 50 $\mu g.l^{-1}$ could be taken as an upper limit under specific circumstances. This variation suggests that the effect of algae on filtration depends not on chlorophyll a concentration, which is a crude indicator, but on algal shape, size and abundance, and the effect these have on filtrate quality, coagulant demand, floc volume and head loss development. Janssens and Buekens (1993) further stated that direct filtration was unsuitable for peak turbidities above 25 NTU and colour beyond 40 Hazen. Logsdon *et al.* (1993) said that waters with turbidity up to 60 NTU could be treated by direct filtration provided the right filter media was used: in their case 2 m deep beds of coarse anthracite (1.5 mm).

Summarising the findings of Wiesner and Mazounie (1989) concerning particle volume concentrations, Janssens and Buekens (1993) said that below 2 ppm contact filtration was the optimum process choice, with conventional separation and filtration optimum above 10 ppm. Direct filtration was generally suitable for waters containing between 2 and 10 ppm of particle volume, although 30 ppm could be accommodated by direct filtration of Wahnbachtalsperreverband reservoir water.

According to O'Melia (1985) contact filtration was suitable for low concentrations of small particles, since the contribution to head loss was acceptable and flocculation would be a slow process. For slightly higher concentrations of small particles direct filtration was desirable as this

would produce larger aggregates, reducing head loss development or, alternatively, permitting higher filtration rates.

5.1.5 THE ROLE OF OXIDANTS IN IMPROVING FILTRATION

Pre-oxidation, especially with ozone, has been demonstrated by several workers to have an important role in improving the removals achieved by a ripened filter. Jekel (1983) and Janssens *et al.* (1985) reported that the use of a coagulant and ozone together produced a filtrate quality better than with either chemical alone. Cleasby *et al.* (1984) found pre-chlorination to be essential to the success of direct in-line filtration during severe blooms of blue-green algae. Janssens *et al.* (1985) found both ozone and chlorine to be effective in improving turbidity removal. Wilczak *et al.* (1992) found that only pre-ozonation improved dual media filtration. No benefit was seen with chlorine, chlorine dioxide, or potassium permanganate.

Logsdon et *al.* (1994) did not need pre-ozonation to meet a 0.1 NTU turbidity goal when alum was the primary coagulant but ozone was required when $FeCl_3$ was used. Tobiason *et al.* (1994a) showed an improvement of up to 0.1 NTU in the filtrate of a conventional treatment plant with ozone instead of chlorine, but the alum dose required for effective treatment was the same.

Petrusevski et *al.* (1994) reported that in-line direct filtration of lowland reservoir water with iron coagulant produced good turbidities (<0.13 NTU), but could not achieve adequate removal of algae without pre-oxidation with ozone or potassium permanganate. Adding 1-2 $mg.l^{-1}$ of ozone increased particle removal efficiency from 60% to well over 90%. Braun *et al.* (1993) demonstrated improved removals of particles from 2 to 15 μm diameter from pre-ozonation, but only marginal benefits for turbidity removal.

Janssens *et al.* (1985) reported improved turbidity removals in direct filtration of reservoir water with pre-ozonation and alum dosing. Ozone was applied at doses from 0.8 to 2.5 $mg.l^{-1}$. They found the beneficial effects of ozone to direct filtration were linked to the presence of algae in the water. Janssens *et al.* (1985) observed a reduction in the required primary coagulant dose. The

lower coagulant dose was reported to reduce loading to the filter, increasing run times and reducing wash water requirements (Janssens *et al.*, 1985). It was also reported that certain blue green algae were partially destroyed by ozonation, whereas diatoms were not broken up by ozone. Removals in excess of 95% resulted from pre-ozonation and dual media direct filtration for species including diatoms and blue green algae.

Tobiason *et al.* (1992) reported that ozone did not permit reduced coagulant doses (alum and cationic polymer). However pre-ozonation increased the filter run length on one run and delayed turbidity breakthrough on another. Logsdon *et al.* (1994) and Tobiason *et al.* (1993a) found that pre-ozonation sometimes increased filter productivity.

Studies of direct filtration at 22 $m.h^{-1}$ by Drago and Thompson (1993) showed an optimum pre-ozonation dose of 1 - 2 $mg.l^{-1}$. They reported that turbidity removal performance decreased with doses less than 0.8 $mg.l^{-1}$ and above 4 $mg.l^{-1}$. They found that adding extra alum could not compensate for not using ozone.

With conventional treatment Richard and Dalga (1993) found that ozone improved removals of algae even when coagulation was optimum, as measured by zeta potential. Increasing ozonation partially compensated for under-dosing of coagulant. An increase in ozone dose of 1 $mg.l^{-1}$ made up for a shortfall in coagulant of 20 $mg.l^{-1}$. At optimised coagulant dosing, an ozone dose of 1.6 $mg.l^{-1}$ improved algal cell removals by 1-log. Singer (1990) noted that it was generally believed that in conventional treatment ozone could save coagulant but would not lead to a superior particle removal performance. Arnac and Burke (1994) reported a 20% saving in coagulant dose using ozone with conventional filtration.

There has been little study into the effect of ozone on ripening behaviour or time. Tobiason *et al.* (1992) examined the use of ozone and hydrogen peroxide with ozone on in-line direct (contact) filtration. Data suggested ozone in addition to coagulants resulted in a lower post-backwash peak, but no change in ripening time, or quality during the main operating period of the filter.

Logsdon *et al.* (1993) and Foellmi *et al.* (1993) reported the lack of effectiveness of ozone at a plant in Vancouver, and interpreted this lack as resulting from the chemical conditions of the water. The water in these studies had a hardness of 5 $mg.l^{-1}$ as $CaCO_3$, and TOC of about 1.5 to 2 $mg.l^{-1}$. Logsdon *et al.* (1993) cited two cases (McBride and Stolarik, 1986, and Langlais et al., 1991) where pre-ozonation was beneficial with waters of hardness of 70 and 300 $mg.l^{-1}$ as $CaCO_3$ respectively. They said Langlais *et al.* (1991) showed that for ozonation to affect particle coagulation the water should have a hardness to TOC ratio of at least 20:1. Singer and Chang (1989) said pre-ozonation only assisted coagulation where the ratio of hardness:TOC exceeded 25 mg $CaCO_3.l^{-1}$ per mg $C.l^{-1}$, and ozone was dosed at 0.4-0.8 mg O_3. l^{-1} per mg $C.l^{-1}$.

Jekel (1994) reported typical ozone doses in drinking water ranging from 0.2 to 2.0 $mg.l^{-1}$, equating to 0.1 - 0.5 mg $O_3.mg^{-1}$ DOC (dissolved organic carbon). Janssens *et al. (1985)* stated that a minimum concentration of organic material must be present for ozone to stimulate coagulation and that ozone should be set at 0.1 - 0.3 g $O_3.g^{-1}$ DOC. A dose not exceeding 0.5 - 1.0 mg O_3 per mg C was suggested by Bernhardt *et al.* (1990). Reckhow *et al.* (1993) stated that pre-ozonation was likely to improve coagulation when raw water contained oxidised iron or certain algal species or high molecular weight NOM.

5.1.6 MECHANISMS OF OZONE ACTION ON PARTICLE REMOVAL

A number of mechanisms for the coagulation or "micro-flocculation" effect of pre-ozonation have been summarised by Janssens *et al.* (1985), Grasso and Weber (1988), Singer and Chang (1989), Singer (1990), Reckhow *et al.* (1993) and Jekel (1994). Saunier *et al.* (1983) showed in a control experiment using air without ozone that the effect was not obtained simply from the action of bubbling a gas through the water, the effect was due to the ozone. The mechanisms were not fully understood, and contradictory results have been reported, depending on raw water characteristics (Jekel, 1994), but briefly summarised the hypotheses were:

the production of carboxylic acid groups resulting in increased interactions with aluminium or calcium and bridging between humic materials;

reducing the adsorbed organic layer on clay particles assisting destabilisation;

precipitation of metal hydroxides from the source water;

polymerisation of NOM catalysed by aluminium floc;

release of biopolymers from stressed or lysed algal cells;

release or formation of polymers from EOM;

changing surface charges of colloidal particles.

Bayley (pers. comm., 1993) suggested that ozone might act by disabling the cilia and flagella of motile phytoplankton so that they were less able to penetrate filters. Petrusevski *et al.* (1994) thought ozone killed or immobilised motile algae. Reckhow *et al.* (1993) presented SEM evidence that ozone could cause algal cell damage, for example the removal of spines from *Scenedesmus*, and that particle size data suggested cell shrinkage occurred with *Chlorella*. Contrary to popular opinion, they said, ozonation up to 3 $mg.l^{-1}$ did not destroy algal cells. Bernhardt *et al.* (1990) reported that ozone reacted with algae to produce hydrophobic colloids. This resulted in considerably enhanced algal removal using flotation.

Ives (1955) found the use of iodine and ozone increased the zeta potential of algae, i.e. made them more negatively charged. Singer and Chang (1989) and Edwards *et al.* (1992) also found that ozone caused particles to become more negatively charged. The reaction of ozone was clearly more complex than could be explained in simple electrostatic terms, since it increased the charge on particles yet acted as a substitute for a coagulant. Petrusevski *et al.* (1993) stated that pre-ozonation encouraged the formation of colloidal particles from dissolved organic matter, encouraged the formation of larger particles, including agglomeration of algae, and the latter appeared to be better with types of algae which produce larger amounts of extra-cellular organic matter (EOM).

5.2 SELECTION AND CONTROL OF COAGULANT DOSE

Since the literature suggests that filtration and ripening are sensitive to coagulation, there was the need to optimise the coagulant dose, defining what constitutes an optimum. Developing the method of determining dose formed part of the experimental work.

JAR TESTS

Early in the project jar tests were carried out to determine the iron dose. The aim was to produce a "pin-head" floc and to evaluate turbidity removal using a filter paper. Initial jar tests indicated that the dose should be around 1 mg $Fe.l^{-1}$.

INSTRUMENTAL TECHNIQUES

Sales demonstrations of both a zeta meter and a streaming current detector both showed no sensitivity to changing doses of iron(III) sulphate within the ranges typically found suitable by the feedback optimisation method (discussed later). The SCD indicated that the dose should be an order of magnitude greater than that actually used. The zeta meter results are presented in table 5.2 below. These instruments were not utilised.

Table 5.2 *Illustration of the poor response of a zeta meter to changes in coagulant dose with reservoir stored river Thames water.*

Water type	Zeta Potential	Standard deviation
Reservoir stored water	-12.4 mV	2.4
After pre-ozonation	-12.4 mV	2.4
After ozone and iron at the correct dose (0.5 $mg.l^{-1}$ as Fe)	-13.1 mV	2.5
After ozone and an excess of iron (5 $mg.l^{-1}$ as Fe)	-12.7 mV	2.4

FEEDBACK TECHNIQUE

Several workers have found the conventional jar test to be inappropriate for direct filtration, and several modified procedures incorporating a filterability test have been proposed (Bernhardt and Schell, 1993). Having established the approximate dose from jar tests the filtrate quality was monitored for a few hours to ensure that the correct dose had been set up. This would have been a labour intensive method, but fortunately adjustments were required infrequently. Major changes in dose followed the progression of algal blooms or were due to changing source reservoir.

Optimisation trials involved making minor adjustments to the dose. A guiding principal was that the dose should not be changed by more than $\pm$ 20% at a time, unless it was known that a new

reservoir was in service. Having used the jar test as an initial guide for setting the dose, the filtrate quality was used to decide whether the dose was optimal. Jar testing was then abandoned.

Many observations of chemical (ozone or iron) dose changes on the pilot plant showed that the effect was usually complete within 30 - 45 minutes, as shown by the on-line turbidimeter. This reflected the time taken for a complete change of water in the ozone contactor, filter shell and turbidimeter body. The new filtrate turbidity was then steady.

In some instances, having established good performance it was hard to understand why, when returning to the pilot plant, the filtrate turbidity was sometimes as it had been when left, and sometimes it was poorer.

It required the examination of logged trends of filter turbidity data to understand what might be happening and develop a model of filtration and a method for dose selection. The importance of using the logged data was discovered in the autumn of 1991, when difficulties were experienced with selecting the iron dose at a time when algal populations were declining and the turbidity load was rising. One of the early rules of thumb developed from jar tests was that iron dose in mg $Fe.l^{-1}$ should be between 75 - 100% of the value of the stored water turbidity. As the influent turbidity rose the iron dose was increased, but the filtrate turbidity showed breakthrough before maximum head loss. In response to this the iron dose was further increased; as a consequence more breakthrough occurred, commencing at lower head losses.

It was hypothesised that this breakthrough curve indicated an overdose of iron, so the iron dose was reduced until the filtrate turbidity curve was flat right up to a maximum head loss of 2.5 m.

These observations produced a coagulant control procedure, using the examination of trends of flow rate, head loss, and inlet and outlet turbidity over the entire filter cycle to determine whether the dose required alteration. The development of this manual feedback system has been discussed

by Chipps *et al.* (1994). Principally the two turbidity trends were examined but disturbances in the trends due to problems of filter operation could be eliminated by checking flow and head loss.

5.3 THE EFFECT OF CHEMICAL DOSING ON RIPENED FILTER PERFORMANCE

Four chemical dosing conditions were possible for columns 1, 5 and 6 i.e. no pre-treatment, pre-ozonation, iron dosing and pre-ozonation plus iron. The calculation of ozone and iron doses was outlined earlier.

Figure 5.1 shows daily sample results taken from column 1 over three years from February 1990 to March 1993 measured by laboratory turbidimeter. For the majority of this period ozone and iron were dosed, but there were samples from times when iron and/or ozone dosing was stopped.

These data are presented to provide an indication of the benefits of chemical dosing. The assumption was that the random timing of sampling with respect to the filter cycle meant that the results reflected ripened filter performance. These results did not differentiate between filters in the ripening, best or breakthrough phases. The order in which iron and ozone were applied was not considered, neither were flow rates, water temperatures or other variables. These are discussed in chapters on physical and biological influences on filter ripening.

Fluctuations in the data from the iron dosed trials could arise if the iron was not dosed correctly. The effect of iron dose in controlling the shape of the filtrate turbidity curve through the filter run is discussed in detail later.

Despite some scatter the data showed very significant statistical associations. With no iron dosing there was a strong relationship between the stored water and filtrate turbidity, irrespective of ozone dosing. Pre-ozonation improved turbidity removal over no dosing, but not as much as with iron dosing. The two chemicals acted synergistically to produce the lowest turbidities. With iron applied the filtrate quality was far less dependent on the applied water turbidity. The regression tended towards a flat line.

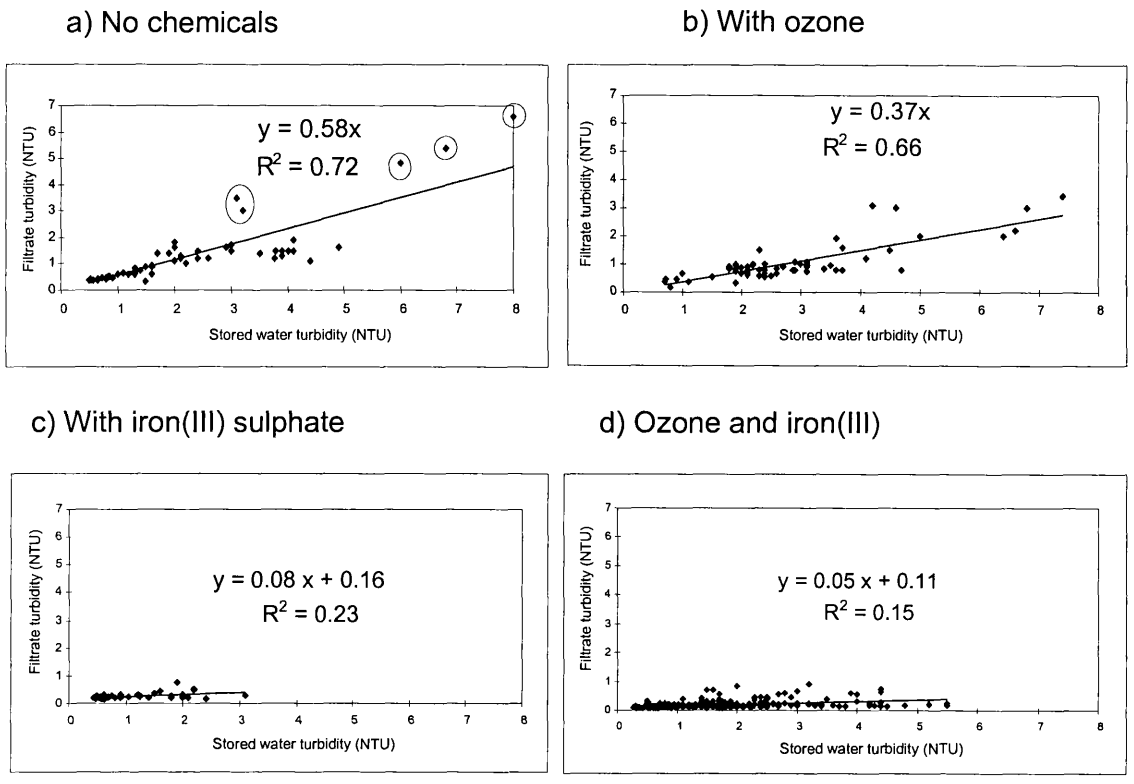

Figure 5.1. *Stored water and filtrate turbidity data from column 1 1990-3 showing the influence of chemical dosing on filtrate quality. Linear regressions have been plotted for each set of chemical dosing conditions.*

Values of the correlation coefficient (r) can range between 1 (perfect positive association) and -1 (perfect inverse relationship) with 0 indicating no relationship (Simpson, 1987). The value of r^2 indicates the degree to which the variation in the y data can be explained by the x value (Devore, 1991). The t-test shows the probability of the correlation coefficient being observed by chance if the variables are not genuinely related (Southampton University, 1979). The t-test was calculated from $t = \frac{r\sqrt{n-2}}{\sqrt{1-r^2}}$ for n-2 degrees of freedom. The data are shown in table 5.3. In each case the association was highly significant at the 0.1% level of chance.

Since the linear regressions for filtration without iron returned a negative y-intercept (x = 0), which is a logical impossibility, the regressions were plotted forcing the intercept through the origin. This reduced the r^2 by very little, suggesting this procedure was justified.

In figure 5.1a there are five data points which have been circled. These were poor removals from the first six weeks of filter operation, possibly indicating immature filter performance. It should be

noted that half of the ozone only data in figure 5.1b were also from this first few weeks and there is less suggestion of two clusters of results representing immature and mature performance. Removing the circled data produced different linear regressions as shown in figure 5.2 ($y = 0.32x + 0.34$, $r^2 = 0.68$, or $y = 0.44x$, $r^2 = 0.54$, $n = 53$).

Table 5.3. *Regression analyses of column 1 turbidity daily sample data 1990-3 based on data shown in figures 5.2 and 5.3. Samples measured using a bench turbidimeter.*

Chemical dosing	Linear regression	Number of samples	r	r^2	t-test	Significance level
None	$y = 0.63 x - 0.15$ $y = 0.58 x$	58	0.85	0.73 0.72	12.21	0.1%
Ozone	$y = 0.39 x - 0.05$ $y = 0.37 x$	60	0.82	0.67 0.66	10.55	0.1%
Iron	$y = 0.08 x + 0.16$	49	0.48	0.23	3.77	0.1%
Ozone & Iron	$y = 0.05 x + 0.11$	400	0.38	0.15	8.22	0.1%

The similarity of the slope in figure 5.2 with the larger data set from the undosed control filter column 4 (figure 5.3) suggests it was correct to exclude these five data points. The data in figure 5.3 were daily samples from column 4 with 900 mm sand, operating from November 1991 to March 1993 (after the after the addition of 300 mm of new sand to 600 mm of mature sand).

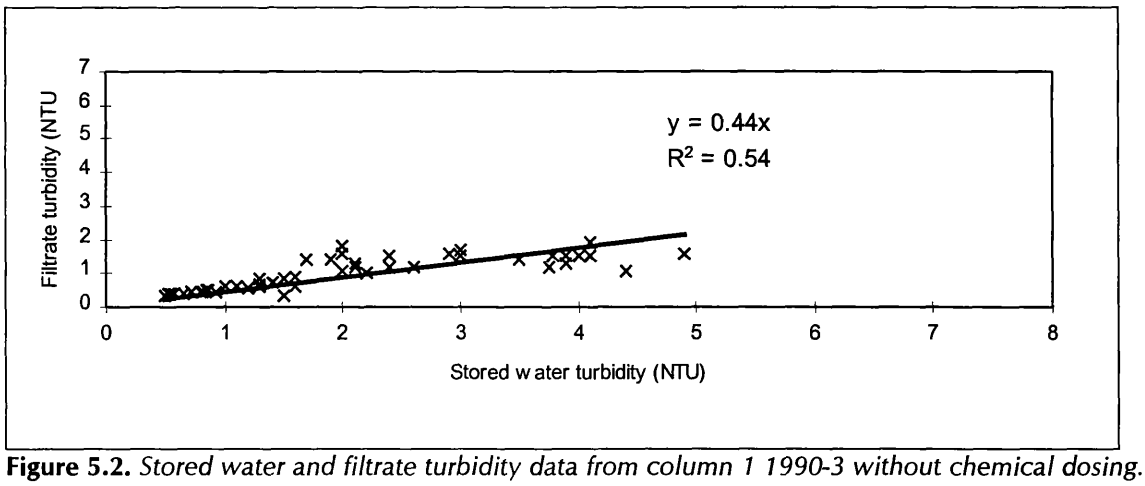

Figure 5.2. *Stored water and filtrate turbidity data from column 1 1990-3 without chemical dosing. The data points circled in figure 5.1a have been removed.*

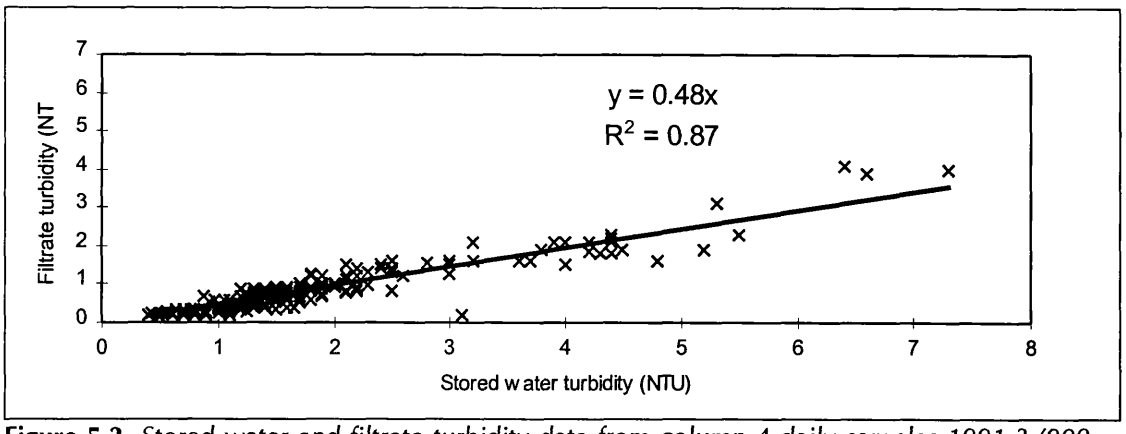

Figure 5.3. *Stored water and filtrate turbidity data from column 4 daily samples 1991-3 (900 mm sand) without chemical dosing (y = 0.48x, r^2 = 0.86, or y = 0.51x - 0.07, r^2 = 0.87, n = 207).*

The results from daily samples in figure 5.4 show that only the combination of iron and ozone was able to produce turbidities of 0.1 NTU, and this combination achieved low turbidity filtrates (<0.2 NTU) from a range of influent turbidity values.

Further analysis using logged data showed the possibility of two distinct trends within the data for ozone and iron dosing. Figure 5.5 shows calculated filtrate turbidity averaged for the period 40 - 50% into the elapsed run time plotted against mean influent turbidity values from each filter run Iron and ozone dosed filter runs from column 1 (December 1991 to March 1993) were used. To avoid results from overdosed filter runs, those runs where mean turbidity removal was <70% after 90% of the filter run time had elapsed were excluded from this examination.

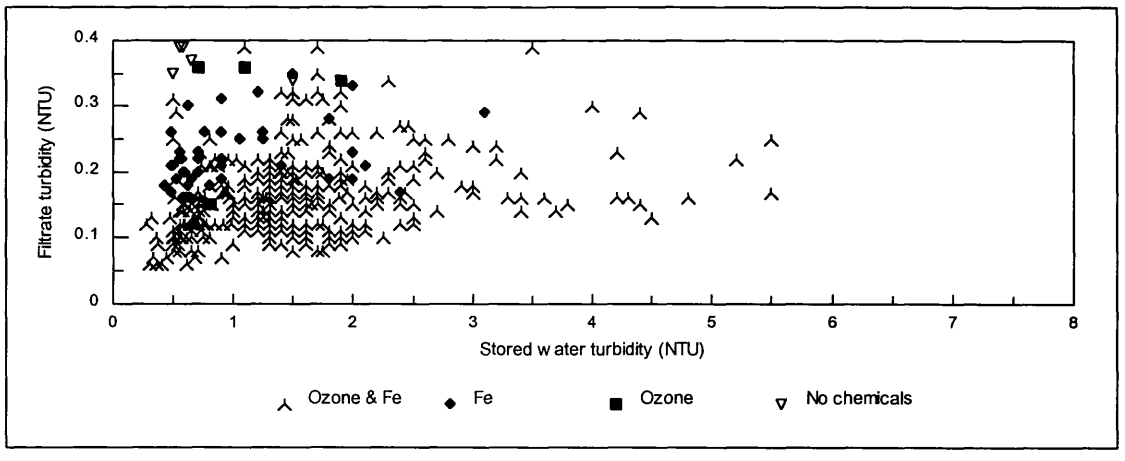

Figure 5.4. *Stored water and filtrate turbidity data from column 1 1990-3 showing how ozone and iron dosing produced the lowest filtrate turbidities with a range of inlet turbidities.*

By dividing the data into "winter" results (December 1991 to March 1992, and November 1992 to March 1993) and "summer" results (April to October 1992) it would appear that the turbidity removal was independent of load during the summer at higher turbidities and dependent on load during the winter. This is examined further in chapter 6.

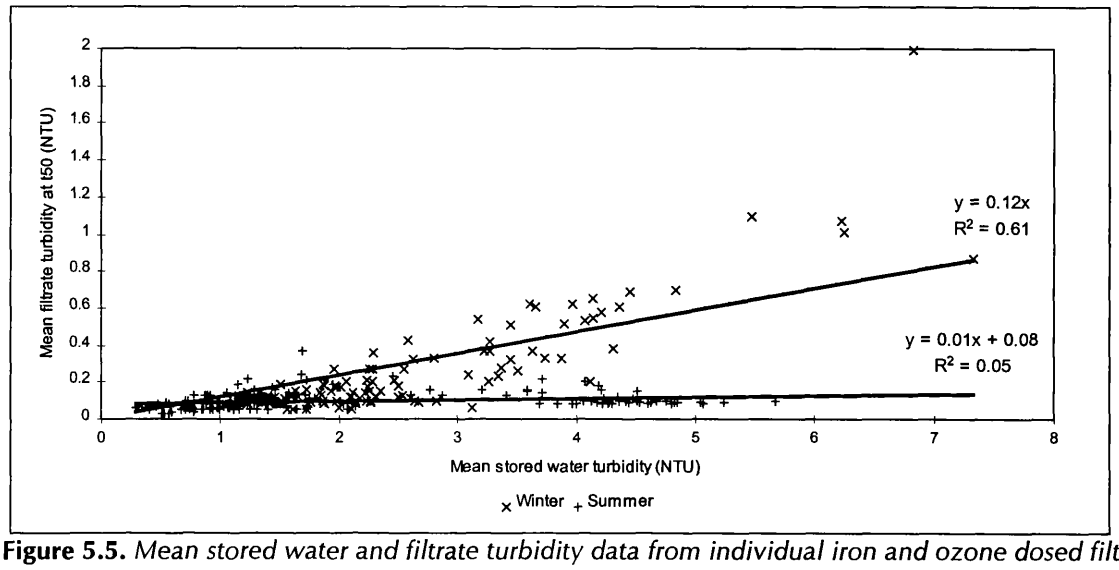

Figure 5.5. *Mean stored water and filtrate turbidity data from individual iron and ozone dosed filter runs (column 1 1991-3), suggesting filter performance in winter (November-March) differed from that in summer (April-October) at higher inlet turbidities.*

The logged data from column 4 (January 1992 - March 1993) confirmed the spot sample observations. Mean inlet turbidity, filtrate turbidity and C/Co values from each valid filter run showed that the spot data were very representative of the entire filter run. A regression analysis of filtrate against stored water turbidity gave the result $y = 0.44 x - 0.02$, or $y = 0.44 x$, with $r^2 = 0.92$ in each case. The data are shown in figure 5.6. Presenting the data this way understates the degree of variation in turbidity removals between filter runs. This is more obvious in figure 5.7.

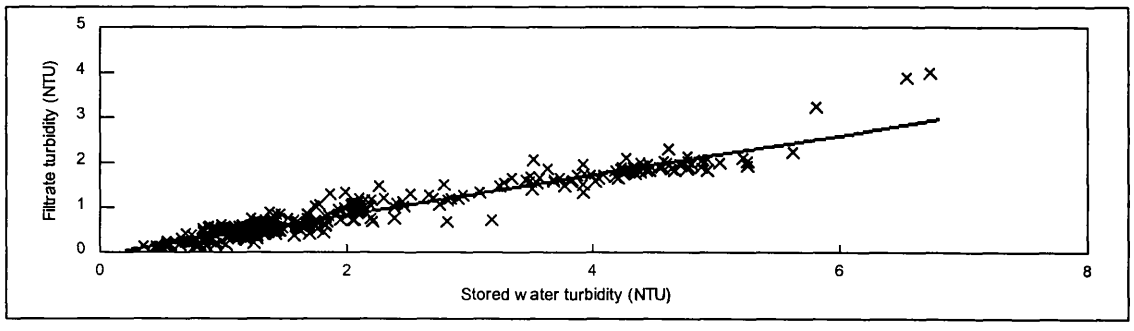

Figure 5.6. *Mean stored water and filtrate turbidity data from each column 4 logged filter run 1992-3 (900 mm sand, no chemical dosing), showing regression $y = 0.44x$, $r^2 = 0.92$, $n = 313$).*

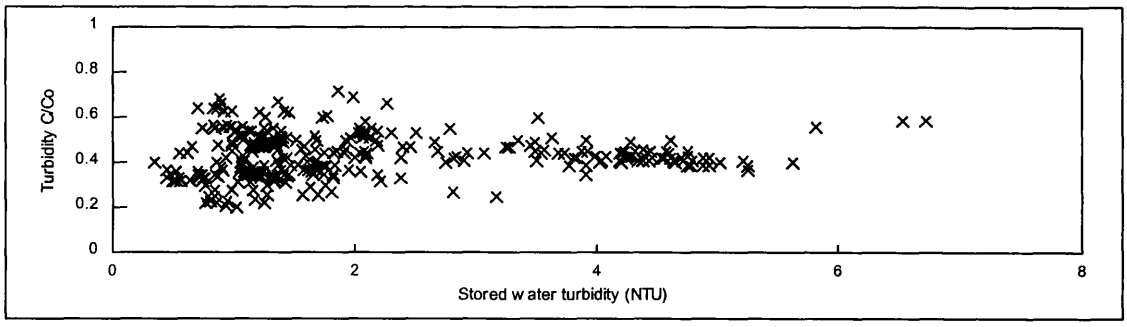

Figure 5.7. *Mean turbidity C/Co data from each column 4 logged filter run 1992-3 (900 mm sand, no chemical dosing.*

Presenting the data from figures 5.6 and 5.7 against time shows some further points. It has been suggested in chapter 8 that ripening may be defined in six ways. However, none of these were appropriate to ripening in a primary filter in the absence of chemical dosing. It can be seen from figure 5.8 that mean filtrate turbidities approached 0.1 NTU only when influent turbidity was very low, but never achieved 0.1 NTU. Figure 5.9 shows that mean turbidity removals never approached 95% removal (C/C_0 = 0.05). Turbidity removal appeared to improve in the summer months. Reasons for this are discussed in chapter 6. When the mean filtrate turbidity was compared with the filtrate values averaged from 40 to 50% into each filter run a regression returned the relationship y = 0.99 x, with r^2 = 0.99. This shows that in this case the mean filtrate turbidity probably accurately reflected ripened filter performance.

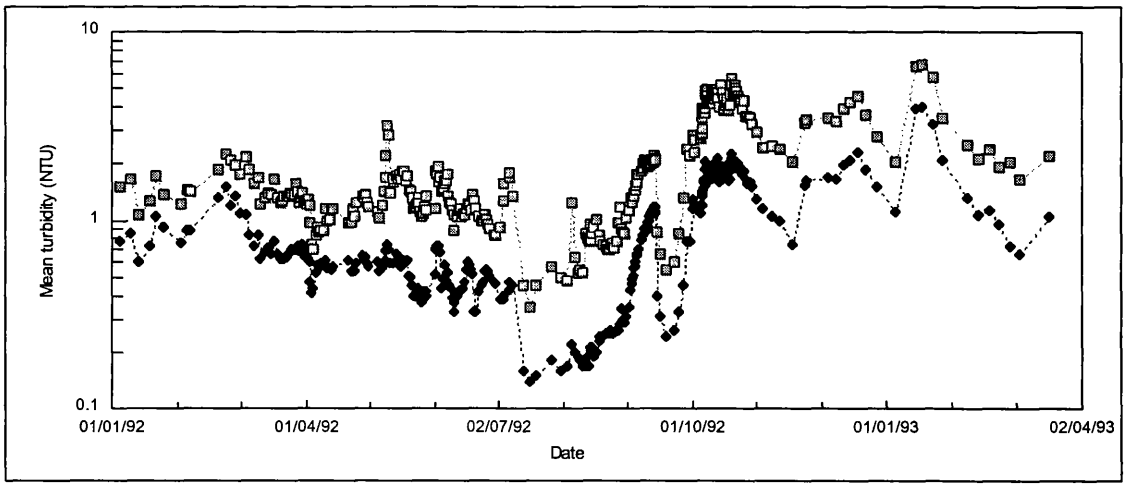

Figure 5.8. *Column 4 mean inlet and filtrate turbidities calculated from each run. The influent values are shown as squares and the filtrates as diamonds.*

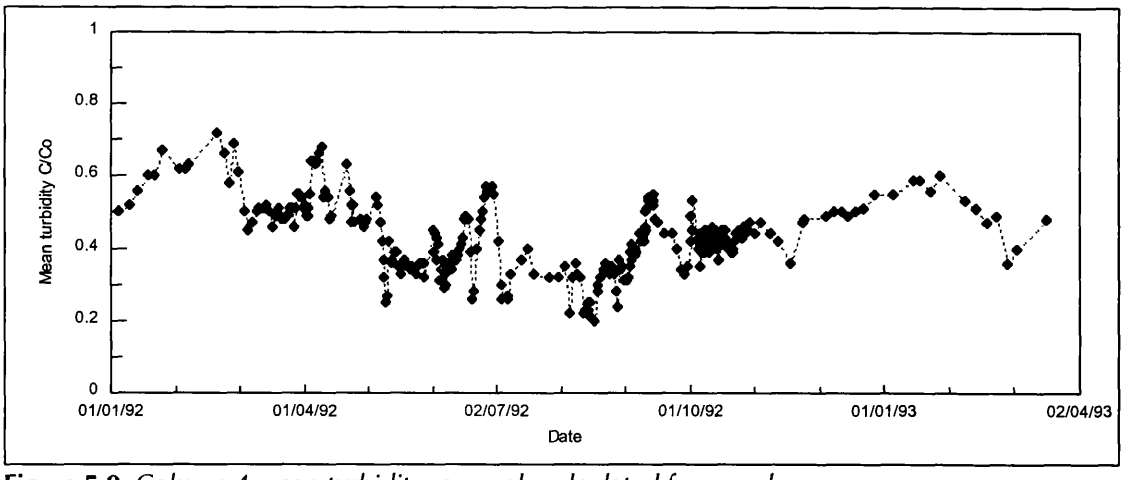

Figure 5.9. *Column 4 mean turbidity removals calculated from each run.*

Determining the minimum turbidity in a run was problematic. Detailed examinations of individual runs showed the minimum turbidity was often at the start or end of the filter runs due to the turbidimeter partially draining, or due to the measurement of the backwash water remnants. A minimum filtrate turbidity could also be due to a minimum influent turbidity value, or a flow rate change. The same objections follow for the use of maximum turbidity removal or the value close to maximum turbidity removal. The objection to using turbidity gradient change is that it gives a ripening point that is at a higher turbidity value if ripening is slow.

5.4 THE EFFECT OF CHEMICAL TREATMENT ON FILTER RIPENING

INTRODUCTION

Essentially ripening involves an initial peak, an ultimate quality or the achievement of a quality target, and a time period. The data from many filter runs have been examined to provide the following observations. Unless specified otherwise filtrate and influent quality refer to on-line turbidity data, sampled and logged at 15 minute intervals. The logged data have been processed to eliminate periods when instruments or backwashing were known to have failed, and when turbidimeter cleaning or calibration took place.

The data presented earlier have shown that turbidity removals were poorer without coagulant dosing. The literature presents evidence of the importance of chemical preconditioning to produce the classic ripening curve in laboratory trials, but the need to examine practical situations

individually remains. In primary filtration without chemicals, what evidence is there of filter ripening and are attempts to manage ripening justified?

The importance of chemical dosing for ripening has been described in laboratory and field trials. In this trial the approach of varying the chemical dosing onto sequential filter runs was used to avoid problems of comparing one filter with an adjacent filter when the second filter was not a proper control due to differences in media type, and potential differences from media ageing, such as abrasion, stratification and biological colonisation. The problem with sequential runs could be that the particles in the water could vary, so that the filtration conditions would not be well controlled. This is a generic problem with environmental studies.

5.4.1 RIPENING IN THE ABSENCE OF COAGULANTS

EXPERIMENTAL

Column 4 contained 900 mm depth of 0.6 - 1.18 mm sand. It has been shown earlier that the sand was being properly backwashed yet was probably biologically mature after 3 - 4 months of operation. The filter was operated with 600 mm sand from October 1990, and 300 mm were added in October 1991. Filter performance is reported from January 1992 when continuous monitoring of inlet and outlet turbidity started, giving 15 months of data for examining ripening.

The filter was operated to a terminal head loss of 1.8 m to 9.6.92, and 2.5 m thereafter. Records for filter runs 554 - 895 and 998 - 1105 were logged every 15 minutes. Records 896 to 997 (29.9.92 to 4.11.92) were logged at 5 minute intervals. In all 311 filter runs produced useful data.

In order to interpret filter performance the trends of influent and effluent turbidity and filter flow rate were required. All the filter runs during which inlet and filtrate turbidities and flow and head loss were recorded properly were plotted individually on a computer screen and examined in detail. A few runs which illustrated key points have been presented. In the turbidity graphs the influent turbidity is represented by the thicker, lighter line. Head loss development was plotted, since in a few cases this revealed information on the behaviour of the filter or the deposits.

Turbidity removal graphs were plotted because they gave a better idea of the filter performance than turbidity data. Turbidity removal was calculated as C/C_0 , where C = effluent turbidity and C_0 = stored water turbidity at the same time. Turbidity removal was greater at lower C/C_0 values. It is acknowledged that the residence time of water in the filter meant that there was a time offset between the inlet sample and the outlet sample (table 3.1) however the relative stability of the stored water, the difficulties in off-setting the data, and the problems in data manipulation of comparing the filtrate at the start of a filter run with the influent towards the end of the previous filter run meant that this was not practicable. Turbidity removals were calculated from simultaneous samples.

One further point concerns the filtrate turbidity value at the end of each run. The pipework design caused the column 4 turbidimeter to drain partially during each backwash. This resulted in a low turbidity value being recorded during the backwash, when flow and head loss fell to zero, and until the filtrate had refilled the sample cell, which took about 1 minute.

It is acknowledged that flow rates and loadings were different for these filter runs. Individual runs may have behaved idiosyncratically due to unmeasured parameters, for instance changes in influent particle size distribution, so it is not intended to discuss individual runs in depth. Groups of filter runs where certain parameters were the same or similar e.g. flow rate, influent turbidity are examined together in later chapters.

RESULTS

The examination of many individual filter runs revealed that in the absence of coagulant dosing filtrate quality was highly influenced by influent quality, confirming the findings indicated by the r^2 values from the spot sample and mean data (figures 5.3, 5.6 and 5.8). This is illustrated by the trend graph from 15 minute samples (figure 5.10).

A key finding deriving from the close relationship between influent and effluent turbidity, was that filtrate trends used in isolation could produce misleading evidence of ripening. It was necessary to use turbidity removal data to determine whether ripening was taking place. This is illustrated in figure 5.10 which shows a filter run where the inlet turbidity changed during the run. It can be seen that, without coagulants, a) the filtrate turbidity data gave no indication of ripening, b) turbidity removals were largely unaffected by the variation in inlet and outlet turbidity, and c) turbidity removals suggested that the filter underwent little or no ripening.

When turbidity removals were adjusted by 2 records (30 minutes), to take the residence time in the filter into account (see table 3.1), there was some evidence of a small amount of ripening taking around 15 hours (figure 5.11).

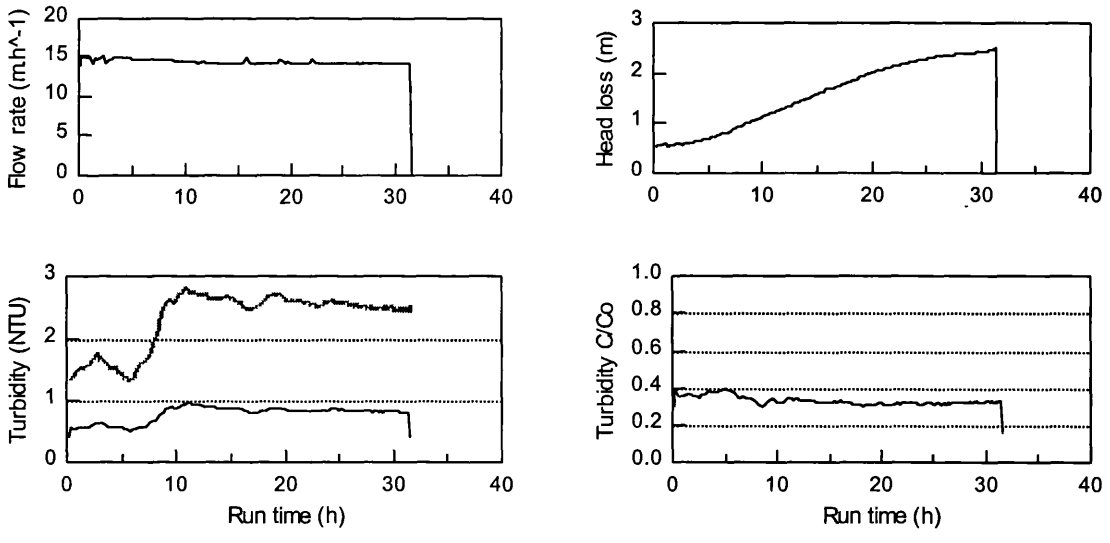

Figure 5.10. *Filter performance data for column 4 run number 895 showing the effect of changes in influent on effluent turbidity and how filter ripening could only be determined using C/Co data. The dates were 28-29.9.92.*

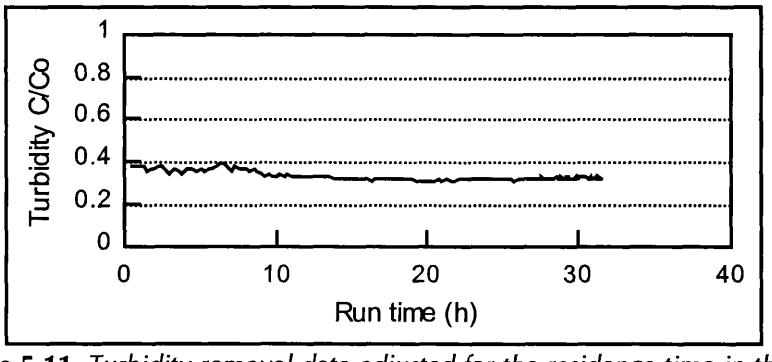

Figure 5.11. *Turbidity removal data adjusted for the residence time in the filter from column 4 run number 895 showing filter ripening.*

From the detailed examination of filter runs four classes of turbidity removal were observed. These are illustrated in figures 5.12, 5.13, 5.14 and 5.15. In each graph the x-axis is elapsed filter run time. Because run 555 lasted several days the turbidity and turbidity removal data have also been plotted with a logarithmic time axis to examine the detail at the start of the filter run. The initial data were assigned to a filter run time of 0.1 h.

Figure 5.12 illustrates classic filter ripening, with an initial low turbidity value, rising to a peak, followed by ripening to a removal value that was sustained for the rest of the run. The ripening illustrates some of the problems that interpretation of trends from pilot plant data presented. If filtrate turbidity was considered, ripening was probably achieved inside 2 h. In the plots with a logarithmic x-axis filtrate turbidity showed an initial ripening period lasting 2 h, followed by a lag phase then a second improving phase from 12 h onwards. At first sight this might seem to be a second ripening phase, however the steady fall in filtrate turbidity from around 12 h was due to the changing inlet turbidity and not ripening.

The removal data suggested that ripening took about 10 h to complete. The removal data with the linear x-axis suggested one ripening curve lasting 10 h, but the data with a logarithmic x-axis suggested a series of steps to 10 h.

The C/Co trends show that the falling stored water turbidity values were the key to understanding filter behaviour. During run 555 the stored water turbidity values were not constant: the initial ripening period was observed during a steady inlet turbidity period, but after 3 h the inlet values climbed and this caused the filtrate turbidity to level off and then rise slightly. Meanwhile removals continued to increase. After 15 h the stored water values dropped, as did the filtrate turbidities. The removal values after 20 h were virtually constant, regardless of falling inlet and outlet turbidity values. This was more clearly seen with the linear time plot. Turbidity removal was therefore a better guide to ripening than filtrate turbidity.

It is apparent in figure 5.12 that the head loss gradient appeared to increase after about 20 h. This was long after ripening of turbidity had finished. In the second case the filter showed no ripening and turbidity removals deteriorated throughout the run. This is illustrated in figure 5.13.

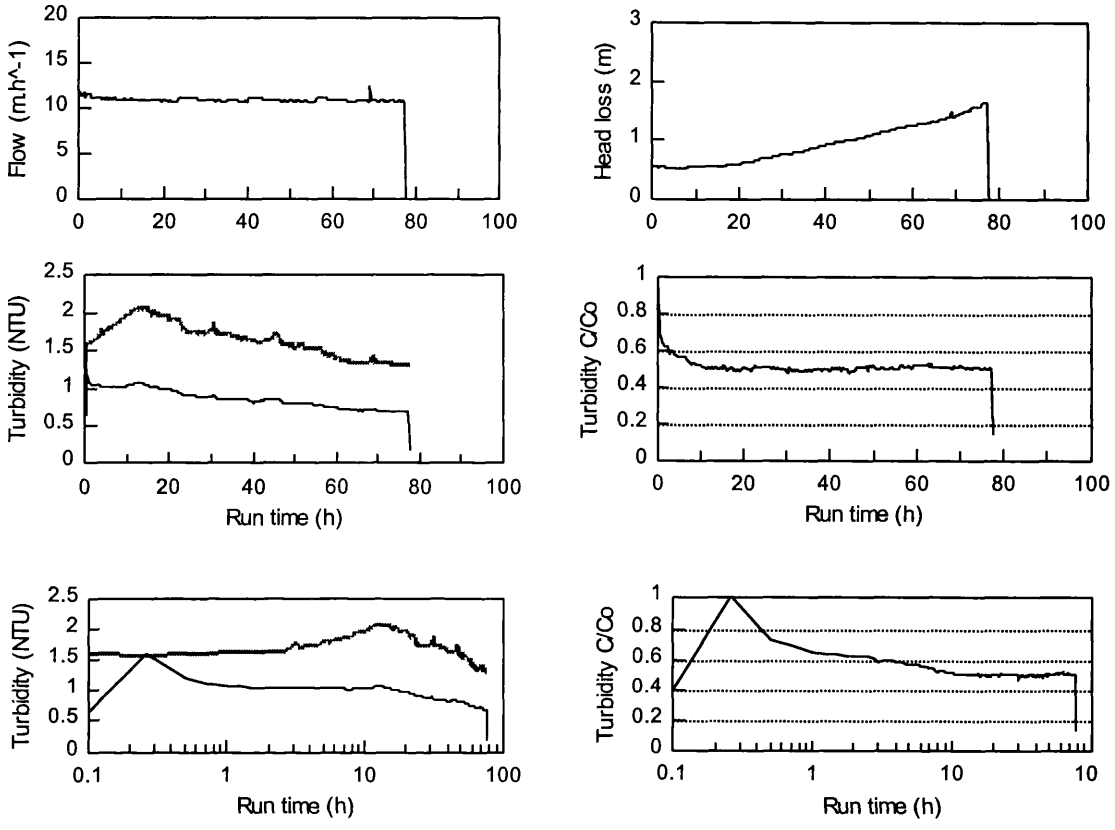

Figure 5.12. *Filter performance data for column 4 run number 555. This filter run showed "classic" filter ripening. The dates were 9-12.1.92.*

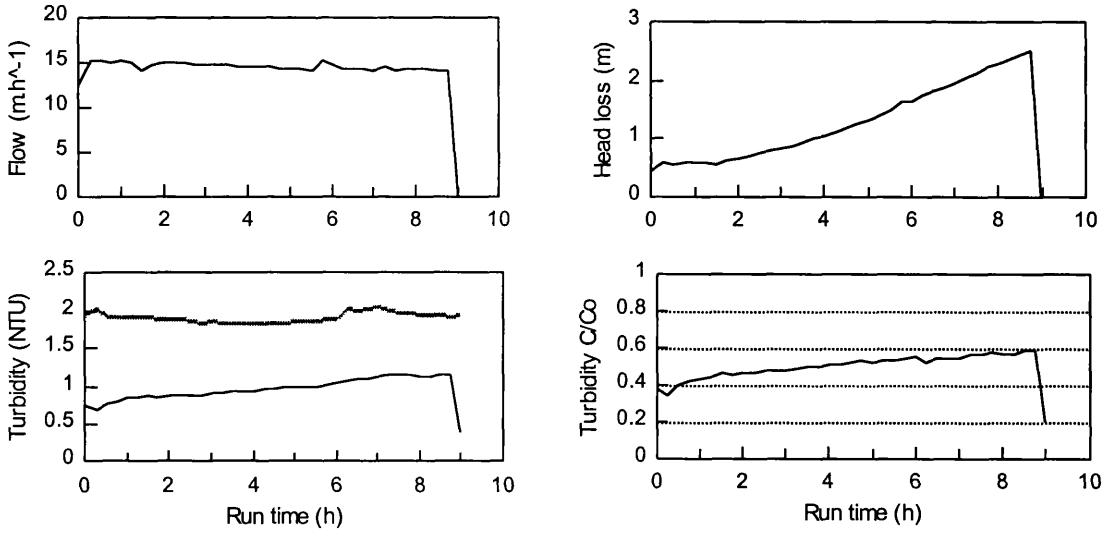

Figure 5.13. *Filter performance data for column 4 run number 878, dated 10.9.92. This filter run showed no ripening, just deteriorating removals.*

In the third case the filtrate showed no ripening and negligible change in removals during the run. This is shown in figure 5.14.

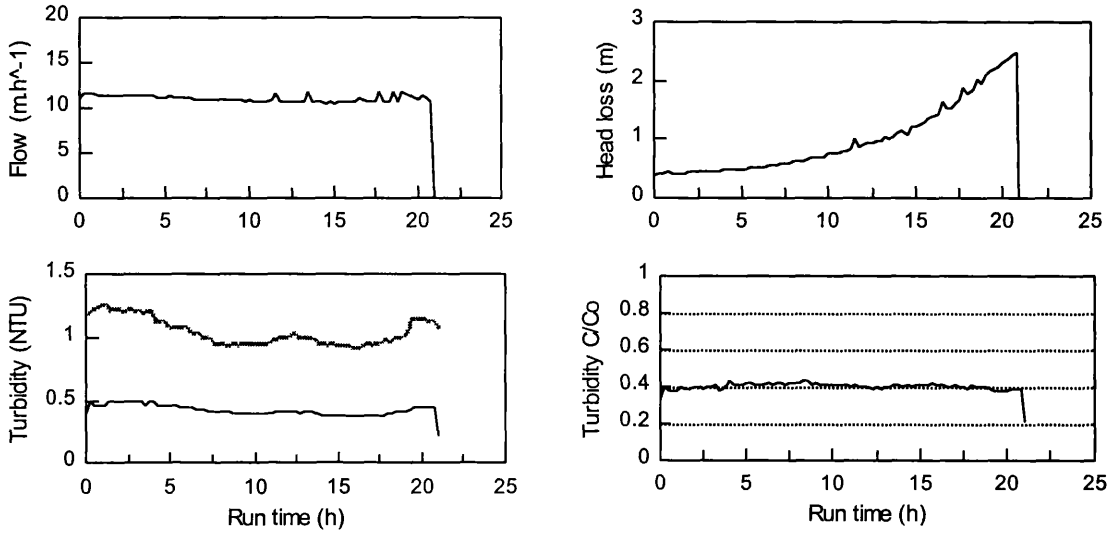

Figure 5.14. *Filter performance data for column 4 run number 782, dated 14.6.92. This filter run showed little evidence of ripening and little change in removals throughout the run.*

In the fourth case there was a steady improvement in turbidity removals throughout the run. This was arguably a constant "ripening", but this does not conform to the definition of ripening as an initial period of improving filtrate quality ("classical ripening"). This is shown in figure 5.15.

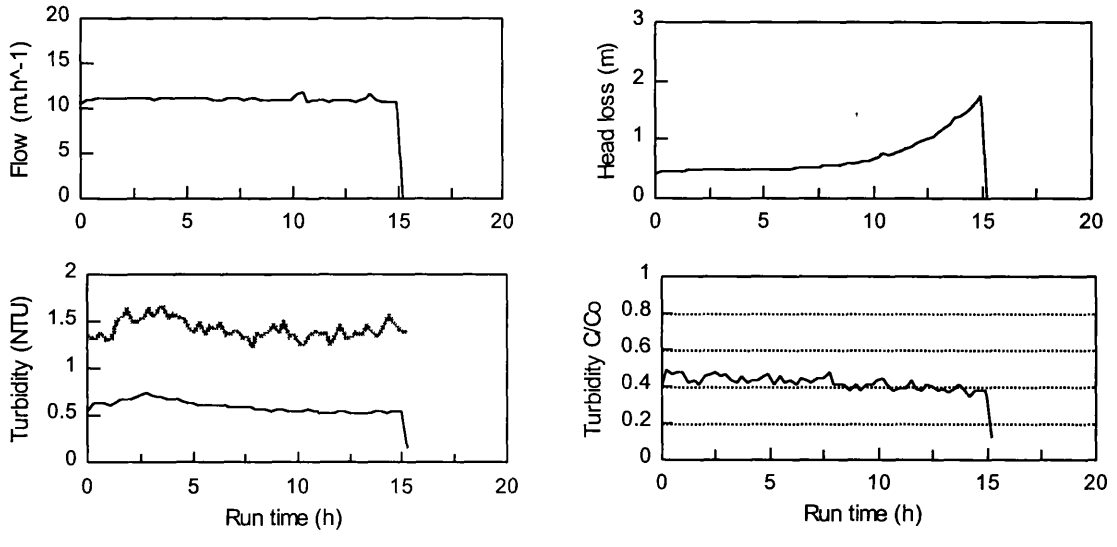

Figure 5.15. *Filter performance data for column 4 run number 729, date 8.5.92. This filter run showed little evidence of ripening but a steady improvement in removals throughout the run.*

The degree to which filtrate and influent turbidities were linked and the resulting relatively constant turbidity removal during each run are shown in figure 5.16, where influent and filtrate turbidities are plotted on separately scaled axes to show how sensitive filtration performance appeared to be. The removal data for this run were shown in figure 5.11.

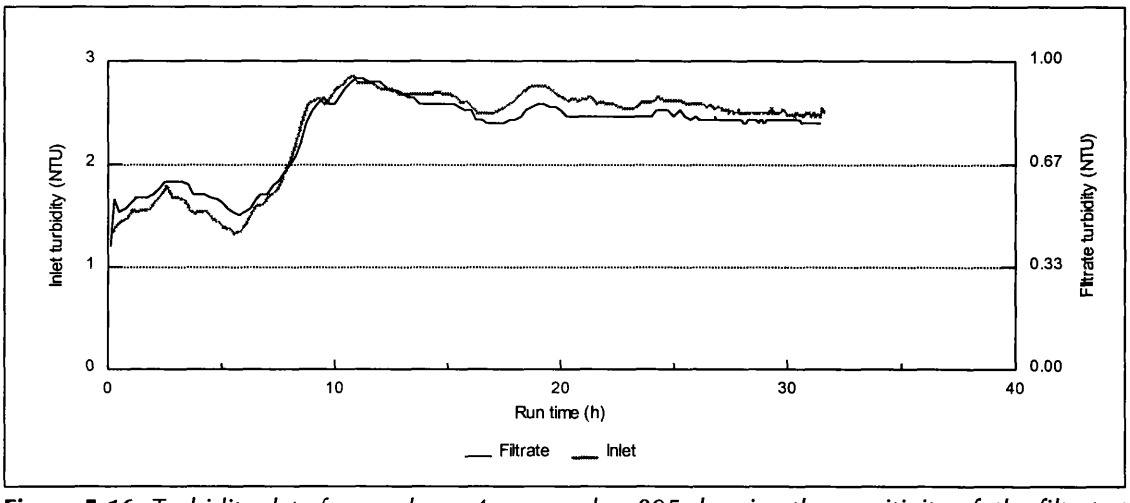

Figure 5.16. *Turbidity data from column 4 run number 895 showing the sensitivity of the filtrate to changes in inlet turbidity.*

5.4.2 RIPENING IN CHEMICALLY DOSED FILTERS

EXPERIMENTAL

Column 5 was operated from 12.9.91 as a dual media filter with 600 mm 1.7 - 2.5 mm anthracite upon 600 mm 0.6 - 1.18 mm sand. There were three further operating conditions added to the ones examined with the undosed column 4: pre-ozonation, iron(III) sulphate addition, and both chemicals added together. There were a number of runs within which ozone and/or iron were started or stopped, and others where doses were changed. It is important to consider these changes in assessing filter performance. The chemical doses were estimated from pump and ozone generator settings as discussed earlier. Data were logged every 15 minutes.

RESULTS

Figure 5.17 shows the results from just under a month of operating the dual media filter column 5 with changes to the chemical pre-treatment (filter runs 554 to 575). The filter had been in service 6 months and should be considered biologically mature. The graphs show stored water and filtrate turbidity and head loss. The flow rate was a constant 17 $m.h^{-1}$. The iron was injected downstream of

the ozone contactor. The ozone dose was high - values between 2.6 and 3.5 $mg.l^{-1}$ being calculated from the daily readings, except where stated. The filter runs were terminated at 1.8 m head loss, except those terminated for backwashing when the iron was turned off.

From time A to B no pre-treatment was applied. At time B pre-ozonation started and the filtrate quality improved. A backwash took place between B and C. Fluctuations in the stored water turbidity made it unsafe to conclude anything about the ripening following this backwash. At time C the filter was backwashed and a new run commenced with iron dosing. A clear ripening period followed this. At time D, during the same run, the ozone was started and a further quality improvement was noted, taking the shape of a rapid ripening period.

Between D and E one complete run with ozone and iron dosing took place. It showed a classic ripening curve. The slight deterioration in turbidity at E was due to turning the chemicals off just prior to the backwash to ensure they were not in the feed water to interfere with the next run.

Between times E and F the filter received no chemical dosing. The tendency of the filtrate turbidity to follow fluctuations in the stored water was clear. This type of filter curve may be termed "S"-shaped. The head loss curve was an exponential type, typical of no coagulant dosing. The flattening of the head loss curve at the end of the run was due to the filter entering declining rate mode once the outlet valve was fully open and the water level in the column at maximum.

At time F the ozone was restarted at the lower dose of approximately 0.7 $mg.l^{-1}$. This assisted with turbidity removal. At time G the iron was restarted. Again this resulted in an improvement, similar to ripening, which was tailing off when the next backwash took place. The run between G and H was ozone and iron dosed, but due to the lower ozone dose the turbidity out of the ripened filter was not as low as that obtained between D and E, and, although the turbidity was clearly higher, the impact of turning the ozone off at time H was not as marked as was seen later at time L. The filtrate turbidity between H and I fluctuated slightly suggesting the iron dose may have been insufficient. The run length was longer with the ozone off reflecting the lower capture efficiency.

At time I the iron was turned off and the filtrate rapidly deteriorated. The deposits already collected on the filter appeared to have an affect on the filtrate turbidity because the filtrate did not reflect the variations in the stored water turbidity until the filter had been backwashed and the next run started. The filtrate turbidity in the run up to time J was clearly S-shaped. At time J the ozone was restarted and the turbidity improved.

Ozone and iron were dosed between from times K to L, and M to N. The filtrate was lower in turbidity than from L to M when the ozone was off, and from N to O when only ozone was applied. The backwash failed at time O so the following run was deleted. At time P the filter received iron only. At Q the ozone was started. Two pieces of evidence suggested that the iron dose was insufficient: the head loss for all the iron dosed or ozone and iron dosed runs had been slightly curved, and the ripening period had been prolonged, particularly between Q and R, where the filter appeared to ripen for the entire time (22 h between the start of the run and R). At R the iron dose was increased, and again shortly afterwards at the two times marked *. The following filter run showed lower turbidities and a shorter ripening time. The following sections examine some of the sequence of runs in figure 5.17 to draw comparisons.

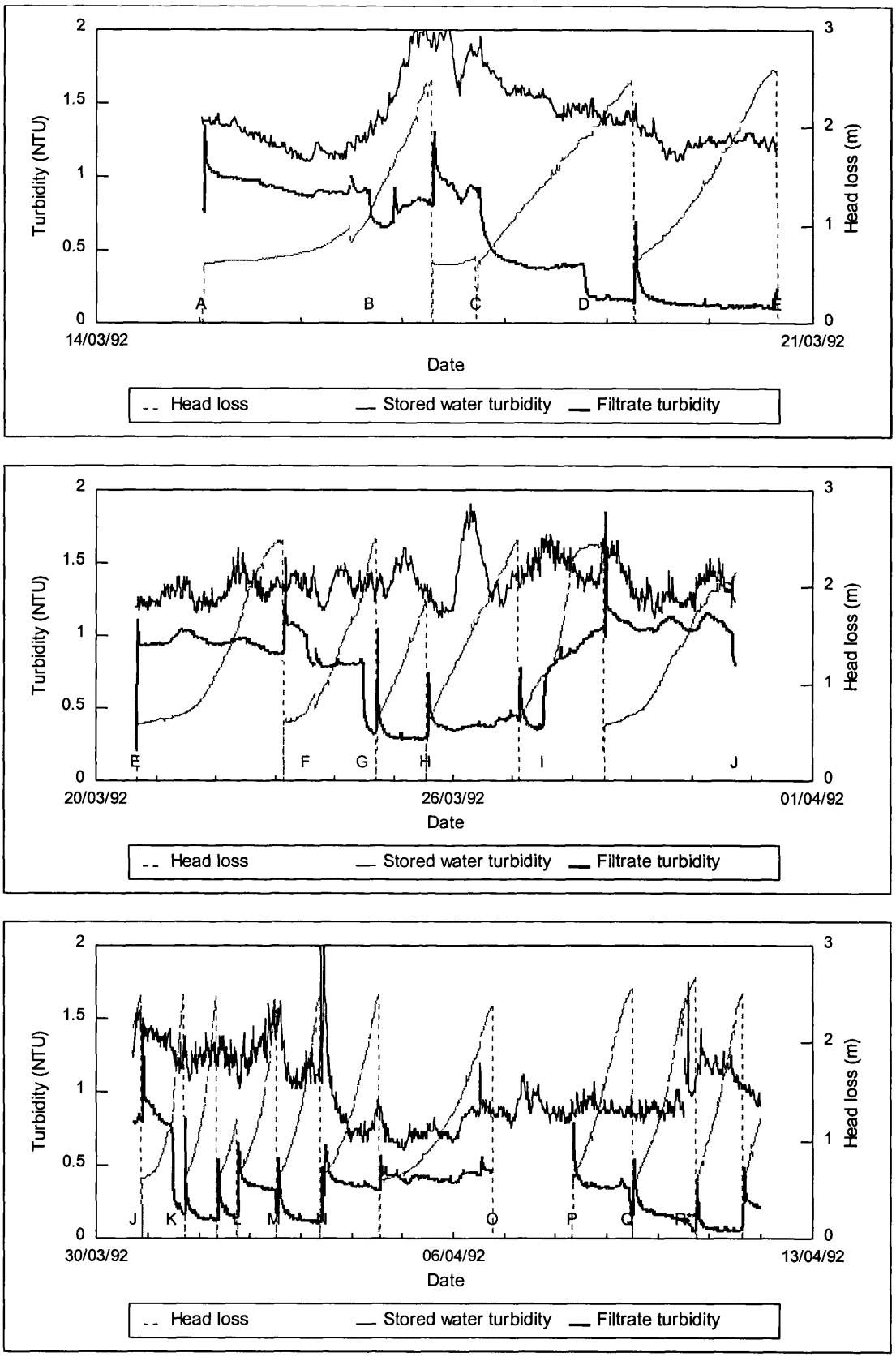

Figure 5.17. *Results of experiments showing the impact of the different applied chemical dose conditions on the filtrate turbidity from column 5 during March and April 1992, runs 554 - 575. The alphabetical characters denote changing chemical events, described in the text.*

5.4.3 THE INFLUENCE OF PRE-OZONATION ON RIPENING

METHOD

Two consecutive filter runs were examined to determine the effect of pre-ozonation on ripening in the dual media column 5 filter. The filter was operated at 15 $m.h^{-1}$ up to 2.5 m head loss. Run 633 was dosed with a calculated 2.2 $mg.l^{-1}$ of ozone for the first 82 h of operation. The ozone was turned off approximately 8 h before run 634 so that no residual effect of ozonation would be carried into the next run. Run 633 took place from 15-19.6.92 and run 634 from 19-24.6.92. Data were logged every 15 minutes.

RESULTS

The details of runs 633 and 634 are plotted in figures 5.18 and 5.19 respectively. The ozone dosed filtrate behaved in a similar manner to the undosed filtrate in that it followed fluctuations in the influent turbidity and produced a fairly smooth turbidity removal trend. As was observed with column 4 only the turbidity removal data were useful to determine whether ripening was occurring.

The influent turbidity averaged 1.14 NTU in run 633 and 1.18 in run 634. The mean filtrate and C/Co for the undosed filter were 0.75 NTU and 0.63, and for the ozone dosed part of run 633, 0.54 NTU and 0.47. The values changed to 0.66 NTU and 0.66, with a mean influent turbidity of 1.00 for the undosed portion of run 633. These observations agree with the findings from spot samples from column 1, suggesting these runs were representative of the effect of pre-ozonation.

The influence of the ozone in improving filtrate turbidity is seen by comparing runs 633 and 634. In figure 5.20 both runs can be seen to show improving removals for at least 30 h and continuing throughout the run until the chemical dosing changed. Figure 5.20 also shows the detail at the start of the run (logarithmic time axis). The ripening behaviour showed that pre-ozonation shifted the quality but did not greatly influence the timing of the ripening changes.

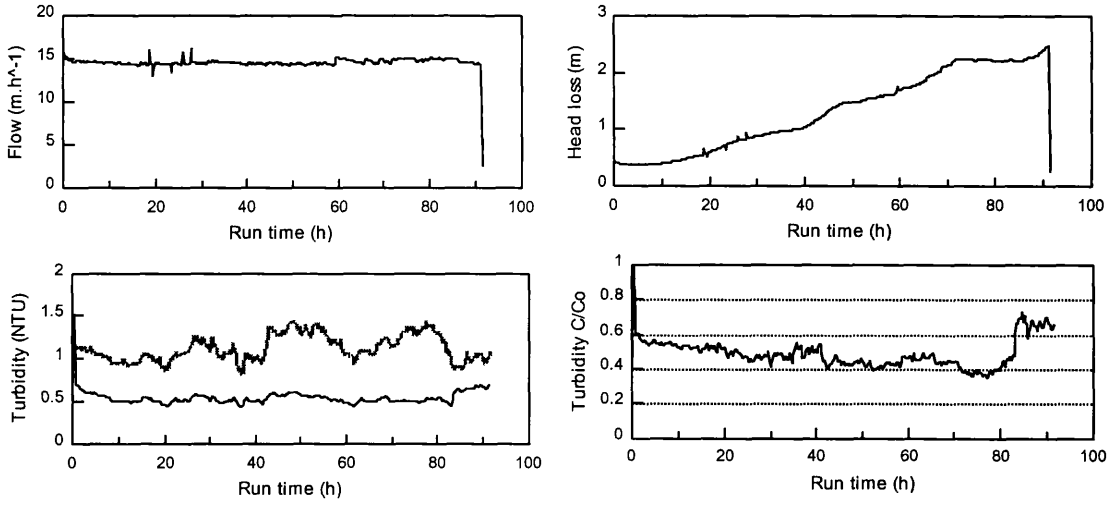

Figure 5.18. *Filter performance data for column 5 run number 633, date 15-19.6.92. This filter run showed the influence of pre-ozonation for the first 82 h of operation.*

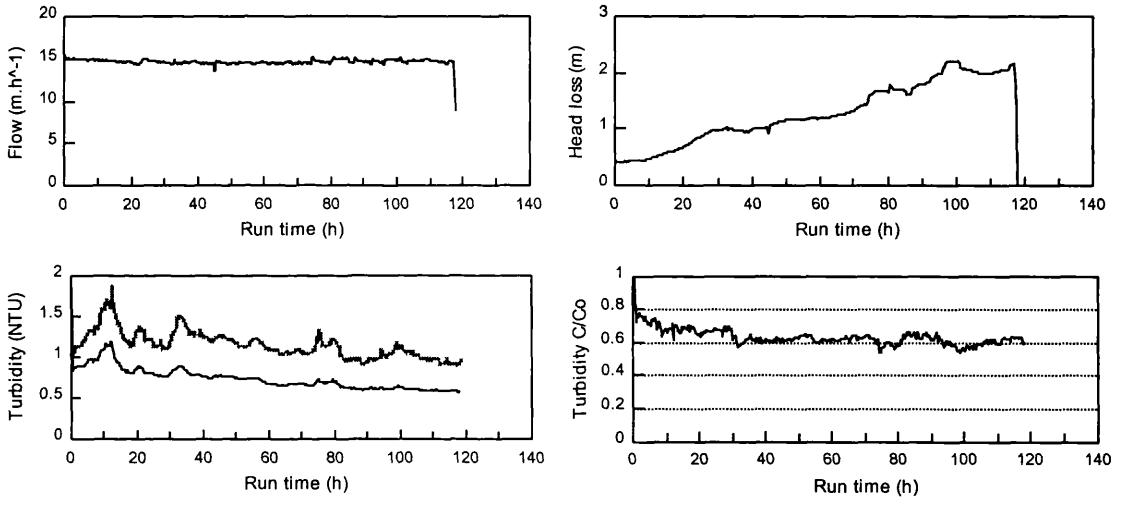

Figure 5.19. *Filter performance data for column 5 run number 634 with no pre-ozonation, date 19-24.6.92.*

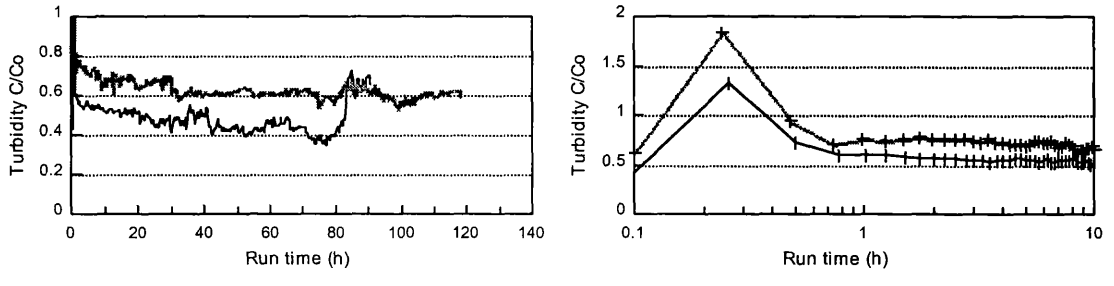

Figure 5.20. *Comparison of turbidity removal data from filter runs 633 and 634. Run 634 is shown in light grey.*

5.4.4 THE INFLUENCE OF COAGULATION ON RIPENING

Figure 5.17 showed that the use of iron, alone or with ozone, produced classic ripening curves. It is necessary to examine how coagulant dosing influenced ripening and what changes in ripening resulted from ozone dosing.

METHOD

A number of filter runs from column 5, closely linked in time, have been examined to provide a comparison of the effects of iron dosing and iron plus ozone on filter ripening in a dual media filter. The filter was operated at 17 $m.h^{-1}$ to 1.8 m head loss. Iron was dosed at a calculated value of 0.31 $mg.l^{-1}$ and remained unchanged for the two runs. In run 556 ozone was not dosed until 24 h filter run had elapsed. It was then started in preparation for run 557. The ozone dose was calculated as 3.3 $mg.l^{-1}$. The ozone was turned off for the last half hour of run 557. Where time was plotted on a logarithmic axis the initial sample time has been changed from 0 h to 0.1 h.

RESULTS

Figures 5.21 and 5.22 show filter runs 556 and 557 (C - E in figure 5.17). In each case the filter flow rate was reasonably constant and head loss development was much closer to linear than had been achieved without coagulant. Ripening was apparent in both the filtrate turbidity data and in the turbidity removal trends. It was noted that the stored water turbidity fell gradually over the first few hours of each run so, bearing in mind the situation when coagulant was not used, it was not possible to say from these results whether turbidity data could be used to indicate ripening when iron is dosed. However when influent turbidity climbed or was constant at the start of the run ripening was measured with both filtrate turbidity and turbidity removal when iron or iron and ozone were dosed. Figure 5.24 shows this for run 574, ozone 3.3 $mg.l^{-1}$, iron 0.62 $mg.l^{-1}$, and figure 5.25 shows run 576 with iron alone, dosed at 0.62 $mg.l^{-1}$.

Figure 5.21 turbidity and removal trends show classic ripening taking up to 10 h to complete. The ripened turbidity was 0.4 NTU, a removal of 75%. The start of ozonation promoted a second "ripening" event, taking 1 - 2 h, as filtrate turbidity fell to 0.16 NTU and removals climbed to 89%. Figure 5.22 shows similar ripened turbidity values and removal values for similar operating

conditions. An initial ripening period of around 5 h can be seen with both turbidity and removal trends, although there was some further slight improvement later in the run. The deterioration in quality at the end of the run was due to the ozone being turned off.

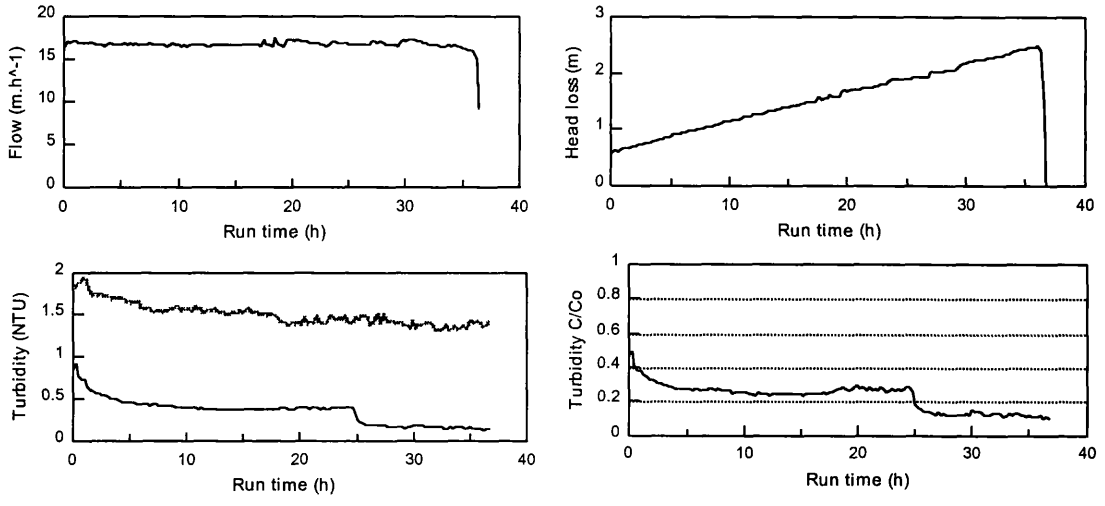

Figure 5.21. *Filter performance data for column 5 run number 556, date 17-19.3.92. This filter run showed the influence of iron dosing for the first 24 h of operation. Pre-ozonation was applied for the remainder of the run.*

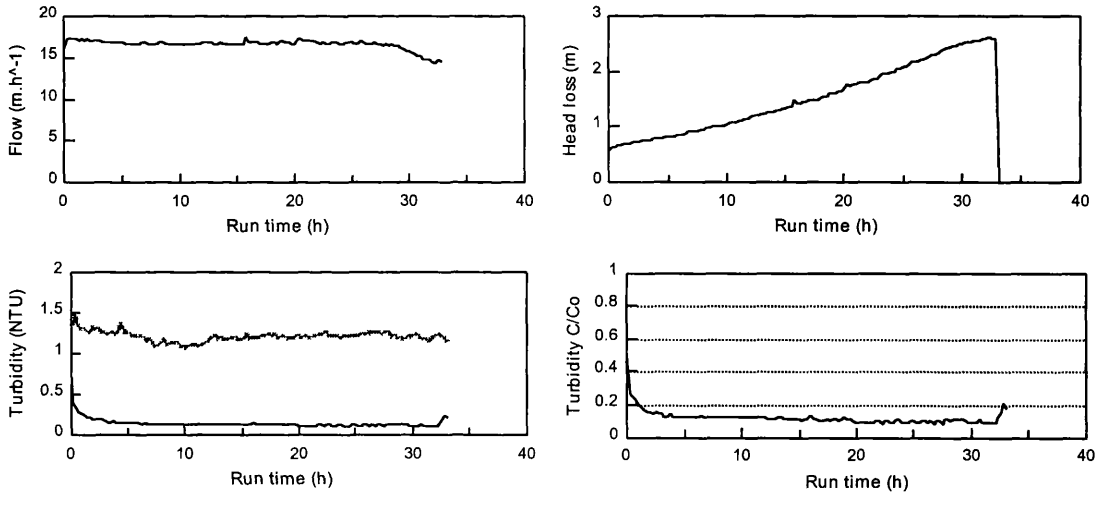

Figure 5.22. *Filter performance data for column 5 run number 557, date 19-20.3.92. This filter run showed the influence of iron and pre-ozonation for the first 32 h of operation.*

The turbidity removal data from these runs are compared in figure 5.23 using linear and logarithmic time axes. Figure 5.23 suggests that the initial spike and the general shape and duration of the ripening curve were similar despite the ozone. The only significant difference was the degree of removal attained was enhanced by pre-ozonation. This was achievable by commencing the ozone at any time during the run.

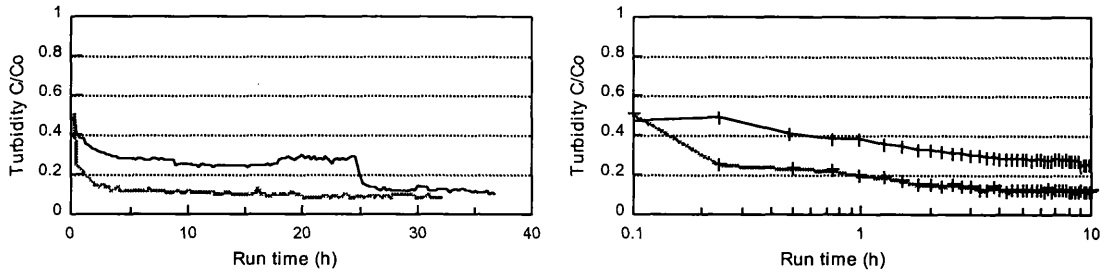

Figure 5.23. *Comparison of turbidity removal data from filter runs 556 and 557. Run 557, with pre-ozonation, is shown in light grey.*

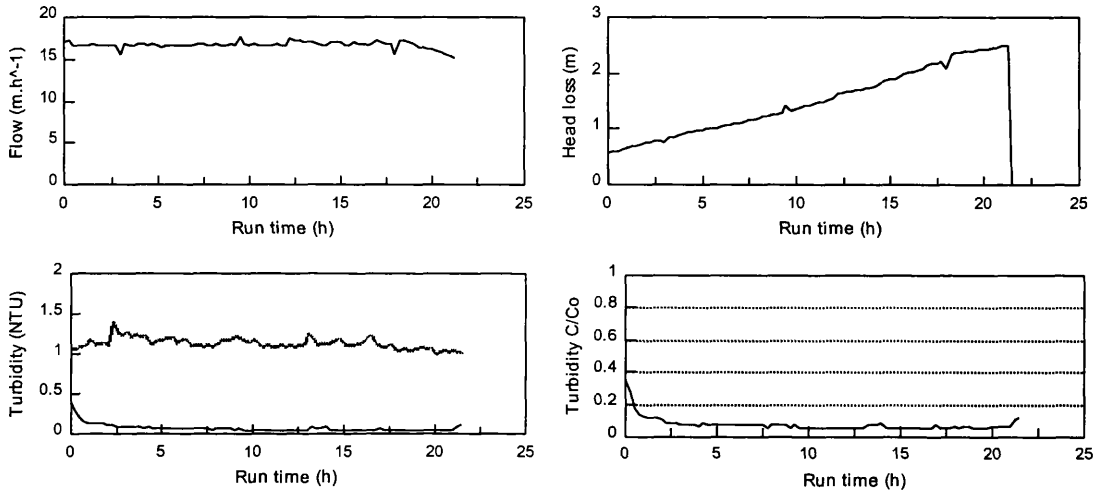

Figure 5.24. *Filter performance data for column 5 with iron and ozone: run 574, date 10-11.4.92.*

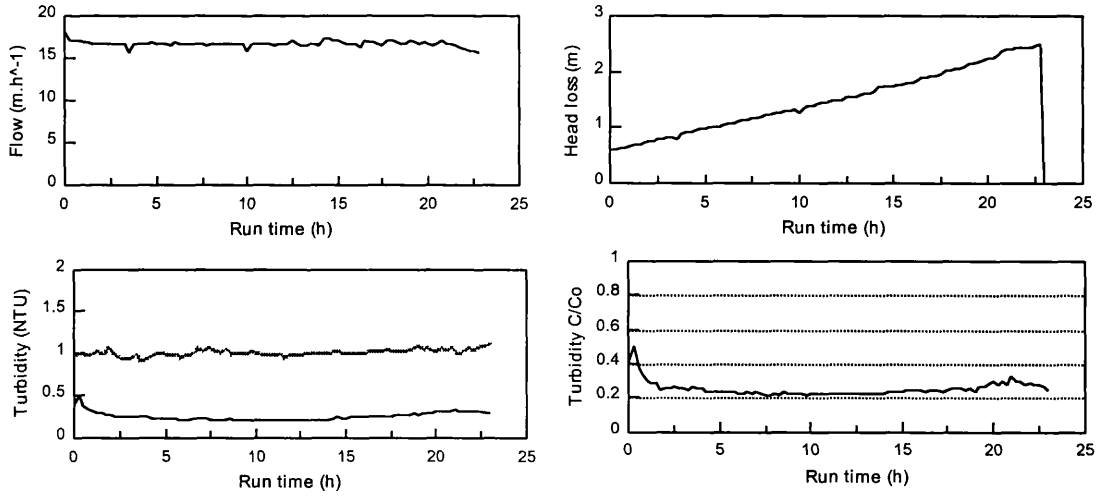

Figure 5.25. *Filter performance data for column 5 with iron only: run 576, date 12-13.4.92.*

5.4.5 THE INFLUENCE OF OZONE COMBINED WITH IRON ON RIPENING

The influence of different ozone doses is shown by figure 5.26, where the data from run 560 (G - H in figure 5.16), which had an ozone dose of 0.7 $mg.l^{-1}$ are compared with run 557 (figure 5.21) with 3.3 $mg.l^{-1}$ of ozone and run 556, which had no ozone (figure 5.20). With the same iron dose in each run (0.31 $mg.l^{-1}$) the impact of increasing the ozone dose was to improve the quality of filtrate throughout the ripening and best operating phases. The ozone dose influenced the ripened removals but not ripening times. Run 560 was backwashed early at 1.2 m, causing the shorter run time.

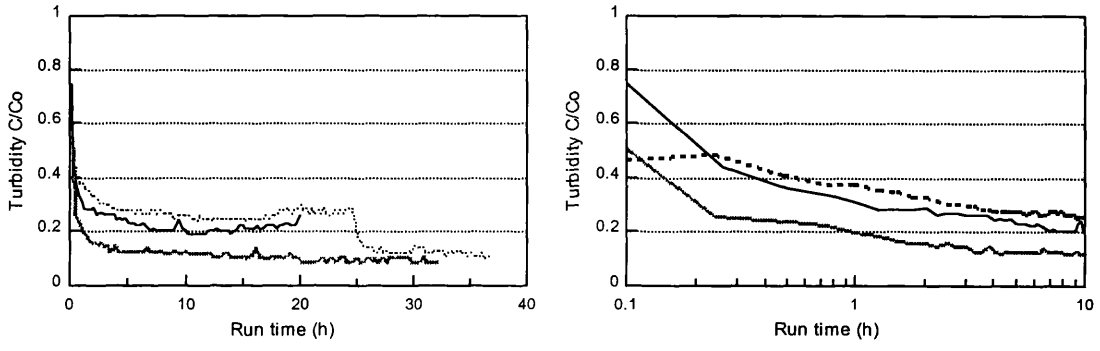

Figure 5.26. *Comparison of turbidity removal data from filter runs 556, 557 and 560. Iron doses were the same. Run 556 (no ozone) is shown as a broken line. Run 557, which had a higher ozone dose, is shown in light grey, run 560 with a low ozone dose is the black line.*

5.4.6 SUMMARY OF THE EFFECTS OF CHEMICAL TREATMENT

Figure 5.27 shows the results from the first six hours of operation from 192 runs with the dual media column 5. The curves show mean turbidity removals for each of the 4 different chemical dosing options. There were 11 runs without coagulant or ozone, 10 with pre-ozonation, 40 with iron alone, and 128 with iron and ozone.

A ripening period of under one hour was common to all four trends. The differences in filtrate turbidity with different chemical treatment, illustrated with individual runs in figure 5.17, was confirmed for turbidity removals in figure 5.27. These means were calculated from data collected throughout the period from autumn 1991 to spring 1993, and differences in coagulant dose,

applied turbidity, and filter flow rate (range 9 - 17 $m.h^{-1}$) were not taken into account. These are discussed in the next section and the next chapter.

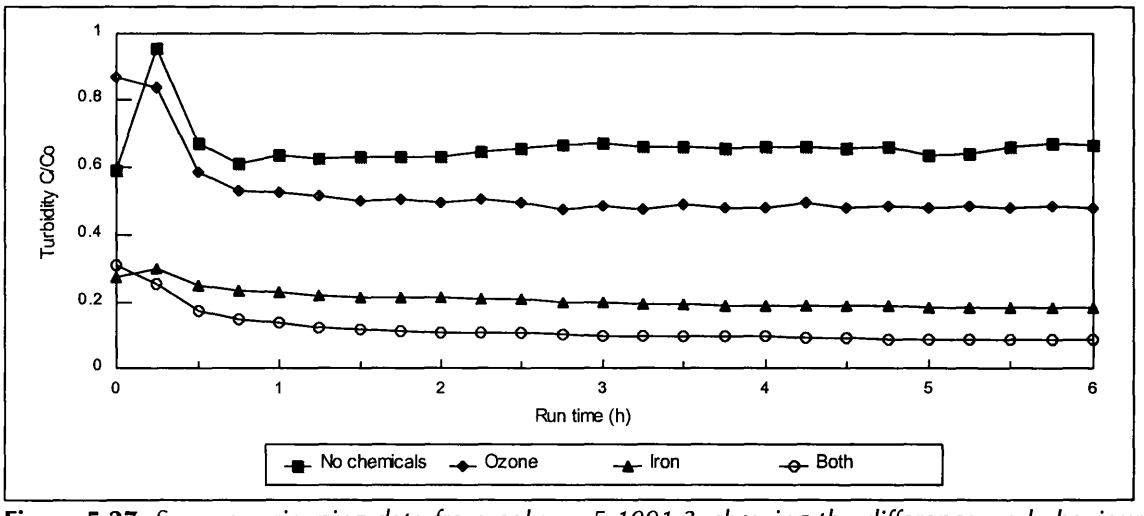

Figure 5.27. *Summary ripening data from column 5 1991-3, showing the difference on behaviour with different chemical treatments.*

5.5 THE INFLUENCE OF THE SIZE OF THE IRON DOSE ON RIPENING

METHOD

The data from a week's operation of filter 1 at three iron doses were examined (filter runs 488 - 496). The ozone dose was held steady at 1.6 $mg.l^{-1}$, flow rate was 15 $m.h^{-1}$. Stored water turbidity was relatively constant over this period. Three filter runs were selected from this week and filtrate turbidity, turbidity C/Co and head loss were plotted against filter run time to examine the effect of different iron doses. Further examination of another week's data was carried out to show that these were not unique results (filter runs 457 - 465), and then confirmed with similar findings from column 5, runs 526 and 527.

The initial inlet flow in column 1 was allowed to pass out with the dirty backwash water to prevent the filter getting an excess ozone or iron dose due to build-up of chemicals in the pipework during the backwash when the inlet valve was closed. This was also the case with column 5, although since it shared a common feed with column 6, and excess influent water was allowed to overflow the column, an initial overdose was not a potential problem. The filters were backwashed at 1.8 m head loss. The data were sampled every 15 minutes. The sample at time 0 h was actually from

between 1 and 15 minutes after the backwash finished, because the backwash sequence took place without reference to the logging intervals.

RESULTS

Figure 5.28 shows the effect of the change in iron dose was a quite marked change to the filtrate turbidity trends.

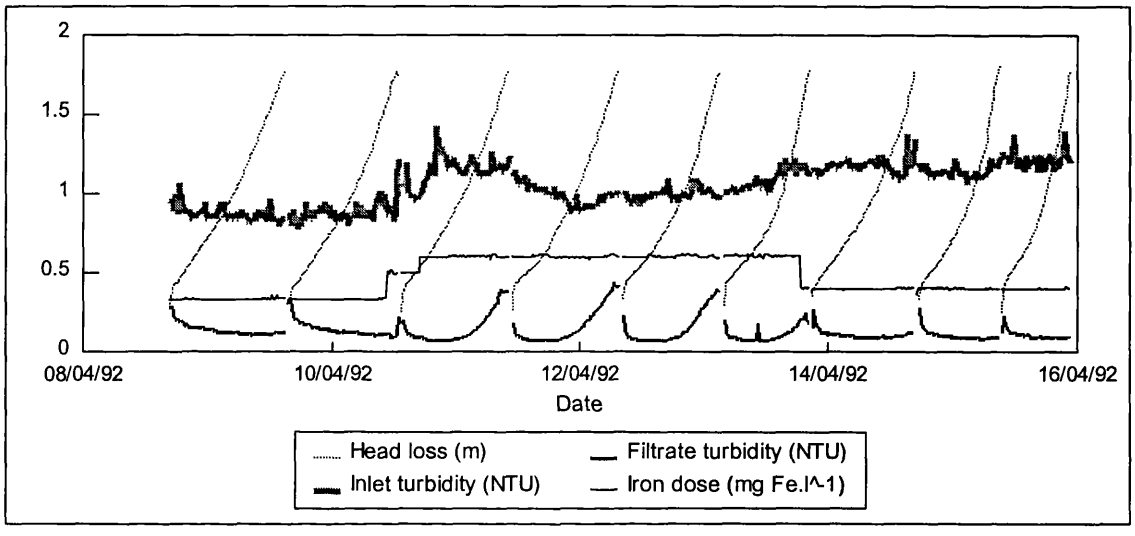

Figure 5.28. *Effect of changing coagulant dose on filtrate turbidity, column 1 runs 488 - 496.*

To clarify this point three filter runs at three different doses were examined. The three selected were those closest together in time, excluding any run during which the chemical dose was changed. These runs were 488, 491 and 494, for which the mean influent turbidities were 0.9, 1.0 and 1.2 NTU, and iron doses were calculated at 0.33, 0.60 and 0.40 mg $Fe.l^{-1}$ respectively. Figures 5.29 and 5.30 show the results for turbidity and turbidity C/Co. The different dose does not appear to have influenced rate of head loss development (figure 5.31).

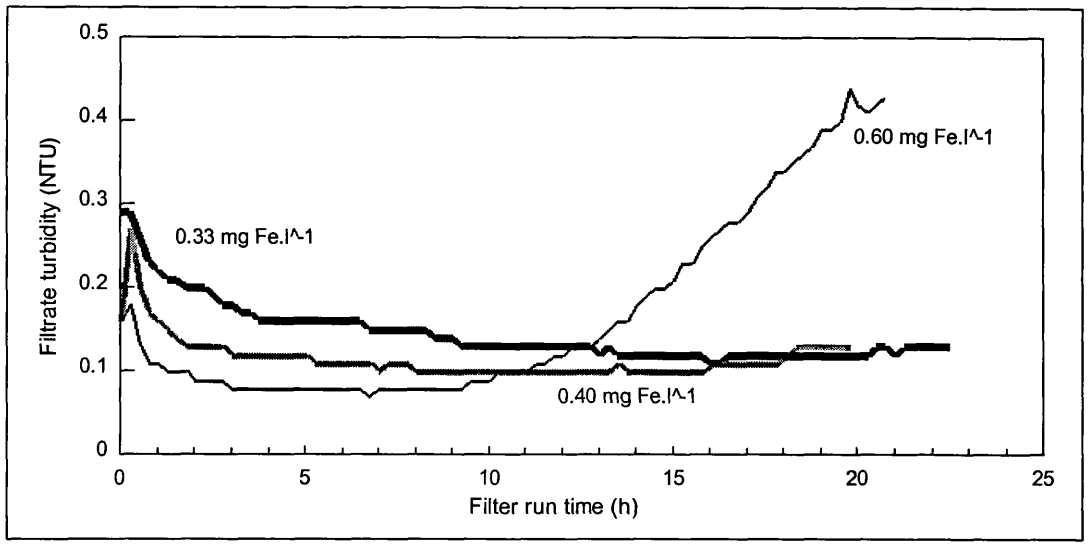

Figure 5.29. *Effect of changing coagulant dose on filtrate turbidity, column 1 runs 488, 491, 494.*

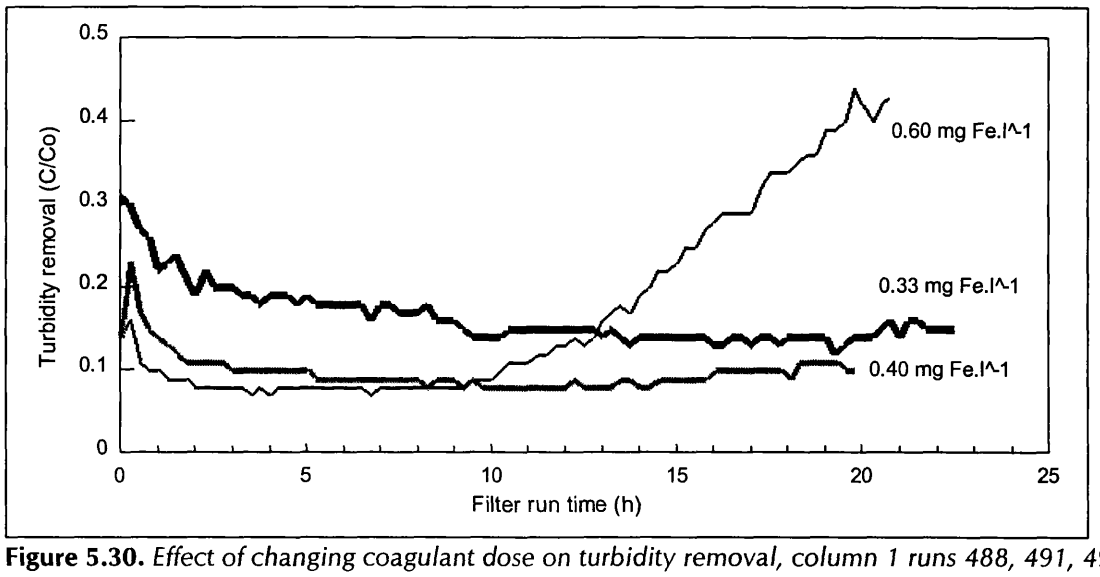

Figure 5.30. *Effect of changing coagulant dose on turbidity removal, column 1 runs 488, 491, 494.*

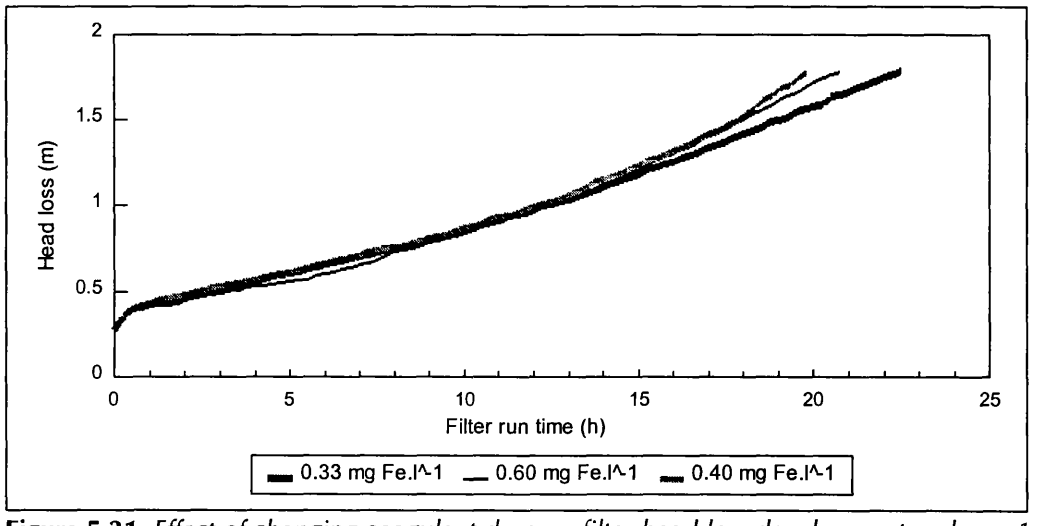

Figure 5.31. *Effect of changing coagulant dose on filter head loss development, column 1 runs 488, 491, 494.*

The impact of the coagulant dose is again illustrated in the following example from column 1 during March 1992, runs 457 - 465 (figure 5.32). The ozone dose was 1.7 $mg.l^{-1}$, and the flow rate 15 $m.h^{-1}$. Stored water turbidity ranged between 1 and 2 NTU.

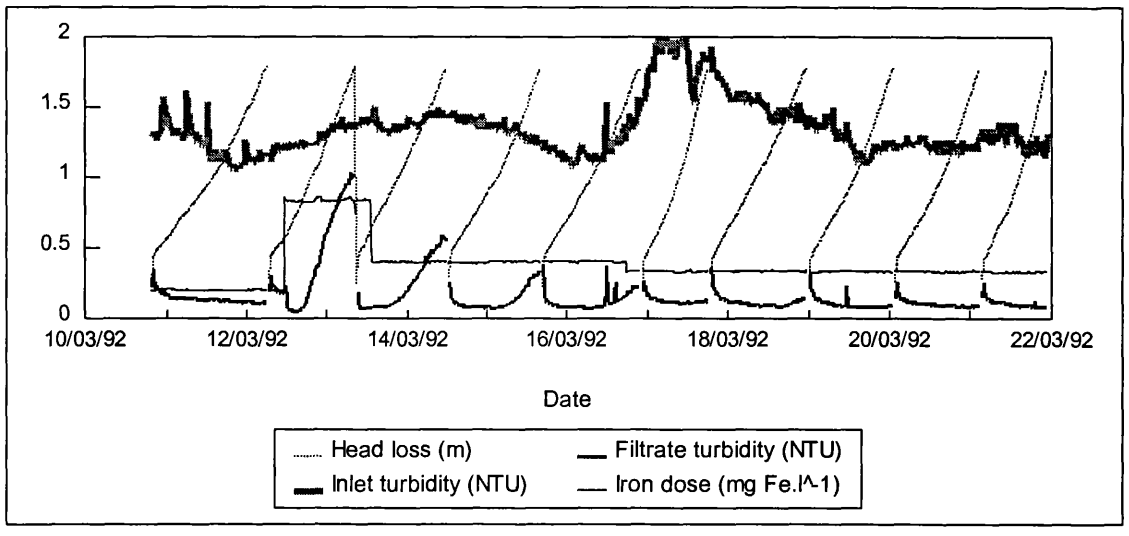

Figure 5.32. *Effect of changing coagulant dose on filtrate turbidity, column 1 runs 457 to 465.*

In run 457 the iron dose was 0.2 $mg.l^{-1}$. After starting the second run with the same coagulant dose as the first, the response to quadrupling the dose from 0.2 to 0.8 $mg.l^{-1}$ was a rapid improvement in filtrate turbidity. The consequences of an iron dose of 0.8 $mg.l^{-1}$ were rapid ripening and severe breakthrough. In the following run, after a short ripening period the iron dose was halved. This reduced the intensity of the breakthrough but did not eliminate it. Subsequent runs showed breakthrough until a dose of 0.33 $mg.l^{-1}$ was used. This had a longer ripening time. The five complete runs at this dose shown in figure 5.32 from 17.3.92 onwards show there was some variation in the trend after ripening with one run showing a slight breakthrough and three runs appeared to give a steady filtrate up to maximum head loss.

The data from the start of each run were combined in figures 5.33 and 5.34 to show the effect of four different coagulant doses on ripening. The short period of dosing at 0.84 $mg.l^{-1}$ is shown together with the mean data from two runs at 0.2 and 0.4 $mg.l^{-1}$ and four runs at 0.34 $mg.l^{-1}$. This showed very similar effects from the chemical dose as were noted earlier for filter runs in April, i.e. better ripening at higher doses, at little cost in head loss (figure 5.35).

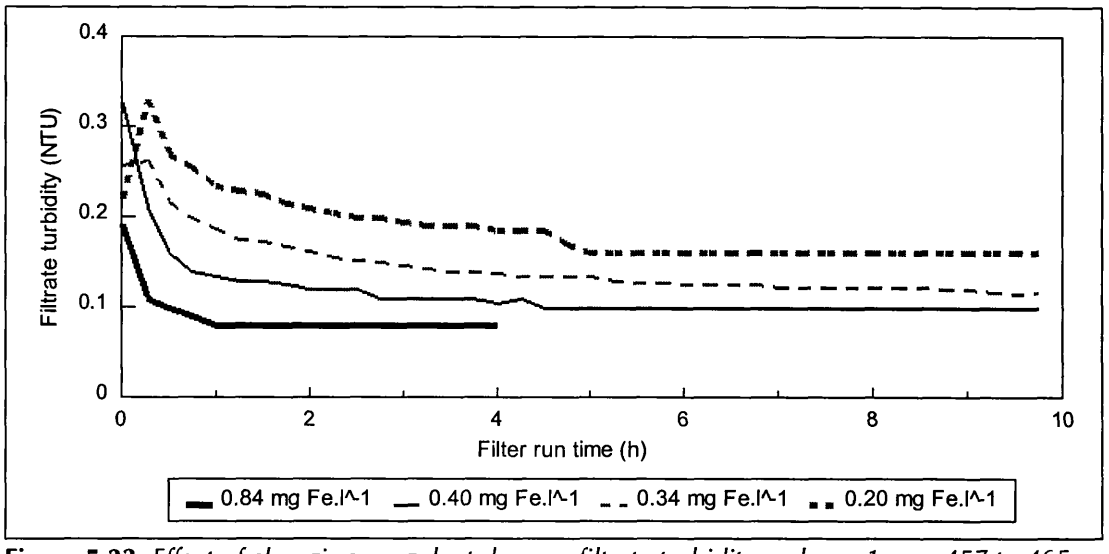

Figure 5.33. *Effect of changing coagulant dose on filtrate turbidity, column 1 runs 457 to 465.*

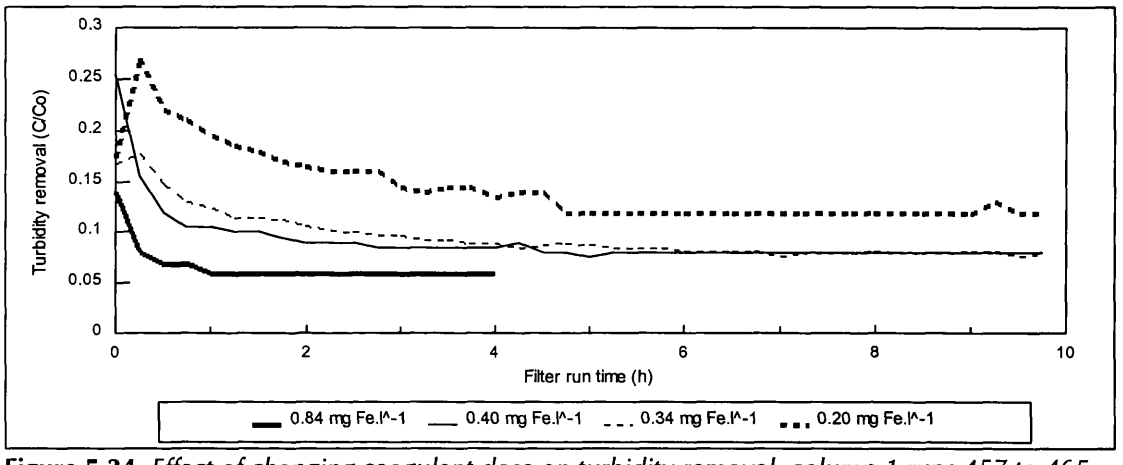

Figure 5.34. *Effect of changing coagulant dose on turbidity removal, column 1 runs 457 to 465.*

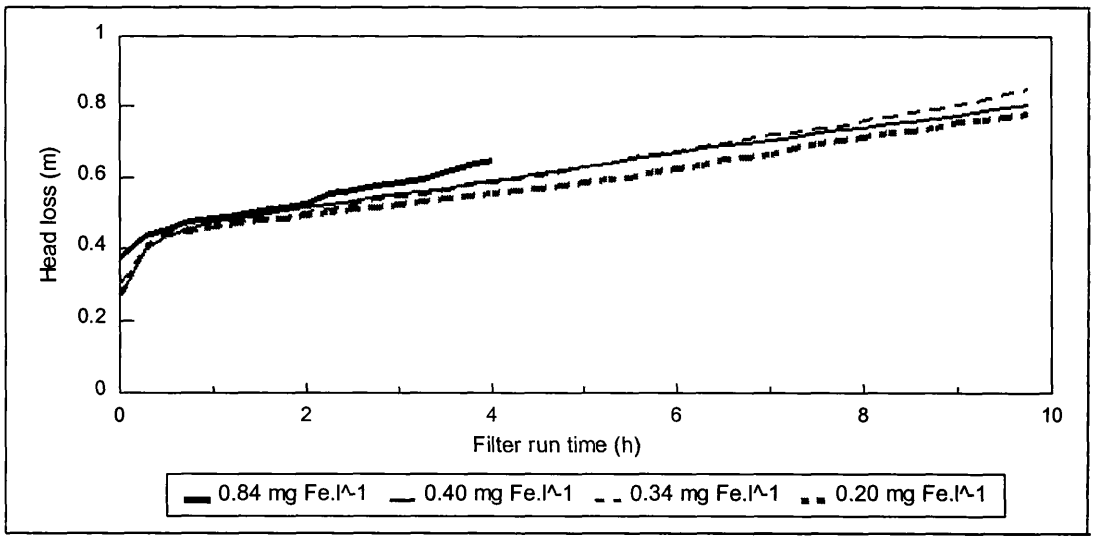

Figure 5.35. *Effect of coagulant dose on filter head loss, column 1 runs 457 to 465.*

Further support was gained from columns 5 over 22-23.1.92. Figure 5.36 shows the results from a very high iron dose of 1.71 $mg.l^{-1}$ on run 526. The dose was reduced for the final hour of run 526 to 0.34 $mg.l^{-1}$, and this continued into run 527 (figure 5.37). Ozone was dosed at 2.3 $mg.l^{-1}$.

The effect of the higher dose was to shorten ripening time to about half and hour, and to produce a low ripened turbidity, but this was followed by breakthrough taking place with less than half the filter run time elapsed, i.e. with less than half the available head used up (figures 5.36, 5.40).

The reduction in iron dose towards the end of the run caused an improvement in filtrate turbidity, even at high head loss and with significant breakthrough. It would appear that breakthrough was a function of the chemical dose rather than the saturation of the bed with respect to particle removal.

In the following run, with the lower iron dose, ripening was slower (figure 5.37). The rapid phase of ripening lasted 1 h, and the slower phase of ripening did not reach a minimum until nearly 10 h were elapsed. This minimum was not as low as the high dose had produced. Breakthrough was eliminated (figure 5.38) and the lower dose also produced a longer filter run, and lower rate of head loss development (figure 5.39).

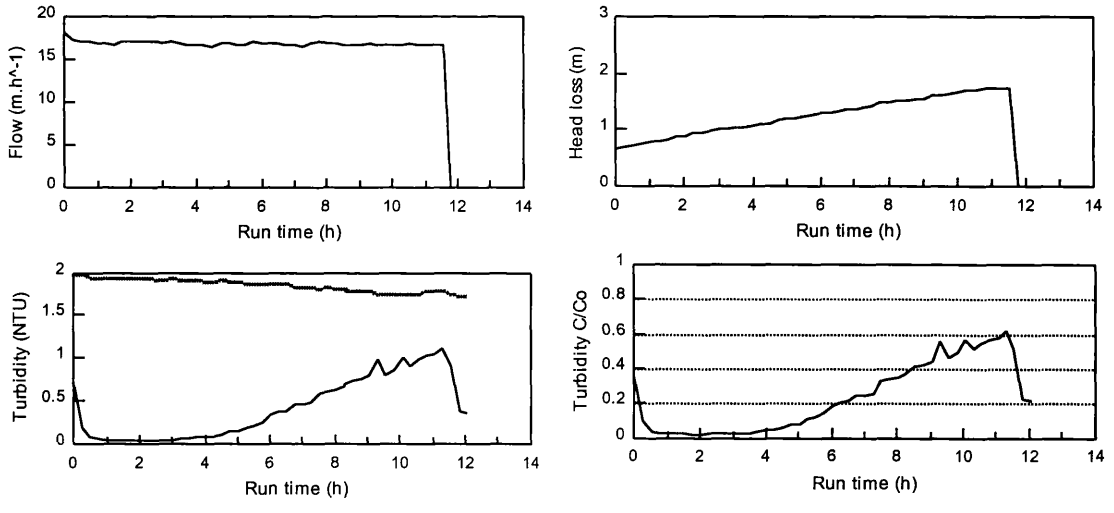

Figure 5.36. *Column 5 run 526 with a high iron overdose, 22.1.92.*

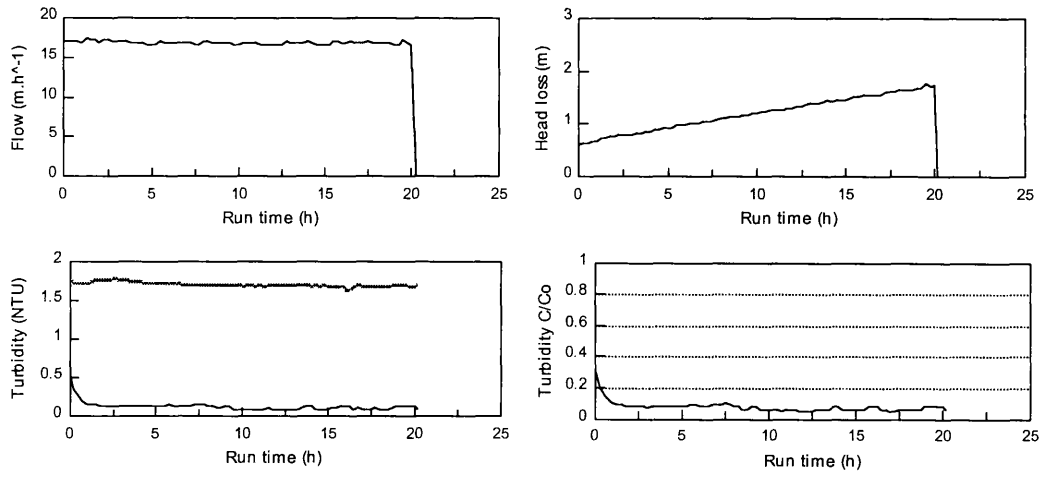

Figure 5.37. *Column 5 run 527 with a low iron dose, 22-23.1.92.*

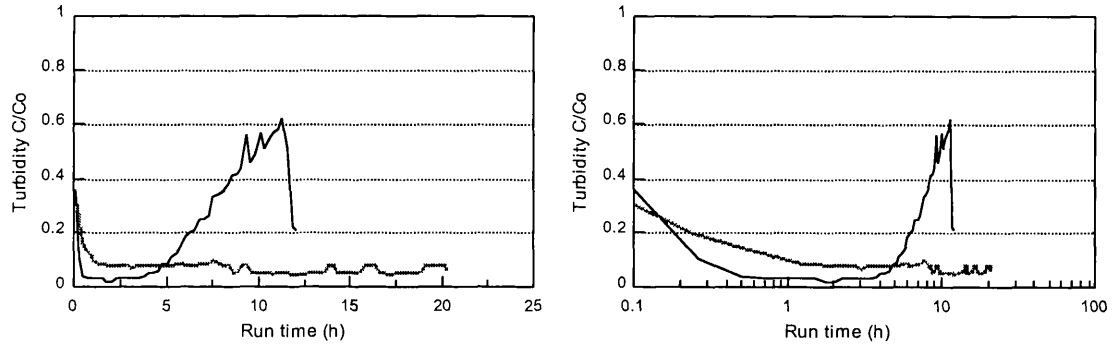

Figure 5.38. *Comparison of turbidity removals at different iron doses. Run 527, with the lower iron dose, is shown by the grey line.*

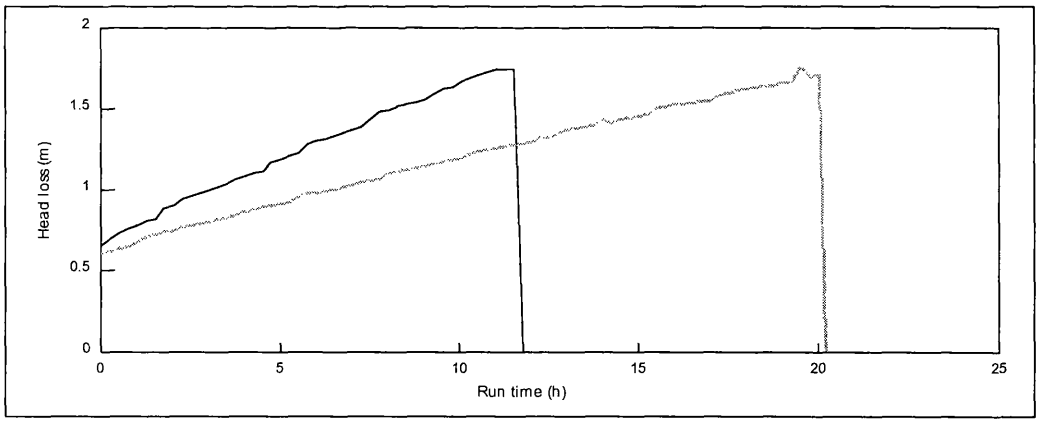

Figure 5.39. *Comparison of head loss development at different iron doses. Run 527, with the lower iron dose, is shown by the grey line.*

It was thought that ripening might be affected by different doses because of the difference in potential for floc formation from the two doses. Despite a dose ratio of 5:1 there was no evidence from figure 5.39 that ripening was five times faster with the higher dose.

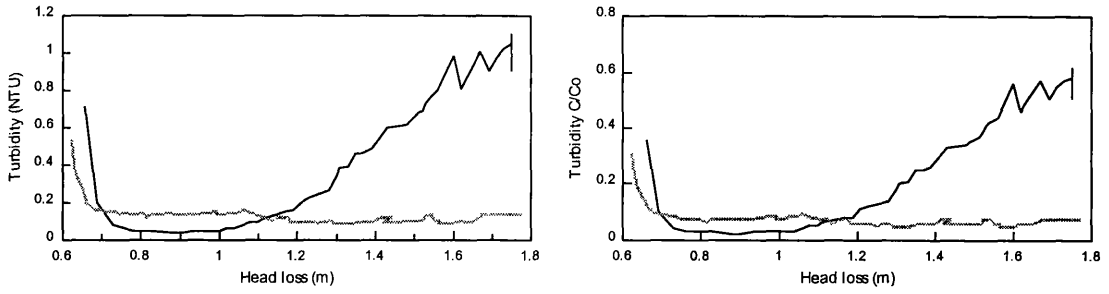

Figure 5.40. *Examination of the relationship between filter ripening and head loss at different iron doses. Run 527, with the lower iron dose, is shown by the grey line.*

An attempt to normalise ripening with respect to deposit quantity, as measured by filter head loss, is shown in figure 5.40. It does not appear that ripening at the different doses was due to the amount of the deposit, measured indirectly from head loss, since the ripening did not take place at the same differential pressure. This does not allow for different deposition sites within the depth of the filter or different deposit densities and morphologies at the two doses.

5.6 DISCUSSION

5.6.1 THE INFLUENCE OF CHEMICAL DOSING ON RIPENING

Ripening has been defined as the improvement in filter performance following backwashing. The data presented have shown that there may be several different types of filter performance after backwashing depending, primarily, on chemical treatment. Some ripening trends were short, others very long, and many showed a rapid ripening phase followed by a slower ripening period. The results in figure 5.17 showed that an improvement in quality, could happen at any time in a filter run when chemical treatment was changed to promote more favourable filtration conditions. It was not only observed immediately following a backwash. Since this form of ripening was not associated with the filter starting up after a backwash, it falls outside of the definition of ripening being used, and is not discussed further.

The literature makes it clear that ripening is sensitive to the chemical treatment of the water: by using coagulants it is possible to reduce filtrate turbidity and pre-oxidation may give extra assistance

in removing turbidity. The objective of this study was to examine to what degree the chemical treatment applied at the pilot plant could influence turbidity removal and filter ripening.

The work reported in this chapter has shown that chemical treatment was a key element in determining filter performance. Without taking operational or seasonal factors into account the results from spot samples, and means from entire filter runs, showed that the degree of turbidity removal was influenced by the chemical dosing. The spot sample data in figure 5.1, together with the statistics in table 5.3, showed distinct differences. The greatest benefit was seen with iron plus a high dose of ozone, then iron with a low ozone dose, iron alone, ozone alone, and finally the poorest performance was achieved by an undosed filter.

It was clear that there was some scatter in the results for all dosing conditions. The r^2 data implied that factors other than chemical dose were influencing turbidity removals. Certain samples may have been taken under sub-optimal dosing conditions whilst the dose control methodology was being established. Some scatter may have resulted from operational differences e.g. flow rate, position in the filter cycle, chemical dose, filter run length and the position of the working layer at the time of terminal head loss, and others from seasonal differences, e.g. water temperature, the impact of the numbers and types of algae present in the stored water, and different degrees of biological activity in the filter or biological response to the loads. Some of these factors are examined in later chapters.

NO DOSING

The degree to which filter ripening might take place without coagulants was not widely discussed in the literature. Under laboratory conditions little ripening might be expected (Tobiason and O'Melia, 1988). Mosher and Hendricks (1988) showed ripening was absent in field trials with undosed filters. In this study classical ripening was seldom observed with undosed filtration on a sand filter, although figure 5.27 shows it was seen in the undosed and ozone dosed runs on the dual media filter. Reasons for this are discussed in chapter 6. The relatively poor filtration performance in the absence of coagulants was to be expected from the literature.

The key point in considering ripening in undosed filters was that filtrate turbidity behaviour could not be properly understood except in the context of influent water turbidity. There was a close relationship between stored water and filtrate turbidity in the un-dosed and ozone dosed filter runs. Hence turbidity removals were the only means of reliably determining whether filter ripening was occurring. Falling inlet turbidity at the start of a run might cause filtrate turbidity to give a misleading appearance of ripening, and a rise in inlet turbidity would result in rising filtrate turbidity, potentially masking ripening.

Without coagulant the column 4 was unable to produce a filtrate of 0.1 NTU, and only produced low turbidity filtrates (< 0.3 NTU) when stored water turbidities were exceptionally low (figure 5.8). Without iron dosing the filter failed to ripen to a target of 0.1 NTU or achieve 95% removals. No ripening targets would be appropriate for undosed filters.

In each of figures 5.12 to 5.15 the head loss development was a curve. It was typical of an undosed filter to show a curved head loss development with an early lag phase, accelerating as time progressed. These examples showed that acceleration in gradient did not necessarily correspond to filtrate turbidity ripening.

Given the range of turbidities measured over the course of the trials, typically from < 1 to 4 NTU (figure 3.6), even the maximum removals of 80% achieved by column 4 (figure 5.9) might not produce filtrate turbidities below 0.8 NTU. Since removals were often as low as 30% and mean filtrate turbidities often exceed 1 NTU (figure 5.8) an un-dosed or pre-ozonated filter could not be considered a reliable barrier to *Cryptosporidium,* based on the observations of Al-Ani *et al.* (1986) and others. The ozone doses and contact times employed were unlikely to be effective against *Cryptosporidium* oocysts (Ives, 1990).

As a consequence, undosed filtration should only be considered as roughing filtration, requiring a subsequent treatment stage to ensure adequate removals of pathogenic particles. This being the

case, the presence of a ripening stage is of reduced significance compared to the situation where the RGF forms part of the primary barrier to pathogens.

It may be concluded that the ripening phase in a roughing filter may be observed for several hours on some filter runs. Any attempt to control ripening is difficult to justify since it was not always present, and the filtrates achieved by "ripened" filters were of such variable quality. Only the "classical" filter runs justify consideration of a filter-to-waste period or a slow start, but there seems little point in losing production from a roughing stage ahead of conventional SSFs. The lack of proper control of filtrate quality in the absence of coagulants makes worrying about the fine detail of filter performance unnecessary.

OZONE DOSING

From the observations of Logsdon *et al.* (1993) and Singer and Chang (1989) ozonation was expected to be of benefit, given the hardness and TOC of the water. Table 3.5 showed that a hardness:TOC ratio of 30:1 was typical for this water. Whether ozone influenced ripening time required investigation since only one study had previously examined this: Tobiason *et al.* (1992) had reported that ozone had no effect on ripening time but reduced the initial peak. However their study was unusual in that ozone did not improve ripened removals.

The data in this study indicated that ozone did not cause any difference in ripening time. The use of ozone alone produced superior filtrate turbidity to the un-dosed case but otherwise the conclusions were the same. A pre-ozonated roughing filter did not produce a high quality filtrate. Ozone enhanced the turbidity removal achieved when dosing iron. The benefit from ozone was related to the size of the ozone dose.

IRON DOSING

When iron was dosed, superior removals were attained and the filtrate turbidity appeared to be far less dependent on the applied turbidity (figures 5.1 and 5.5). The magnitude of the iron dose determined the ripening time and the ripened filter turbidity removal. With iron it was possible to

measure ripening in both the filtrate turbidity and removal trends. These findings were the same with iron and ozone dosed together, only the filtrate quality was even better, regularly achieving 0.1 NTU. Because this turbidity was reached irrespective of influent turbidity, at least in the summer (figure 5.5), the use of percent removal as a ripening criterion was inappropriate. The apparent link between influent and filtrate turbidity in the winter required further investigation.

With iron and ozone, figures 5.28 to 5.38 showed that the initial peak reduced as the iron dose increased, and was earlier in the run, and the minimum consistently achieved turbidity (ripened filter performance) was lower for the higher iron doses. The time to achieve ripened performance was shortest with the highest dose and was longer with lower doses. From figures 5.30 and 5.31 this was judged by eye as approximately 3 h, 7 h and 13 h for the three doses, although the highest dose showed considerable breakthrough, commencing half way through the run, and the middle dose showed slight evidence of breakthrough at the end of the run. The lowest dose showed no evidence of breakthrough. All three runs had vary similar head loss development curves. Ripening time with an extremely high dose was under 1 h (figure 5.38), and in this example head loss development rate was changed (figure 5.39).

Further regression analyses, showing similar benefits from ozone and iron on turbidity removal for all the different media investigated, were given by Chipps (1992). Although, when dosed singly, iron was more beneficial than ozone for enhancing turbidity removal, Chipps *et al.* (1993, 1994) reported the reverse was true for POC, chlorophyll *a* and particle numbers and volume. When both iron and ozone were dosed the removals of both turbidity and algae were greater than with either chemical alone.

CONTACT AND DIRECT FILTRATION

The evidence in this chapter, along with the criteria given by Janssens and Buekens (1993), suggested that contact filtration was a suitable method for attaining high quality primary filtrates for much of the year. The results in figures 3.8 and 3.9 show that the stored water occasionally had chlorophyll *a* and particle volume concentrations outside the range specified by Janssens and

Buekens (1993), but within the values which were successfully treated by direct filtration of Wahnbachtalsperreverband reservoir water.

Janssens and Buekens (1993) have pointed out that, at plant scale, the distinction between contact and direct filtration may become blurred. In this study the injection of iron close to the filter, in the outlet of the ozone contactor and in the feed to the ozone contactor did not affect filtration, beyond changing the back pressure on the dose pump. Ives (1981) has said that bubble contactors do not act as flocculators, so this study should be considered to be contact filtration. A later trial on the same water at another pilot plant confirmed the need for ozone and iron dosing for high quality filtrates, and confirmed that only low iron doses were required (Chipps *et al.*, 1995a). There was no difference in filtration performance between contact and direct filtration with stored Thames water.

COAGULANT DOSE AND HEAD LOSS DEVELOPMENT

Mackie and Bai (1993) reported that the effect of particle size on head loss was complicated because particles of different sizes had different attachment efficiencies, causing variations in the location of deposition and the probable morphology of the deposits. The production of different sized flocs, or a variation in capture efficiency, caused by different iron doses might explain why there was little difference in the head loss development gradients in the coagulant dose optimisation trials shown in figures 5.31 and 5.34, despite the variation in turbidity removal.

5.6.2 COAGULANT DOSE SELECTION AND CONTROL

The results which showed how significant the magnitude of the coagulant dose was on filter ripening provide a critical discussion point for this study. It is often assumed that there is one suitable coagulant dose for any given treatment situation and that it is simply a matter of performing a jar test or using an instrument to determine this.

JAR TEST

The concept of a jar test was flawed for contact filtration where floc formation was not required. Cleasby (1990) stated that direct filtration required a "barely visible ... pinpoint floc" and that jar

tests were not appropriate. Having established the initial dose range, jar tests were abandoned as time consuming and misleading, since the production of flocs produced early breakthrough of turbidity and iron.

INSTRUMENTAL TECHNIQUES

Dentel (1994) warned that SCD may not be effective in waters where pH > = 8 or where coagulant demand is low. Watts (1994) found that SCD showed poor sensitivity to coagulant dose in high conductivity waters (>600 $\mu S.cm^{-1}$). A salesman confidently used the results from an on-site SCD demonstration to predict an optimum dose which was ten times greater than the pilot studies showed to be optimum, and would have caused extreme breakthrough. The SCD was not suitable for this application due to the high conductivity and high pH of London's reservoir stored water. Ives (pers. comm., 1993) said that the cause was the compression of the electrical double layer of particles in River Thames water (table 5.1).

A commercial on-line dose control instrument, based on colour measurement, was thought to be inappropriate for London's low colour water. It had a high purchase cost, and cost of ownership was high as there were several instruments to calibrate and maintain. The absence of regular dramatic changes in reservoir water quality meant that constant monitoring of coagulant dose was not justified (Chipps *et al.*, 1994).

It was therefore necessary to devise a coagulant control method which was simple to apply, effective, requiring minimal operator time and skill, and low in material cost. It should not compromise the performance of the downstream SSFs.

FEEDBACK METHOD

Optimising iron dose was achieved using feedback from an individual filter turbidimeter and the logged data. From late 1991 the logged data curves helped set the correct dose. Four typical filtrate trends were discovered. They have been termed S, L, U and V-shaped (after Mosher and Hendricks,

1988). These were used to determine whether the dose was optimum. Figure 5.41 illustrates the key trends of chemical dose and filter behaviour (the SLUV model).

The S-shaped curve (the S is lying sideways, figure 5.41a) was typical when no iron was applied. This curve could also be produced by coagulant dosing, and some ripening would result, but it was considered an under-dose, since the filter behaved similarly to an un-dosed filter and was not achieving its full potential for turbidity removal. Head loss development was typically curved upwards towards the end of the run.

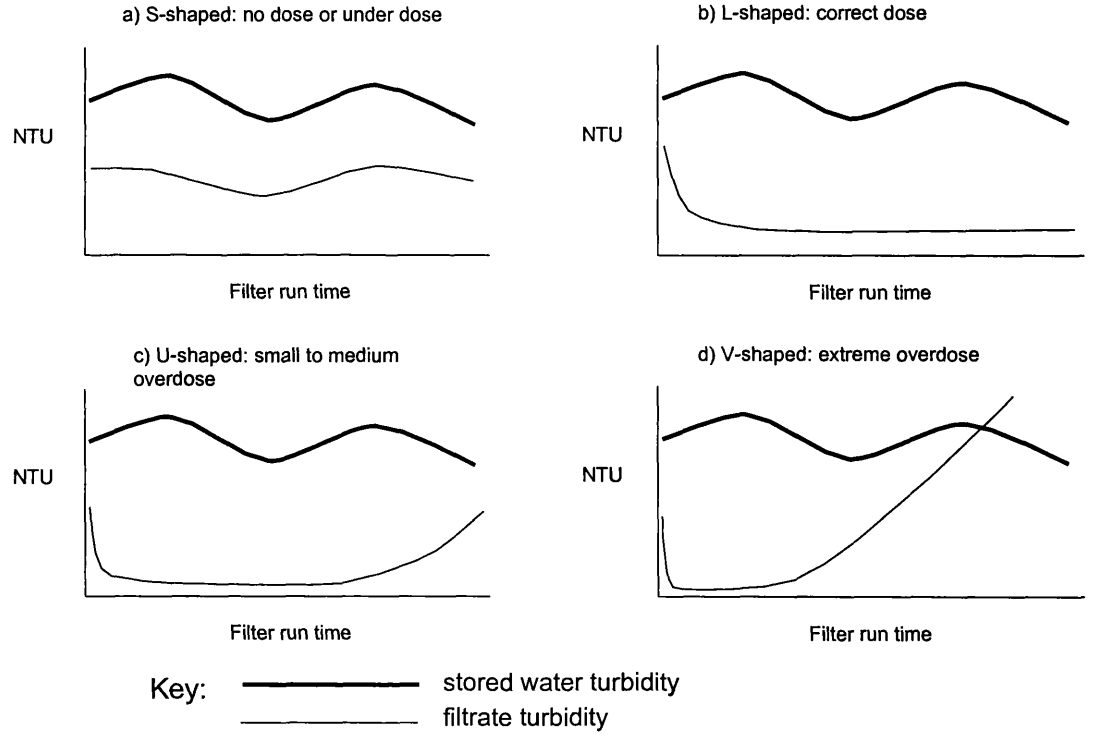

Figure 5.41. *Schematic of turbidity trends over the course of a filter run, illustrating the effect of different doses of coagulant.*

Figure 5.41b shows the L-shaped curve, regarded as indicating the optimum dose. The filter underwent a ripening period, then reached a steady state turbidity which was independent of changes in applied turbidity. This value was maintained within acceptable limits until the end of the run. Head loss was typically linear, indicating depth filtration.

Figures 5.41c and d illustrate the effects of excessive dosing, described as U-shaped and V-shaped trends. Ripening was achieved rapidly, to a higher quality than by L-shaped runs. It was followed by a short period of good performance then a progressive deterioration in filtrate quality. The more excessive the dose, the earlier and more extreme the breakthrough. Head loss development was linear. Filter runs were shorter than they should be, and not only turbidity but residual coagulant passed through the filter. Filtrate turbidity could be poorer than stored water in extreme cases. Table 5.4 summarises these four types.

This method requires at least one full-scale filter to have the instrumentation to feed back the information for the control of chemical dosing. A filter run could be many hours long, yet the filtrate could show a response to a change in dose after around one bed volume. If only one filter was being monitored it could be receiving an overdose and showing excellent ripening behaviour, whilst other filters could be in the breakthrough phase, with the operator unaware of the problem for several hours. To ensure an operator could swiftly determine the impact of a coagulant dose change, several filters in a plant should have on-line trend monitoring so that all parts of the filter cycle could be examined at the same time.

Table 5.4. *Typical filtrate turbidity trends used to develop the SLUV coagulant dose control model.*

Trend type	Coagulant dose	Turbidity trend	Head loss trend
S	No dose or low dose	Filtrate follows fluctuations in influent turbidity. Negligible ripening or breakthrough.	Curved line
L	Correct dose	After ripening filtrate is constant to maximum head loss.	Straight line
U	Moderate over dose	Accelerated ripening, good best phase, breakthrough at moderate-high head loss.	Straight line
V	Extreme over dose	Rapid ripening, excellent best phase, severe breakthrough commencing at low head loss	Straight line

5.6.3 COAGULANT DOSE OPTIMISATION

When iron(III) sulphate was dosed, with or without ozone, ripening was clearly demonstrated with filtrate turbidity and turbidity removal. The rate of ripening was governed by the iron dose. The value of the ripened filtrate turbidity was determined by both the iron dose (figures 5.29, 5.30, 5.33, 5.34, 5.38) and the ozone dose (figures 5.23, 5.26). Rapid ripening to a low turbidity value required both higher ozone and higher iron doses, but higher iron doses caused breakthrough of

turbidity and iron (U- and V- shaped curves, figures 5.29, 5.38). The drawback of the L-shaped curve was a relatively long ripening time. The dilemma in determining the dose was that ripening could be reduced if breakthrough was acceptable, and breakthrough could be avoided at the cost of a long ripening time.

Reducing ripening whilst promoting breakthrough would not be an acceptable operating practice. Whilst there is some evidence in the literature that a filter is vulnerable to the passage of cysts and oocysts during ripening, Logsdon *et al.* (1981) found *Giardia* cysts also appeared in filtrate during the breakthrough phase. There is evidence that particle counters may measure filter breakthrough before any change in turbidity is measured (Chipps *et al.*, 1995a). If turbidity is breaking through a filter then larger particles will almost certainly be penetrating. The primary particles are likely to be accompanied by coagulant, giving further quality problems. It is important to minimise both ripening and breakthrough phases.

The literature suggests that an optimised filter produces a filtrate turbidity with a slight ripening and breakthrough phase, where the critical breakthrough value is reached simultaneously with maximum head loss (Ives, 1982). It would not be an optimum use of the filter if overdosing resulted in the filter being backwashed early due to poor quality .

It follows that successful filter operation requires careful optimisation of the coagulant and ozone doses to provide a least cost means of achieving turbidity removal as well as removal of algae and other larger particles. This should minimise ripening time and product losses during filter-to-waste, and maximise the amount of time the filter is achieving the desired quality target or targets by avoiding breakthrough. The filter run should be as long as possible to maximise productivity, minimise backwash costs and wash water handling costs. Some of these goals are unfortunately mutually exclusive.

In these trials it was decided that the dose should produce an L-shaped turbidity graph. The dose philosophy in setting the iron dose was to achieve a flat ripened filtrate turbidity curve, avoiding

breakthrough, with the lowest possible dose to minimise rate of blockage. With iron and ozone dosing a target of 0.1 NTU was achieved in many cases, but not all conditions. A small breakthrough to 0.2 NTU was considered acceptable as Janssens *et al.* (1982) had considered 0.2 JTU to be a target to supply high rate SSFs. This would make a very suitable criterion for the ripening and breakthrough point in these experiments, to judge whether dosing was optimum.

However from the discussion above it may be incorrect to assume that only one coagulant dose was satisfactory. It would appear that ripening could be addressed by using a higher initial dose, as in the V-shaped run, but long before the filter entered the breakthrough phase this should be cut back to a dose suitable to sustain an L-shaped curve. This method was used with some success by Janssens *et al.* (1982) who showed that overdosing the coagulant for a limited period at the start of the filter run to reduced the ripening peak and duration. For similar reasons the use of coagulants or polymers in backwash water has some benefit in reducing ripening (Harris, 1970, Cranston and Amirtharajah, 1987).

5.6.4 OPTIMISING OZONATION

The ozone plant presented the greatest challenge to pilot plant operation. Compressor and drier failures led to pipework blockages from nitric acid corrosion. Ozone leaks from pipework were a health and safety problem. The diffusers were found to foul over time. Leakage of ozone meant that it was not possible to tell how much of the generated ozone had been applied to the water. Large leaks to the 2 stage column 5 and 6 contactor meant that at times only one of the contactors was used (contactor 1 was counter current, 2 co-current). The pilot plant did not have expensive ozone gas concentration monitors. All ozone figures were therefore based on estimates from the calibration trials. Measurement of ozone transfer efficiency from off-gas concentrations was not possible because of the design of the pilot plant. Since there was an ozone residual in the water leaving the contactor the ozone and water contact time should include residence time above the filter.

The ozone doses were initially set to achieve a residual after 5 minutes contact of <0.5 mg.l^{-1}. This required an estimated dose around 2 mg.l^{-1}. Because of the beneficial effect on filtration performance the ozone was set for optimum filtration which meant running the ozonisers near their maximum capacity, i.e. a dose of some 1.5 - 3.5 mg.l^{-1}. The minimum contact time was 4 minutes, which was the hydraulic residence time in the column 1 contactor.

Chipps *et al.* (1993) presented data which showed that, with ozone and iron dosed together, an estimated minimum ozone dose of 1 mg.l^{-1} was necessary to achieve enhanced turbidity removal, and that the optimum ozone dose for turbidity removal was 2 mg.l^{-1}.

CHAPTER 6. PHYSICAL INFLUENCES ON FILTER RIPENING

6.1 LITERATURE REVIEW

6.1.1 INTRODUCTION

In addition to the chemistry of particles and collectors there are a number physical factors potentially influencing filter ripening. These include media size, bed depth, flow rate, temperature, clean collector morphology, deposit morphology and the influence of deposit particle size and shape. Although much work had been performed on initial removal in clean beds, Darby and Lawler (1990) said that theoretical and experimental consideration of microscopic (within bed) processes during the dynamic phase of filtration had been limited. A number of filtration mechanisms may influence ripening, and models of these have been tested with experimental data, often under idealised laboratory conditions.

O'Melia and Ali (1978) thought that particle-particle interactions were more favourable to filtration than particle-collector interactions, i.e. media with retained deposits provided a more effective filter than the clean collector. They proposed the development of chains of particles (dendrites) into the stream lines as a mechanism of ripening.

As deposits start to build up they may have the effect of increasing the collector surface irregularities. Camp (1969) thought that ripening was due to the increased surface area for filtration caused by the presence of deposits. According to Gimbel (1989) deposition changed the geometry of the filter bed, modifying the hydrodynamic forces and torques which affect filtration.

It is possible to envisage three stages of filter condition (after Tien and Payatakes, 1979) :

i) Clean media where the presence of deposits was of negligible benefit to filtration;

ii) Media with a few deposits where particle-grain and particle-particle filtration processes may take place (the dendrite growth stage);

iii) Media covered with deposits, where only particle-particle interactions can result in filtration (the "open structured solid growth stage" where dendrites became merged and interconnected and lost their distinctive shape).

There is a fourth change which may occur to filter media.

iv) Media subjected to long-term maturation where chemical deposition (Galvin, 1992) or biofilm growth may modify filtration conditions.

Ripening may be considered as the passage through stages i) and ii) above. The fully ripened stage being very poorly defined, but somewhere after the start of stage ii) or into stage iii). It is likely that initially the top of the filter will reach stage iii) whilst below it the media is at stage ii) or i). As the filter run progresses sequential depths of filter media will pass from stage i) to stage iii) causing a "working layer" to move down the bed.

Vigneswaran and Chang (1989) showed that sites of particle deposition could become saturated with a finite concentration of particles. Adin and Rajagopalan (1989) described filtration taking place predominantly in a working layer, which had a high removal rate and a steep concentration gradient. The working layer appeared to move progressively through the filter as each previous layer became "saturated" with particles. When the working layer passed the last of the clean filter media breakthrough commenced. This is too simple an explanation for breakthrough because it implies breakthrough is all or nothing, whereas it is usually a gradual deterioration. There may be different working layers active for different particle sizes, since large particles break through a filter before small ones. Evidence of different breakthrough curves for different sized particles in the same suspension was presented by Mackie (1989). The picture is complicated by the change in particle size distribution within a filter caused by particle detachment (Moran *et al.*, 1993a).

6.1.2 MORPHOLOGY OF COLLECTOR AND DEPOSIT

Both collector structure and pore structure change together during ripening. It has not yet been demonstrated whether one or other of these changes is more significant. Smaller collectors have

been shown to be more efficient filters (Goldgrabe *et al.*, 1993) but during filtration the effective size of the collector increases. Concurrently the pore size reduces. Smaller collectors have smaller pore sizes than larger media of the same porosity.

There is disagreement as to how much of the pore space is used by the deposit. Gimbel (1982) said that only a small fraction of the available pore space was actually utilised for particle deposition. Hunt *et al.* (1993) estimated that this was a maximum of 1.4% of the pore volume. Visual studies of filtration reported by Stein (1940, discussed by Camp, 1964), Maroudas and Eisenklam (1965), Ison and Ives (1969), Payatakes *et al.* (1981), Ives and Clough (1985) and Ives (1989) have given a subjective impression that more than a few percent of the pore volume is blocked. Certain pores appear to become completely blocked. There is also a suggestion that some pores stay open, forming wormholes through even very clogged filters (Baumann and Ives, 1987). These studies have shed little light on filter ripening.

Using a scaled-up filter model a number of different deposit morphologies and varying degrees of pore blockage were reported depending on the degree of destabilisation of particles (Payatakes *et al.*, 1981). Initially the deposits formed "snow caps". Parts of these were observed to become re-entrained and deposited deeper in the filter. Deposits were described as forming pendants and pouches when weakly destabilised; pillars and pendants when well destabilised; and well spaced deposit patches when overdosed with polymers to re-stabilise particles. Ferric flocs tended to form sheaths around collectors, but $CaCO_3$ particles formed domes on the upper collector surface (Payatakes *et al.*, 1981).

Payatakes *et al.* (1981) described throat clogging in filter pores by either gradual bridging of deposits or sudden embolism with detached particle clusters. Payatakes *et al.* (1981) reported that Maroudas and Eisenklam (1965) had seen complete blockage of some channels. These data offer support to François and Van Haute's (1985) concept of ripening caused by deposits blocking small pores and dead spaces in the filter, but it is hard to envisage how flow passes into "dead spaces" and how this affects ripening.

Ives and Clough (1985) and Ives (1989, 1990a) used endoscopes to reveal deposition processes at the scale of individual filter grains. Deposits were observed to build up like "snow caps" and occasional small "avalanches" were observed when "considerable amounts of deposit were present". Ives and Clough (1985) did not observe the formation of dendrites. Later, Ives (1989) reported that dendrites were seen on rare occasions in 100 h of recorded video observations, but not during the ripening period. Ives (1989) did not observe the deposits blocking any whole pores.

Gimbel (1989) summarised three typical deposit morphologies reported in the literature:

i) the formation of a deposit of constant thickness, uniformly around the collectors, particularly with hydroxide flocs;

ii) denser particles, such as kaolin, formed a homogeneous layer on the surface of the collector facing the flow;

iii) heterogeneous morphology caused by agglomeration of deposits.

Based on evidence from research by Billings (1966) on aerosol filtration, O'Melia and Ali (1978) suggested a fourth deposit morphology :

iv) deposits form chains or dendrites.

A fifth morphology arises from a mechanical effect. Tien (1989) described a theoretical "shadow" effect where the capture of a particle made the area around it unavailable for deposition, causing:

v) deposition in discrete territories and a non-homogeneous coating. Any particles arriving in the immediate vicinity would be collected by the deposit itself, resulting in particle aggregates or possibly dendrites.

Since Gimbel (1989a) noted that particles could move around the collector, it is possible this reduces the likelihood of a shadow effect occurring. Darby and Lawler (1990) thought that Brownian motion could bring small particles into the shadow of larger particles. In this case the shadow effect might improve filtration in a similar manner to having cavities or enhanced surface

roughness in the media. This would tend to create smooth deposits rather than dendrites. These two observations suggest that filtration of monodisperse and polydisperse suspensions may result in different deposit structures.

6.1.3 MORPHOLOGICAL CHANGES IN DEPOSIT OVER TIME

As filtration proceeds the pore space is changed and transport streamlines may change and also shear forces may change, both affecting attachment. Camp (1964) reported that some deposits showed a creeping or kneading action as they moved around the collector. It is possible that flocs undergo some internal rearrangement. Sembi and Ives (1982) suggested that deposit structure became compressed as deposition in the filter pores continued. How this occurred was not made clear. Camp (1964) stated that there was no evidence that water was lost from flocs even under high shear forces. Hunt *et al.* (1993) modelled their filtration data by considering a porous deposit in which the porosity of the deposit reduced as the pore space became filled up. This was due to flow through the deposit resulting in further particle deposition in the deposit itself. Payatakes *et al.* (1981) were said by Hunt *et al.* (1993) to have shown visually that flow could pass around and through deposits at pore restrictions.

Baumann and Ives (1987) inferred three types of flow pattern from the work of Coad and Ives (1981), including flow through deposits:

i) very slow seepage flow through deposits;

ii) slow flow through constricted pores (the majority of the flow);

iii) open pore or wormhole flow, consisting of a small proportion of the total flow travelling at high velocity.

Ojha and Graham (1993) took the view that Coad's (1982) data had shown that flow velocities through the interstitial spaces in the deposit were negligible, in which case deposit pores were unlikely to be filled internally. They thought that deposits could be compressed by external forces but did not elaborate on this suggestion.

Choo and Tien (1993) described a mathematical filtration model where the deposit was not assumed to be of uniform thickness, and was considered to be porous, allowing the flow of liquid through the deposit. It was suggested that this could enable significantly enhanced particle deposition compared with flow around an impermeable deposit layer.

The problem with these ideas is if deposits only occupy a small percentage of the pore volume (Gimbel, 1982) it is hard to understand how or why water streamlines should pass through them.

Changes to flocs over time have been reported by François (1987). He found that over 6 - 12 h aluminium hydroxide flocs changed size and strength, and electrophoretic mobility reduced. Initially, aggregation of individual flocs took place, and over the course of a few hours "cementation" followed, where inter-particle attachments developed leading to shrinkage and strengthening of the floc. There was no evidence that the shrinkage was caused by break-off of particles from these flocs. There is no clear evidence that these chemical changes influence ripening since they take a long time and new flocs will always be entering the filter.

6.1.4 FILTER MEDIA GRAIN SIZE

The size of the media grains has an impact on filter efficiency. Experiments reported by Goldgrabe *et al.* (1993) with five media sizes from 0.44 - 1.38 mm showed that the smaller collectors produced the best filtrate turbidities and particle counts. Clark *et al.* (1992) showed better removals with 0.8 mm media. Most of the particle removal and almost all the ripening took place in the top half of the bed. With 1.85 mm media Clark *et al.* (1992) found that removal was more evenly distributed throughout the bed and this resulted in a slower rate of ripening. Removals with 3.68 mm media were said to have been minimal or nil at all depths.

Vigneswaran and Ben Aïm (1985) and Vigneswaran *et al.* (1990) showed that ripening time and removal for large pollen grains and coarse filter media were dependent on media size and particle size. With 2.5 mm media ripening took about 60 minutes for 3 different pollen sizes. With 4 mm

media not only were removals poorer at the first sample but the filters were still ripening when the trial stopped after 120 minutes.

Moran *et al.* (1993) found that initial and ripened removals were greater with smaller media for the same bed depth. When they controlled for media surface area by taking samples from a greater depth of the coarser media, Moran *et al.* (1993) found that different diameter media had the same ability to remove particles when an equivalent surface area of collector was ripened by an equivalent surface area of deposit.

6.1.5 COLLECTOR SURFACE STRUCTURE

The morphology of the media surface is important in filtration. Gimbel and Sontheimer (1978) and Gimbel (1982, 1989) produced experimental evidence that the surface roughness of clean media influences filtration. Gimbel and Sontheimer's (1978) filter model showed an improved fit to experimental data if it incorporated surface roughness. Gimbel (1989) showed experimental data of how clean bed filter efficiency, for a range of applied particle sizes, changed depending on the filter medium used. Materials such as pumice or filter coke, with surface cavities and high degrees of surface roughness, were far more effective filters than quartz sand, which was in turn superior to glass spheres. Kau and Lawler (1995) showed that sand was a better filtration medium than glass spheres, observing larger clean bed and ripened particle removals and faster ripening times, irrespective of media size, flow rate or particle size class.

Scanning electron microscope (SEM) studies revealed, even for relatively clean filter sand, a heterogeneous structure, with material (probably biological in origin) inhabiting depressions on the grains (Bayley, pers. comm., 1993). The complex surface structure of quartz sand from a beach in Florida was well illustrated by the SEM photographs of Mack and Leistikow (1996).

Gimbel (1989) reported improvements in filter efficiency when sintered quartz particles were used to add protrusions onto smooth 1.5 - 2 mm diameter glass spheres. Cavities in filter media grains enhanced filtration by providing a region of relatively low hydrodynamic forces (Gimbel, 1989).

Using an endoscope Ives (1989) observed the filling of surface crevices and depressions which he suggested supported Gimbel and Sontheimer's (1978) model.

6.1.6 EFFECT OF TEMPERATURE ON FILTER RIPENING

The effect of temperature will be to reduce filter effectiveness by a number of means, yet no studies have been carried out on the effect of temperature on ripening. From the trajectory analysis viewpoint the reduction in water temperature will reduce transport efficiency by increasing fluid drag and hydrodynamic retardation and reducing Brownian motion. Equations 5.25 - 5.29 given by Tien (1989) show that temperature is not a consideration in the London-van der Waals force, but that it is a factor in determining the double layer force. It is known that the speed of chemical reactions and the activity of poikilothermic organisms are diminished at lower temperatures. This latter effect may improve filtration by reducing the activity of motile organisms.

In reviewing the work of Morris and Knocke (1984), Haarhoff and Cleasby (1988) noted that one effect of lower temperatures was to produce smaller flocs, although it did not affect the rate of hydroxide precipitation. With very cold water (<3°C) ripening took place over the first 2 to 3 hours of the run. Whilst there were no comparable data from warmer water this time was not exceptionally long.

Al-Ani *et al.* (1986) demonstrated that neither water temperature (5 - 18°C) nor flow rate (5 - 20 $m.h^{-1}$) had a detrimental effect on removals of bacteria or cysts. However, at 25 $m.h^{-1}$ some deterioration in performance and sensitivity to temperature was observed: at the higher flow rate better turbidity removals were recorded at 18°C than 5°C.

6.1.7 EFFECT OF FLOW RATE ON RIPENING

Cleasby and Baumann (1962, 1962a) presented data showing ripening in water filtering precipitated iron. Effluent quality was measured as iron concentration. Their results showed a highest initial peak with the highest flow rate (15 $m.h^{-1}$) with smaller peaks for flow rates of 10 $m.h^{-1}$ and 5 $m.h^{-1}$, the latter being slightly lower than 10 $m.h^{-1}$. The peak took longer to appear for slower flow rates. Cleasby and Baumann (1962a) said that the peak in filtrate quality for turbidity or iron was reached

"at approximately the theoretical detention time to displace the water from the filter and the turbidimeter". The curve at 10 $m.h^{-1}$ produced overall the best quality filtrate and appeared by eye to have the smallest area under the curve, i.e. the fastest ripening rate and best overall quality. Ripening was therefore delayed by having too high and too low a flow rate.

The filtration of large particles (>40 μm) was shown by Darby *et al.* (1991) to be affected by flow rate. Removals and ripening were poorer at 13.3 $m.h^{-1}$ than 6.5 $m.h^{-1}$. The removal of the smaller particles was less sensitive to filtration velocity. At the higher rate the particle capture tended to be distributed more evenly in the depth of the bed. This could have important consequences in selecting media size and flow rates for the efficient use of deep bed filters.

Moran *et al.* (1993) showed that clean bed removal removals were dependent on flow rate, being slightly poorer at higher velocity. In order to clarify this Moran *et al.* (1993) compared percentage removals at different flow rates normalising detention time in the filter and normalising surface area of particles captured. Presenting the data this way showed very similar removals were obtained with all particle sizes at two filtration velocities.

Watson (1990) showed that ripening of POC took 70 minutes on a lower rate filter and less than half this, 30 minutes, on a filter flowing at double the rate. Watson (1990) thought her data could support either Cranston and Amirtharajah's (1987) or O'Melia and Ali's (1978) view of ripening. From the Cranston and Amirtharajah (1987) viewpoint, the higher flow rate would have the effect of driving the backwash water remnant through the bed more quickly, eliminating the solids from the previous run (although this might cause a higher peak) and diminishing the dilution effect of the clean backwash water on the new influent, allowing the properly destabilised water to influence the quality of the filtrate sooner. Alternatively, using the O'Melia and Ali (1978) view, the higher rate would mean a higher loading of particles and a more rapid coating of the filter media with particles.

6.1.8 EFFECT OF SUSPENDED SOLIDS CONCENTRATION ON RIPENING

Experiments by Clark *et al.* (1992) showed that there was a complex interaction between media size, flow rate and influent suspended solids. Deposit location and morphology were influenced by flow rate and solids concentration in different ways with different sizes of media. O'Melia and Ali (1978) noted the dependence of ripening time and removal efficiency on influent particle concentration. Darby *et al.* (1992) found with two concentrations of 2 μm particles that clean bed removals were similar (30%) as were the ripened removals (98%) but ripening was quicker with the higher suspension concentration. It was suggested that this was due to the higher number of particles to act as additional collectors available earlier in the run.

Clark *et al.* (1992) suggested a possible mechanism for the difference in filter behaviour due to particle concentration: their assumption was that a particle did not immediately come to rest after making contact with the collector, and that its position determined whether it was available as an aid to capture. They presumed that a particle could migrate along the surface to the bottom of the media grain. In a lower concentration suspension the distance a particle could migrate from the initial collection point before another particle was collected would be greater than with a more concentrated suspension. Therefore the probability of one particle assisting with the removal of another would increase with increasing particle concentration, thus increasing ripening and removal rates.

Whilst not mentioned by Clark *et al.* (1992), a similar logic could be applied to collector size and filter flow rates. With larger collectors deposits could be more widely spaced than with smaller collectors. Higher flow rates could cause more rapid movement of particles across the collector to sites where they could not facilitate particle-particle ripening. Migration could be explained by a mixture of fluid shear forces, gravity and particle kinetic energy. A theoretical consideration of the forces acting to move particles across a collector, and support that a particle does not immediately come to rest, were provided by Gimbel (1989, 1989a).

Visual observations of the behaviour of flocs support the concept of deposit movement across the filter (Payatakes *et al.*, 1981) but there is a need for endoscope observations to look for the migration of individual particles.

A standard practice in filtration has been to increase the post backwash flow rate incrementally over time until the desired maximum flow rate is achieved - the "slow start". Since there is some evidence that flow rate influences ripening the use of slow starts has been considered in the section on controlling ripening.

6.1.9 THE EFFECT OF PARTICLE SHAPE AND SIZE ON RIPENING

Many studies have investigated filtration of Kaolin suspensions or latex spheres, but even in laboratory filtration experiments particles or flocs may not behave as simple spheres: Ives and Clough (1985) reported that aggregates could enter much smaller pore openings without blocking them. They said the particles deformed and broke up. In full scale filter plants a wide diversity of particle shapes and sizes must be filtered, the use of Kaolin or latex may be an over-simplification, especially for contact filtration. Examples of morphological factors which are biological in origin are discussed in the section on the influence of biology on filtration.

The data from a number of laboratory investigations into filter ripening have been examined for the insights they give into ripening. The experimental conditions are summarised in table 6.1.

Table 6.1. *A summary of the experimental conditions under which laboratory experiments were conducted to investigate filter ripening.*

Author (date)	Water source	Particle material	Particle concentration ($mg.l^{-1}$)	Particle diameter (μm)	Coagulant	Superficial velocity ($m.h^{-1}$)	Media material	Media diameter (mm)	Media depth (mm)
O'Melia and Ali (1978), using Habibian's (1971) data. (Also used by Tare and Venkobachar, 1985)	NR	Latex spheres	9.7 - 52	0.1 1.0 7.6	Cationic polymer	4.9	NR	0.38	140
Vigneswaran and Ben Aïm (1985)	NR	Pollen grains	28 - 75	12.8 26 42	WAC-2 (no further details)	5 10	Glass beads	2.5 4	NR
Vigneswaran *et al.* (1990)	NR	Pollen grains	20 - 100	22 50 85	NR	5	Sand	1.1	20
Vigneswaran and Chang (1986)	NR	Kaolin	25, 50 & 100	5	Alum	4.7 - 14	NR	1.19	300
Vigneswaran and Chang (1989)	Tap water	Kaolin	80	12	NR	5 - 17	Glass beads	0.3 - 0.4	20
Mackie *et al.* (1987), Mackie (1989), Mackie and Bai (1992, 1993)	NR	Emulsified PVC powder spheres	90 - 264	<0.5 - 15	None	2.4 - 7.2	Glass beads	0.42 - 0.50 0.60 - 0.71 0.71 - 0.85 1.10	375
Darby and Lawler (1990), Darby *et al.* (1992)	Particle free water	Latex spheres	3.6 - 16.2	0.6 2 6 21	$Ca(NO_3)_2$ 0.03 M pH 2	6.1	Glass beads	0.5 - 0.6	140

Table 6.1. *(continued)*

Author (date)	Water source	Particle material	Particle concentration ($mg.l^{-1}$)	Particle diameter (μm)	Coagulant	Superficial velocity ($m.h^{-1}$)	Media material	Media diameter (mm)	Media depth (mm)
Darby *et al.* (1991)	Activated sludge plant effluent	Natural particles	6.7	From 1.3 - 80	None	6.5 - 13.3	Glass beads	1.55 1.85	220
Clark *et al.* (1992)	Reservoir water	Natural particles	1.2 - 3.3	From 1 - 20	Cationic polymer	6 - 20	Glass beads	0.8 1.85 3.68	220 949
Veerapaneni and Wiesner (1993)	Buffered de-ionised particle free water	Latex spheres	5 - 15	0.09 0.94 3.2 7.0 7.6	$Ca(NO_3)_2$ 0.01 M	4.8	Glass beads	0.37	135
Tobiason *et al.* (1993)	Buffered de-ionised water	Latex spheres	10 & 20	0.27 1.24 1.32 10.0 10.1	Cationic polymer or $CaCl_2$ > =0.05M	4.8	Glass beads	0.4	170
Moran *et al.* (1993)	Natural water after lime softening and settlement	$CaCO_3$	3.0 - 6.3	Hetero-disperse	None	6.5 13.3 19.8	Glass beads	0.78 1.85	946
Kau and Lawler (1995)	Natural water after lime softening and settlement	$CaCO_3$	3.9 - 4.9	Hetero-disperse	None	6.5 19.8	Sand	0.39 0.78 1.85	220 961

NR = not reported in the cited reference. N.B. Pollen grain shapes and surface characteristics were not described, but they can be more complex than simple smooth spheres. The pollen grains were *Broussonetia* (12.8 μm), *Cuppressus arizonica* (26 μm) and *Sorgho* (42 μm). Trials with monodisperse, bimodal and trimodal suspensions of 22, 50 and 85 μm average diameter (d_{50} by weight) pollen grains from *Platane*, *Sorgho* and *Mais* respectively.

RIPENING WITH MONODISPERSE SUSPENSIONS

The effect of particle size on clean bed removal of monodisperse suspensions of latex spheres was clearly demonstrated by Yao *et al.* (1971) and confirmed by the work of O'Melia and Ali (1978), Veerapaneni and Wiesner (1993) and Tobiason *et al.* (1993), summarised in tables 6.2 and 6.3. The results agreed with expectations from trajectory theory (Yao *et al.*, 1971) as particles < 0.5 μm and > 5 μm diameter were well removed, but particles around 1 μm diameter were comparatively less well removed. The trend observed with clean media persisted through ripening to the ripened performance. However, despite the poor clean bed removals of particles around 1 μm, it was clear that after ripening they were extremely well removed.

Table 6.2. *Ripening of monodisperse latex suspensions. (After Veerapaneni and Wiesner, 1993).*

Particle size (μm)	Particle concentration ($mg.l^{-1}$)	Clean bed removal (%)	Ripened removal (%)	Ripening time (minutes)	Removal after 200 minutes (%)
0.09	5	46	95	45	>98
0.94	6.54	30	90	70	95
7.04	11	79	95	25	>98

Table 6.3. *Results from filtration of monodisperse latex suspensions by Tobiason et al. (1993).*

Particle size (μm)	Coagulant type	Clean bed removal (%)	Ripened removal (%)	Ripening time (minutes)	Head loss development ($mm.h^{-1}$)
0.27	polymer	67	>98	70	150 [1]
1.32	polymer	20	>98	70	20
10	polymer	85	>98	40	5
0.27	calcium	64	>98	150	60
1.24	calcium	52	>98	100	18

[1] calculated after deducting initial 1 h lag phase

Darby and Lawler (1990) obtained ripening curves with 2 μm and 0.6 μm particles which were virtually identical, although ripened removal was marginally poorer for the smaller particles. Two ripening rates were apparent; the faster rate lasted for the first 40-60 minutes, then ripening continued more slowly until the experiment was terminated after 240 minutes.

Vigneswaran and Ben Aïm (1985) and Vigneswaran *et al.* (1990) showed clean bed removals and ripened removals were ranked in sequence of particle size. The largest particles showed

deteriorating removals towards the end of the trial, when two smaller sizes were still improving marginally.

RIPENING WITH POLYDISPERSE SUSPENSIONS

A number of experiments have reported the behaviour of one particle size in the presence of another, and comparing this with the particle removal in a monodisperse suspension. The results have produced contradictory conclusions.

a) LARGER PARTICLES HAVE NO EFFECT ON THE RIPENING OF SMALLER PARTICLES

Veerapaneni and Wiesner (1993) found that combining 0.944 μm particles with larger particles produced no improvement in the ripening of the 0.944 μm particles compared to the monodisperse case. They also found ripening of 0.09 μm particles was unaffected by the presence of larger particles. Darby and Lawler (1990) concluded that a particle is more likely to be captured by a particle of similar size rather than larger size.

b) LARGER PARTICLES HINDER THE RIPENING OF SMALLER PARTICLES

Tobiason et al. (1993) found 10 µm particles caused a slight deterioration in clean bed removals of sub-micrometre particles, lasting 20 minutes, but no other change in removals. This was possibly due to interference with destabilisation, or unfavourable hydrodynamic interactions as the small particles approached the larger ones (Tobiason *et al.*, 1993).

Darby and Lawler (1990) found that ripening of 2 μm particles occurred in the presence of 6 μm particles but took twice as long as in the 2 μm monodisperse experiment. Similarly, Darby and Lawler (1990) showed that ripening of the 0.6 μm particle removal in the presence of 6 μm was poorer than the single size experiment and took twice as long. However in the bimodal experiment both the concentration and the surface area of the 0.6 μm particles were an order of magnitude lower, and this might have explained the poorer removal of the 0.6 μm particles. Whilst Darby and Lawler (1990) found ripening with 6 μm particles on their own, there was no ripening of 6 μm particles in the presence of 21 μm particles. Darby and Lawler (1990) suggested several possible reasons for this: i) ripening was enhanced by more small particles than fewer, larger particles; ii) charges on the 6 μm particles caused particle interactions between the 2 μm particles to be

preferred due to different particle stabilities; and iii) the 6 μm particles somehow hindered the removal of the smaller particles.

With polymer destabilised suspensions of 0.27 and 1.32 μm particles Tobiason *et al.* (1993) found that the larger particles caused a large decrease in the removal of the sub-micrometre particles for the first 20 minutes, but thereafter the sub-micrometre particles were removed to the same extent as in their monodisperse experiment. In a mixture of 0.27 and 1.24 μm particles, destabilised with calcium, there was no change in the initial removal of the sub-micrometre particles compared to the monodisperse situation. This contradiction led Tobiason *et al.* (1993) to suspect that, with polymer, the poorer removal of 0.27 μm particles with 1.32 μm particles was due to chemical rather than physical causes.

c) LARGER PARTICLES ASSIST THE RIPENING OF SMALL PARTICLES

The results of Vigneswaran and Ben Aïm's (1985) experiments on filter ripening showed the presence of 26 μm pollen grains impeded the clean bed removals of 12.8 μm pollen but assisted their ripened removal, and reduced the ripening times.

For similar mass suspensions Vigneswaran *et al.* (1990) showed the removal during ripening, the ripened removal, and the ripening time of 22 μm pollen improved as the ratio of 85:22 μm pollen grains increased. In trimodal experiments, where 50 μm pollen was also included, the filtration of 22 μm particles was not as efficient as in the bimodal experiments.

Tobiason *et al.* (1993) reported the effect of 10.0 μm particles on 1.32 μm particles was to improve marginally the poor removal of these particles during the ripening phase. There was no change in the removal of the 10.0 μm particles.

Veerapaneni and Wiesner (1993) said that, if the small particles assisted with the removal of larger particles, but the resulting ripening helped remove further small particles, this might account for the observations of Vigneswaran *et al.* (1990).

d) SMALLER PARTICLES ENHANCE THE RIPENING OF LARGER PARTICLES

Veerapaneni and Wiesner (1993) found that, compared with the monodisperse case, the combination of larger (0.944 μm) particles with smaller (0.09 μm) particles improved clean bed removals of the former from 50% to 80% after 3 minutes, and accelerated ripening time and improved the ripened particle removal. Removals of 7.04 and 7.6 μm particles improved from 80% to "complete removal" when combined with the 0.09 or 0.944 μm particles. The 0.09 μm particles were more beneficial than the 0.944 μm particles in this respect.

Tobiason *et al.* (1993) reported that 0.27 μm particles assisted the clean bed removal of 1.24 and 1.32 μm particles and ripening was greatly accelerated (compare table 6.4 with table 6.3). The presence of 0.27 μm particles enhanced the removal of 10 μm particles, eliminating the ripening phase.

Darby and Lawler's (1990) experiments showed the ripening of 6 μm particles was assisted by 0.6 μm particles but not 2 μm particles. During ripening the 0.6 μm particles served as better collectors for 6 μm particles than the original media. However lower ripened filter removals showed 0.6 μm particles were poorer collectors of 6 μm particles than other 6 μm particles in a ripened filter.

Darby and Lawler (1990) found that, with 21 μm particles and 6 μm particles, the clean bed removals of 21 μm particles were slightly enhanced compared to the monodisperse experiment. Ripening of 21 μm particles did not occur in either case.

In contrast Vigneswaran *et al.* (1990) reported that there was no significant improvement in the removal of 50 and 85 μm particles when 22 μm particles were present.

Veerapaneni and Wiesner (1993) suggested the improved removal of larger particles in the presence of smaller ones was due to the relatively large extra surface area for deposition created by the attachment of the small particles to the collectors. Ripening was dependent mainly on retained particle surface area and independent of suspension particle size composition.

Tobiason *et al.* (1993) suggested that improved removals could have been due to the formation of a different shape or size of larger particle by the two sizes combining ahead of filtration, or to prior deposition of the small particles making a rougher collector surface.

Table 6.4. *Results from filtration of bimodal polydisperse latex suspensions (10 mg.l^{-1} each) by Tobiason et al. (1993). The particle size to which the performance reported in the table relates is given first, with the other particle size in suspension in parentheses.*

Particle size (μm)	Coagulant	Clean bed removal (%)	Ripened removal (%)	Ripening time (minutes)	Head loss development (mm.h^{-1})
0.27 (+1.32)	polymer	37	>98	100	110
1.32 (+0.27)	polymer	20	>98	20	110
0.27 (+10.0)	polymer	60	>98	180	140
10.0 (+0.27)	polymer	98	>98	5	140
1.32 (+10.0)	polymer	25	>98	70	18
10.0 (+1.32)	polymer	85	>98	10	18
0.27 (+1.24)	calcium	65	96	120	60
1.24 (+0.27)	calcium	63	>98	40	60

e) SMALLER PARTICLES IMPEDE THE RIPENING OF LARGER PARTICLES

Darby and Lawler' s (1990) results showed the presence of 2 μm particles appeared to hinder the removal of the 6 μm particles.

INTERACTIONS BETWEEN PARTICLES OF THREE OR MORE SIZES

A number of studies have been performed where the filtration of a broad range of particle sizes has been analysed by breaking down the data into a number of size classes.

Mackie (1989) and Mackie *et al.* (1987) and presented filtration results for overall particle removals which showed the usual ripening - best quality - breakthrough sequence. However, when different particle size classes were examined it was found that the overall picture was made up of components behaving very differently. Particles from 0.63 - 1.26 μm were initially poorly removed, but ripened progressively over the course of a six hour experiment. As particle size increased the clean bed removal improved, ripening was more rapid, and better ripened filtrate quality resulted. The largest particles (5.04 - 10.08 μm) were well removed (>90%) after almost no ripening time.

However particles of this size group started to break through half way through the experiment and ultimately achieved poor removals (<40%).

Similar size related ripening and breakthrough trends were observed by Darby *et al.* (1991) and Moran *et al.* (1993). The same pattern was reported from a tri-modal experiment by Darby and Lawler (1990). However it was not seen in turbidity or suspended solids measurements (Clark *et al.*, 1992). Figure 6.1 summarises this relationship between particle size and the filter cycle.

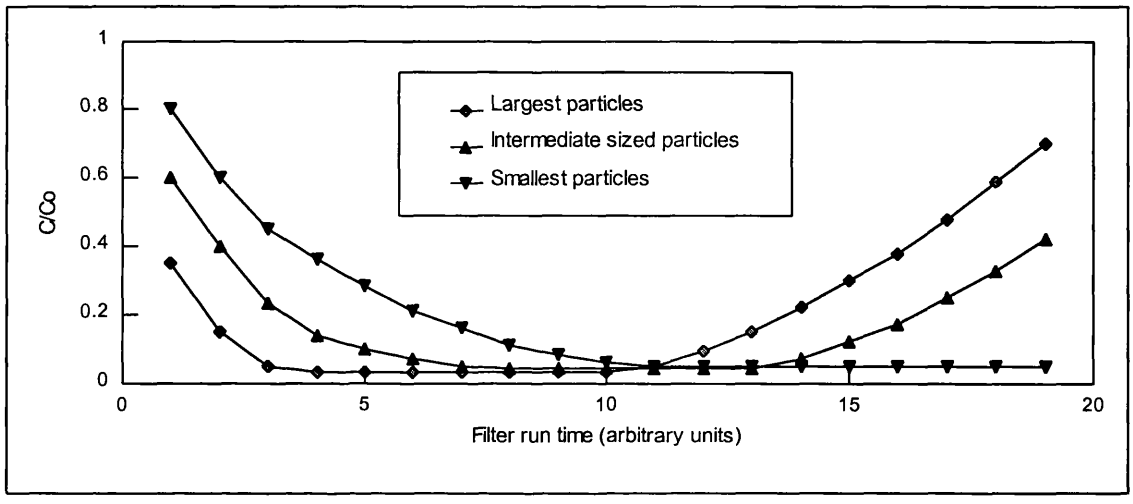

Figure 6.1. *Idealised summary diagram of the relative ripening behaviour of different sized particles in a mixed suspension, based on the findings of Mackie (1989), Mackie et al. (1987), Darby and Lawler (1990), Darby et al. (1991), Clark et al. (1992) and Moran et al. (1993).*

THE DYNAMICS OF FILTRATION

Mackie (1989) and Mackie *et al.* (1987) combined three filtration mechanisms to model their findings:

i) the deposited particles acted as collectors;

ii) there were macroscopic and microscopic effects resulting from the build up of deposit;

the macroscopic effect was that the overall deposit structure acts on the flow field around it, and the media grain size is considered to have increased (Choo and Tien, 1993);

the microscopic effect was due to individual particles which protruded from the overall deposit structure (a "dendrite" could be one particle long), giving a surface roughness, similar to individual deposits on a clean media grain;

iii) interstitial velocity and shear forces increased as deposition built up.

The model considered variations in different regions of the media grain to deal with particles of different sizes. Shear forces would be different at different positions of the grain with respect to the flow stream and also higher in the centre of the pore than at the grain / deposit surface. Larger particles would protrude further into the flow stream and experience larger shear forces. As a consequence, at a given time a part of the collector might have been able a) to collect particles of all sizes, or b), only collect particles of some sizes, or c), be unable to collect any particles.

Based on this model Mackie's (1989) interpretation of filter behaviour was as follows: the initially deposited particles protruded into the fluid streamlines, promoting ripening; removal eventually deteriorated because the increasing size of the deposit resulted in increased shear forces acting on the deposits. Since large particles protruded further into the flow field they experienced greater shear forces than the smaller particles, and hence removal of these fell whilst smaller particles were still being well removed. Some of these detached larger particles may have been aggregates of smaller particles (Clark *et al.*, 1992).

Mackie (1989) predicted that removing the larger particles from the influent suspension would reduce removals and prolong ripening, since they would not be available to act as collectors for the smaller particles. Mackie and Bai (1992, 1993) confirmed experimentally that large particles (10 - 15 μm) assisted the removal of smaller ones and hastened their ripening.

Mackie (1989) speculated that a practical application of this observation would be to add larger particles to a suspension of fine particles to increase filtration efficiency. Alternatively Mackie (1989) said that it might be important not to remove all the larger particles prior to filtration. It can be seen in the former case that large particles could be dosed or alternatively larger particle sized flocs could be built out of the smaller particles by applying correct coagulants or filter aid polymers. In the second case performance of a clarification stage (settlement or flotation) might require being downgraded a little to ensure the total process behaved optimally. The observation is also

consistent with adding a coagulant to backwash water or leaving a little dirty backwash water above the filter to mix with influent.

Mackie and Bai (1992) found an interesting dynamic behaviour in the depth of a filter. In tests with water containing a wide range of influent particle sizes the larger particles were so effectively removed in the top 75 mm of the bed so that removals between 75 and 375 mm depths after 3 h were poorer than those of a separate finer sized suspension. At these depths the finer suspension actually contained a higher proportion of larger sized particles. After 5.5 h the position reversed because the largest particles were breaking through from the top of the bed, ripening the lower section.

METHODOLOGICAL CRITICISM

The major problem with these experiments is in creating comparable experimental conditions. Chemical destabilisation was of paramount importance. Yao *et al.* (1971) and Tobiason and O'Melia (1988) showed that ripening was only seen when a suitable coagulant dose was applied to the latex suspension. Tobiason *et al.* (1993) and Darby and Lawler (1990) ascribed anomalous behaviour in some laboratory experiments to unfavourable chemical interactions.

There may be methodological problems to be resolved before the relationships between particles are understood. The research does not make it clear whether suspensions of different sized particles should be compared with other suspensions of equal particle masses, particle volumes, or particle surface areas. For example Veerapaneni and Wiesner (1993) showed that ripening behaviour was the same for 0.944 μm particles in several polydisperse suspensions when plotted against the retained particle surface area but not against retained particle mass. Similarly Tobiason *et al.* (1993) showed that head loss development by different particle size suspensions was very similar if plotted against surface area removed.

6.2 EXPERIMENTAL INVESTIGATION

6.2.1 METHODS

Logged turbidity data from columns 1, 4, 5 and 6 were used to show the influence of flow, temperature and particle loading variables on biologically mature filters with and without coagulant or ozone dosing.

Column 4 was a sand filter, operated without chemicals. The filter started operating with 600 mm 14-25 sand in November 1990 although it was topped up to 900 mm 14-25 sand on 30.10.91. Three dual media filters were operated with ozone and iron available. Column 1 contained 660 mm coarse cut grade 2 anthracite (1.7-2.5 mm) above 340 mm 14-25 sand. It was operated with its own iron pump and ozone generator and contactor, commencing 22.2.90. It was operated at 15 $m.h^{-1}$ to 1.8 m head loss, except from 16 May to 31 December 1991 when backwashing started at 1.5 m. Columns 5 and 6 were operated at flow rates from 12 to 22 $m.h^{-1}$. They were backwashed at 1.8 m prior to 28.1.92 and 2.5 m thereafter. Column 5 was a dual media filter with 600 mm 1.7-2.5 mm anthracite over 600 mm 14-25 sand and started operating on 12.9.91. Column 6 contained 600 mm 1.2-2.5 mm anthracite over 600 mm 14-25 sand. This filter commenced service on 28.8.91. Columns 5 and 6 could be iron dosed separately, or receive pre-ozonated and iron dosed water from a common manifold.

The ozone and iron figures were calculated from daily readings as discussed earlier. The logged data from each filter run were combined with the ozone and iron estimates in the database. The logged data were used in three ways: to calculate means from whole runs, to plot trends of individual filter runs, and to calculate mean trend data from groups of filter runs operated under similar conditions.

Summary values of flow rate, chemical dose and inlet and outlet turbidities were calculated for each filter run based on means of the logged data for the entire filter run. These gave an approximate idea of ripened filter performance. Mean figures from filter runs which had a

corresponding daily water temperature value were used to examine whether there was an effect of temperature on ripening.

It has been demonstrated earlier that the chemical dose was critical to ripening although the factors influencing the appropriate dose to achieve the L-shaped trend were not fully understood. In the examples where data from iron and ozone dosed filter runs have been used it was necessary to eliminate runs during which dosing started or stopped. Runs where the dose was not considered optimal were discarded if the filter performance trend and the iron dose indicated unusual behaviour compared to the immediately preceding and following runs. Filter runs where excess iron was dosed were excluded from the summary data by removing runs where the turbidity C/Co value averaged >0.3 over the period of 80-90% of elapsed filter run time.

FLOW RATE TREND DATA

A series of flow rate changes was performed on filter 4 during June and August 1992. Logged trend data were examined for the impact of flow rate on turbidity removal and ripening over these periods.

A sequence of flow rate trials was undertaken with columns 5 and 6 in November 1991. Column 5 had been operating for 2 months since the media had been installed. Column 6 had been in service 3 months. Unfortunately both trials were undertaken before the logging of the stored water turbidimeter data commenced, so only filtrate turbidity data were available. Although the filter runs where flow rate was changed were grouped closely together in time it was necessary to assume that the influent water did not change significantly over the period examined. Daily and weekly data were presented to confirm this. Trend data were presented to examine the effect of flow rate on ripening.

For the trial period column 5 was operated with iron, but not ozone, at 3 flow rates stepping up from 12 $m.h^{-1}$ to 17 $m.h^{-1}$ then 22 $m.h^{-1}$. The filter was fed by its own iron pump. Changing the filter flow rate caused a change in the coagulant dose, and the pump had to be reset to compensate. The

filter was operated at one flow rate for a number of runs to gain reliable data and to allow corrections in chemical dose, as a consequence of changing the flow rate, to be effected.

A reverse sequence of flow trials was undertaken with column 6. The water for this trial was passed through the ozone contactors and was common with the feed to column 5, minimising the impact of the change in flow rate on iron dose. Ozone was dosed at 2.4 $mg.l^{-1}$. In the logged data trend graphs the x-axis tic marks denote midnight at the start of the day labelled. In the combined flow and head loss graphs the latter is depicted by the darker, thinner line. The filtrate is the darker, thinner line in the graphs depicting both inlet and outlet turbidities.

CONTOUR PLOTS

Groups of filter runs operated under similar conditions had the first 6 h of turbidity data averaged to produce contour surface plots. These runs were grouped into classes of stored water turbidity, filter run time and a particle index. The stored water turbidity classes were each 0.5 NTU in breadth, based on the mean stored water turbidity calculated for each initial 6 h period of operation. The run length classes spanned 6 hours. The particle index was created from the weekly sampling of chlorophyll *a*, POC, and particle size and number. Trend data from the runs which were in progress as the weekly samples were taken were used to examine the effect of the loading measured by these parameters.

An attempt to determine the loading due to the biological particles in the water was made by taking the weekly POC, chlorophyll *a*, particle volume and particle number (4-80 μm) data and combining these to produce a particle index. The weekly data from March 1989 to March 1993 were examined as cumulative frequency distributions. The data were then divided into quartiles. The weekly POC, chlorophyll *a*, particle volume and particle number results from 1992 were then assigned a ranking of 1 if they fell in the lowest quartile and 4 for the highest quartile. A weekly total of ranks for each of the four measures was calculated. This was used as a particle index for every filter run which corresponded to the samples being taken.

The mean C/Co data from runs which were grouped in the same class were plotted against estimated filter run time elapsed. The time was in nominal 15 minute intervals, since the actual time since backwashing was not recorded. The first sample after backwashing was taken as 0 minutes, the second 15 minutes and so on. For runs when the logging interval was 5 minutes the database was modified to select only every third measurement after the end of backwashing.

6.2.2 THE EFFECT OF FLOW RATE ON FILTER RIPENING

RESULTS FROM FILTRATE SAMPLE MEANS

UNDOSED FILTER

The mean filtrate data from column 4 (January 1992 - March 1993) were examined to determine if there was a relationship between flow rate and turbidity removal (figure 6.2). The graph suggests that filter flow rate was not influential in determining turbidity removal i.e. ripened filter performance. The variation within each flow rate was greater than the variation between flow rates.

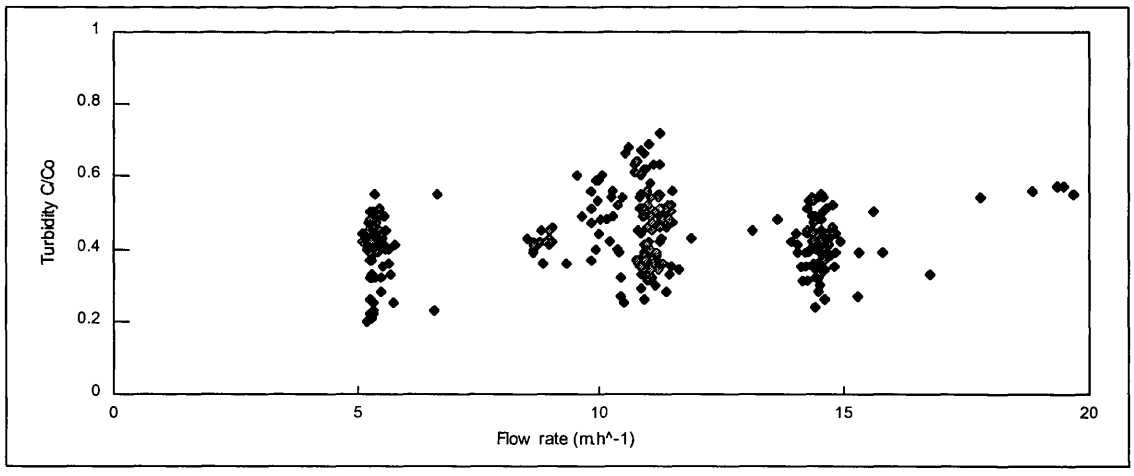

Figure 6.2. *Column 4 mean turbidity removal against mean flow rate calculated from each run.*

IRON AND OZONE DOSED FILTER

The data from column 5 and column 1 with optimised iron and ozone dosing showing mean filter run turbidity removals and flow rates plotted over time (figures 6.3 and 6.4) suggested that seasonal variations influenced turbidity removal more than flow rate.

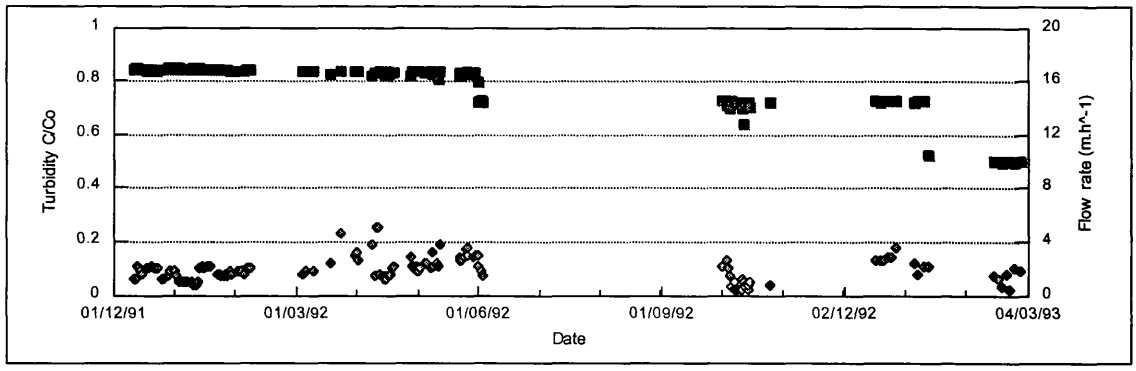

Figure 6.3. *Column 5 mean filter flow rate (squares) and mean turbidity removal data (diamonds) for optimised iron and ozone dosed runs.*

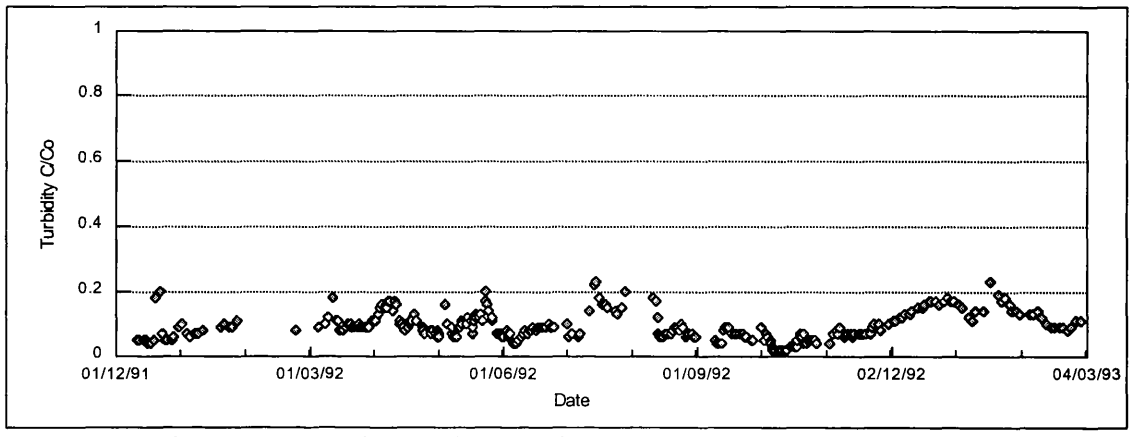

Figure 6.4. *Column 1 mean turbidity C/Co data for optimised iron and ozone dosed runs at 15 m.h^{-1}.*

RESULTS FROM FILTRATE TRENDS

UNDOSED FILTER

Figure 6.5 shows the effect of a change in flow rate on filter performance. The preceding run at 5 m.h^{-1} and the following run at 15 m.h^{-1} are also shown (runs 840 and 842). Remarkably little disturbance in filtrate turbidity resulted from a very large step change in flow in run 841. The effect of changing the flow rate was more noticeable on turbidity removal than on turbidity itself.

Figure 6.6 shows the turbidity removals of these three runs. Of the two runs at 5 m.h^{-1}, run 840 appeared to show some ripening, yet there was no evidence of this in run 841. Although the removals were different, the lack of ripening was also evident in run 842. Runs 840 and 842 both showed long term ripening, i.e. a gradual improvement in removals over the entire run. Whilst removals differed, initial removal behaviour was similar in both runs 841 and 842, suggesting a slow start would not have been beneficial to the higher rate filter.

An earlier sequence of runs, with target flow rates varied between 5 and 21 $m.h^{-1}$, is shown in figures 6.7 and 6.8. The dependence of turbidity removal on filter flow rate is clearly seen, together with evidence of that this relationship held during periods of declining rate and during step changes in flow rate. Declining rate is illustrated by run 789, where the automatic backwash initiation was disabled and declining rate was caused by the clogging of the filter, and runs 797 and 798, where 21 $m.h^{-1}$ was too great a flow rate for the equipment to sustain. A step increase flow rate was initiated in runs 792, 793, 795 and 796. Each was accompanied by a jump in head loss and a corresponding fall in turbidity removal.

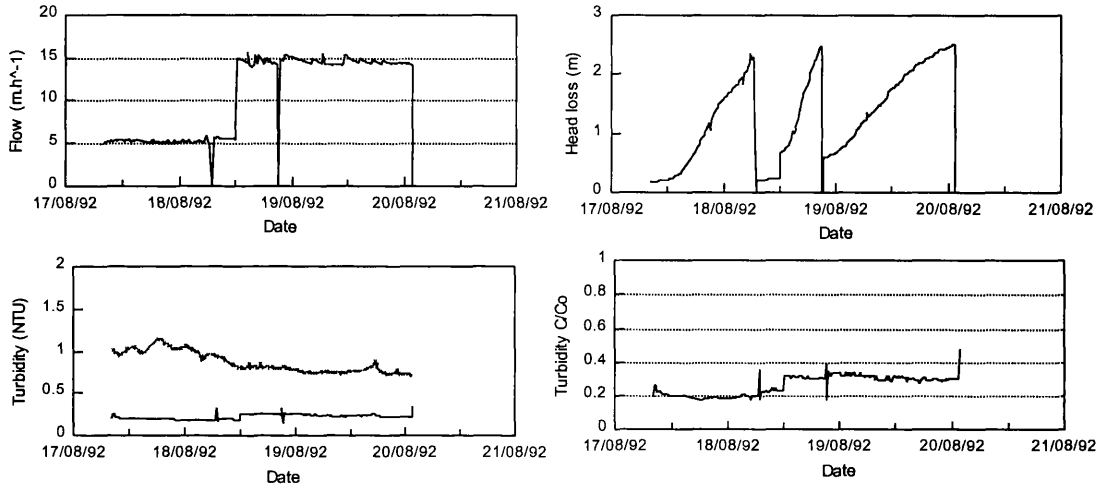

Figure 6.5. *Filter performance data for column 4 run number 840-842. Run 841 showed the a relatively small effect on filtrate turbidity from a large step change in flow rate. 17-20.8.92.*

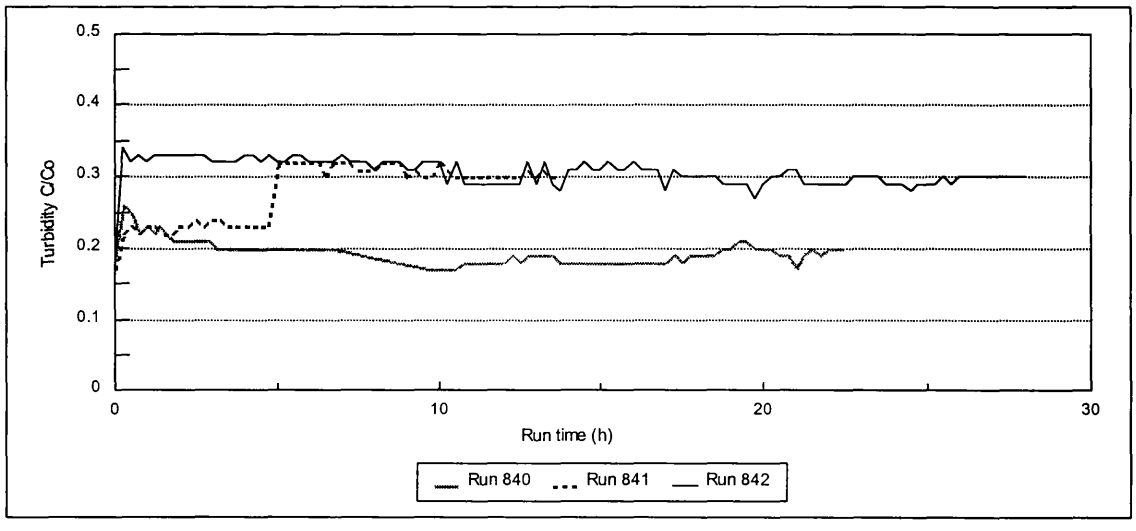

Figure 6.6. *Turbidity removal for column 4 run number 840-842, operated at 5 $m.h^{-1}$ and 15 $m.h^{-1}$, 17-20.8.92.*

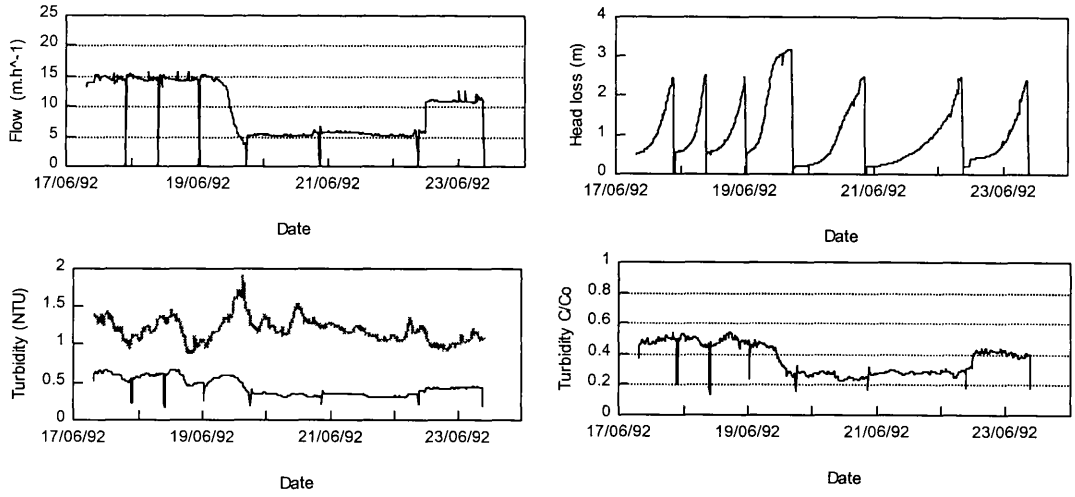

Figure 6.7. *Column 4 data from runs 786 to 792, 17-23.6.92.*

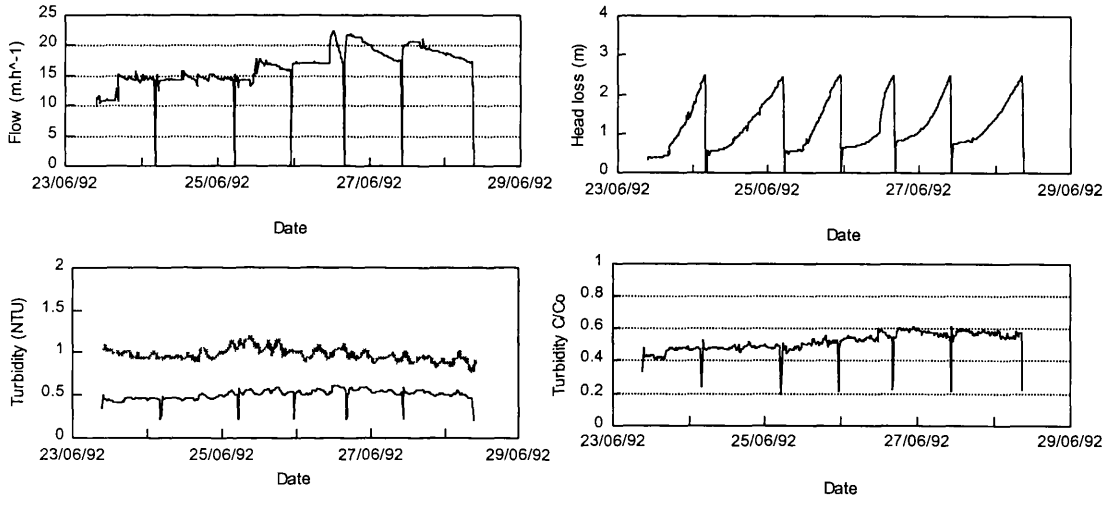

Figure 6.8. *Column 4 data from runs 793 to 798, 23-28.6.92.*

IRON DOSED FILTER

The background water quality parameters and the calculated iron doses for the experimental period are shown in tables 6.5 and 6.6. Stored water turbidity and other water quality parameters varied slightly over this trial. The chlorophyll *a* and total particle volume data indicated falling algal population sizes. The iron dose was reduced as the coagulant demand due to the algae diminished.

Iron doses were adjusted to maintain the optimum L-shaped trend. Run details are shown in figures 6.9 to 6.11, indicating that the doses applied produced some L-shaped runs, but a slight overdose on some of the runs at 12 and 17 $m.h^{-1}$ was apparent.

Table 6.5. *Parameters measured weekly over the period of the flow rate trials on column 5.*

Date	POC ($\mu g.l^{-1}$)	Chlorophyll *a* ($\mu g.l^{-1}$)	Total Particle Volume 4-80 μm (ppm)	Total Particle Number 4-80 μm (per ml)
21.11.91	464	3.4	1.097	4869
28.11.91	317	2.4	0.775	3309
05.12.91	361	1.3	0.683	4012

Table 6.6. *Parameters measured daily over the period of the flow rate trials on column 5. Times when coagulant dose was changed are shown.*

Date	Flow rate ($m.h^{-1}$)	Stored water turbidity (NTU)	Water temperature (°C)	Iron dose ($mg.l^{-1}$)
21.11.91	12	2.5	7.0	0.8 after 11:30
22.11.91	12	1.8	8.0	0.8
25.11.91	17	1.8	8.0	0.8 1.1 after 12:00
26.11.91	17	2.1	9.0	1.1 0.4 after 10:40
27.11.91	17 22	1.5	8.0	0.4 0.6 after 13:00
28.11.91	22	1.5	7.8	0.6 0.4 after 10:50
29.11.91	22	1.3	8.5	0.4

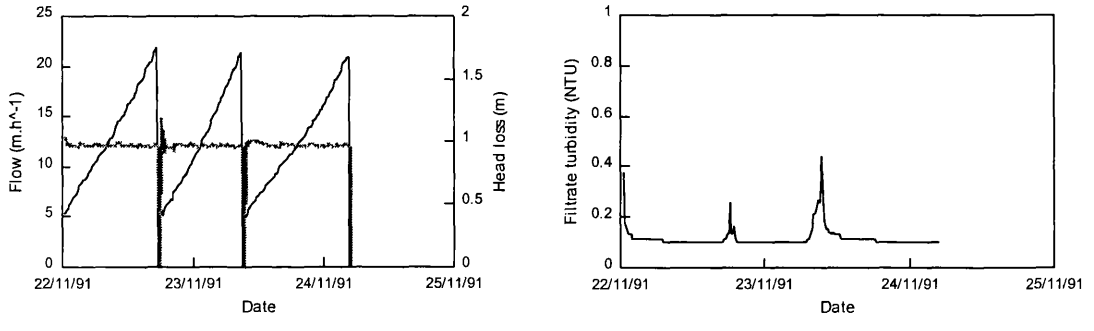

Figure 6.9. *Column 5 data from runs 456 to 458, at 12 $m.h^{-1}$.*

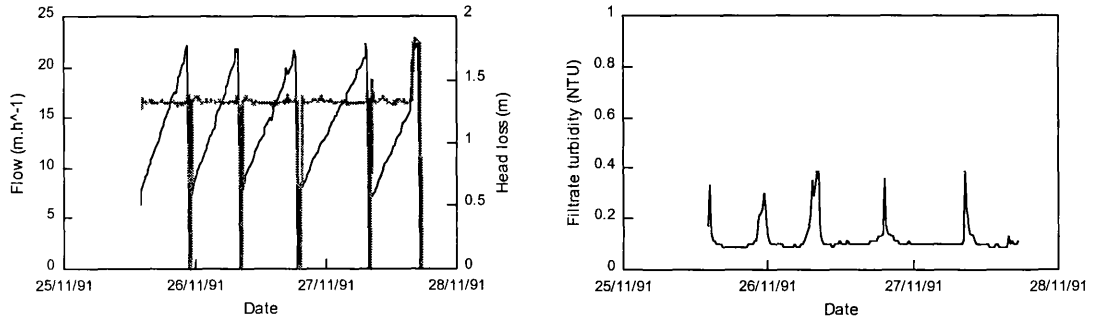

Figure 6.10. *Column 5 data from runs 460 to 464, at 17 $m.h^{-1}$.*

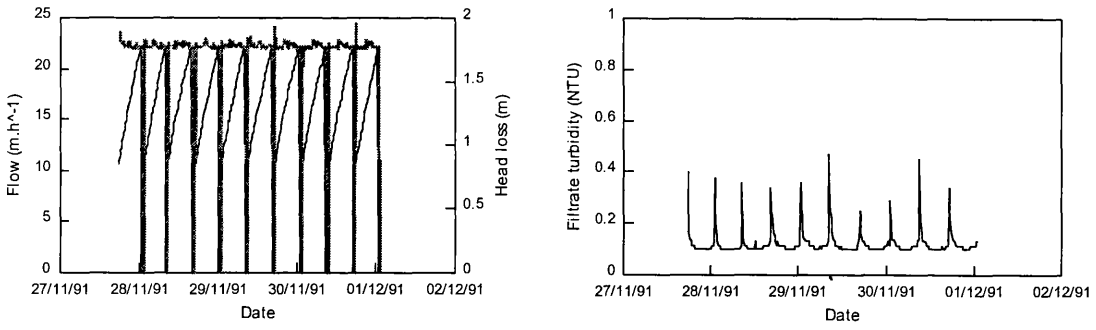

Figure 6.11. *Column 5 data from runs 465 to 474, at 22 m.h^{-1}.*

A summary of the results from individual runs is shown in figure 6.12. The turbidity data for the first 6 hours from 3 runs at 12 m.h^{-1} (456-458, 22-24.11.91), 5 runs at 17 m.h^{-1} (460-464, 25-27.11.91) and 10 runs at 22 m.h^{-1} (465-474, 27-30.11.91) are plotted as a mean for each flow rate. The iron pump air locked during run 459 so these data were excluded. It appeared that flow rate had negligible impact on the ripening behaviour of an iron dosed dual media filter.

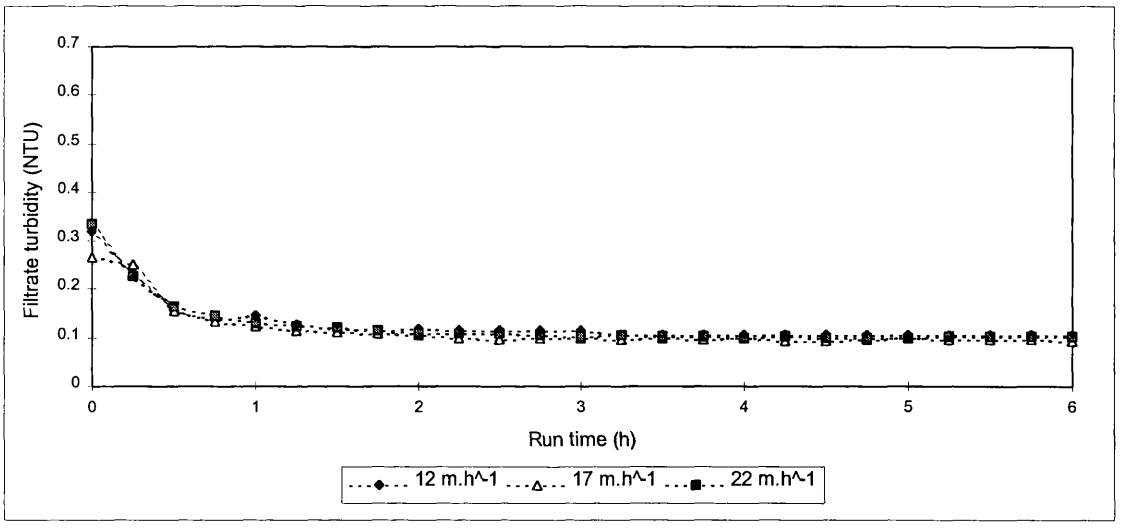

Figure 6.12. *Column 5 mean filter ripening data from iron dosed runs 456 to 474, 22-30.11.91, showing negligible effect of filtration rate on ripening times and filtrate turbidity profiles.*

IRON AND OZONE DOSED FILTER

Filter runs 683 to 706 (4-20.11.91) were examined where flow rate was stepped down from 22 m.h^{-1} to 17 m.h^{-1} and finally 12 m.h^{-1}. The background water quality parameters and the calculated iron doses are shown in tables 6.7 and 6.8. The data showed the decline of the autumn algal blooms. The iron dose was relatively high as a consequence.

Table 6.7. *Parameters measured weekly over the period of the flow rate trials on column 6.*

Date	POC ($\mu g.l^{-1}$)	Chlorophyll a ($\mu g.l^{-1}$)	Total Particle Volume 4-80 μm (ppm)	Total Particle Number 4-80 μm (per ml)
31.10.91	465	4.0	2.161	9254
07.11.91	500	10.8	1.073	5731
14.11.81	467	5.3		
21.11.91	464	3.4	1.097	4869

Table 6.8. *Parameters measured daily over the period of the flow rate trials on column 6. Times when coagulant dose was changed are shown.*

Date	Flow rate ($m.h^{-1}$)	Stored water turbidity (NTU)	Water temperature (°C)	Iron dose ($mg.l^{-1}$)
04.11.91	22	2	11.0	1.1
05.11.91	22	1.5		1.1
06.11.91	22	1.5		1.1
07.11.91	22 17	1.5		1.1 0.8 after 15:30
08.11.91	17	1.5		0.8
11.11.91	17	1.6	13.5	0.8
12.11.91	17	1.9		0.8 1.0 after 12:00
13.11.91	17	1.7	9.5	1.0 0.8 after 16:45
14.11.91	17 12	2.3	9.0	0.8
15.11.91	12	1.2	9.5	0.9
18.11.91	12	1.5	10.5	1.0
19.11.91	12	1.5	8.0	1.0
20.11.91	12	2.4		1.0

The flow, head loss and turbidity trend data are shown in figures 6.13 to 6.15. The turbidity data for the first 6 hours from 8 runs at 22 $m.h^{-1}$ (683-690, 4-7.11.91), 8 runs at 17 $m.h^{-1}$ (694-701, 10-14.11.91) and 5 runs at 12 $m.h^{-1}$ (702-706, 15-19.11.91) are plotted as a mean for each flow rate in figure 6.16. It appeared that flow rate had little impact on the ripening behaviour of an iron and ozone dosed dual media filter.

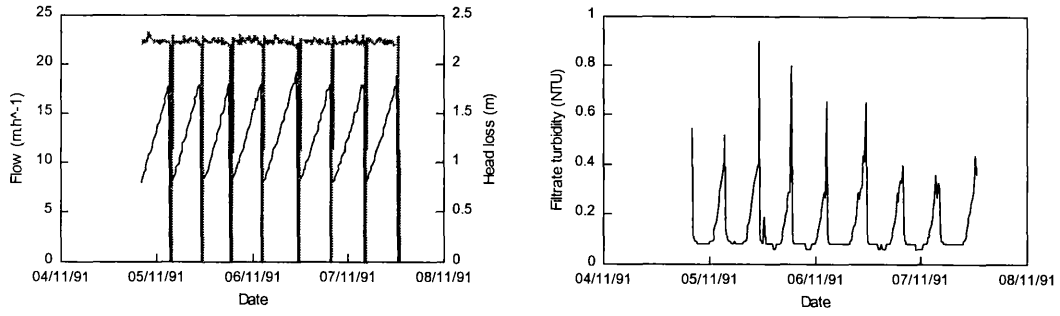

Figure 6.13. *Column 6 data from runs 683 to 690, at 22 m.h^{-1}.*

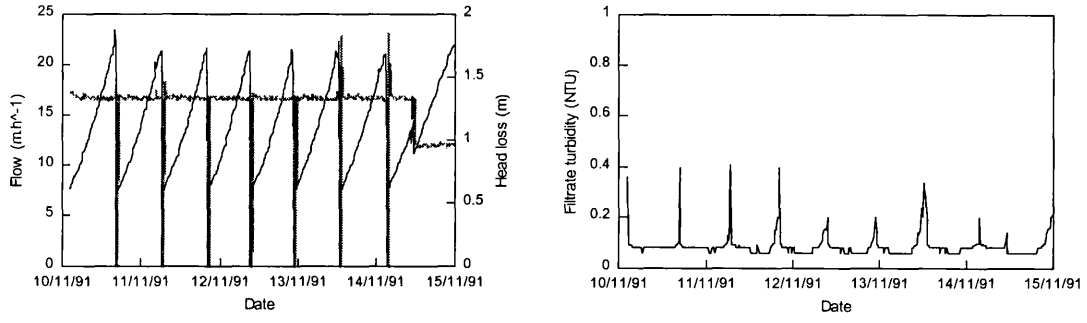

Figure 6.14. *Column 6 data from runs 694 to 701, at 17 m.h^{-1}.*

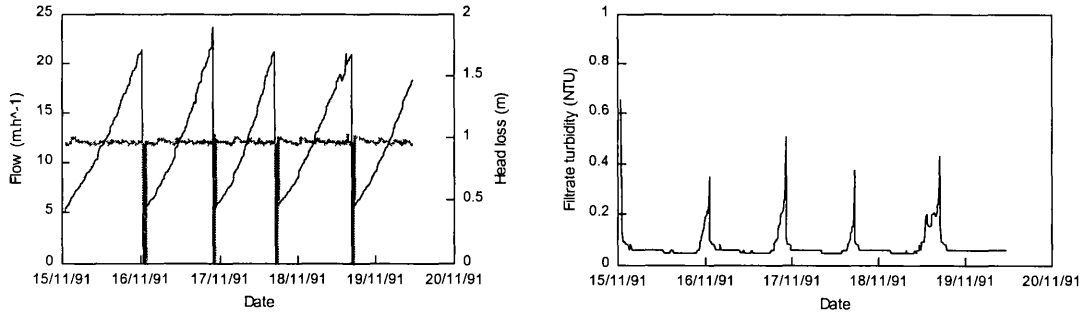

Figure 6.15. *Column 6 data from runs 702 to 706, at 12 m.h^{-1}.*

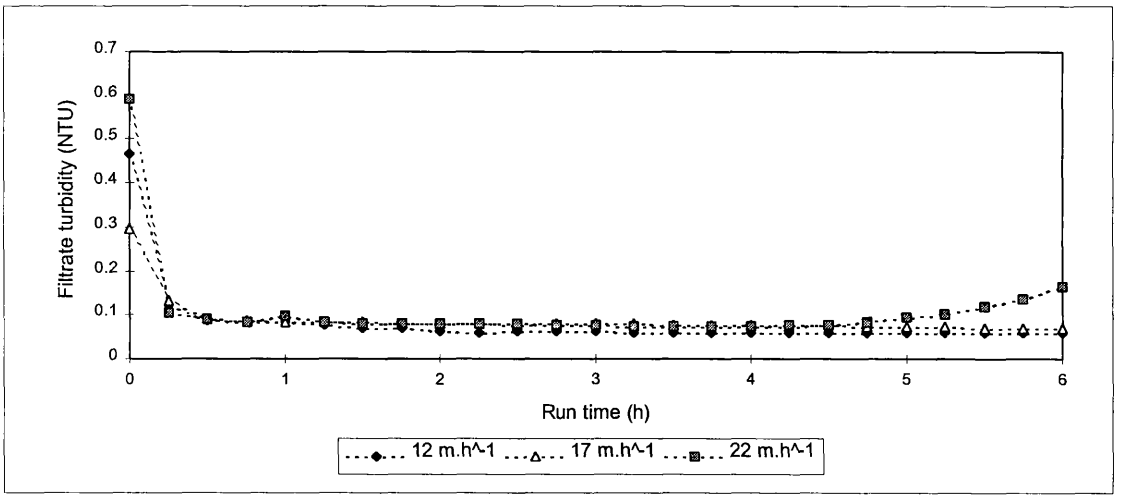

Figure 6.16. *Column 6 mean filter ripening data from iron and ozone dosed runs 683 to 706 (691 to 693 excluded), 4-20.11.91, showing negligible effect of filtration rate on ripening times and filtrate turbidity profiles.*

6.2.3 THE EFFECT OF TEMPERATURE ON FILTER RIPENING

UNDOSED FILTER

A water temperature result was available for some of the filter runs. These data have been plotted in figure 6.17 to show mean turbidity removal at different temperatures. The data have been divided into three flow rate classes. Whilst there was some scatter the graph suggests that temperature, or one or more factors correlated with temperature, was influential in assisting turbidity removal. The linear regression $y = -0.02x + 0.67$, $r^2 = 0.65$, was calculated without regard to flow rate. The r^2 value for the relationship between removal and temperature suggested 65% of the variation in removals was accounted for by temperature. Figure 6.17 showed that flow rate did not appear to influence the removals.

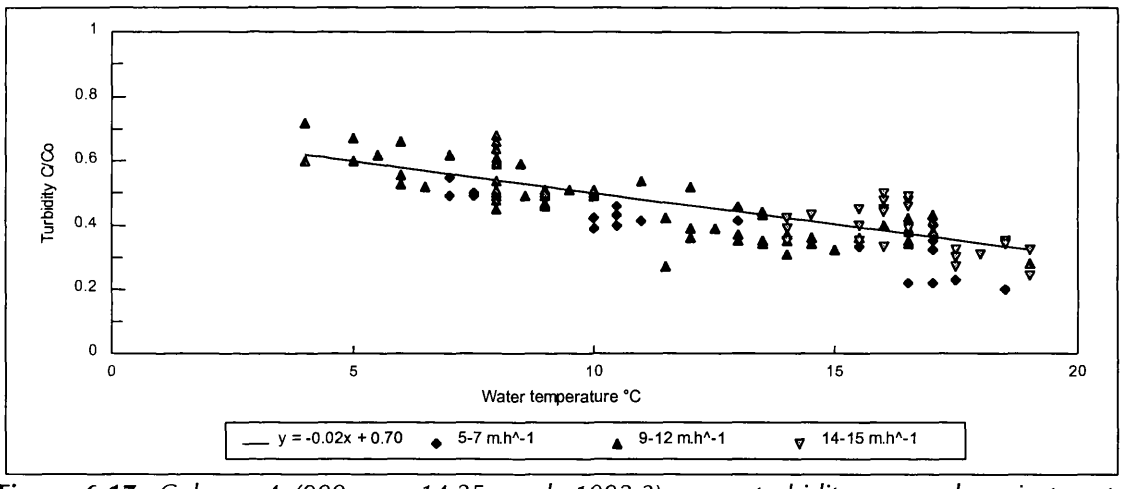

Figure 6.17. *Column 4 (900 mm 14-25 sand, 1992-3) mean turbidity removal against water temperature, with mean filter flow rate range indicated. The linear regression was calculated disregarding flow rate.*

IRON AND OZONE DOSED FILTER

Mean filter run turbidity removal data from column 1 operated over 14 months at 15 $m.h^{-1}$ plotted in figure 6.18 suggest that the iron and ozone dosing made the filter performance independent of water temperature.

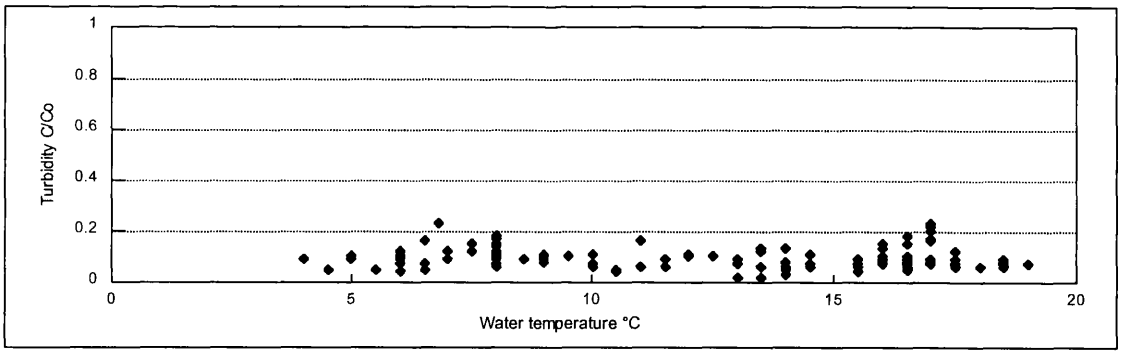

Figure 6.18. *Column 1 mean filter run turbidity removal against water temperature, for iron and ozone dosed water, flow rate 15 m.h^{-1}, December 1991 - January 1993.*

6.2.4 THE EFFECT OF FILTER LOADING ON RIPENING

RESULTS FROM FILTRATE SAMPLE MEANS

UNDOSED FILTER

Stored water turbidity data were used as a measure of applied load. Figure 6.19 shows the mean turbidity removal plotted against inlet turbidity with the flow rate illustrated, based on the three clusters in figure 6.2. There was no clear relationship between flow rate, inlet turbidity and turbidity removal. Mean filter turbidity removal was essentially independent of mean inlet turbidity.

IRON AND OZONE DOSED FILTER

Figures (6.20 and 6.21) show column 1 data (15 m.h^{-1}) for mean filtrate turbidity and turbidity removal against mean stored water turbidity for individual runs dosed with ozone and iron. The data have been divided into autumn/winter and spring/summer graphs. The evidence of these graphs is that the influence of load on filter performance was threefold, depending on seasonal factors: i) in the autumn and winter removals were generally poorer than in the summer - both removals and filtrate turbidities showed a tendency towards proportionality with the load; ii) in the spring and summer the filtrate turbidity was good yet, below 2 NTU, removals tended to be inversely proportional with inlet turbidity, despite filtrate turbidity remaining good; iii) at loadings over 2 NTU the removals and the filtrate turbidity values showed less dependence on the inlet turbidity than was the case in the autumn and winter.

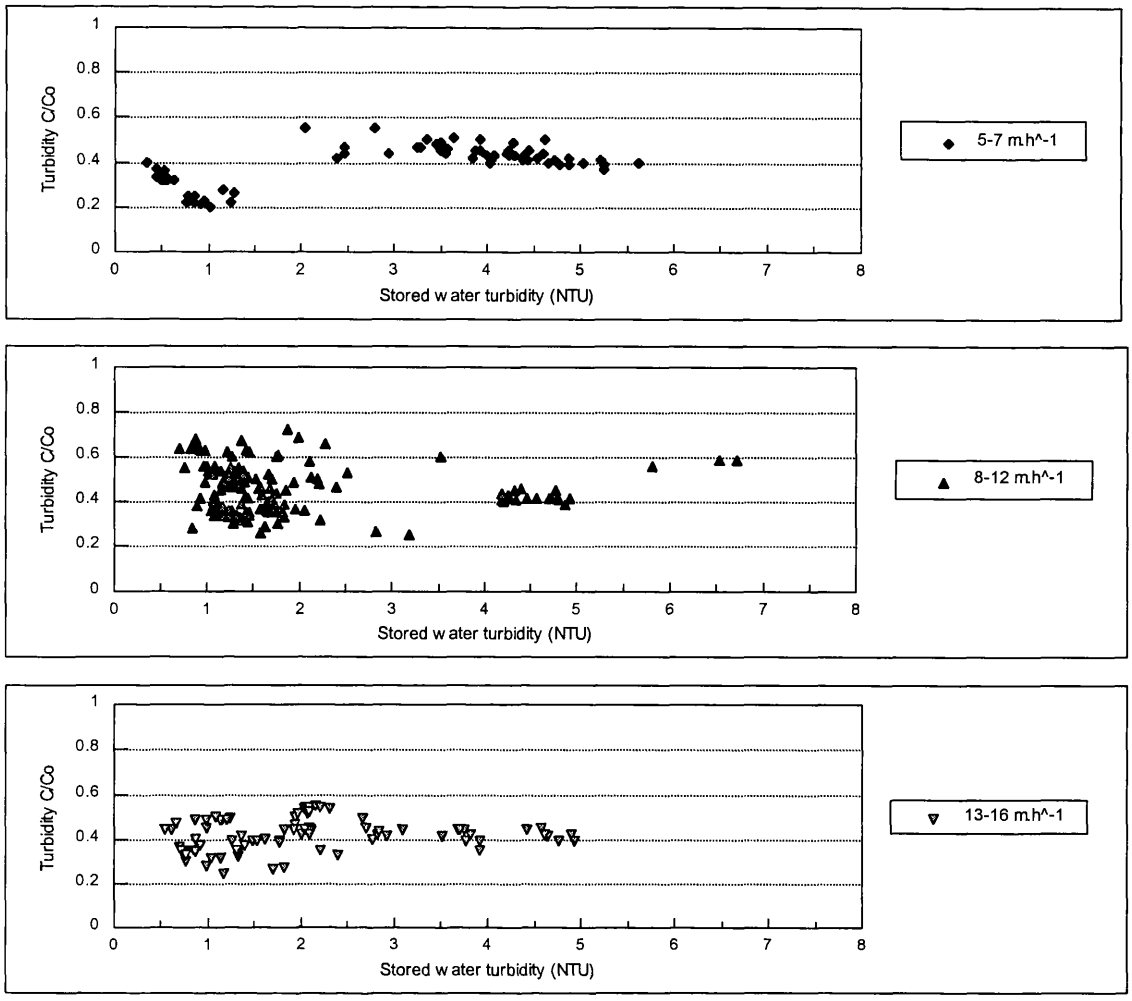

Figure 6.19. *Column 4 mean turbidity removal against mean influent turbidity calculated from each run, divided into flow rate groups.*

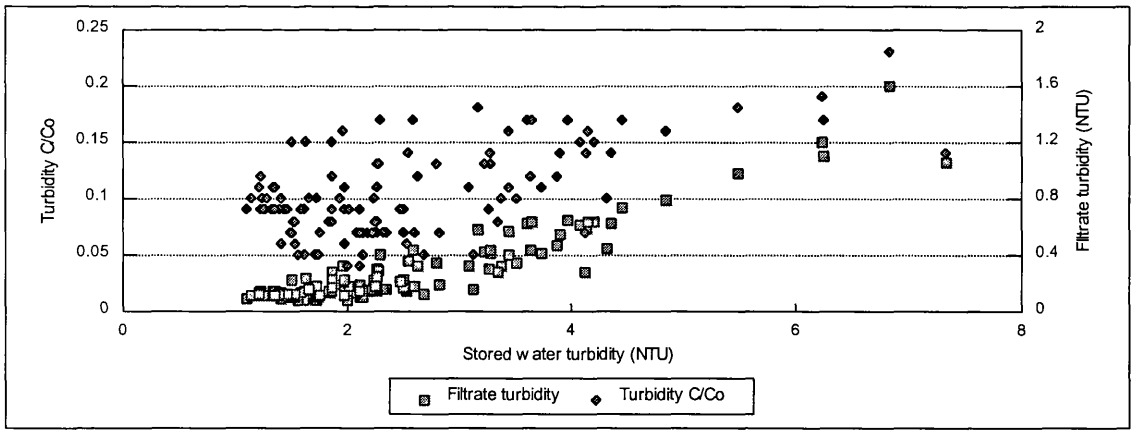

Figure 6.20. *Column 1 mean autumn and winter filter run turbidity removal and filtrate turbidity against mean stored water turbidity, for iron and ozone dosed water, flow rate 15 m.h^{-1}, December 1991 - March 1992, November 1992 - March 1993.*

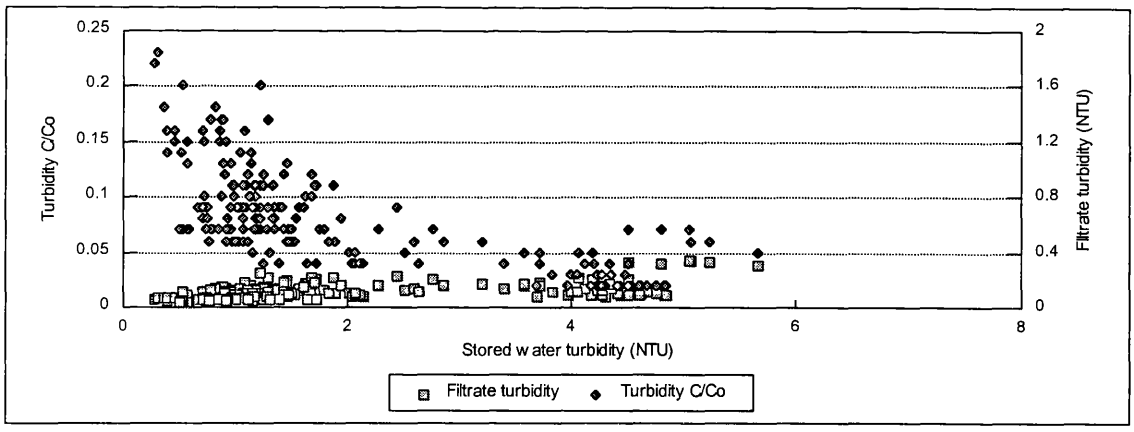

Figure 6.21. *Column 1 mean spring and summer filter run turbidity removal and filtrate turbidity against mean stored water turbidity, for iron and ozone dosed water, flow rate 15 m.h^{-1}, April - October 1992 inclusive.*

RESULTS FROM TURBIDITY TRENDS

UNDOSED FILTER

Figures 6.22 and 6.23 have been plotted to show whether the mean stored water turbidity calculated over the first 6 hours of each run had any influence on turbidity removal. Figure 6.22 shows results from filter runs at 5-6 m.h^{-1} taking place from July to December 1992. Figure 6.23 shows results from filter runs at 8-12 m.h^{-1} which were recorded from January 1992 to January 1993. The general impression from both graphs is of a lack of influence of inlet turbidity on ripening time, minimum initial turbidity removal, or ripened removal. The effect of flow rate on ripened removals also appeared minimal.

IRON AND OZONE DOSED FILTER

Figures 6.24 and 6.25 show filtrate turbidity and turbidity removal trends from 237 iron and ozone dosed runs on column 1. The data were means for all the filter runs within the size classes indicated. The higher turbidity runs tended to take place in the autumn and winter. The number of runs in each size class ranged from 88 (1.0-1.5 NTU) to 8 (<0.5 NTU).

The results showed that up to inlet values of 4 NTU the initial peak turbidity and the ripened filtrate turbidity both increased with inlet values. Surprisingly, these figures were lower beyond 4 NTU. The reason for this was due to the runs with stored water turbidities >4 NTU being concentrated in October 1992, when good filter performance was achieved. Poorer removals were measured in

December 1992 and January 1993, and these were reflected in the runs with stored water turbidity ranging from 2-4 NTU.

For each inlet class ripening was shown to be rapid for the initial hour, continuing more slowly for two more hours. Removal data showed little influence from turbidity load. Two turbidity classes have been plotted in detail in figures 6.26 to 6.29. There were 88 runs in the 1-1.5 NTU class and 18 in the 3.5-4 NTU class. Error bars showing one standard deviation around the mean were included. They showed the range of turbidity values around the mean tended to be larger for the higher inlet turbidities, but this did not cause a broader spread of removal values. The error bars tended to be larger in the first two sample periods. This reflects the greater variations in the data due to the samples at time 0 h and time 0.25 h actually being collected at random with respect to the pre-ripening and ripening phases during the 15 minute period after the nominal time.

Original in colour

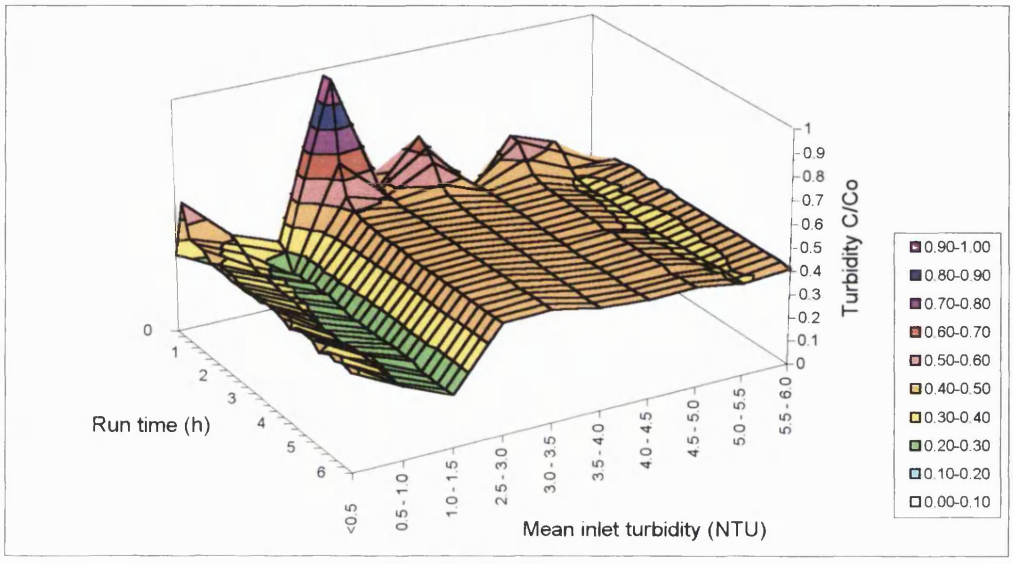

Figure 6.22. *Influence of stored water turbidity on turbidity removal, from means of column 4 filter runs at 5-6 m.h^-1, July - December 1992. No chemicals dosed.*

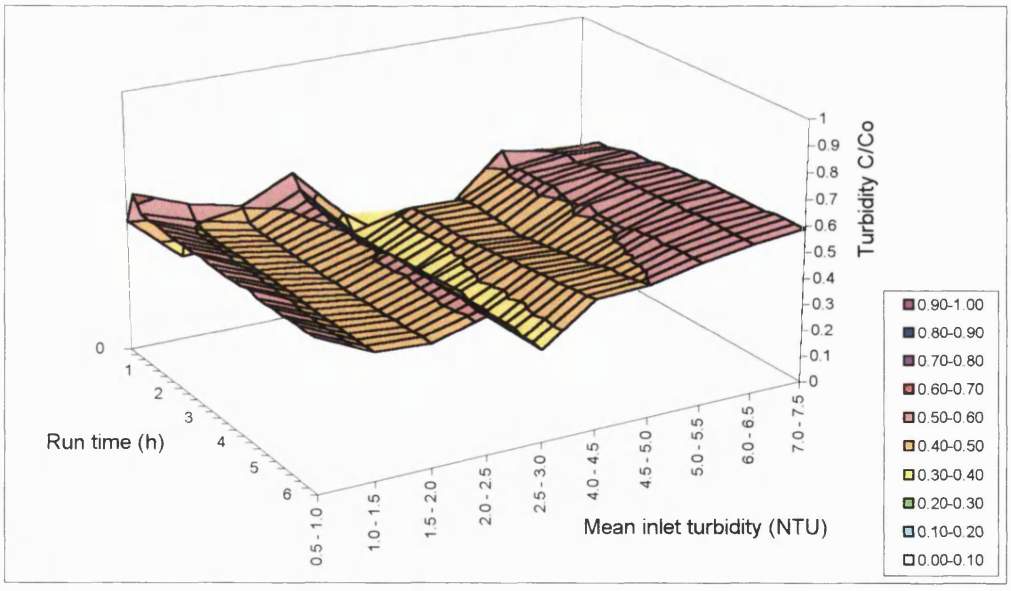

Figure 6.23. *Influence of stored water turbidity on turbidity removal, from means of column 4 filter runs at 8-12 m.h^-1, January 1992 - January 1993. No chemicals dosed.*

Original in colour

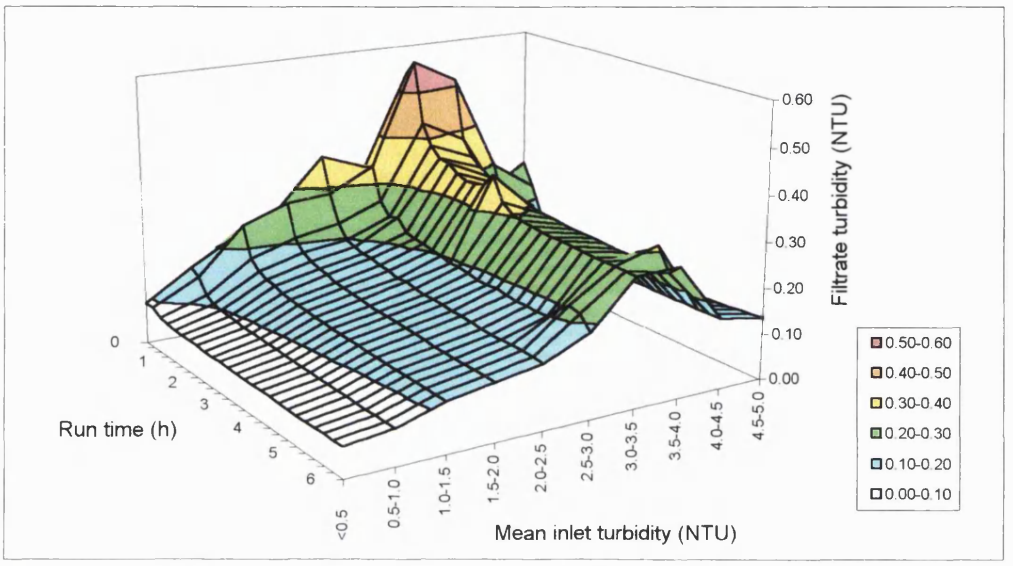

Figure 6.24. *Influence of stored water turbidity on turbidity ripening, from means of column 1 filter runs, dosed with iron and ozone, December 1991 - March 1993.*

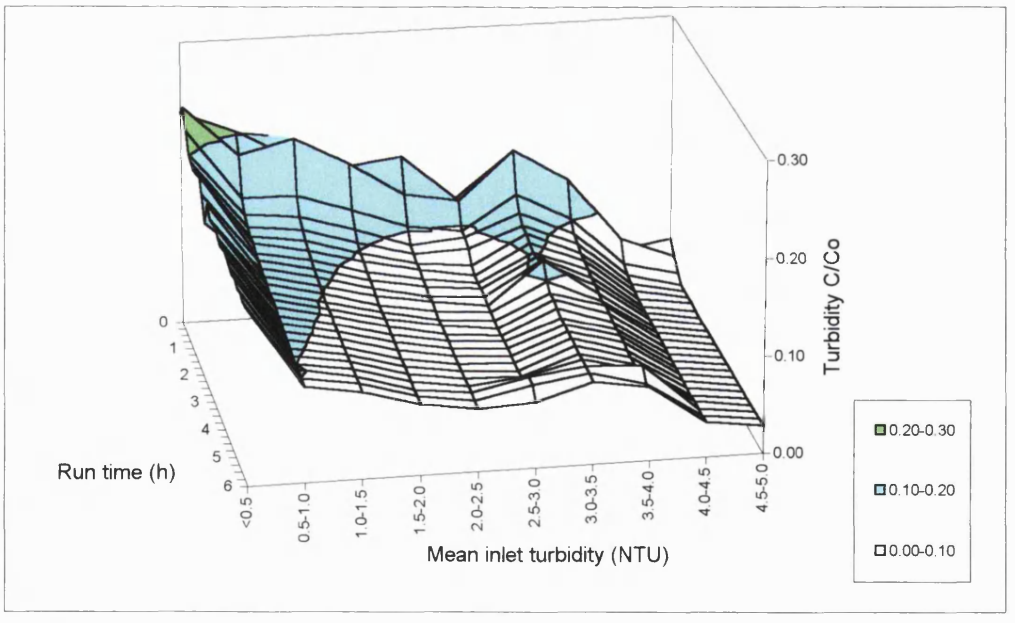

Figure 6.25. *Influence of stored water turbidity on filter turbidity removal, from means of column 1 filter runs, dosed with iron and ozone, December 1991 - March 1993.*

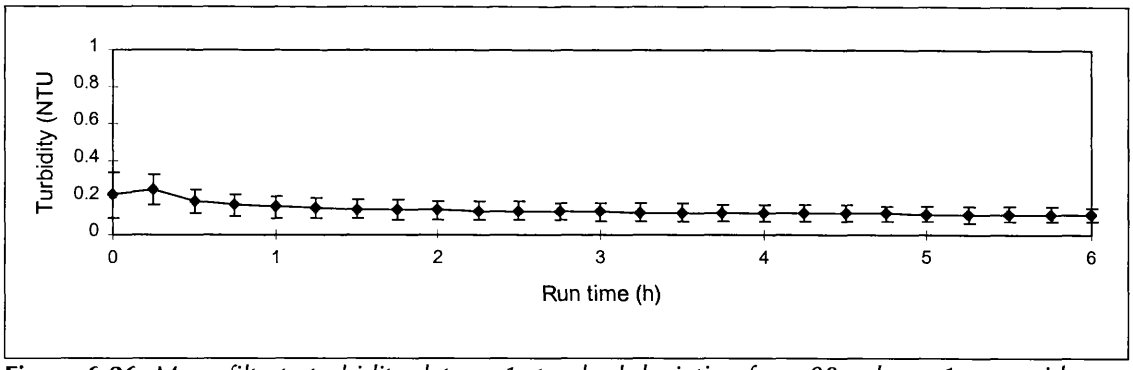

Figure 6.26. *Mean filtrate turbidity data ± 1 standard deviation from 88 column 1 runs with mean inlet turbidity of 1 - 1.5 NTU.*

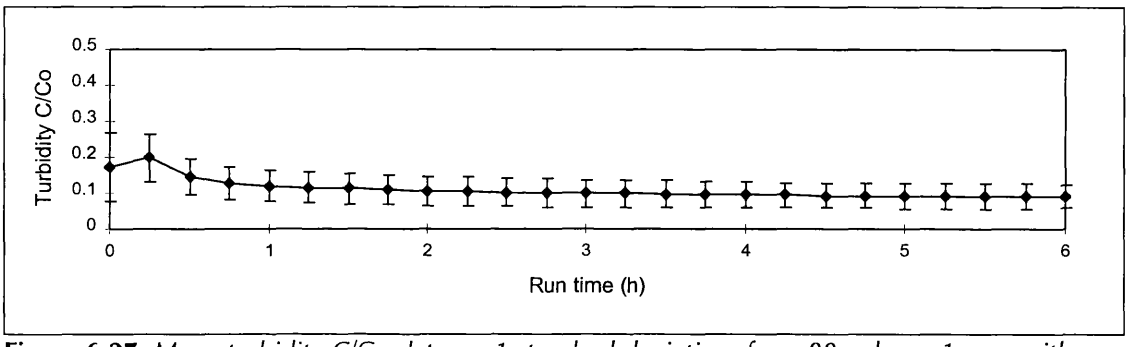

Figure 6.27. *Mean turbidity C/Co data, ± 1 standard deviation, from 88 column 1 runs with mean inlet turbidity of 1 - 1.5 NTU.*

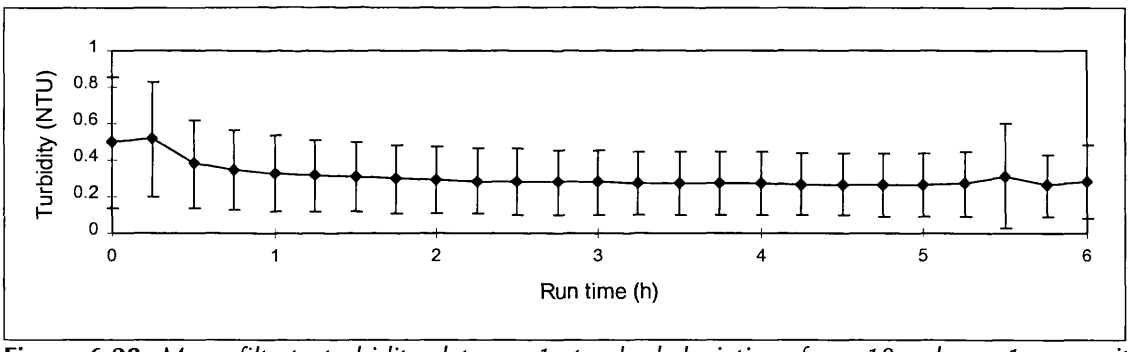

Figure 6.28. *Mean filtrate turbidity data, ± 1 standard deviation, from 18 column 1 runs with mean inlet turbidity of 3.5 - 4 NTU.*

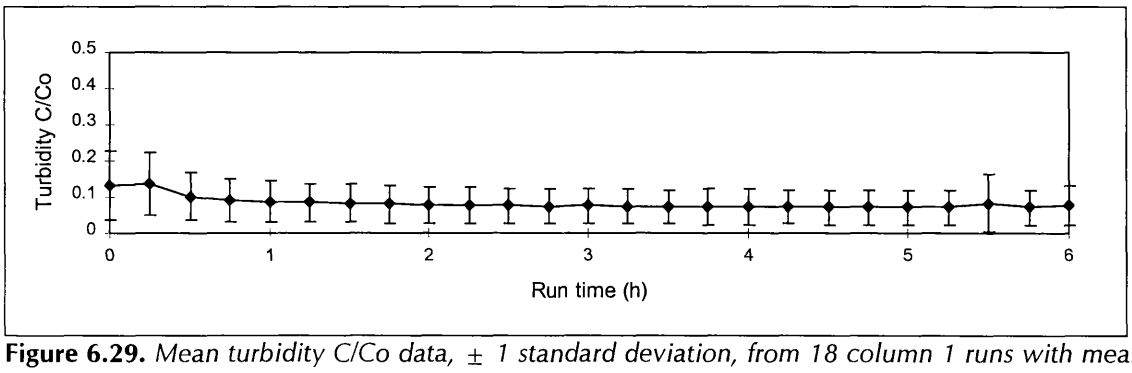

Figure 6.29. *Mean turbidity C/Co data, ± 1 standard deviation, from 18 column 1 runs with mean inlet turbidity 3.5 - 4 NTU.*

6.2.5 RIPENING AND PARTICLE INDEX

Four years' weekly stored water sample chlorophyll *a*, POC, particle number and particle volume data were plotted as cumulative frequency distributions (the original data were shown in figures 3.3 to 3.6). The values of each parameter below which 25%, 50% and 75% of samples were measured were used to divide the data into four ranks. Table 6.9 shows the value ranges of these quartiles, used to generate the rank value. The four rankings were added to produce a particle index. Graphs were plotted showing the mean turbidity removal (C/Co) data for all the runs which were undertaken with the same particle index value, plotted against particle index. When iron was dosed turbidity data were also presented.

Table 6.9. *Particle index rank ranges calculated from weekly stored water sample fequency distribution data.*

Parameter	Rank 1 range	Rank 2 range	Rank 3 range	Rank 4 range	Number of samples
Chlorophyll a ($\mu g.l^{-1}$)	< 2.7	2.7 - 5	5 - 10	> 10	195
POC ($\mu g.l^{-1}$)	< 400	400 - 530	530 - 700	> 700	194
Particle Number 4-80 μm (per ml)	< 3500	3500 - 5300	5300 - 8000	> 8000	218
Particle Volume 4-80 μm (ppm)	< 0.8	0.8 - 1.5	1.5 - 2.5	> 2.5	218

UNDOSED FILTER

The relationship between mean turbidity C/Co for the first 6 h of operation for groups of undosed column 4 filter runs and the particle index of the stored water is examined for runs carried out at 10-11 $m.h^{-1}$ in figure 6.30. The distribution of particle index data by date for the runs at this flow rate are shown in figure 6.31. The graph indicated that there was little in the way of ripening occurring at any of the particle loadings.

Original in colour

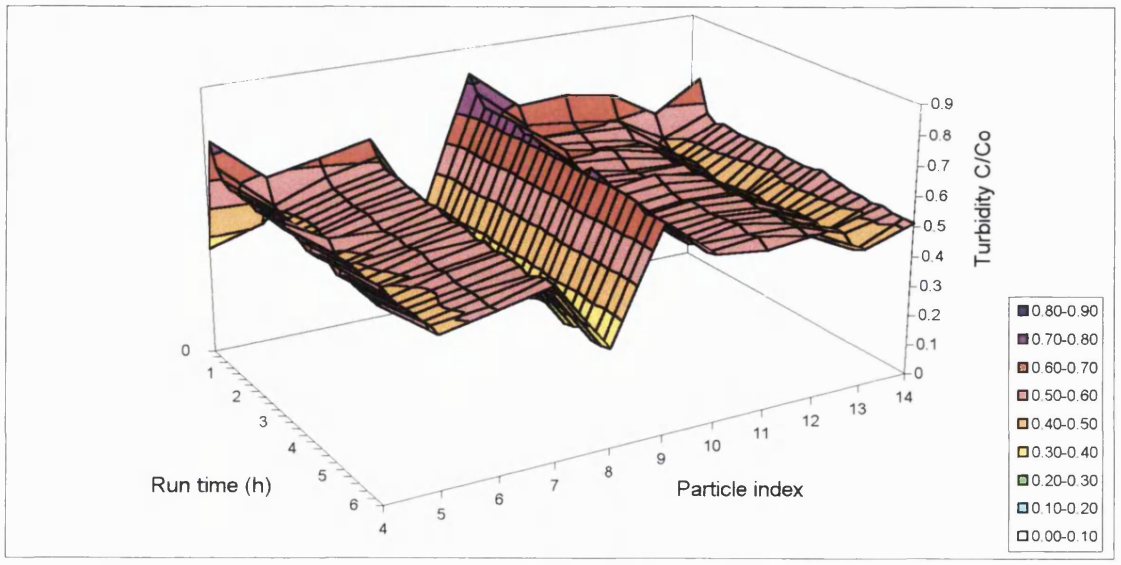

Figure 6.30. *Mean filter ripening behaviour and particle index, column 4, 10-11 m.h^-1, January - June 1992.*

Some irregularities in the trends were noticeable but formed no systematic pattern. The unusually good removals with a particle index of 8 were due to the two samples in this class being in late May and early June. The Thames Water biology laboratory reported the presence of *Aphanizomenon* sp. in their "frequent" and "abundant" categories in these two weeks. This is a filamentous blue-green alga, which appears as bundles of filaments under a microscope. The filaments probably conditioned the filter surface to improve turbidity removals by straining.

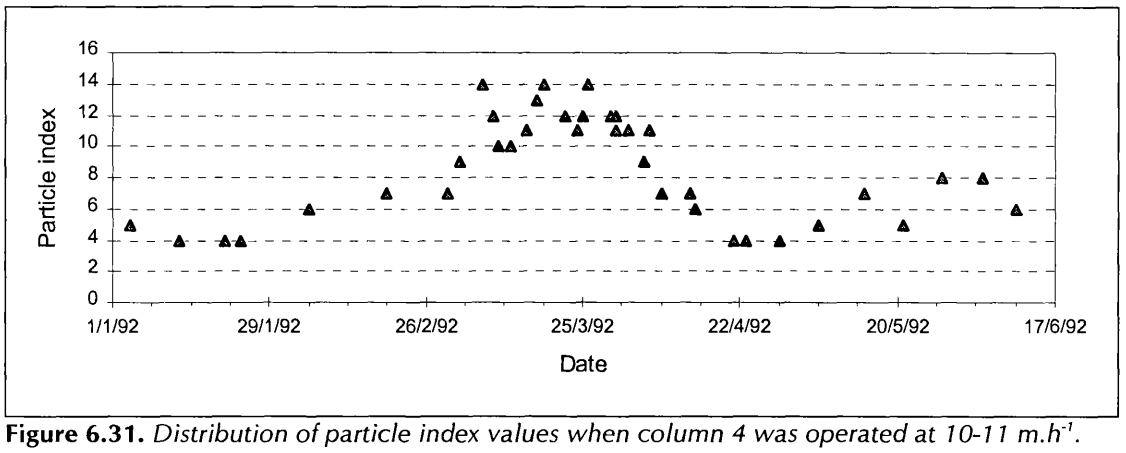

Figure 6.31. *Distribution of particle index values when column 4 was operated at 10-11 m.h^{-1}.*

As can be seen in figures 6.32 and 6.33 the particle index showed quite a spread in filter run times and turbidity removals, so was probably not a good means of estimating filter loading and predicting ripening.

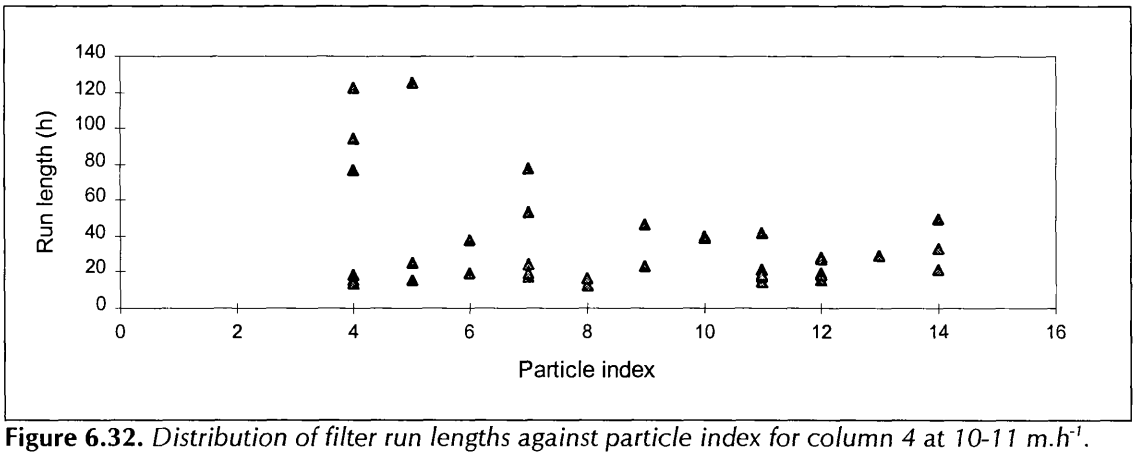

Figure 6.32. *Distribution of filter run lengths against particle index for column 4 at 10-11 m.h^{-1}.*

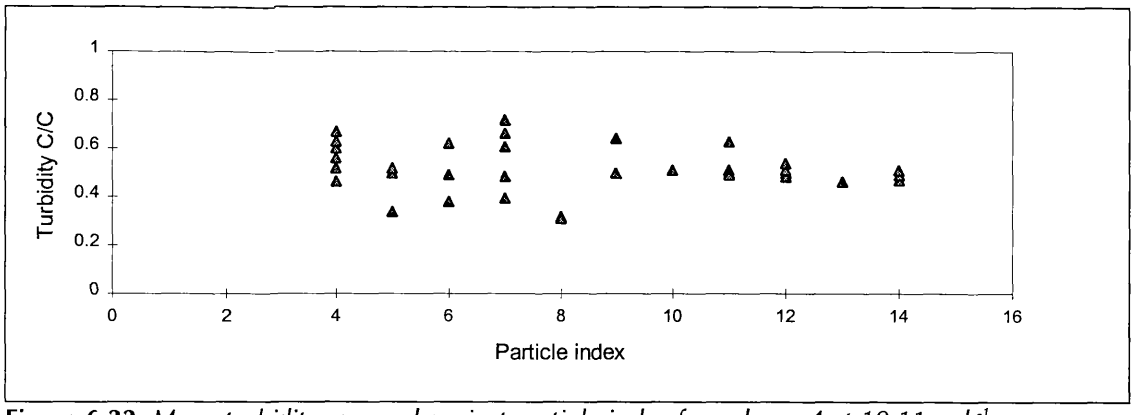

Figure 6.33. *Mean turbidity removal against particle index for column 4 at 10-11 m.h^{-1}.*

IRON AND OZONE DOSED FILTER

The relationship between particle index and mean filter ripening trends is examined for column 1 in figure 6.34, which shows turbidity, and in figure 6.35, with turbidity removal. In each case there was a small initial peak which lasted under 1 h followed by a gradual slow ripening. There appeared to be generally little to differentiate between ripening at different particle loadings. Further details are contained in the mean summary data in figures 6.36 to 6.38. Once again the spread of values within each particle index class made the particle index way of classifying loading unable to show any effect on ripening behaviour.

Original in colour

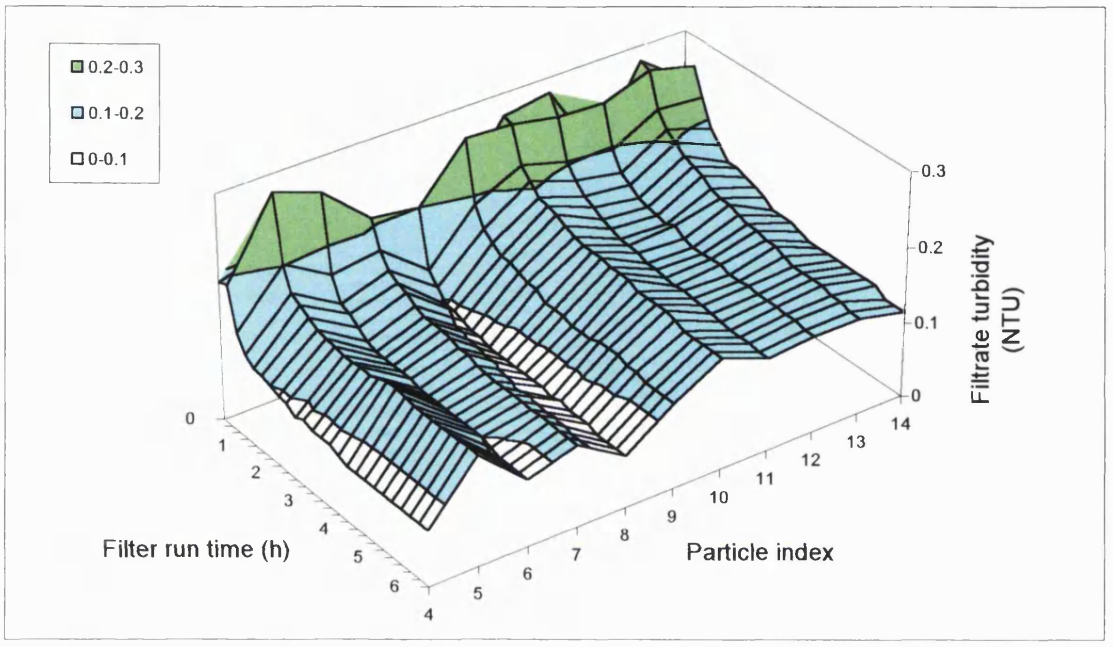

Figure 6.34. *Influence of stored water particle data on filter turbidity ripening, from means of column 1 filter runs, dosed with iron and ozone, December 1991 - December 1992.*

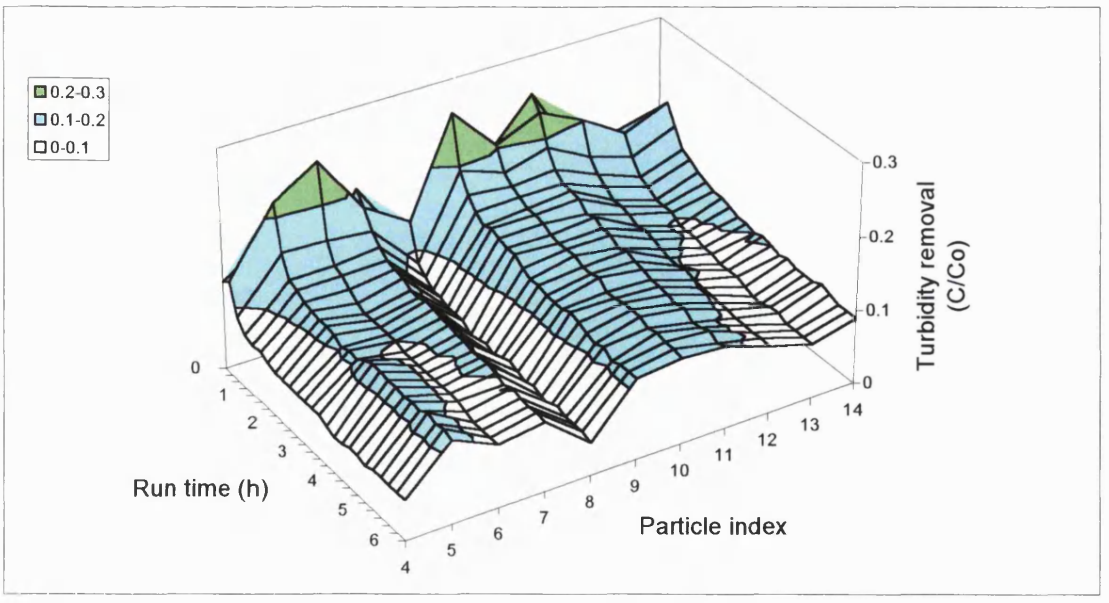

Figure 6.35. *Influence of stored water particle data on turbidity removal, from means of column 1 filter runs, dosed with iron and ozone, December 1991 - December 1992.*

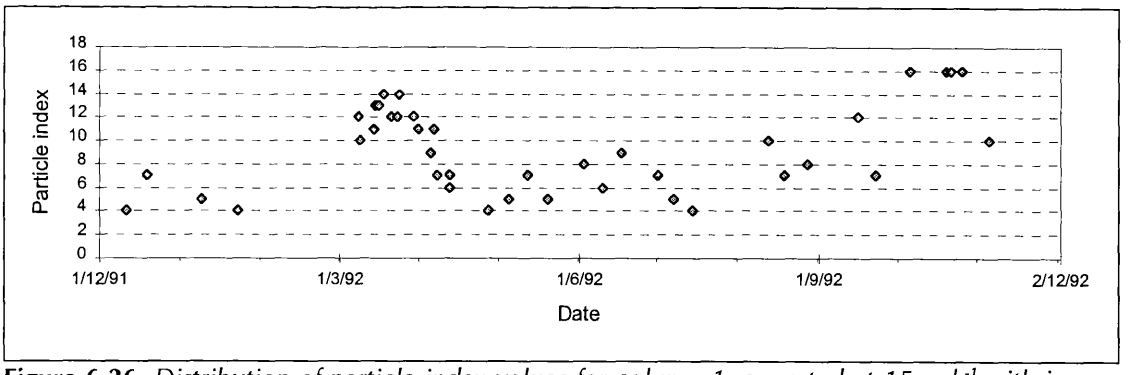

Figure 6.36. *Distribution of particle index values for column 1, operated at 15 m.h^{-1} with iron and ozone.*

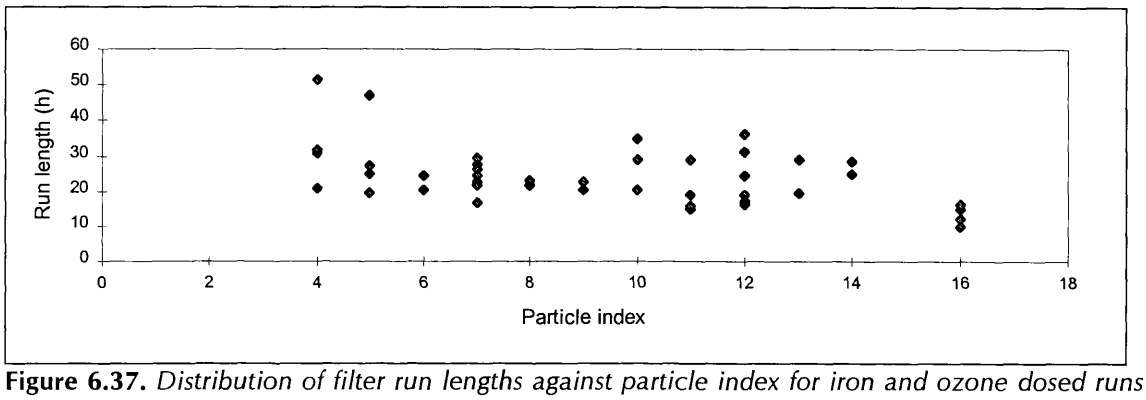

Figure 6.37. *Distribution of filter run lengths against particle index for iron and ozone dosed runs on column 1 at 15 m.h^{-1}.*

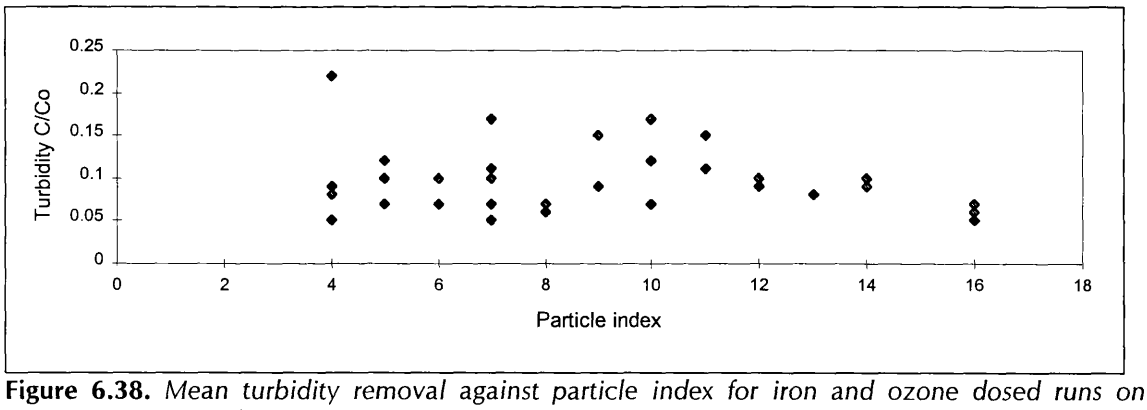

Figure 6.38. *Mean turbidity removal against particle index for iron and ozone dosed runs on column 1 at 15 m.h^{-1}.*

6.2.6 RIPENING AND FILTER RUN LENGTHS

An alternative way of considering load was by examining the filter run lengths. It is likely that these are governed by the mass, particle size and surface area of the load, the attachment efficiency, and the location, concentration and structure of the deposits.

UNDOSED FILTER

Figures 6.39 to 6.41 show turbidity removal as a function of filter run length at different flow rates over different time spans. There was a tendency for mean removal over the first 6 h of the runs to be poorer with longer filter runs. However there was no consistent indication of different rates of ripening. The data reflected the observation noted previously that there were some runs in which classic ripening curves were measured. These appear to have taken place in some of the longer filter runs. No categorical explanation for this can be made. The most likely explanation would be a failure of the rinse phase of the backwash, which went unrecorded over a period of unmanned plant operation. Backwash flows were not logged reliably as they could fall in between the 15 minute sample intervals.

IRON AND OZONE DOSED FILTER

Figures 6.42 and 6.43 show that there was a slight relationship between the initial spike, the ripened removal and the run length. This was in line with intuitive expectations - a more efficient filter would be expected to clog more rapidly.

Further evidence of a relationship between turbidity removal and run length came from the linear regression shown in figure 6.44, where $y = 0.0017 x + 0.047$, with a correlation coefficient $r^2 = 0.21$.

Figures 6.42 and 6.43 indicated that the length of the ripening period was weakly related to the run length, although the rapid ripening phase was limited to approximately 1 h in most cases.

Original in colour

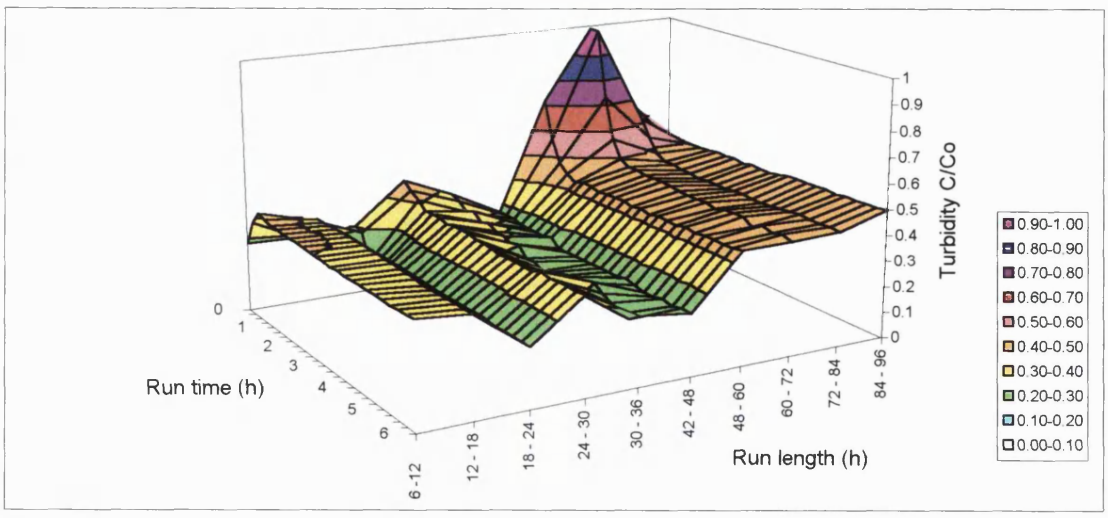

Figure 6.39. *Column 4 turbidity C/Co as a function of filter run length at 5-6 m.h^-1, July - December 1992.*

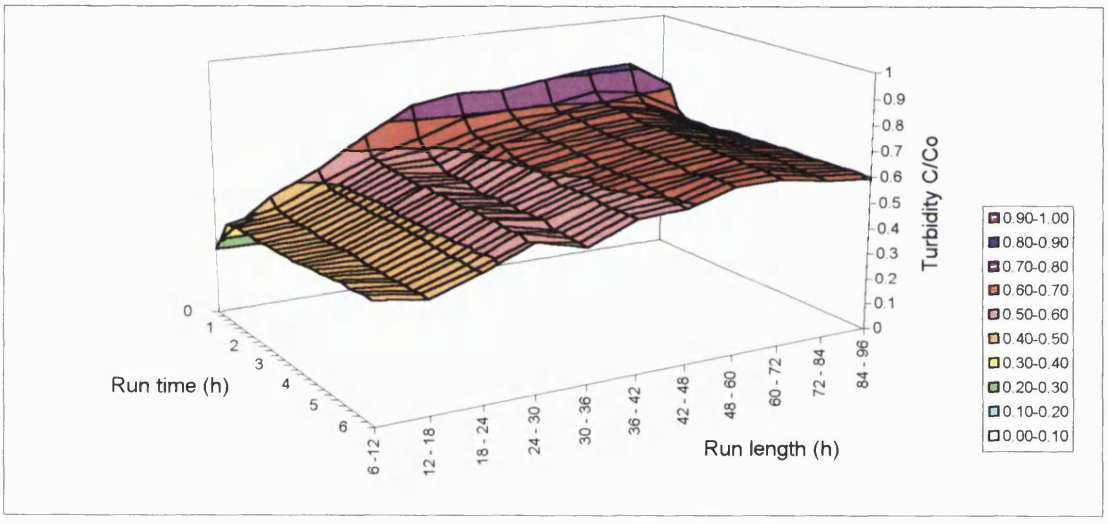

Figure 6.40. *Column 4 turbidity C/Co as a function of filter run length at 8-12 m.h^-1, January 1992 - February 1993.*

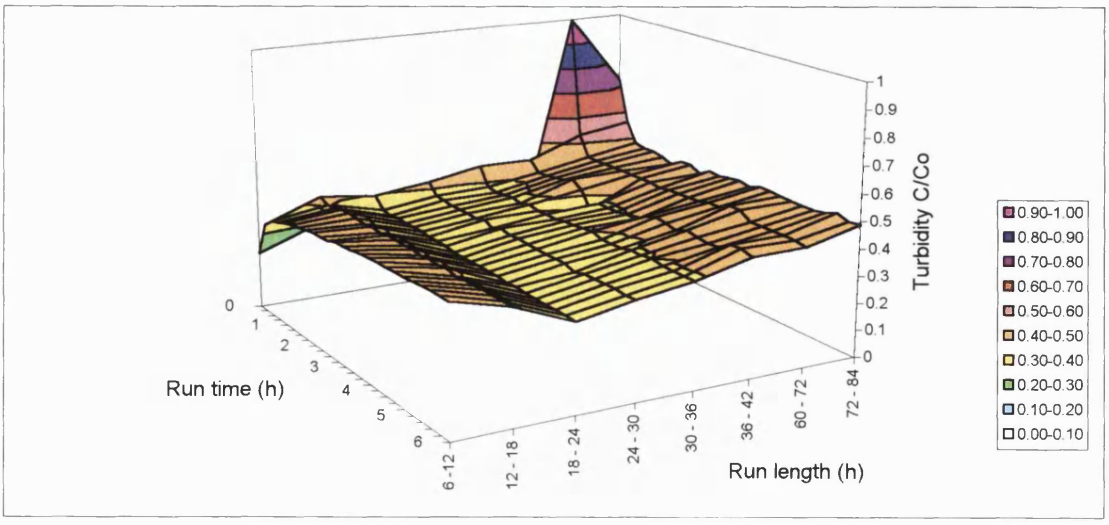

Figure 6.41. *Column 4 turbidity C/Co as a function of filter run length at 14-15 m.h^-1, August - September 1992.*

Original in colour

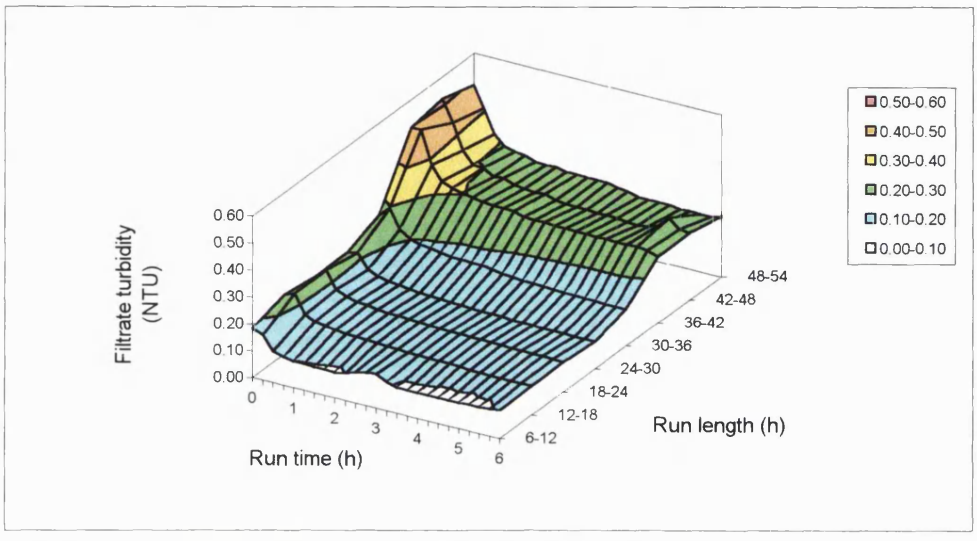

Figure 6.42. *Column 1: mean filtrate turbidity trends against run length class for iron and ozone dosed filter runs at 15 m.h^-1.*

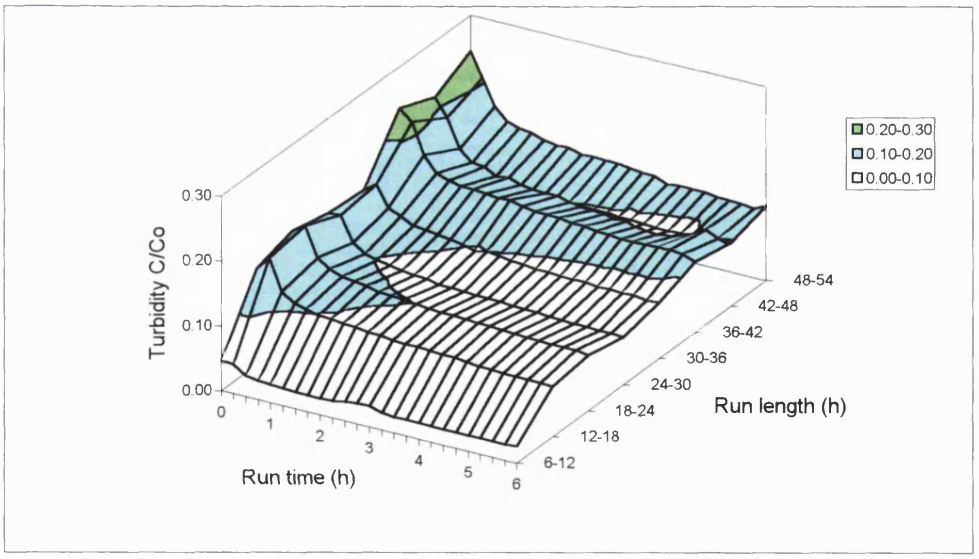

Figure 6.43. *Column 1: mean turbidity removal trends against run length class for iron and ozone dosed filter runs at 15 m.h^-1.*

The relationship between filter run length and filter efficiency was not entirely clear. Figure 6.45 shows that the iron dose was higher for the shortest runs and this might partly account for the shorter runs and more rapid ripening. Figure 6.46 shows that stored water turbidity could not be used to predict filter run length or efficiency. The particle index data in figure 6.47 suggest a weak relationship between the suspended particles in the water and filter run length, but that the generalised measures of chlorophyll *a*, POC and total particle count and volume did not provide sufficient information to understand filtration performance fully. The particle index did not relate well to the coagulant demand of the water, as figure 6.48 shows. The pattern of annual changes in stored water turbidity, particle index and the iron dose employed is illustrated in figure 6.49.

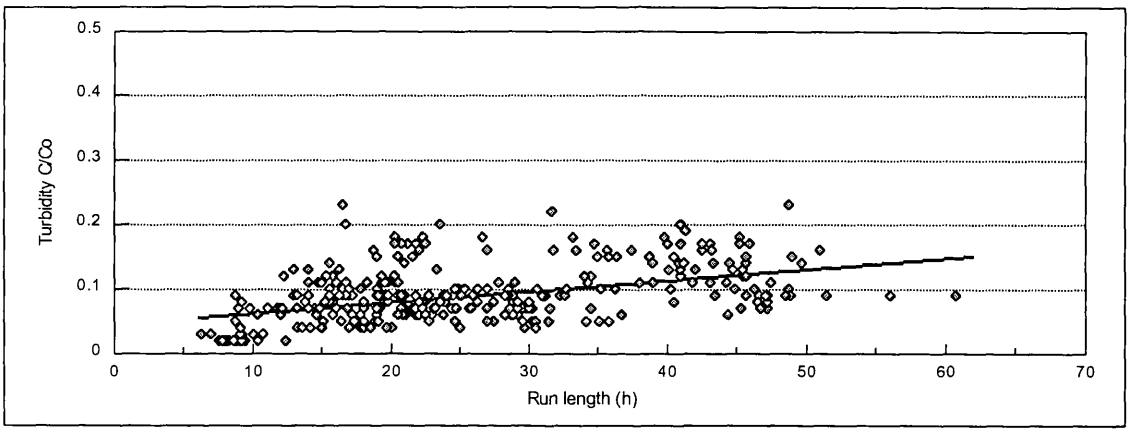

Figure 6.44. *Column 1: mean turbidity removal against run length for iron and ozone dosed runs, 1991-3.*

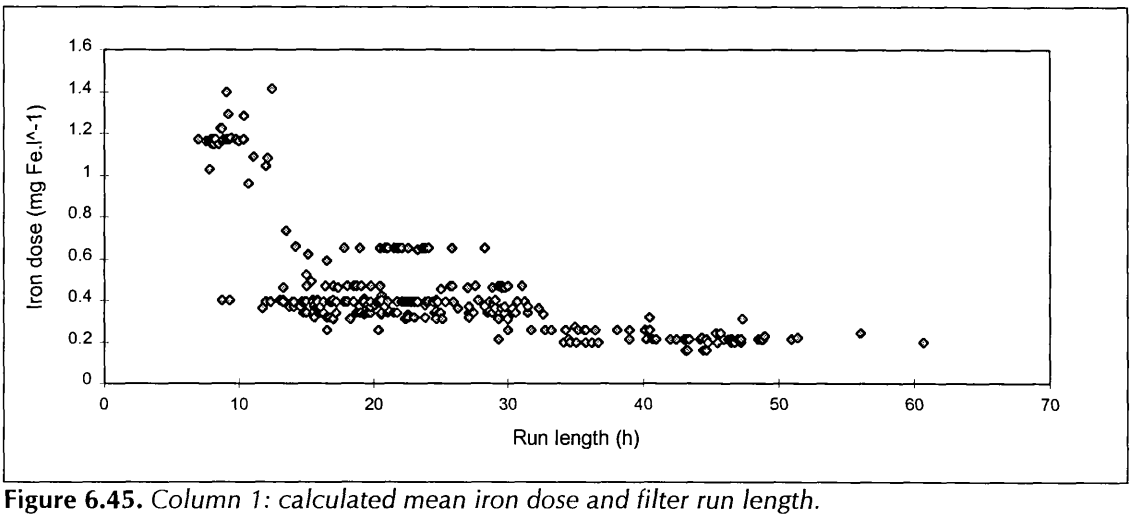

Figure 6.45. *Column 1: calculated mean iron dose and filter run length.*

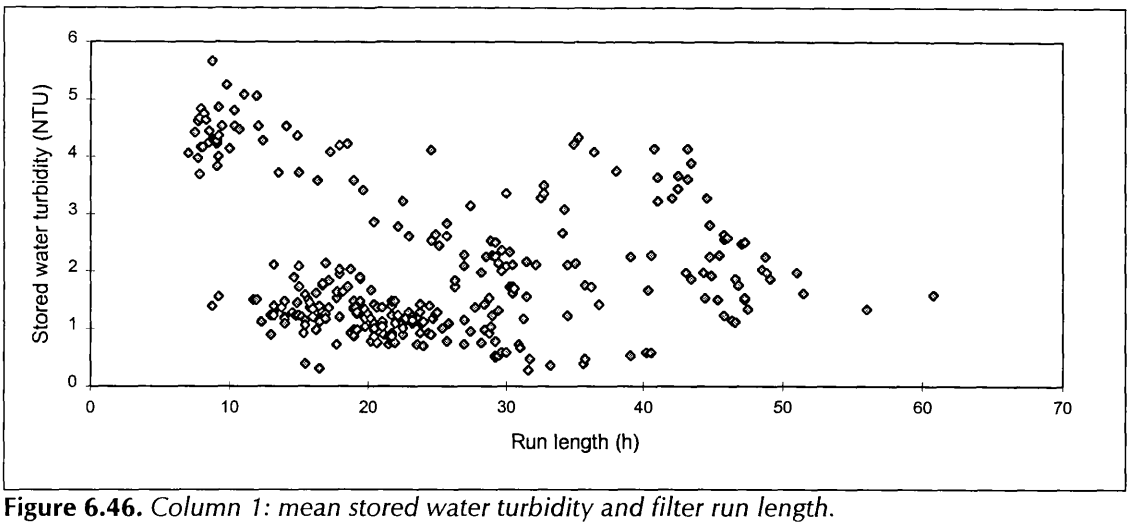

Figure 6.46. *Column 1: mean stored water turbidity and filter run length.*

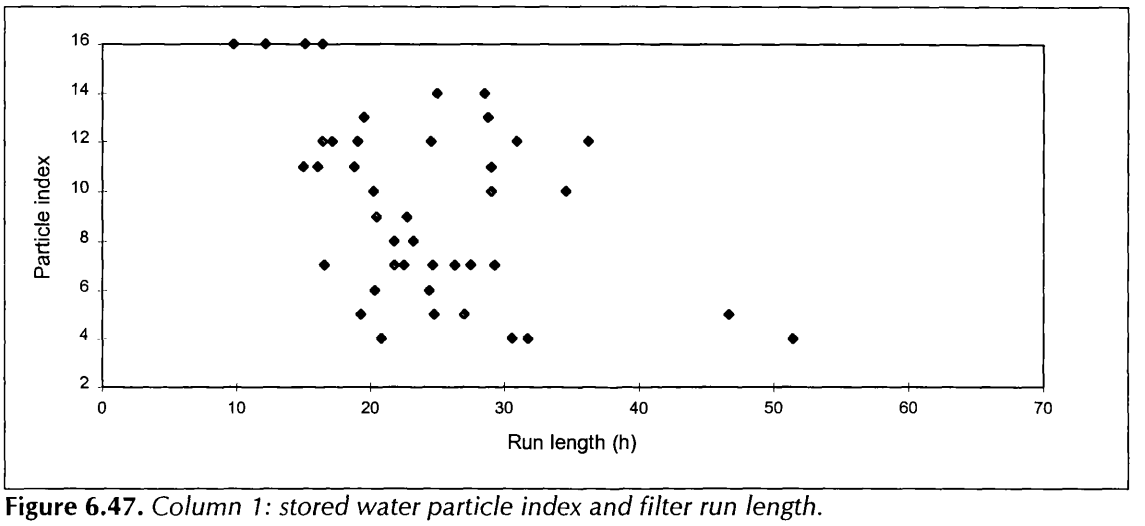

Figure 6.47. *Column 1: stored water particle index and filter run length.*

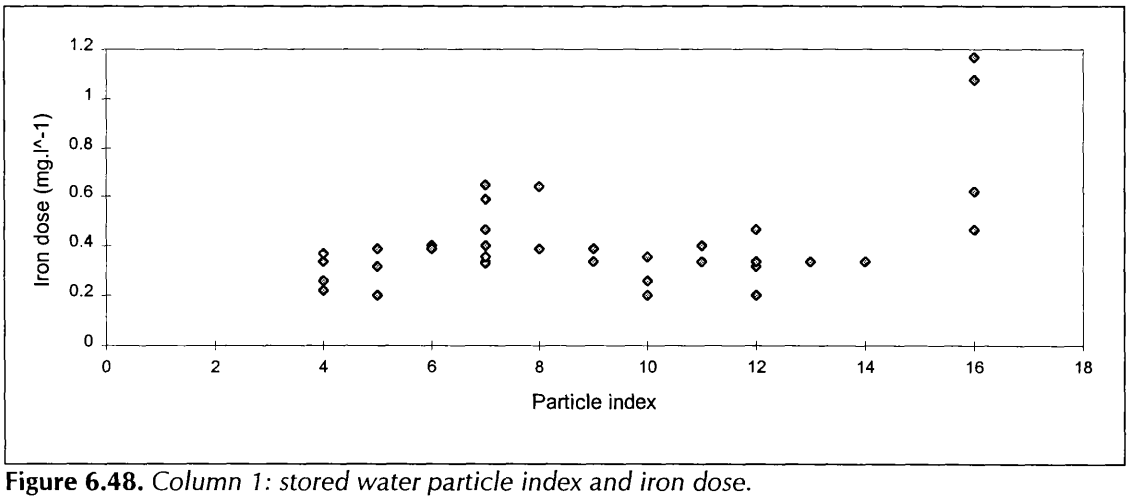

Figure 6.48. *Column 1: stored water particle index and iron dose.*

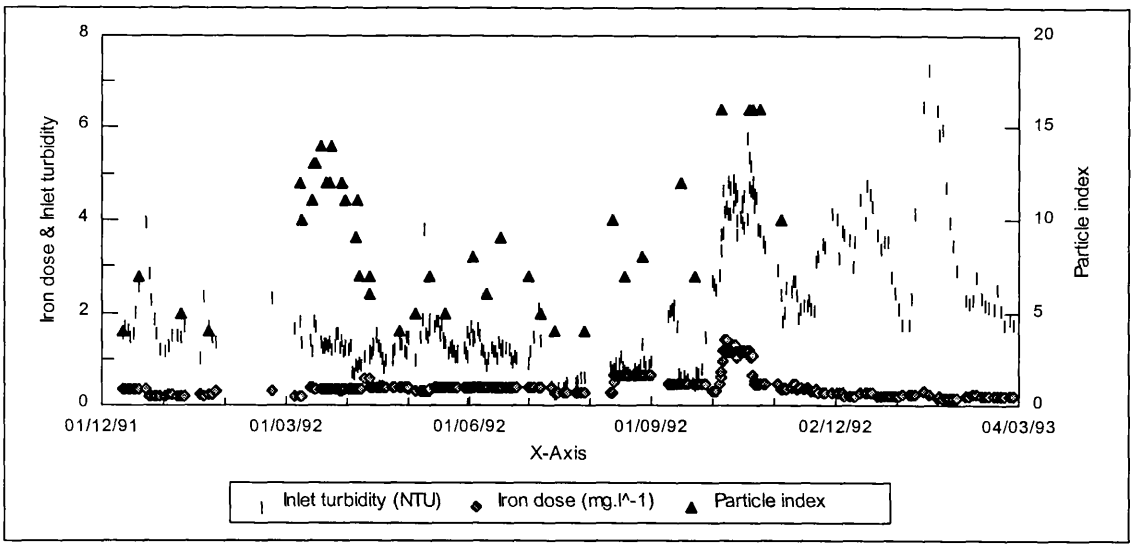

Figure 6.49. *Column 1: stored water turbidity, particle index and iron dose plotted against date.*

6.2.7 RIPENING AND FILTER MEDIA

The experimental design did not allow for parallel studies comparing different depths of the same media, operated under the same flow and backwash conditions and media age.

One finding of interest came from the comparison of four undosed filter runs the sand filter (column 4 at 15 $m.h^{-1}$ from 17-26.6.92), with a dual media filter operated at the same flow rate without chemicals over the same time period (column 5 run 634, 19-23.6.92). The column 4 data were presented in figures 6.7 and 6.8. Run lengths of 14, 11, 14 and 24 h were achieved. Run 634 on column 5 lasted nearly 120 h and was detailed in figure 5.19.

The removals of the four runs on column 4 are summarised in figure 6.50 as one mean removal line (black line). The data from column 5 are shown as the thicker, lighter line. No ripening was noticeable in the sand filter data, whereas a long slow ripening period was observed on the dual media filter. The difference in runs lengths was considerable, and the ripened removals were also markedly different.

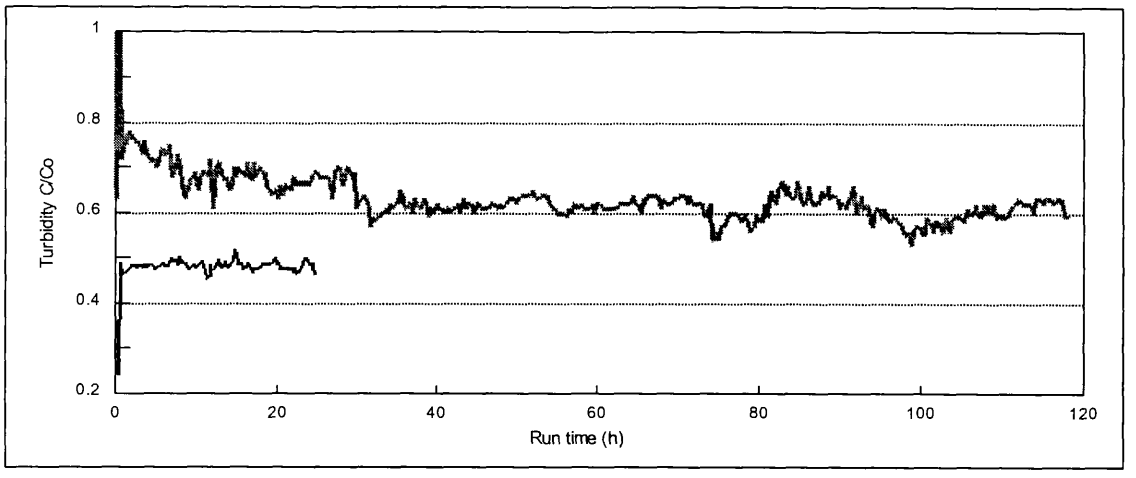

Figure 6.50. *Comparison of turbidity removals by undosed single media and dual media filters, June 17-24, 1992.*

6.3 DISCUSSION

6.3.1 INTRODUCTION

The physical factors measured during the trials were the flow rates, which were logged at 15 minute intervals, and daily temperature values. The physical loading to the filters comprised suspended material in the water, both organic and inorganic. This was measured indirectly by logged influent turbidity, and weekly measurements of chlorophyll *a* and POC concentrations, and particle sizes and numbers.

Chemical flocs may have contributed a physical load, but the contribution made to loading, i.e. particle volume, structure and surface area, from the coagulant addition could not be determined. It was noted that the rate of head loss growth was greater with chemical dosing. This was due to greater particle capture efficiency and, probably, to the formation of floc within the filter. Although no visible flocs were observed in jar tests with the correct contact filtration chemical dose, this did not necessarily mean microscopic flocs were not produced. It must be assumed that some iron(III) hydroxide floc species were formed during contact filtration since flocs were observed to settle in samples of dirty backwash water. However the microscopic analysis of floc size and shape is not easily performed without risk of distorting or breaking the floc.

This chapter examined whether physical variables influenced filtration. Three methods were used. Firstly, small groups of runs operated close together in time were examined. The second method

involved taking means from larger data sets. This assumed that variations in seasonal factors would tend to cancel each other out, rather than distorting the data. In the third method groups of filter runs were examined using coloured contour plots. This method examined the effect of loading on ripening over longer time periods.

6.3.2 THE EFFECT OF FLOW RATE ON RIPENING

Flow rates were changed periodically to gain information in the longer term on how satisfactory different flow rates were in terms of filtrate quality and filter run length.

The mean filter run data suggested that over the period of a year no consistent relationship between flow rate and turbidity removal could be determined for undosed or iron dosed filters. In figure 6.4, where flow rates were constant, removals were found to vary, even when runs without optimised iron dosing were discarded. When individual runs were examined in figure 6.5 and 6.6 the undosed filter showed that higher flow rates resulted in poorer filtrate turbidities and turbidity removal, as expected from many literature references. The lack of a ripening curve was noted on many undosed filter runs.

Whether flow rate affected turbidity removal was dependent on how the data for the undosed column 4 were examined. When the data were examined from a long period in time other factors tended to hide the influence of flow rate (figure 6.2). At the level of individual runs a change in flow rate caused a clear change in turbidity removal (figure 6.5), although the response of the filter to a very large change in flow rate was remarkably small.

Whilst the step change in flow rate was not representative of a slow start, the fact that conditioning the bed at a low rate made no difference to the removal at the higher rate, compared to the run started at high rate (figure 6.6), suggests that there was little to be gained from a slow start.

The examination of the effect of flow rate on individual filter runs of iron dosed filters was carried out shortly after the methodology for controlling coagulant dose had been established. Ripening

was observed as the runs generally produced L- and U-shaped trends. Whilst the U-shaped trends were not ideal, the data were viable in that the degree of breakthrough was not excessive. A slight U-shaped curve was indicated as typical of optimum filtration by Ives (1982) and 0.2 JTU was thought acceptable for SSFs by Janssens *et al.* (1982). Tables 6.5 and 6.7 showed a range of influent turbidities from 1.6 - 2.0 NTU was typical. Since the breakthrough seldom exceeded 0.4 NTU, good removals, in excess of 75%, were obtained, showing that the runs were not overdosed.

The data in figure 6.12 can be interpreted simply as showing that there was no effect of filter flow rate on the shape of the ripening curve. However, the change in the flow rate on column 5 caused the coagulant dose to require adjusting, and this was not carried out with immediate accuracy. It is a criticism of this work that the chemical dose changed for different runs. It was intended that this should result in each run being optimised, but it is not ideal to change two variables at the same time. In addition there was no control for what might be changing in the reservoir water. It appeared from tables 6.4 and 6.6 that algal populations in the water may have been in decline over this period. Whilst figure 6.48 showed no clear relationship was established between particle index and coagulant dose, it was one of the operational rules of thumb that higher algal populations required higher coagulant doses to produce optimum filtrate turbidity.

Considering the curves in figure 6.12 in the light of the earlier chemical dosing work, a higher iron dose should have produced a better ripening curve than the lower doses, yet this was not apparent. This might have been because the effect was countered by the lower flow rate, delaying ripening. There were two potential effects of flow rate on ripening: higher rates might benefit ripening by raising the solids loading per unit time, but it might disadvantage ripening by reducing capture efficiency and possibly changing the distribution of the deposits within the filter.

In the column 5 results it might be thought that a higher iron dose, and higher biological loading and higher capture efficiency might have accelerated ripening at low flow rates. A lower dose, lower capture efficiency, and fewer algae in the water might have delayed ripening at higher rates

by reducing the loading rate. These different behaviours might have cancelled each other out and produced similar ripening trends.

The column 6 trials however did not support this interpretation of the column 5 data. There were a smaller spread of iron doses, and in this trial the highest iron dose accompanied the highest flow rate. The three trends in figure 6.16 showed that the ripening timings and ripened turbidity were not sensitive to the flow rate, confirming the simple interpretation of the column 5 data.

This finding was unexpected as the literature suggested that flow rate was an important parameter in determining ripening rate, although there were inconsistencies in the reported findings. Cleasby and Baumann (1962a) showed that ripening was better at 10 $m.h^{-1}$ than with higher or lower flow rates. Watson (1990) reported higher flow rates accelerated ripening times, but Darby *et al.* (1991) and Moran *et al.* (1993) showed poorer removals at higher flow rates.

The significance of the findings in this study that ripening times and turbidity removals were not influenced by flow rate was that slow starts would not be a suitable strategy to control ripening as measured by turbidity.

It was not possible in this study, but a better methodology might have been to study two identical filter beds, acclimated at the same flow rates, then operated in parallel at different flow rates after a period to allow the biofilm to adjust to the new flow rates. It would be necessary to backwash the filters at the same time and ensure media grades were identical, so parallel filter studies would not be without their difficulties.

6.3.3 THE EFFECT OF WATER TEMPERATURE ON RIPENING

The influence of water temperature on filtration has received little attention in the literature. In the results from the undosed filter (figure 6.17) the impact of temperature on filtration appears to have been more significant than flow rate. Figure 4.9 indicates that the filter was biologically mature by the time these trials started. The reduction in removals around week 208 confirms that the

maturation process did not continue indefinitely and confirms the temperature effect. The data would have been more confusing if results from the maturation period had been included.

Figure 6.18 showed there was little impact of temperature on turbidity removal when iron and ozone were dosed. Since the temperature was not recorded automatically the data set was limited to the runs coinciding with daily readings. The fuller data set shown in figures 6.20 and 6.21 suggested distinct seasonal difference in filter behaviour between autumn / winter and spring / summer.

The studies were not able to state how temperature affected mean filtrate removals. With the undosed filter higher water temperatures would correlate with three factors: i) a combination of lower fluid shear forces acting on deposits, and reduced hydrodynamic retardation, due to lower water viscosity, ii) higher rates of biological activity in the filter biofilm, and iii) changes in particulate load arising from the different shapes and sizes of algae in the water, due to seasonal changes of algal taxa in the reservoir community, causing variations in the morphology of the media-deposit structure. The succession from unicellular forms in cold water through to filamentous and colonial forms in the warmer water was typical.

The lack of effect of temperature on turbidity removal with the iron and ozone dosed filter was expected. Coagulation reaction times are extremely fast. Amirtharajah and Tambo (1991) said that the formation of hydrolysis products with singe Al^{3+} ions occurs within microseconds, those containing multiple Al^{3+} ions form within 1 second. The formation of aluminium hydroxide precipitate $Al(OH)_{3\,(s)}$ is also a rapid reaction, taking from 1 to 7 seconds. Although these reactions might be slowed by lower temperatures, the time available in the water above the bed and within the filter means this was unlikely to affect filtration. Lower temperatures would increase mixing energy from fluid shear forces, although it is beyond the scope of this study to speculate how this might impact on flocculation effects within the filter.

Water temperature was not a variable under experimental control. In field studies it may not be possible to isolate the effect that temperature has on physical filtration processes, on chemical coagulation and flocculation processes, and on biological growth in the water and on the filter media to examine them separately. Laboratory conditions would be required to set up filtration studies where the effects related to temperature could be examined alone.

6.3.4 THE EFFECT OF LOADING ON RIPENING

The very nature of pilot plant trials set in the field meant that it was not possible to control individual physical variables in isolation from other environmental factors (physical, chemical or biological) which could have a bearing on filtration. Information gathered must be considered in the light of other variables that may be influencing the results.

It has been shown by Tate and Trussell (1978) and Lewis and Manz (1991) with turbidity and particle counts, Logsdon *et al.* (1985) with turbidity and *Giardia* cysts, and LeChevallier and Norton (1992) with turbidity, particle counts, *Cryptosporidium* oocysts and *Giardia* cysts, that relationships exist between turbidity and other measures of suspended particles in the water. LeChevallier and Norton (1992) demonstrated statistically significant correlations and, like Al-Ani *et al.* (1986) and Ongerth (1990), suggested that turbidity or particle counts could be used to predict *Cryptosporidium* oocyst and *Giardia* cyst removals. However they thought these relationships were site specific and could not be used reliably as a predictive tool at other works with different source waters. In general, these relationships do not stand up to mathematical analysis beyond the fact that turbidity trends bear some resemblance to the trends of other particles.

In order to examine the effect of loading on filtration under these conditions three general indicators of applied loading were used. The first was stored water turbidity, the second was a particle index based on particle counts and chlorophyll *a* and POC concentrations in the reservoir water. The third used the filter run length to indicate the loading.

In the light of the diversity of experimental findings on ripening, from laboratory studies reported in the literature, it was likely that there would be considerable variation in filter performance with varying inorganic and organic particulate loads from the reservoir water. These methods of data analysis were performed to determine whether turbidity would be a useful indicator to pick up these factors systematically and return mean data which could be used to manage the ripening process, or whether a simple particle index could produce a meaningful way of linking filter performance to the changing algal community in the reservoirs.

TURBIDITY

Results from the mean filter run turbidity removals from the undosed filter in figure 6.19 showed that, for the stored water turbidity values with a large number of corresponding filter runs a range of removals was found. Taking the data from the three flow rates as a whole it was not possible to determine any relationship between loading and removal other than a tendency for an overall removal performance in the region of 50-60%, as seen in figures 5.3, 5.6 and 5.7. Figure 5.6 demonstrated that, with seasonal and flow rate influences on turbidity removal ignored, the filtered water turbidity was largely determined by the stored water turbidity. The regression plotted was $y = 0.44x$, $r^2 = 0.92$, $n = 313$, suggesting turbidity removal was typically 56%. Figure 5.7 showed that there was a range of turbidity removals at any stored water turbidity, but that removal was not dependent on applied loading, although appearing more varied than in figure 5.6. A calculated linear regression (not plotted) was $y = 0.005x + 0.42$, $r^2 = 0.005$, $n = 313$, i.e. virtually a flat line, suggesting removal was independent of stored water turbidity. The regression indicated that removal averaged 58%. From the evidence in figure 6.17, the variations in turbidity removal at any stored water turbidity value were likely to be due to an effect linked to water temperature.

The results for mean filtrate turbidity and turbidity removal from iron and ozone dosed runs on column 1 were more complex than the undosed filter. There was an apparent relationship between filtrate turbidity and load in the autumn and winter (figure 6.20) that was much less obvious in the spring and summer (figure 6.21). The scarcity of runs with mean inlet turbidity above 5 NTU means the relationships should be treated with some caution. The apparent deterioration in turbidity

removals at low inlet turbidities in figure 6.21 could have had several causes. Firstly, the target effluent turbidity was 0.1 NTU. The operating strategy for coagulant dose was to minimise the use of coagulant, as long as L-shaped curves with a ripened turbidity of 0.1 NTU were achieved. That way coagulant and backwash costs would be minimised and plant throughput maximised. So whilst better removals could possibly have been achieved with very low inlet turbidities, the operating strategy did not allow this to be demonstrated. Two less likely possibilities were that very low loadings delayed ripening, as Gray (1991) had warned, although figures 6.24 and 6.25 showed no evidence of this, or that the linearity claimed by the turbidimeter manufacturer was not realised below 0.1 NTU.

It is recommended that care is taken with turbidity removal data for the iron and ozone dosed filter when the influent turbidity was <1 NTU. The choice of whether to use filtrate turbidity or removal trends may influence the impression gained of filter performance, depending on the loading.

The presentation of contour plots, produced by grouping filter runs into stored water turbidity, particle index, and run length classes, was an attempt to establish trends in filter performance from means of filter runs with one potentially load-related variable in common. In this study the stored water turbidity was not under experimental control, hence the variation in numbers of runs in each class, and the times of year from which they derived. Accompanying parameters such as algal population density and morphology, and particle volume and surface area in the water were also not under experimental control.

With the undosed filter, figures 6.22 and 6.23 confirmed the impression from mean filtrate samples that neither filtrate turbidity nor turbidity removal were influenced by inlet turbidity. They also showed that ripening was not consistently observed, and where it was, it was accomplished within an hour.

In contrast, figures 6.24 and 6.25 showed that ripening occurred regularly with iron and ozone dosed filters, and that the intensity of the peak, the duration of the ripening time and the ripened

removal all tended to give poorer filtrate quality with increasing mean inlet turbidity, up to 4 NTU. The reason this pattern was not found with higher inlet turbidities was that the runs above 4 NTU were from October 1992 when high algal loadings and the highest iron doses were associated with the high turbidity (figure 6.49).

If it is assumed that a clogged filter contains the same amount of material at the same head loss, regardless of influent turbidity, so that the backwash had to remove a similar amount of material, and that the backwash was of equal effectiveness throughout the year, the data in figure 6.24 indicated that the size of the initial backwash peak was dependent on the influent turbidity and not the backwash water remnants. This agreed with the findings of François and Van Haute (1985). However the reversal of the trend at the highest inlet turbidities suggests that the types of biological particles in the water and the higher coagulant dose both had a significant effect on ripening.

PARTICLE INDEX

The effect of algae in the water assisting the ripening of turbidity has not been reported in the literature although results with other particles have been presented. The range of spheres examined e.g. 0.09 - 7.6 μm (Veerapaneni and Wiesner, 1993), 0.27 - 10.1 μm (Tobiason *et al.*, 1993), <0.5 - 15 μm (Mackie and Bai, 1992, 1993) and pollen grains 12.8 - 42 μm (Vigneswaran and Aïm, 1985) and 22 - 85 μm (Vigneswaran *et al.*, 1990) do not necessarily correspond to the range of algal shapes and sizes, organic matter, and mineral particles, found in reservoir water. The Coulter counter measured particles between 4 and 80 μm equivalent sphere diameter in inlet water samples, and, on occasion, filaments and zooplankton visible to the naked eye were present. The particles in the reservoir tended to be present in larger numbers in the smaller size classes. The turbidimeters were most sensitive to sub-micrometre particles (Gregory, 1994).

Mackie and Bai (1992, 1993), Vigneswaran and Aïm (1985) and Vigneswaran *et al.* (1990) reported that, within a narrower range of particle diameters, larger particles assisted the ripening and removal of smaller particles. Their observations were supported by the change in turbidity trends and improved removals when inlet turbidity in excess of 4 NTU was coupled with high particle

index values in October 1992, as demonstrated in figures 6.24, 6.25 and 6.49. The opposite finding of larger particles hindering the ripening of smaller particles, obtained by Darby and Lawler (1990) and Tobiason *et al.* (1993), was not supported.

Figures 6.30, 6.34 and 6.35 showed that the particle index did not provide a tool for predicting ripening behaviour for either the undosed or iron and ozone dosed filtration conditions. The initial peaks, ripening times and ripened turbidities and removals showed no pattern in response to increasing particle index values.

RUN LENGTH

The filter run length was likely to show a relationship with turbidity removal since runs lengths would be expected to be shorter with higher capture efficiencies, for example if blockage of the upper layers of the filter was caused by large algal filaments, or if higher doses of coagulant were used. Conversely, if filter capture efficiencies were low, longer run lengths would result as the rate of head loss development would be slowed down. Results from both the undosed filter, figures 6.39 to 6.41, and the iron and ozone dosed filter runs (figures 6.42 and 6.43) indicated that there was a relationship between better turbidity removals and shorter run lengths. This was apparent for all three groups of flow rates in the column 4 data. The poorer removals seen on column 4 with the middle flow rate (8-12 $m.h^{-1}$) was due to the inclusion of more data from the winter period. The tendency for a ripening period with a higher peak and longer duration to appear in the longer filter runs was noticeable with both chemical dosing and no chemicals applied. With the iron and ozone dosed runs the weak correlation coefficient in figure 6.44 ($r^2 = 0.21$) nevertheless returned a t-test value of 8.0 which showed the relationship between mean turbidity removal and run length was highly statistically significant ($p < 0.001$).

The question of what caused the link between filter performance and run length could not be answered exactly. The trials indicated a link between iron dose and run length (figure 6.45), and hinted that some stored water conditions may have shown a link between turbidity and run length (figure 6.46) and particle index and run length (figure 6.47). More detailed work is required to

develop a model that allows ripening to be predicted from the particle loading and capture efficiency under plant conditions.

6.3.5 THE EFFECT OF MEDIA TYPE AND DEPTH ON RIPENING

Media types and depths were two other physical variables. Details of when these were changed are in the appendix. Unfortunately the discovery of the chemical dose control method came towards the end of 1991. The inspection of logged data trends, with hindsight, showed that much of the earlier work was carried out with excess coagulant dosing. The discovery of the importance of logged filtrate turbidity trends on the dosed filters led to the logging of stored water turbidity trends, starting 10.12.91, and undosed filter turbidity trends starting 2.1.92. This meant that removal data for the undosed filter were not available until January 1992.

The lack of good data, and inadequate chemical dose control, means the investigation of ripening curves had to be confined to the data gathered after late 1991. This was after all but two final changes in media had been accomplished. During the last year of operation there were very few opportunities to compare different filters operating as matched pairs. Therefore, for several reasons the operation of the plant did not permit the investigation of the effect of media size, depth or maturation on ripening.

There was one opportunity to compare the ripening behaviour of undosed sand and dual media filters. The results of this comparison, shown in figure 6.50, are intriguing. The dual media filter had more media (1.2 m total) but less sand (0.6 m) than the sand filter (0.9 m). The poorer turbidity removal by column 5 suggests that the anthracite was contributing little to turbidity removal. In addition the dual media filter showed a long ripening period, when there was no ripening in the sand filter.

Ripening was not typically seen with the sand only filter. Although the number of filter runs was limited, the data in figure 5.27 confirmed the presence of a short, classic ripening period in the undosed and ozone dosed runs on the dual media filter. This contrasting behaviour may be

explained by the effect of different media size on the structure and location of the deposits of different sized particles.

Observations of head loss development through the dual media filter (not presented here) showed that, for much of the year, little head loss developed on the anthracite, even with iron and ozone dosing. The bulk of the head loss development took place in the upper layers of the sand. Only under exceptional algal loads did the head loss development switch location to the anthracite layer. The poorer ripening behaviour of coarser media filters was reported by Clark *et al.* (1992), Goldgrabe *et al.* (1993) and Moran *et al.* (1993).

However, deposits were frequently observed on the anthracite surface. Based on Mackie and Bai's (1992, 1993) findings, it is therefore possible that the poorer turbidity removals, the presence of a ripening period, and the long run length on the dual media filter may have been due to larger particles being removed from the water in the anthracite layer. They were therefore no longer available to ripen the sand layer. The resulting poorer removal of finer particles in the dual media filter sand layer showed as lower turbidity removals and less rapid head loss development. With the sand-only filter the possibility that the shorter runs and better removals were due to surface straining was indicated by curved head loss trends (Ives, 1980), although the undosed dual media filter also showed curved head loss trends indicating straining on the sand layer.

CHAPTER 7. BIOLOGICAL INFLUENCES ON FILTER RIPENING

7.1 MORPHOLOGY OF BIOLOGICAL PARTICLES

Ridley (1967) warned against considering algae simply as particles with certain physico-chemical characteristics since some were motile. Some eukaryotic unicellular algae and the gametes of multicellular forms are motile, possessing one flagellum or more (Bell and Woodcock, 1971). Many Protozoa achieve movement by the rhythmic beating of flagella or cilia (Barrington, 1967). Other biological particles in water, for example the amoeboid Protozoa, change shape to effect locomotion (Barrington, 1967). These passive and active structures might affect transport and capture mechanisms in filtration. For instance the locomotion of muticellular organisms could displace deposits from filter grains, potentially reducing filter effectiveness, but their presence in rapid filters is rarely reported, and this mechanism is purely speculative.

Martin *et al.* (1992) said that bacteria differed from non-biological particles in ways which could interfere with transport and attachment mechanisms. These included their electrophoretic mobility, hydrophobicity, and the presence of capsules and appendages such as flagella, fimbriae and pili. Bacterial metabolic activity could alter the cell surface chemistry and that of their local environment.

The physical properties of algae may influence filtration. Clarke (1988) reported differences in filtration effectiveness of two genera of centric diatoms of similar shape and size, *Stephanodiscus* and *Cyclotella*. The reason given for poor removals of *Cyclotella* was that these cells possessed many fibrils up to 112 μm long radiating from the centre. These fibrils prevented the cells from being transported close enough to the media for attachment to take place. *Stephanodiscus*, which did not possess fibrils, was well removed. Ives (1955) cited Fuller (1904) as having said that *Anabaena* was not easily retained by a filter because it possessed a smooth surface texture.

The different lengths of rigid spines on two rotifer species was identified by Watson (1990, citing the work of Bernhardt, 1988) as causing different filtration efficiencies. Each species was over 300 μm long. The one with a spine approximately 360 μm long was more effectively removed than the one with a shorter spine, roughly 50 μm long. The overall size of the organism (including the spine), rather than the presence of spines, was critical to its removal.

7.2 BIOLOGICAL MODIFICATION TO FILTER MEDIA

In conditions other than ideal laboratory experiments the media grains may be coated with a biofilm, comprising bacterial cells and fungal hyphae. These will alter the media surface structure and may exude extra-cellular matter such as enzymes, influencing the microscopic environment and potentially affecting filtration. It is a mechanism of slow sand filtration that aerobic bacteria exude a sticky matrix of extra-cellular biopolymers (Ives, 1990) which helps trap particles.

Because of concerns that the presence of biological growths might impair filtrate quality, Goldgrabe *et al.* (1993) performed experiments on filters receiving pre-ozonated, alum-coagulated and settled river water. Filter 1 received water with an additional chlorine dose. Filter 2 was backwashed with chlorinated water. Filter three received no exposure to chlorine. No biomass developed on the pre-chlorinated filter, some growth was recorded on the filter 2, and seven times more growth was measured on filter 3 after 12 weeks compared with filter 2.

The effect of biological growth on filter ripening was examined after 8 weeks acclimation time. After backwashing an initial deterioration phase was reported, lasting up to 6 minutes. This corresponded to one bed volume, containing the remnant of the backwash process. During this pre-ripening phase the numbers of the smallest sized particles in the filtrate increased, accounting principally for the deterioration.

Comparing filter 1 with filter 2 there were greater numbers of particles in the filtrate from the more biologically active filter in the first 20 minutes of operation, although the mean particle size was

smaller from this filter. In both filters the duration of ripening for 20 - 50 μm particles was about 10 minutes. After 10 minutes the maximum particle size measured was in the range 20 - 25 μm. After 1 h the maximum particle size range was 10 - 15 μm and this was consistent for the remainder of the filter run.

Following the ripening period there was no difference in filtrate turbidities attributable to the biological condition of the filter. Measurements of particle counts showed the lowest values came from filter 1. It was not determined whether this was due to pre-chlorination improving the destabilisation of particles in filter 1, or the shedding of particles of biological origin from filters 2 and 3. Filter 2 produced poorer quality than filter 3. This was attributed to the chlorine in the backwash water disrupting the biofilm.

Huck (1994) reported that differences in turbidity removal between biological and non-biological filters were only significant in the ripening period. He said that turbidity peaks were significantly higher for the biological filter and it only showed a single peak, whereas a pre-chlorinated filter displayed twin peaks.

7.3 EFFECT OF BIOLOGY ON COAGULANT DEMAND

Rodman (1982) showed from electrophoretic mobility data that different algae had different surface charges and these varied with solution pH and even with the age of the algal culture. Ives (1955) reported that the zeta potentials of all algae were similar, despite large variations in organism sizes, but varied with pH. Ives (1955) did say that the charge density around an algal cell might vary and thus the coagulant demand would vary. However, in practice a monoculture of algae was rare, and the mix of genera outweighed any variations in individual species' coagulant demands, and hence the required dose was dependent on algal concentration but not species composition.

The use of ozone at doses up to 1.05 $mg.l^{-1}$ was reported by Ives (1955) to increase the size of the negative zeta potential of cultured *Chlorella* and *Aphanizomenon*, *Melosira* and *Stephanodiscus*

astraea from the Queen Mary reservoir. Petrusevski *et al.* (1994) reported an increase in coagulant demand after ozonation of an algal-laden water. Microscopic examination showed little damage to algal cells although a marked definition of the chloroplasts indicated *Stephanodiscus astraea* had been stressed (Ives, 1955).

Haarhoff and Cleasby (1989) examined the direct and contact filtration of *Chlorella* with cationic polymer. They said that algae behave similarly to colloidal particles, but extra-cellular organic matter (EOM) exerted a coagulant demand. EOM behaved as an anionic polymer bound to the particles at low concentrations (up to 1 mg $C.l^{-1}$), so that when $FeCl_3$ coagulant was added the EOM was capable of causing bridging flocculation. At higher EOM concentrations the EOM was present in solution and competed for the iron(III) ions interfering with filtration. Between 5 and 95% of photosynthesised carbon could be released as EOM - the higher values for stressed algae.

7.4 RESULTS

Figure 7.1 shows the results from column 1 for the stored water turbidity, particle index and the iron dose used to achieve predominantly L-shaped runs. The peak in particle index in the spring was not accompanied by a change in turbidity or coagulant dose. In October 1992 the peak in iron dose was linked to a peak in particles and turbidity. The turbidity peaks in December 1991 and January 1993 respectively did not require higher iron doses. The higher doses shown for December 1991 had actually produced V-shaped filtrate turbidity curves. The subjective impression was that the iron demand was related to the presence of higher numbers of algae in the water and not to higher stored water turbidities of inorganic origin.

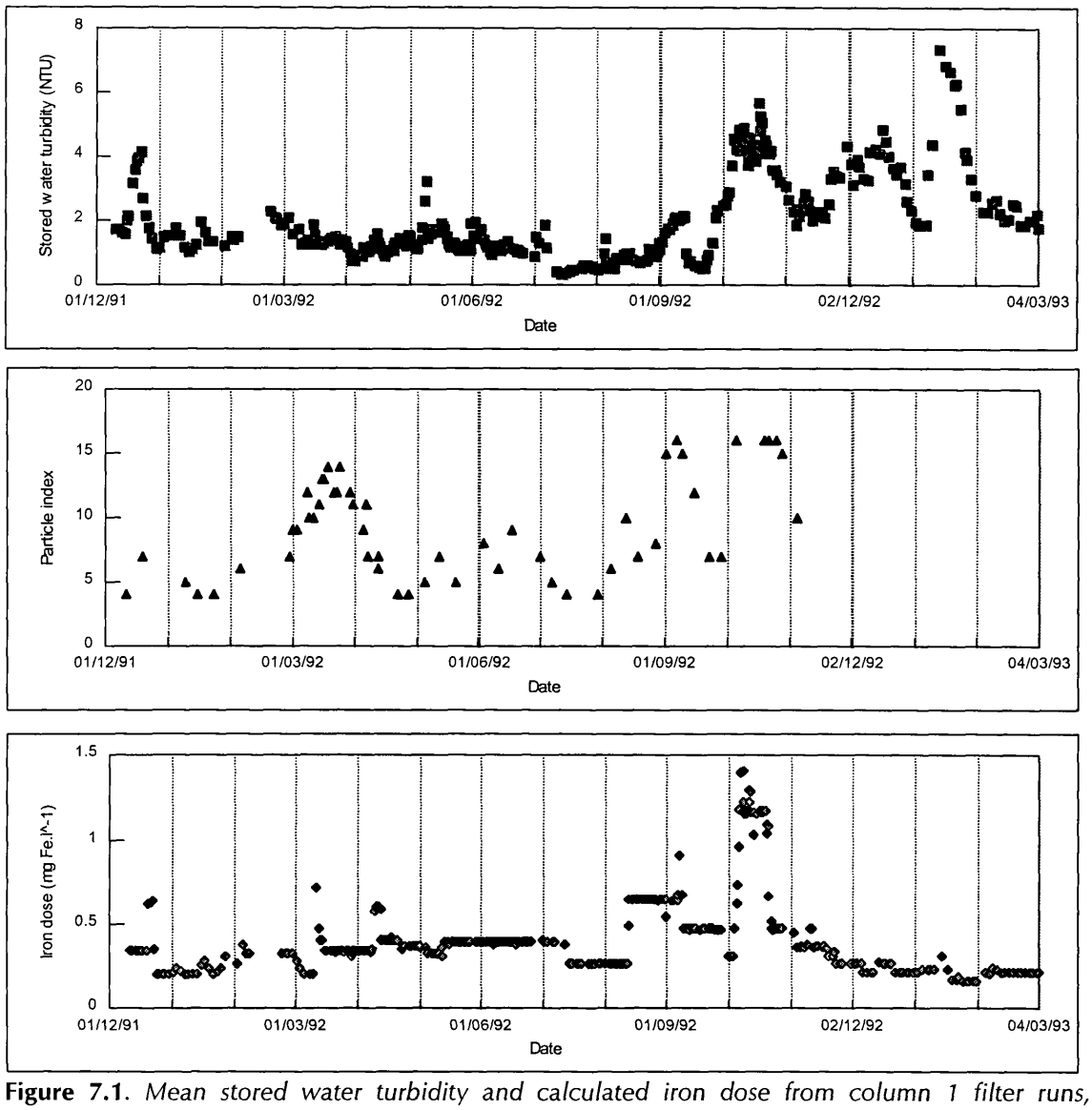

Figure 7.1. *Mean stored water turbidity and calculated iron dose from column 1 filter runs, together with particle index values, showing no clear relationship between particulate or turbidity loading and iron dose.*

In figure 7.2 the daily sample data from column 6 are plotted for the times when iron and ozone were dosed. The filter started with new media on 28.8.91. The difficulty in determining whether filter maturation took place can be seen. The turbidity removal data implied maturation was taking place, but this was due to the low removals achieved with low stored water turbidities. The filtrate turbidity data suggested no maturation effect was apparent.

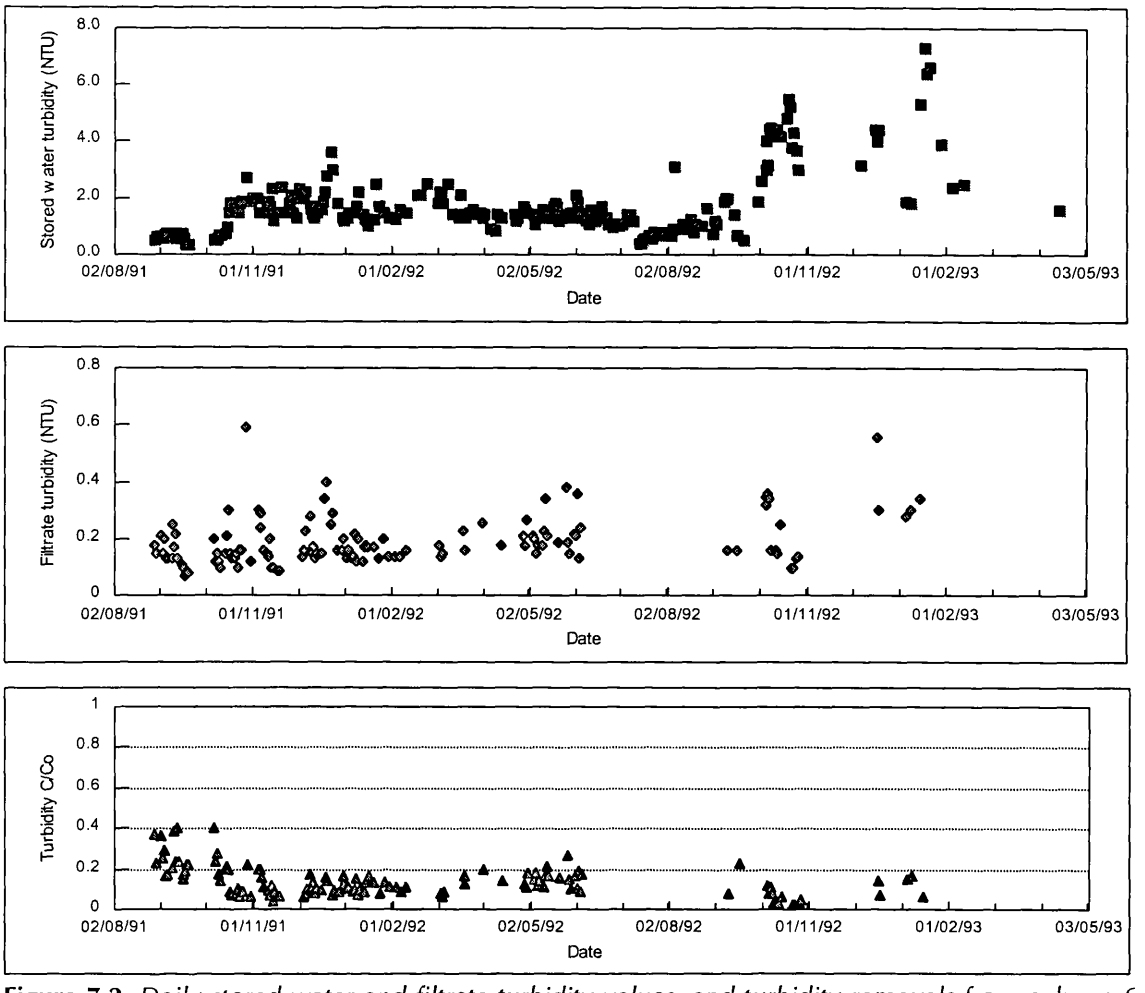

Figure 7.2. *Daily stored water and filtrate turbidity values, and turbidity removals from column 6, with iron and ozone, 1991-3.*

7.5 DISCUSSION

The particle index work discussed in chapter 6 did not reveal clear effects from algae and other microscopic particles on filter ripening. The generalisations used to create the particle index were too broad for it to be useful. Watson (1990) discussed in detail the difficulties with using POC, chlorophyll a, and particle size analysis to produce meaningful estimates of algal biomass. Further studies are required to determine how biological factors influence filter ripening.

The presence of algae in the stored water was recorded by Thames Water biologists from microscopic analysis of weekly reservoir samples taken from a boat. Identification to genus or species and a subjective abundance ranking was carried out. The dominant taxa were identified and classified as abundant, common, frequent, occasional, rare or none, described as the ACFORN

scale by Ta and Woodward (1997). These samples were used to indicate which reservoirs contained algae at concentrations which could cause treatment problems. This helped to ensure London's works' output was not compromised.

Unfortunately it was not possible to produce a systematic assessment of the effect of algae on ripening. It is questionable how representative a single weekly sample was of a whole reservoir for a whole week, given rates of algal population growth, and the influence of wind direction in relation to the position of the draw-off tower. Additionally the water at the pilot plant could be a blend of reservoirs. Resources were not available to determine an ACFORN score for pilot plant samples. The problem with this method was that it was highly subjective, and over the course of a year ACFORN scores showed no good relationship with chlorophyll a or POC concentrations. Whilst the reservoir water might be dominated by one type of organism, there were always others of different shape and size present in lower concentrations. Within any one taxon the shape and size of algae could be variable for those algae which formed fragile chains or clusters.

The improved turbidity removals achieved by mature undosed filters (see chapter 4) was likely to be a biological effect, from the evidence of POC on the media, and the biological structures reported by Kay (1991).

It is surprising that some authors have not reported improved removals with biological filters e.g. Goldgrabe *et al.* (1993), Huck (1994), or Huck *et al.* (1997), as cited by Urfer *et al.* (1997). The explanation for this would be if, like Goldgrabe *et al.* (1993), Huck had been filtering destabilised inorganic material like alum flocs, as it is possible that biological organisms discriminate between removing organic and inorganic particles from the water, with only the former being assimilated into new biomass.

The data in figure 7.2 support the view that maturation was not an influence on filtrate quality in chemically dosed filters. This view is not unequivocal due to the variability of the data from the daily samples. One reason for the variation in the data could be when the samples were taken in

relation to each filter cycle. A second reason was that if the target of 0.1 NTU was easier to achieve due to maturation the operational consequence would have been a reduction in coagulant dose.

The release of particles from biological filters during the ripening period reported by Goldgrabe *et al.* (1993) may indicate a need to optimise the backwash procedure rather than be a deficiency of biological filters. The data from Goldgrabe *et al.* (1993) did not suggest that biological maturation was beneficial to filtration when coagulants were used. They reported poorer filtration from the biologically active filters. There was no data on whether the biological coating had any impact on optimising coagulant doses. It is known that extracellular organic matter, produced by algae, exerts a coagulant demand (Haarhoff and Cleasby, 1989).

Suggestions that biological filters, with the presence of fungal hyphae (Kay, 1991) giving structures analogous to dendrites, were poorer filters than clean media, appears to conflict with the O'Melia and Ali (1978) view of ripening. However, it is possible that the surface charges on biofilm organisms, and possible release of EOM under the stress of backwashing, could have interfered with particle capture.

CHAPTER 8: DEFINING FILTER RIPENING POINTS

In order to examine the ripening of the pilot plant data some consideration was given to devising a systematic way of interpreting ripening curves. By defining quantitative ripening points the subjectivity usually associated with filter ripening could be eliminated. It was realised that some filter runs would show different ripening behaviour depending on which definition of ripening was used, and that ripening might occur but not be regarded as significant by certain ripening definitions. Figures 8.1, 8.2 and 8.3 illustrate how different ripening definitions appear on different ripening curves.

Figures 8.1, 8.2 and 8.3 have been constructed from the formula $y = e^{-mx} + 0.1$ with values of $m = 0.05$, $m = 0.1$, $m = 0.2$ and $m = 0.5$. Six ripening points have been considered, these are described in table 8.1, together with the markers used in figures 8.2 and 8.3 to show these achievement of these ripening points. For this example filtrate turbidity has been chosen as the filtrate quality parameter, but it could have been any other parameter that is known to show ripening. Two influent turbidities were considered for the removal performance, 2 NTU and 4 NTU.

Table 8.1. *Definitions of ripening and the markers used to illustrate these ripening points in figures 8.2 and 8.3. Ripening points judged at 2 decimal places are marked by capital letters. The suffix i for markers D, E and F represents the influent turbidity value, which is constant in these examples.*

Ripening definition	**Marker (2 decimal places)**
Turbidity = 0.1 NTU	A
Difference between adjacent tubidity values < 0.01 NTU	B
First occurrence of minimum turbidity value	C
First occurrence of 95% turbidity removal	Di
First occurrence of maximum turbidity removal	Ei
First occurrence of maximum turbidity removal * 0.95	Fi

To illustrate the final ripening definition, if the maximum turbidity removal achieved was 90%, this would consider ripening to have been achieved the first instance removal was >= 0.95*90 % removal, i.e. >= 85.5% removal.

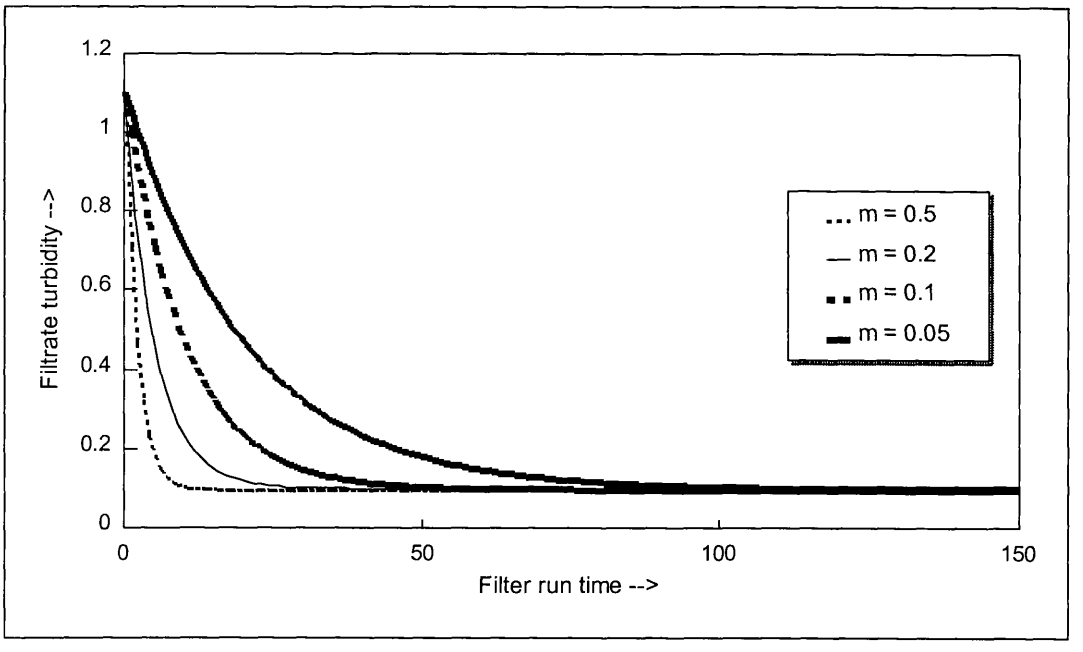

Figure 8.1. *Illustrative ripening curves* $y = e^{-mx} + 0.1$, *for 4 values of m.*

Figure 8.1 shows that in each example the filtrate turbidity starts at 1.1 NTU and levels out at 0.1 NTU. The different values of m result in different times taken to reach 0.1 NTU and consequently the differences between adjacent turbidity values vary. For clarity the x-axis has been plotted up to 150 time units. This slightly truncates the data for the curve where m = 0.05, since this curve does not reach 0.100000 NTU until 291 time units have elapsed. The other ripening parameters similarly take long periods of time if considered to 6 decimal places where m = 0.05 (i.e. E4 is reached at t = 263, E2 and D2 t = 277, A and C t = 291). In practical situations a turbidity meter might be considered to give worthwhile data to 2 decimal places. Consequently the points A and C are probably good enough indicators of ripening, and running a filter to waste for an almost infinitesimally small improvement in turbidity would be hard to justify.

It is clear from figure 8.1 that if the ripening point was $>=0.1$ NTU each curve would be considered to have ripened sooner or later. If the ripening point was defined as <0.1 NTU none of the filter runs would actually have ripened, despite the obvious improvement over time. Similarly any target filtrate value below the value of c for a curve $y = e^{-mx} + c$ would never be reached despite the clear ripening behaviour simulated by the curves.

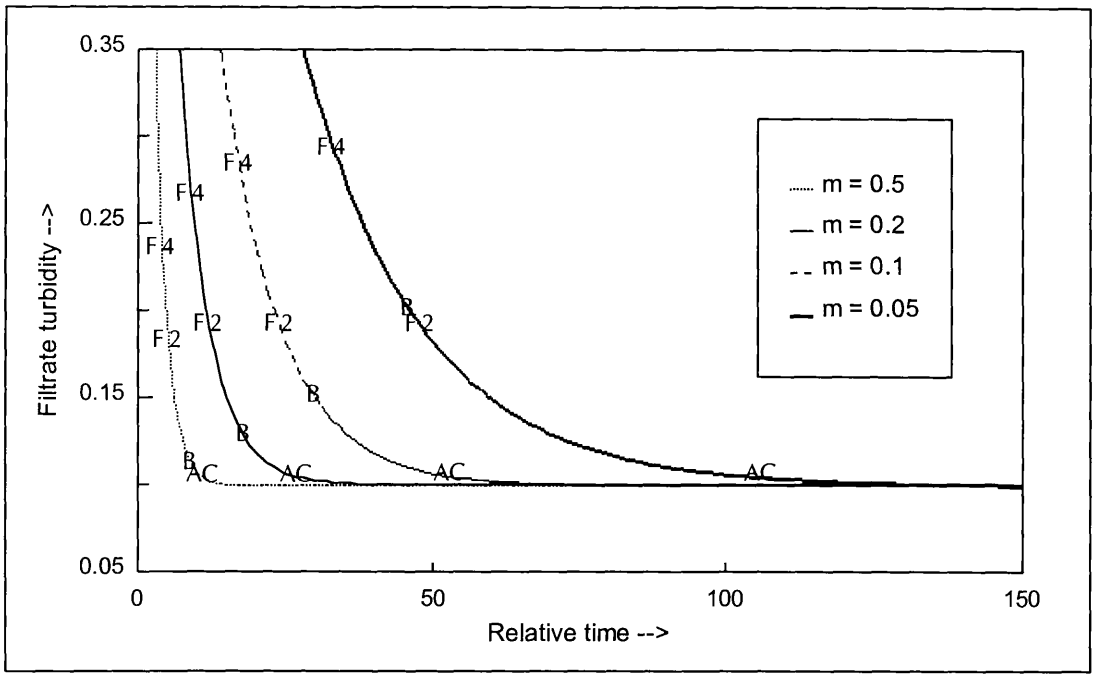

Figure 8.2. *Illustration of 4 different classes of ripening point, showing how they vary according to the slope of the ripening curve and the raw water turbidity value. The markers are described in table 8.1.*

Figures 8.2 and 8.3 illustrate the time taken to reach the different ripening points described in table 8.1, and the filtrate quality at these points, for the same four curves shown in figure 8.1. The axes have been adjusted to see the ripening points more clearly. It is clear that in the worked example y-values of 0.1 NTU represent both the minimum attained turbidity and the 0.1 NTU target, hence the values of A and C coincide. This would not necessarily be true of other curves.

It is noticeable how the gradient based ripening indicator (B) changes position with respect to the other markers and to filtrate turbidity depending on the value of m. Where ripening is rapid the values of turbidity and time for B are near enough to those of A and C to suggest that they are not particularly different. This is in marked contrast to the ripening curve which shows slow improvement (m = 0.05).

The removal based indicators (F in figure 8.2 and D and E in figure 8.3) show how sensitive these are to the inlet water quality. Even with removals calculated for inlet turbidities of 2 and 4 NTU the positions of the markers changes noticeably. It is purely an artefact of the selected equation that where raw water turbidity was 2 NTU 95% removal coincided with 0.1 NTU filtrate, and was the

maximum removal achieved (E2). If the constant (c) had been a larger number 95% removal (marked by D2) would not have been achieved with a raw water turbidity of 2 NTU. When the removals were based on a raw water turbidity of 4 NTU the 95% removals (D4) were not the same as the maximum removals (E4).

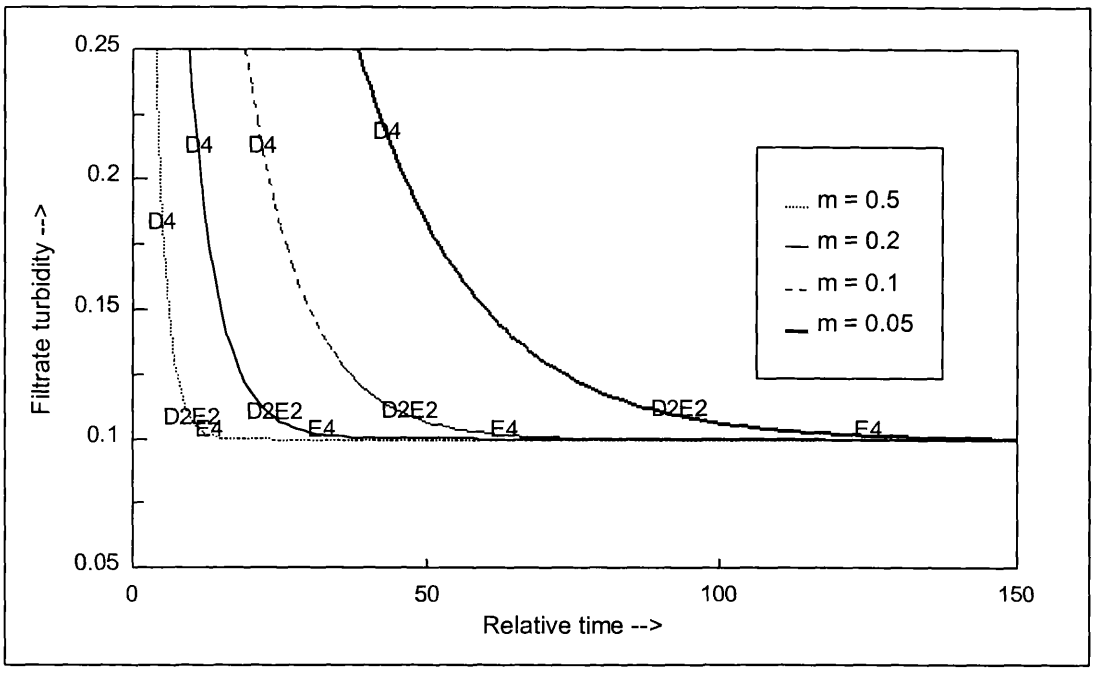

Figure 8.3. *Illustration of 2 different classes of ripening point, showing how they vary according to the slope of the ripening curve and the raw water turbidity value. The markers are described in table 8.1.*

In conclusion, no one method of defining ripening appears to provide a tool to sort through a database and objectively pick out the filter ripening point, and hence ripening time and ripened water quality. Filtrate turbidity may not reach a target. Turbidity removals will vary if filtrate quality is unaffected by influent water quality. The rate of ripening, together with sample frequency will determine the turbidity gradient change, such that a slowly ripening filter will meet the gradient criterion earlier and probably with a poorer quality water than where ripening is actually more rapid. In the pilot plant unusual turbidity removals were caused by changes in source stirring up sediments in the stored water tunnels; checking, cleaning and calibration of turbidimeters, algal filaments or zooplankton crossing the turbidimeter light path.

The resolution of the data logger could also produce a peak removal or minimum turbidity at a time other than the point of ripening. Assuming the ripened removal was around 0.1 NTU. The turbidimeter displayed a value to 3 decimal places and sent out a variable current between 4 - 20 mA. This was converted to 0 - 4095 bits, and the computer converted this to turbidity with 2 decimal places. Whilst this number of decimal places is not highly meaningful, a change in turbidity from 0.096 to 0.094 NTU would mathematically, at 2 decimal places, be a step change in quality. This might be the filter run minimum. Human assessment would attach little significance to this change. An experienced filtration scientist would examine the entire filter run and pay little attention to small fluctuations within the overall trend. The use of rolling averages would reduce but not eliminate this change.

This method of discerning ripening was abandoned. It is unsurprising that nobody has published an objective means of determining the completion of ripening.

CHAPTER 9. SUMMARY

9.1 LITERATURE REVIEW

The literature on ripening in rapid gravity filtration of potable water was reviewed in Chapter 2. Filtration comprised transport, attachment and detachment processes, and the mechanisms involved were summarised. The net effect of the capture of particles was a filter cycle made up of seven phases. The ripening phase took place at the start of the cycle. It was poorly defined, but consisted of a period of improving in filtrate quality. A few authors had attempted to define acceptable ripened filtrate turbidities, giving values ranging from 0.1 to 0.5 NTU. A few experimental data appeared to show a link between ripening and a change in head loss gradient. This would have made a useful indicator, but it was only rarely observed.

The need for water filtration to remove particles of concern to public health was discussed. Recent outbreaks of cryptosporidiosis had shed doubt on the efficacy of conventional water treatment practices. Some outbreaks were linked to filtration plant problems, but even well operated plants were vulnerable to the passage of large numbers of particles during the ripening phase.

It was clear that ripening had been known about for many years but had been ignored by many authors. In the area of phenomenological modelling, Ives (1975a) showed that there was no general agreement on ripening behaviour, or if it occurred at all. Many of the trajectory modelling studies summarised by Tien (1989) concentrated on the behaviour of clean beds, and ripening only came to be considered in studies following O'Melia and Ali's (1978) lead.

Data on filter ripening came firstly from idealised laboratory investigations, often conducted to help mathematical model verification, and secondly, from observations of pilot or full scale filters, usually carried out to provide site-specific data. As ripening was reported from operational water treatment plants it was not an artefact of laboratory studies.

Following the work of O'Melia and Ali (1978) it has been widely accepted that the presence of retained deposits improved the performance of the filter over the initial clean bed removal. The magnitude of the initial ripening peak was shown to be related to the influent turbidity (Janssens *et al.*, 1982, François and Van Haute, 1985), but backwash remnants also contributed to the ripening process (Amirtharajah and Wetstein, 1980, Cranston and Amirtharajah, 1987).

From the practical view point, when ripening was looked for, it was found with a wide range of water quality parameters, although chemical conditions could be set up so that ripening was not observed (Tobiason and O'Melia, 1988). The significance of ripening has, historically, been underestimated, but could no longer be treated lightly given the recent history of outbreaks of giardiasis and cryptosporidiosis. The literature suggested that turbidity was a useful general indicator of filter performance, although there was no general agreement that it could be used as a surrogate for *Giardia* cysts or *Cryptosporidium* oocysts.

During the ripening phase filtrate is likely to fail water quality targets. In many practical situations, blending with other filters has smoothed out the effect of individual filter ripening on water works' final water. This might have been acceptable with chlorine disinfection to deal with bacteria or viruses penetrating filters. It would not be satisfactory if *Giardia* or *Cryptosporidium* were in the water, because of their low infective dose, and the inability of chemical disinfection to inactivate these cysts and oocysts at the doses and contact times used in water treatment (Ives, 1990).

The presence of a ripening phase has not necessarily influenced how filters are operated. Ways of operating filters after backwash have included returning to operation at full rate (ignoring ripening), or, in recognising ripening as an issue, filters may be run to waste for a period, or be returned to service at a progressively increasing rate (slow start). This latter approach inevitably involves flow rate step changes of some magnitude and frequency and thus potential risk of breakthrough. Particle counting studies have shown that slow starts were not fully effective. Chemical treatment methods to control ripening have also been proposed. None of these approaches have eliminated the ripening period.

Wherever possible references were taken from studies of "direct in-line" or "contact" filtration, since this mirrored the experimental investigations in this thesis. It was also foreseeable that a single barrier treatment method (contact/direct filtration) might be more vulnerable to particle penetration, including pathogenic protozoan cysts and oocysts, than the double barrier of sedimentation or flotation plus filtration, although Nieminski and Ongerth (1995) reported that ripened filters in conventional treatment and direct filtration achieved removals of *Giardia* cysts and *Cryptosporidium* oocysts that showed no significant difference for either treatment process. This was, however, yet another study which did not examine the ripening period.

9.2 PILOT PLANT DESIGN AND OPERATION

Chapter 3 described the design and operation of the pilot plant. The pilot plant was designed to examine ripening under realistic conditions of chemical dosing, filter bed depth, maximum available head loss, flow rate and method of backwashing. The pilot plant was operated with six fixed flow rate, variable head loss filters, with either water level or outlet valve position changing to accommodate the effect of filter clogging on flow rate. Unfortunately, the pilot plant could not perform flow rate changes within a run or slow starts.

The filter columns were operated as single or dual media filters. Chemical pre-treatment with pre-ozonation and iron(III) sulphate was applied to dual media filters. The calibration of the chemical dosing equipment was detailed in chapter 3.

Stored water quality data were presented. The water was low in turbidity but prone to algal blooms. It was a hard water, with little colour.

Automatic monitoring of filter ripening was carried out using on-line turbidimeters. The turbidimeters were a relatively low cost means of obtaining continuous filter performance data. The instruments worked reliably in the field, subject to routine cleaning. It was found that absolute

turbidity values need to be treated with caution due to differences between results when measuring turbidity on-line and in the laboratory, and small signal loss errors. A long delay in getting signal wiring installed to log the stored water turbidity and the filtrate from columns 3 and 4 limited the available data.

Logging at 15 minute intervals provided a reliable general picture of whether filter ripening took place since, in many instances, the ripening period took longer than 15 minutes. This time interval was suitable for the type of turbidimeter used. On-line particle counters and other makes of turbidimeter have been used by others to give a more detailed picture of ripening.

Much of the time in the study was taken up with achieving sustainable plant operating conditions in order to limit the number of variables acting on the filter performance over time. It was necessary to determine the correct backwash strategy to maintain filter media in a good condition in a biological environment. This work is reported in chapter 4. The results in chapter 5 show how it was also necessary to devise a procedure to optimise the dosing of chemical coagulants. Iron(III) sulphate was chosen as the coagulant to avoid the use of aluminium based coagulants which were the subject of scares concerning Alzheimer's disease. It was unfortunate that the development of the chemical control procedure was not achieved earlier in the study as it meant that many experiments with different filter media had to be discarded due to coagulant overdosing.

DATA PROCESSING

Several hundred filter runs were examined to evaluate factors influencing ripening. Many filter runs were deleted due to the failure of the data logger, the backwash cycle or an instrument, or if a filter run had been terminated early to modify the filter plant, or if the filter run lasted less than 3 hours - indicating manual intervention or equipment failure. Care was required with runs where coagulant dosing or pre-ozonation was started or stopped during the run.

Both iron and ozone dosing plants were subject to failures when unattended. Since neither the operation of the iron dosing plant, nor the ozone generator were logged by the computer, and there

was no on-line logging of iron or ozone concentrations in the water, it was necessary to compile over a thousand changes in pump settings and generator settings from daily readings and site log books into the logged database. The pump and generator calibrations were incorporated into the database calculations to give iron and ozone dose estimates.

The modified database was used to examine ripening. Mean filter run data and spot sample data were used to provide general indications the effect of different chemical treatments. It was found that the best turbidity removals were achieved with iron and ozone dosed together. The next best removals were achieved with iron, then ozone, and finally the poorest removals were from undosed filter runs. Spot samples also showed how undosed filters matured biologically over time to achieve better turbidity removals.

All the filter runs were plotted in rough form and examined. The full set may be seen by arrangement with the author. Examples of characteristic filter runs with and without coagulant dosing and pre-ozonation were described in chapter 5.

Summary data from the first six hours of operating time were presented in chapter 6 to examine the effect of physical variables on filter ripening. The examination of filter runs when iron was dosed was limited to runs with optimised dosing of coagulant.

METHODOLOGICAL CRITICISM

An enormous number of filter runs were produced. Each run was subject to many variables, some of which were under experimental control and others not. An assessment of filter ripening needs to consider the influence of many factors on filter performance.

The variables under experimental control were: filter media types, size grades and depths utilised; flow rate; terminal head loss; backwash protocol and timings; ozone dose; iron dose; and discharge of influent to waste when ozone or iron could build up in the pipework. Unfortunately the design of the pilot plant control software did not allow investigation into start up strategies. The backwash

was kept as constant as possible to eliminate a further variable, although there could be merit in investigating the effect of varying this on ripening.

The variables not under experimental control were: the community succession in the stored water i.e. different algal and zooplankton populations; reservoir source; poor quality water in "dead legs" when reservoir was changed; water temperature; seasonal or diurnal pH changes induced by algal photosynthesis; failure of dosing plant - meaning some filter runs may well not be receiving the chemicals they should; failure of the data logger; occasional turbidity spikes in the stored water.

Other factors which had to be taken into account in examining ripening behaviour were cleaning and calibration of turbidimeters; the logging of turbidity data to two decimal places causing apparent steps changes at low values; sensitivity differences between laboratory and on-line turbidimeters; and turbidimeter signal losses to the computer.

9.3 BACKWASHING

Chapter 4 showed the backwash method was critical to the long term management of filter media. The development of a biological layer on the filter media resulted in improved removals achieved by roughing filters, i.e. without chemical coagulants, termed maturation. This biofilm could not be kept under control with conventional separate air and water backwashing, resulting in very short filter runs. The use of combined air and water backwashing was essential for the management of both sand and dual media filters. The collapse-pulsing backwash described by Amirtharajah *et al.* (1991a) and Fitzpatrick (1993) was used to provide a practical solution to managing filter media over one or more years, confirming observations based largely on laboratory trials and theoretical considerations.

The backwash performance was assessed by sampling the filter sand for attached solids and POC, and by examining the logged data. The combined air and water backwash enabled almost constant flow and temperature normalised filter starting head loss to be maintained for up to three years (figures 4.2 - 4.8).

The evidence from turbidity removal data for maturation on the undosed filters was shown in figures 4.9 and 4.10. There was no evidence that maturation aided the filter performance achieved by iron and ozone dosed filters (figure 7.2). A successful backwash strategy maintained low starting head losses but did not inhibit maturation. The elimination of controlled maturation was neither desirable, nor feasible with an economic volume of backwash water.

The volumes of wash water used varied according to filter run length and flow rate. In some calculations they were well below the 5% of production said to be excessive by Ives (1981a). With 8 h runs at 15 $m.h^{-1}$ volumes in the region of 6-8% were used. This was said by Janssens *et al.* (1990) to be satisfactory for direct filtration. The use of conventional volumes of backwash water would be inadequate in some circumstances.

The first year's trials, with poor backwashing, did not include on-line turbidity logging, so there was no data on the impact of poor backwashing on the shape of the ripening curve, beyond the maturation effects noted for ripened filter performance.

9.4 CHEMICAL TREATMENT

Chapter 5 examined the influence of chemical treatment on filter ripening. The literature showed there were several mechanisms by which chemical coagulation and ozonation assisted particle removal. Different coagulants gave different ripening behaviour. Low turbidity waters could be treated by contact or direct filtration. According to the definitions of Janssens and Buekens (1993), the stored water in this study was suitable for direct filtration, as it was low in colour, turbidity was < 25 NTU, and particle volume and chlorophyll *a* concentrations were generally < 10 ppm and 10 $\mu g.l^{-1}$, respectively. Although the trials actually used contact filtration, Janssens and Buekens (1993) said that the distinction between contact and direct filtration became blurred at plant scale. The hardness:TOC ratio was over 20, so ozone was expected to assist particle removal.

The literature demonstrated the need for coagulation and ozonation in order to achieve high quality filtrates, and this was confirmed by the experimental results. A number of methods for optimising the coagulant dose were described. Individual filter runs, presented in chapter 5, showed the sensitivity of the filtrate turbidity to turning the iron or ozone on and off (figure 5.17). The turbidimeters responded predictably to changing chemical treatments allowing the development of a chemical control model and dosing strategy. This was vital to the success of the project, as other chemical control methods reported in the literature, and tested at the pilot plant, did not prove satisfactory.

Until the iron dosing method was fully developed there was the tendency to adjust the dose repeatedly on some days when filtrate turbidities were not as expected - with hindsight the effect this had was to move between under dosing and overdosing.

The filtrate data in figures 5.1 and 5.4 showed that the best turbidity removals were achieved with iron and ozone dosing. Values below 0.1 NTU were recorded. Slightly poorer turbidities were achieved if the ozone was not applied. When iron was dosed the filtrate turbidity showed a low correlation with the applied turbidity. Without iron poorer removals were achieved, and correlation data showed they tended to be a fixed proportion of the inlet turbidity (figure 5.6). Pre-ozonated water produced better removals than water with no chemical treatment. These observations were confirmed with spot samples, mean filter run data and using complete filter run trends. The tendency for the filtrate to follow the stored water turbidity values in undosed filters (figures 5.16, 5.19) and ozone dosed filters (figure 5.18) meant that ripening could only be determined from turbidity removal data (figures 5.12, 5.20). Ripening was not always be seen with undosed sand filters (figures 5.10 to 5.15), though it was observed with an undosed dual media filter (figure 5.27). With optimum coagulant dosing it was found that ozone improved filtrate turbidity during and after ripening, but did not influence the length of the ripening period (figure 5.26).

The data from chemically dosed dual media filters (figures 5.28 - 5.38) showed the definite presence of a ripening phase. The entire filter cycle, including the ripening phase, was very

sensitive to small changes in the coagulant dose. Higher iron doses reduced the initial peak, and the ripening time and gave better ripened filtrates, though this was at the cost of earlier breakthrough. Four generalised filtrate turbidity trends were observed. They were described in figure 5.41 as S, L, U, and V-shaped. This was termed the SLUV model. From this model it was suggested that the optimum coagulant dose should produce an L-shaped trend, although this meant longer ripening times. The relationship between chemical dose and the head loss at which breakthrough commenced meant that the choice of optimum dose partly depended on head loss that the filters were run to before backwashing.

9.5 PHYSICAL INFLUENCES ON RIPENING

In chapter 6 examples were presented from the literature showing how different physical variables acted to influence filter ripening. As a filter run progresses a mass of deposits builds up on the filter media. A number of different deposit morphologies have been described, and it was possible these changed over the course of a filter run. Smaller media grain size and rough surface textures improved filter efficiency, as did higher influent suspended solids concentrations, so it was unsurprising that the presence of deposits caused filter ripening. Limited studies on the effect of flow rate on ripening were found, although it was generally considered that higher flow rates caused poorer filtration. Few data existed on the influence of temperature on filtration.

A number of studies had been conducted to examine the influence of particle size distribution on ripening. Although the results showed many complexities, and contradictions, a general picture emerged of larger particles initially being well removed. Subsequent breakthrough was due to increasing fluid shear forces causing poorer attachment of the original particles and the detachment of aggregates of deposited particles. Smaller particles displayed prolonged ripening periods and less tendency to breakthrough (figure 6.1).

The range of seasonal variations in the stored water caused problems with drawing general conclusions from the pilot plant data. For example, the experimental results showed that, with the undosed filter, turbidity removals were clearly poorer at higher flow rates (figure 6.6). However the

scatter in a year's removal data showed the influence of other variables obscured any relationship between removal and flow rate (figure 6.1).

An unexpected finding was that the filters dosed with iron or iron and ozone showed no difference in ripening time or ripened filtrate turbidity at flow rates of 12, 17 and 22 $m.h^{-1}$ (figures 6.12 and 6.16). This suggested that slow starts were not likely to control ripening.

There was a strong relationship between turbidity removal and water temperature seen with the undosed sand filter (figure 6.17). Reasons for this could have been both physical and biological. No relationship was discernible when iron and ozone were dosed (figure 6.18).

With the undosed filter the removal of turbidity was largely independent of applied turbidity load and flow rate (figures 6.19, 6.22, 6.23). This was not the case with the iron and ozone dosed filter. Figures 6.20 and 6.21 showed a complicated picture of poor removals at low loads, and a different relationship between turbidity removal and filtrate turbidity in winter and summer was observed. The summary data in figures 6.24 to 6.29 showed that removals were better, filtrate turbidities were poorer, and ripening was more apparent with higher inlet turbidities, although there was more scatter in the data with higher turbidities suggesting this was an over-simplification. It was suggested that the effect of different algal concentrations and morphologies was the cause. The particle loading index, based on the weekly stored water samples, did not shed further light on the effect of algal loading on filter ripening, although there was a link between a high particle index, high inlet turbidity and high iron doses in October 1992. This was illustrated in chapter 7, where a brief discussion of the influence of algae on filtration was given.

One indirect indicator of loading was the filter run length, although this was also related to particle capture efficiency. It was shown for the dosed and undosed filters that the ripening behaviour was poorer and more pronounced on longer filter runs. The evidence from the literature would suggest that this reflected the complex relationship between particle load in terms of volume, surface area, shape and size, the structure and location of the deposits formed. With the iron dosed filter the

additional influence of the chemical dose on attachment efficiency and floc formation would need to be considered. Different ripening behaviours were noted for undosed dual media and sand only filters, and an explanation based on similar reasoning was proposed.

9.6 EXAMINING FILTER RIPENING

Chapter 8 returned to the subject of defining ripening. Using idealised curves it was demonstrated that there were problems inherent with each of six different ways of defining ripening. This would be further complicated by the fluctuations in turbidity values measured experimentally.

9.7 CONCLUSIONS

It is clear from the literature that the ripening period presents an opportunity for pathogenic micro-organisms to penetrate the solids-liquid separation process in conventional water treatment. As Al-Ani *et al.* (1986) Gertig *et al.* (1988), Mosher and Hendricks (1988) and Saterdal *et al.* (1988) showed roughing filtration alone was not a satisfactory means of removing particles from water. In London the slow sand filters should be retained after roughing filters. Due to the reservoir storage, the mean stored water and filtrate turbidities recorded from the un-dosed and ozone dosed filters were well below the 5 NTU value regarded by Cleasby *et al.* (1984) as being suitable for treatment by SSFs. However the filtrates exceeded the 0.2 JTU value given by Janssens *et al.* (1982) as being desirable for high rate SSFs (assuming 1 NTU = 1 JTU).

Only iron and ozone dosed filters were capable of achieving filtrates below 0.2 NTU. Thus iron and ozone dosing should be used for pre-treatment of waters ahead of high rate SSFs. The results suggested that it was not always possible to achieve filtrates this low with the higher inlet turbidities. This was particularly noticeable in the winter when low doses of coagulant produced long ripening times and breakthrough still took place. A better strategy would have been to use a higher coagulant dose and backwash on filtrate turbidity quality not head loss, since the U-shaped runs that were recorded lasted several days.

The filtrate quality produced by the iron and ozone dosed filter was not likely to be of the same standard as that of SSFs. It should be remembered that the study sought to assist the SSF process not replace it. From this point of view the ripening period was probably not of significance. The impact on SSF performance of poor ripening removals from iron and ozone dosed filters has not been investigated. Whilst the SSFs should ensure that *Cryptosporidium* oocysts or *Giardia* cysts did not enter the public water supply, the extra load on to the SSFs might be of significance, especially if they were shaded from sunlight to reduce in-bed algal growth.

Turbidity data showed ripening was a real effect with iron and ozone dosed filters. Defining filter ripening was difficult as it appeared to start rapidly, but could continue slowly for a prolonged period. Filtering all this water to waste would use up a significant part of the filter run. The initial peak duration was usually less than an hour and this could be filtered to waste if desired. The lack of an unambiguous indicator to say a filter is ripened means that operating practises to deal with ripening will have to be based on a pragmatic risk management approach rather than mathematical exactness. For example, it is not sensible to filter to waste the water from a filter which ripens to 0.11 NTU even if the target was 0.10 NTU. RGFs are not an absolute barrier to particles in the way membranes are, and turbidity is a very general indicator of filtrate quality.

Stored water turbidity data did not provide sufficient information to predict filter performance or coagulant dose. More needs to be known about the influence of algae on all aspects of filtration.

In spite of the difficulties with trying to unravel all the influences on filter ripening, the pilot plant study provided the process design for new RGF plants which have been operating successfully at Ashford Common and Walton on Thames in Thames Water Utilities Ltd. The Ashford Common study and parallel dissolved air flotation trials (Eades *et al.* 1994) were the first to result in chemical coagulation being used at a full-scale surface water treatment plant in Thames Water's London area.

Having examined a large body of scientific literature on the subject of filtration and filter ripening, and conducted a study which revealed that filter behaviour was subject to a number of factors both

within experimental control, and under the influence of environmental variables, it is fascinating to note the comments of Prof. W.E. Stanley in the discussion of Iwasaki's (1937) classic paper initiating the era of filter modelling. Despite countless hours of research by academics and practical water treatment scientists and engineers his comments are still valid after 61 years. He said:

> "It is within the realm of possibilities that in due time a team of highly trained and properly equipped scientific men may be able to analyze the water characteristics, the filtering media characteristics and determine with mathematical exactness the results which may be obtained under a given set of conditions or for any number of different sets of conditions.
>
> It is not so easy for the practical minded engineer to anticipate with confidence that these scientific developments can be applied on a practical scale so that the many complications due to the fickleness of mother nature may be neutralized or provided for and a filtered water produced with greater comparative economy than with less refined methods."

CHAPTER 10: FURTHER WORK

10.1 BACKWASHING

In the pilot plant trials it was necessary to produce a workable backwash, but a further study is required to determine precisely the optimum duration and flow rates of the all the phases of a backwash.

Assuming there is sufficient cleaning achieved by collapse-pulsing backwashing what should the backwash objectives be? By careful control of the collapse-pulsing duration, can media be encouraged to mature in a way that assures improved filtrate quality, whilst maintaining acceptable run lengths? It is not known whether the media can be allowed to mature, but then be effectively backwashed clean again to extend the available head, e.g. if filter run times were too short. If it were possible, this might be a strategy to manage filter media as a part of optimising plant performance. Is there an economic relationship between rinse duration and filter production? Do backwash water remnants assist ripening or accentuate the need for a filter to waste period?

Further work is required to determine how maturation assisted filtration and why it appeared not to be a benefit when coagulants were dosed. Laboratory studies could investigate whether biofilms act in the same way as polymer dosing of the backwash water (Cranston and Amirtharajah, 1987) in the structural and chemical effect they have on ripening. It is possible that biological growths on the media surface influenced filter efficiency by a) a geometrical mechanism, as in dendrite formation (O'Melia and Ali, 1978), b) changing the surface roughness (Gimbel, 1982), c) improving attachment efficiency due to the production of exuded bio-polymers by bacteria, as reported for slow sand filtration by Ives (1990), or d) whether predation mechanisms were contributing to turbidity removal, as reported for bacterial removal in slow sand filters by Ellis (1985).

Interestingly, Amirtharajah (1978) stated "In the early years of filtration study it was believed that a dirty coating on the sand grains was essential for effective filtration; and backwashing, which

removed all of this coating, was assumed to be detrimental to filtration". During the 20th century the use of pre-chlorination prevented biological growth in RGFs, as Goldgrabe *et al.* (1993) showed, and it will be interesting to see whether abandoning or reducing pre-chlorination to control trihalomethane formation leads to biological growths. If so, will they cause filter problems or be of benefit? Fulton (1988) reported the development of slimes on filter media as a consequence of ceasing pre-chlorination.

It must be questioned whether a backwash has ever removed *all* the dirty coating from filter media. Bayley (pers. comm., 1993) has said that 20 minutes backwashing using collapse-pulsing conditions kept filter media so clean that filtrate quality was very poor; the biological maturation which assists primary filtration had been prevented. A similar effect of over-washing was noted by Colton *et al.* (1996)

Whether there is any difference in filter maturation in the summer and winter requires further investigation. It might be important to only replace filter media in warm water temperature periods when rapid biological growth is possible, since media changed in the winter could take several months to mature and be under-performing for this time.

Further research on optimising backwashing should confirm the limited studies of the use of chemicals in backwash water, or overdosing the inlet water with chemicals. There may be a case for adding an inert material as "bulking agent", or retaining some backwash remnants above the filter to control ripening.

ANTHRACITE CONDITION

It was not possible to quantify the condition of the anthracite. In year 1 samples of anthracite operated for several months had a brownish appearance. With the better backwash the anthracite maintained a silvery sheen, suggesting it was cleaner. Scanning electron micrographs confirmed that the grains of sand and anthracite lacked the colonisation usually found on dirty primary filter

sand grains (Kay, 1991). Whilst SEM can be used to examine anthracite, further work is required to develop a low cost technique for determining the degree of cleaning taking place on anthracite.

MUD DISCS

The appearance of aggregates of material, potential future mud balls, was of concern. However no mud balls were found, and it is possible that the backwash coped with them in the long term. Further work would be required if these were found elsewhere.

10.2 CHEMICAL TREATMENT

COAGULANT DOSE OPTIMISATION

The sensitivity of the filtrate to coagulant dose, shown by the SLUV model suggested that using one coagulant dose per run was not a satisfactory concept and that ripening could be controlled by using more than one coagulant dose during a filter run. Trials should be conducted with a high initial dose to minimise the ripening period, followed by a trimming back to the optimum to sustain long runs without turbidity breakthrough.

Engineering a separate second coagulant dose on a filter plant might present difficulties, based on the need to supply extra dosing pumps, pipelines and coagulant injection and mixing points to each filter, timed to operate at the start of the filter run. A sparge pipe in each filter inlet channel might be a practical solution. The double dose approach could be retro-fitted and does not require breaking into the backwash system. The alternative of filtering to waste can also be complex to engineer and impossible to retro-fit.

Whilst modifying the backwash water chemistry is feasible, it has similar problems of supplying and controlling the equipment, and there is a risk that water with a high coagulant or polymer concentration remains on the clean water side of the filter. Cranston and Amirtharajah (1987) only reported turbidity ripening data. No data on alum or polymer in the filtrate were given. Unless dilution with filtrate takes place this water might need to be filtered to waste!

The iron was added either before or after the ozone, with elbows, orifice plates or weirs to induce turbulent mixing conditions. Static mixers were employed if the water did not pass through ozone contactors. To achieve the low doses in the winter the pumps were dosing only 5 to 10 times a minute. Coagulant dosing clearly improved filtration and seemed to be very flexible in terms of installation requirements. There was room for further work optimising the location of coagulant injection and control of mixing energy (G).

OZONE OPTIMISATION

For the reasons outlined in chapter 5 there was further scope for optimising the ozone dose and contact times for turbidity and particle removal. It has been demonstrated (Chipps *et al.*, 1994) that ozone was more beneficial to the removal of algae than turbidity. Perhaps in the same way that on-line turbidity monitoring was used to optimise iron dose, on-line particle monitoring could be used to optimise ozone dose and contact time. It should be borne in mind, however, that practical situations may require ozone to be optimised for pesticide reduction (Foster *et al.*, 1991) rather than particle removal.

10.3 FUNDAMENTAL UNDERSTANDING OF CHEMICAL TREATMENT

These trials contributed to the knowledge of practical water treatment yet left the fundamental mechanisms of how the iron and ozone promoted turbidity removal for further investigations. The fundamental relationship between the water chemistry, biology, and particle loading, and the optimum iron dose requires further understanding to enable iron dose to be predicted. As figure 7.1 showed, the mean stored water turbidity and the mean iron dose in each run, and the particle index measured weekly, showed no consistent relationship.

Whilst a short trial with a zetameter produced no evidence of any change in particle charge with a wide range of coagulant doses, electrophoretic mobility measurements have been successfully carried out on algae in filtered river Thames water (Rodman, 1982). This is a time consuming

procedure but it might enable the relationship between particle charge, iron dose and ozone dose to be better understood.

The influence of pH on electrophoretic mobility was demonstrated by Rodman (1982) on algal cells in a culture medium. It is usual to optimise pH along with coagulant dose in water treatment. This could form the basis of future trials to see whether it has an impact on ripening. The problem with river Thames water is that it is highly buffered and attempts to modify pH at operational scale would be expensive.

Ta and Woodward (1997) have developed a treatability index based on the roughing filtration of cultivated suspensions of lake algae. This could be extended to include ozonation and coagulation of algal suspensions and examination of the ripening period.

THE MECHANISM OF PARTICLE REMOVAL WITH OZONE

These trials do not indicate how ozone caused the better removal of turbidity. It may have had a direct action on the particles measured by the turbidimeter. Alternatively it may have acted indirectly by improving the removal of larger particles in the water (shown by Chipps *et al.*, 1994) which in turn assisted with the removal of turbidity. Furthermore, it might have stressed the biofilm on the filter media, causing the release of extracellular organic matter, promoting better turbidity removals. It might also have directly influenced the chemistry of the water. The response of the filtration system to turning ozone on and off was very swift, approximately one filter volume. The number of mechanisms proposed in the literature for how ozone assists turbidity removal means that the true cause of this effect will not be easy to determine in field trials.

THE MECHANISM OF PARTICLE REMOVAL WITH COAGULANTS

Determining the mechanism by which the coagulant worked in these experiments was beyond the scope of the study and requires further work so that coagulation might be further improved. O'Melia (pers. comm., 1996) was surprised that such low coagulant doses were effective, given the TOC in the water. He expected the coagulant to form complexes with the soluble organics before

any contact was made with particle surfaces. It was presumably possible that a metal-TOC precipitate could become attached to the media and prepare the media for particle capture. Understanding the fundamentals of the interaction between low coagulant doses and filtration in a biological environment requires further work.

If particle charge can be measured two possible explanations of the SLUV behaviour might be investigated. In the first mechanism the coagulant causes charge neutralisation at lower doses and restabilisation at higher doses. Under conditions which produced the L-shaped curve, the low dose of iron, coupled to an infrequent pump stroke rate, might be insufficient to cause charge neutralisation of all the particles. This could result in a suspension of stabilised and destabilised particles. With clean media some particle - collector encounters would be subject to repulsive forces when stabilised particles approached a negatively charged media surface. As the media became coated with destabilised particles the net charge on the collectors would reduce and removal of stabilised particles would improve. Even if the particles were all only partially destabilised, a coating of deposits would exhibit lower repulsive forces than the clean media.

The higher dose could cause some particles to be destabilised and others restabilised (i.e. carrying excess positive charge). In encountering clean media favourable interactions would take place with all particles. Once a coating of deposits became established some particle - deposit interactions would be between restabilised particles, reducing attachment efficiency and leading to breakthrough.

The second explanation derived from the similarity between the curves of turbidity removal at different doses and the curve of removals of different sized particles given by Mackie (1989). The low doses produced filter behaviour like that of the smaller particles in his trials (0.63 - 1.26 and 1.26 - 2.52 μm). The fast ripening and early breakthrough produced by the higher doses was similar to filter behaviour seen with the suspension containing 5.04 - 10.08 μm particles.

Whilst it is possible that larger coagulant doses could promote more particle aggregation and produce larger particles there are problems with this explanation. The doses were well below sweep flocculation doses. It is not certain that the turbidimeter would have been sensitive to the penetration of larger particles out of the filter. This explanation awaits the development of on-line particle size monitors capable of measuring flocs.

The fact that breakthrough was a gradual process could be consistent with a charge model where negatively charged sites deeper in the bed become progressively covered by restabilised particles. It was also consistent with Mackie's (1989) observations where different working layers existed for different sized particles, with the more numerous smaller particles (to which the turbidimeter is more sensitive, Gregory, 1994) appearing later in the breakthrough.

10.4 PHYSICAL INFLUENCES ON FILTRATION

Neither stored water turbidity nor the particle index could not be used to predict ripening behaviour reliably. Principally the chemistry and the temperature related effects seemed most influential in determining ripening. For the future a full understanding of ripening will require knowledge of the shape and size of particles in the water and the factors that influence coagulant demand. Mackie (1989) and others showed that detailed monitoring of head loss and particle removal at different depths within the bed important to understanding ripening behaviour. All these things can be more easily achieved in the laboratory than in the field. The development of a low-cost, robust, on-line particle counter, measuring shape and size, is awaited.

Experiments are required to determine why water temperature appeared to affect ripening. It would be necessary to devise experiments to control chemical, physical and biological factors separately. Chemical reaction rates and biological activity are slower in colder temperatures and the viscosity of the water affects physical filtration processes.

10.5 BIOLOGICAL INFLUENCES ON FILTRATION

There is much more work to be done to understand the details of the filtration of algae. The examination of water to count and type algae is a labour intensive process, subject to sampling error (Watson, 1990). Ives (1955) commented that further work was needed to understand the relationship between algal taxa and chemical coagulant demand, and this is still true. It was stated that this might not produce meaningful data in waters containing mixtures of algae. Dolejs (1992) showed a further complication in that the age and numbers of algae influenced coagulation conditions. The mechanisms of ozone's action on algae is still unclear. Ives (1955) showed that ozone increased the negative charge on algae. Chipps *et al.* (1994) showed that ozonation was essential to prevent breakthrough of *Microcystis* with contact filtration.

The uncertainties over chemical treatment, coupled with the diversity of life forms and the difficulties in categorising the shapes and sizes of particles in the water makes the development of fundamental models of algal filtration very difficult. Furthermore the behaviour of colonial forms of algae under filtration conditions may add extra complications if the structures identified in the influent deform or break up within the filter.

A further interference biological particles had over simple depth filtration came from observations through the filter windows which showed that some algae, and dead zooplankton, formed surface mats on the top of the filters. This was true even with coarse anthracite (1.7-2.5 mm) and clearly seen as a white layer on the black coal.

Some work has been carried out in measuring the different rates head loss development in filtration of individual algal taxa, and modelling the effect of combinations of taxa on head loss in undosed primary filters (Ta and Woodward, 1997). Limited trials with single species showed that the effect of algal morphology on removals was consistent with filtration theory (Watson, 1990). Further work on ripening with algae, rather than latex spheres, is still required.

Empirical models of filtration may be developed, perhaps using neural networks, if the shape and size classification of particles in the water can be carried out automatically. Data from the filtrate could also be incorporated so that filter performance could be "learned", and adjustments for biological filter maturation could be made.

The trajectory models of filtration will require further development to incorporate the subtleties of biological influences on both the morphology of the particles and their effects on process chemistry if they are to be of practical benefit to water filtration plant design or operation. Furthermore trajectory modelling must also incorporate a new field into filter modelling by taking into account the complex structures produced by biofilms on the filter media (Coghlan, 1996). Whilst existing work has shown the significance of media shape and roughness on filtration (Gimbel and Sontheimer, 1978, Kau and Lawler, 1995) the effect that biofilms have on water filtration has yet to be considered in water filtration. At present it is difficult to determine biofilm structures on the filter media without resort to destructive SEM techniques. The production of mathematical descriptions of flow paths through and around complex structures might be difficult. It was a measure of how far trajectory theories were from operational water treatment plants that Tien (1989) deliberately excluded biological factors in his major review of filter modelling work!

APPENDIX - OPERATING CONDITIONS

This appendix presents details of the media types and depths, flow rates, backwash initiation criterion, and chemical treatment for each column. All flow rates are nominal, based on an approximate value of 5 $l.min^{-1}$ = 1 $m.h^{-1}$. This actually underestimates flow rate as the true ratio was 4.5:1, based on a filter internal diameter of 584 mm. The dates for chemical dosing are for extended trial periods (i.e. many days). Failures or deliberate cessation of chemical dosing of short duration are not included in these tables. These details are in separate log books and data sheets but are not presented here. When two media are described together it denotes a dual media filter. Where the same media appears twice, a fresh charge of media was used.

Ozone was available to column 1 from July 1989. Iron was dosed only to column 5 in 1989. In 1990 ozone and iron were available to column 1. A temporary ozone contactor, suitable for low flows was provided to column 2 but experiments on this column concentrated on iron, acid and polyelectrolyte dosing. Column 5 was iron dosed.

On a chemically dosed filter a change in flow rate occurred when the filters were washing too frequently, or, if optimum iron dose could not be found at the high rate, then rates were reduced to re-establish correct dose.

In 1991 ozone and iron were maintained to column 1. Ozone and iron dosing were started to columns 5 and 6, and column 2 was briefly provided with column 5 or column 6's ozone and iron dosed supply. Column 2 and column 5 could be individually iron dosed without ozone. Towards the end of 1992 column 3 was fed with the same ozone and iron dosed water as columns 5 and 6.

DESCRIPTION OF FILTER MEDIA CONFIGURATIONS

Table A1.1. *Filter media configuration 1989.*

COLUMN	FILTER MEDIA	DEPTH (m)	START DATE	END DATE
1	GAC Chemviron F400	1.0	13/02/89	11/07/89
1	GAC Chemviron F400	1.0	11/07/89	11/01/90
2	10/18 sand	1.0	13/02/89	11/07/89
2	GAC Chemviron F400	1.0	11/07/89	11/01/90
3	Anthracite Grade 2 14/25 sand	0.34 0.66	13/02/89	11/01/90
4	14/25 sand	1.0	13/02/89	11/01/90
5	Anthracite Grade 2 16/30 sand	0.66 0.34	13/02/89	11/01/90
6	Anthracite Grade 2 16/30 sand	0.66 0.34	13/02/89	11/01/90

Table A1.2. *Filter media configuration 1990.*

COLUMN	FILTER MEDIA	DEPTH (m)	START DATE	END DATE
1	Anthracite Grade 2 Coarse 14/25 sand	0.66 0.34	22/02/90	continuing
2	Anthracite Grade 2 Coarse	2.0	22/02/90	05/11/90
3	Anthracite Grade 2 14/25 sand	0.34 0.66	22/2/90	05/11/90
4	14/25 sand	1.0	22/02/90	05/11/90
5	Anthracite Grade 2 Coarse 14/25 sand	0.34 0.66	22/02/90	21/11/90
6	Anthracite Grade 2 Coarse 14/25 sand	0.34 0.66	22/02/90	22/11/90

Table A1.3. *Filter media configuration 1991 - 1993.*

COLUMN	FILTER MEDIA	DEPTH (m)	START DATE	END DATE
1	Anthracite Grade 2 Coarse	0.66	22/02/90	12/04/93
	14/25 sand	0.34	(continued)	
2	6/14 sand	1.8	19/11/90	14/05/92
		0.5 *	15/07/92	30/07/92
3	10/18 sand	0.9	19/11/90	12/05/92
3	Anthracite Grade 2 Coarse	0.4	25/09/92	12/04/93
	14/25 sand	0.4		
4	14/25 sand	0.6	19/11/90	12/04/93
		(0.9 **)	(30/10/91)	
5	Anthracite Grade 3	0.66	26/11/90	16/08/91
	10/18 sand	0.34		
5	Anthracite Grade 2 Coarse	0.6	12/09/91	12/04/93
	14/25 sand	0.6		
6	Anthracite Grade 3	2.0	11/12/90	19/08/91
6	Anthracite Grade 2	0.6	12/09/91	12/04/93
	14/25 sand	0.6		

*Column 2 had 1.3 m depth of media removed for this trial.
** Column 4 media was not replaced. An additional 0.3 m depth of new sand was added to the bed and the fines removed.
Column 5 was topped up with 200 mm anthracite to replace lost media 20/02/92.

BACKWASH INITIATION

Table A2.1. *Backwash initiation 1989.*

COLUMN	COMMENT	VALUE	START DATE	END DATE
ALL	Initiated by floatswitch	Column full (see logged data files)	13/02/89	02/10/89
ALL	Initiated by dp value	1.2 m	02/10/89	11/12/89
ALL	Columns extended by 1.2 m (type A) or 0.77 m (type B)	1.8 m	11/12/89	11/01/90

Table A2.2. *Backwash initiation 1990.*

COLUMN	COMMENT	VALUE	START DATE	END DATE
ALL	Initiated automatically by computer measurement of differential pressure	1.8 m	22/02/90	26/11/90

Table A2.3. *Backwash initiation 1991 - 1993.*

COLUMN	COMMENT	VALUE	START DATE	END DATE
1	Continued 1990-1 trial	1.8 m	22/02/90	16/05/91
1	Deliberately set at 1.5m under programme control, then, after changing program continued at 1.5m due to floatswitch being set too low (programme changed 1/10/91)	1.5 m	16/05/91	31/12/91
1	Floatswitch problem resolved	1.8 m	01/01/92	12/04/93
2 - 6		1.8 m	Nov 1990	as below
3	Increased for 3	2.5 m	Dec 1992	12/04/93
4	Increased for 4	2.5 m	09/06/92	12/04/93
5 & 6	Increased for 5&6	2.5 m	28/01/92	12/04/93

All filters: Outlet valve opened fully after backwash up to 17/7/92, closed to produce correct flow rate. Thereafter partially opened till correct flow rate achieved.

EXPERIMENTAL PROGRAMME 1989

Table A3.1. *Chemical dosing 1989.*

COLUMN	CHEMICAL	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
1	Ozone 2 $mg.l^{-1}$, 4 mins	10	11/07/89	11/01/90
5	Iron (III) sulphate	8	05/05/89	28/06/89
		10	28/06/89	25/07/89
5	Iron (III) sulphate	10	11/11/89	11/01/90

Table A3.2. *Flow rates 1989.*

COLUMN	COMMENT	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
ALL	Commissioning period	6	13/02/89	10/03/89
ALL	Data logging commenced	8	10/03/89	28/06/89
ALL		10	28/06/89	11/01/90
ALL	During 72 hour survey	12	19/06/89	22/06/89

EXPERIMENTAL PROGRAMME 1990 - 1993

Table A4.1. *Operating conditions for filters without chemical dosing during 1990: dual media in columns 3 and 6, and monomedium sand in column 4.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
3 & 4	None	10	22/02/90	21/06/90
3 & 4	None	5	21/06/90	02/07/90
3 & 4	None	12	02/07/90	10/07/90
3	None	15	10/07/90	06/11/90
4	None	15	10/07/90	08/11/90
6	None	15	10/07/90	22/11/90

Table A4.2. *Operation of dual media filter in column 1, 1990* - 1993.

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
1	Ozone	10	27/02/90	11/04/90
1	Ozone and Iron	10	11/04/90	21/06/90
		5	21/06/90	02/07/90
		12	02/07/90	05/07/90
1	Iron	10	05/07/90	06/07/90
1	None	10	06/07/90	10/07/90
1	None, ozone contactor by-passed	15	10/07/90	26/07/90
1	Ozone	10	26/07/90	30/07/90
1	Ozone and Iron	10	30/07/90	08/08/90
1	Iron (III) sulphate	10	08/08/90	21/08/90
1	Ozone and Iron	10	22/08/90	11/10/90
1	Ozone (many failures over this period)	10	11/10/90	15/03/91
1	Ozone and Iron	10	15/03/91	22/04/91
1	Iron and Ozone 22/4/91 moved iron injection upstream of ozone contactor 7/5/91 mixer removed 14/5/91 diffuser repaired also - October	10	22/04/91	16/05/91
1	Iron and Ozone	13	16/05/91	24/01/92
		11	24/01/92	09/03/92
		13	09/03/92	12/04/93

Table A4.3. *Operation of deep coarse anthracite filter in column 2, 1990.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
2	Iron (III) sulphate	10	12/04/90	21/06/90
		5	21/06/90	28/06/90
2	Ozone and Iron	5	28/06/90	02/07/90
2	Iron (III) sulphate	12	02/07/90	10/07/90
2	None	15	10/07/90	30/07/90
2	Iron, LT25, LT31, H_2SO_4	15	30/07/90	05/11/90

Table A4.4. *Operation of dual media filter in column 5, 1990.*

COLUMN	COMMENT	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
5	Iron (III) sulphate	10	22/02/90	21/06/90
		5	21/06/90	02/07/90
		12	02/07/90	09/07/90
5	None	15	10/07/90	30/07/90
5	Iron (III) sulphate	15	30/07/90	21/11/90

Table A4.5. *Flow rates through non-dosed filters, columns 3 and 4, monomedium sand 1991 - 1993.*

COLUMN	COMMENT	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
3		16	19/11/90	24/04/91
3		5	24/04/91	21/11/91
3	Failed outlet valve meant 7 $m.h^{-1}$ for some of this period (replaced valve 6/1/92)	10	21/11/91	14/05/92
4	Media depth increased 30/10/91	10	19/11/90	16/06/92
		13	16/06/92	19/06/92
	Turbidimeter draining during wash prevented after 22/1/93	5	19/06/92	22/06/92
		10	22/06/92	23/06/92
		13	23/06/92	25/06/92
		15	25/06/92	26/06/92
		20	26/06/92	01/07/92
		5	01/07/92	02/07/92
		10	02/07/92	06/07/92
		13	06/07/92	07/07/92
		15	07/07/92	10/07/92
		5	10/07/92	19/08/92
		15	19/08/92	08/10/92
		8	08/10/92	13/10/92
		5	13/10/92	08/01/93
		9	08/01/93	12/04/93

Table A4.6. *Operation of coarse sand filter in column 2, 1991 - 1993.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
2	None	20	19/11/90	27/12/90
		25	27/12/90	24/04/91
		5	24/04/91	19/06/91
		10	19/06/91	22/08/91
2	Iron and Ozone and Iron (double injection)	20	22/08/91	23/10/91
2	Iron (III) sulphate	20	23/10/91	02/03/92
2	Iron (III) sulphate Stored water pump failed	10	02/03/92	14/05/92
2	Iron and PAC trials - shallow bed	10	16/07/92	31/07/92
2	Iron and PAC trials - shallow bed	10	13/10/92	16/10/92

Table A4.7. *Operation of dual media filter in column 3, 1992 - 1993.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END D
3	None	10	04/11/92	16/12/9
3	Iron and Ozone (shared with feed to columns 5 & 6)	10	16/12/92	12/04/9

Table A4.8. *Operation of coarse dual media filter in column 5, 1991.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
5	None	20	26/11/90	16/01/91
5	Iron (III) sulphate	20	16/01/91	24/01/91
5	Ozone and Iron, LT31	20	24/01/91	11/03/91
5	Ozone and Iron, LT31 many ozone failures 19/3/91 iron moved to contactor outlet	15	11/03/91	09/04/91
5	Ozone and Iron	20	09/04/91	10/04/91
		15	10/04/91	11/04/91
		11	11/04/91	30/04/91
		15	30/04/91	13/05/91
5	Iron and Ozone 13/5/91 injection upstream of contactor	15	13/05/91	28/05/91
5	Iron and Ozone	20	28/05/91	16/08/91

Table A4.9. *Operation of dual media filter in column 5, 1991 - 1993.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
5	None - media fines removed 20/9/91	10	16/09/91	08/10/91
		20	08/10/91	22/10/91
5	Iron and Ozone	20	22/10/91	07/11/92
		15	07/11/91	14/11/91
		11	14/11/91	25/11/91
		15	25/11/91	27/11/91
		20	27/11/91	04/12/91
		15	04/12/91	13/01/92
5	Ozone and Iron (13/1/92 dosing point moved to outlet, via injector nozzle)	15	13/01/92	01/06/92
		13	01/06/92	03/06/92
5	Iron and Ozone (3/6/92 moved injector to inlet of contactor)	13	03/06/92	13/01/93
		11	13/01/93	14/01/93
		9	14/01/93	12/04/93

Limited trials with H_2O_2 in May 18-22 1992, and polymers in August 3-24 1992, and LT31 from January 22 - February 26 1993. Period of experimentation no dosing / ozone / iron / ozone and iron from March 1992. Anthracite topped up 20/2/92.

Table A4.10. *Operation of deep, coarse anthracite filter in column 6, 1991.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
6	None	20	11/12/91	24/01/91
6	Ozone and Iron, LT31	20	24/01/91	11/03/91
6	Ozone and Iron, LT31 many ozone failures 19/3/91 iron moved to contactor outlet	15	11/03/91	09/04/91
6	Ozone and Iron	20	09/04/91	10/04/91
6	Ozone and Iron	15	10/04/91	11/04/91
6	Ozone and Iron (supplementary iron from 29/4/91)	11	11/04/91	16/05/91
6	Iron and Ozone (13/5/91 injection upstream of contactor) (single and double dosing)	15	16/05/91	28/05/91
6	Iron and Ozone and Iron (double dosing)	20	28/05/91	19/08/91

Table A4.11. *Operation of dual media filter in column 6, 1991 - 1993.*

COLUMN	CHEMICAL DOSING	FLOW RATE ($m.h^{-1}$)	START DATE	END DATE
6	Iron and Ozone	20	28/08/91	07/11/91
6	Iron and Ozone	15	07/11/91	14/11/91
		11	14/11/91	25/11/91
		15	25/11/91	27/11/91
		20	27/11/91	04/12/91
		15	04/12/91	13/01/92
6	Ozone and Iron	15	13/01/92	01/06/92
	(dosing point moved to outlet, via injector nozzle)	13	01/06/92	03/06/92
6	Iron and Ozone	13	03/06/92	13/01/93
	(3/6/92 moved injector to inlet	11	13/01/93	14/01/93
	of contactor)	9	14/01/93	12/04/93

Limited trials with H_2O_2 in May 18-22 1992, polymers in August 3-24 1992, and PAC and LT31 in January 21-29 1993. Period of experimentation no dosing / ozone / iron / ozone and iron from March 1992.

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