The use of coagulation-flocculation as a treatment for starch factory effluent

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THE USE OF COAGULATION - FLOCCULATION AS A TREATMENT FOR STARCH FACTORY EFFLUENT.

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B.Sc. (HONS)

Being a thesis submitted in fulfilment of the requirements of the degree of

Master of Engineering

School of Civil Engineering

University of NSW

Submitted June 1995

CERTIFICATE OF ORIGINALITY

I hereby declare that this submission is my own work and that, to the best of my knowledge and belief, it contains no material previously published or written by another person nor material which to a substantial extent has been accepted for the award of any other degree or diploma of a university or other institute of higher learning, except where due acknowledgement is made in the text.

I also declare that the intellectual content of this thesis is the product of my own work, even though I may have received assistance from others on style, presentation and language expression.

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ABSTRACT

Bench-scale jar test experiments using chemical coagulation followed by sedimentation were conducted with effluent generated from a starch and gluten manufacturing factory. The objectives of these trials were to achieve maximum removal of the colloidal suspended solids fraction of the effluent by optimising physical and chemical parameters of the jar test and the various coagulating agents.

Aluminium sulphate (alum) and ferric sulphate were the two metal salts assessed for their ability to reduce the suspended solids. A range of cationic (Zetag) and anionic (Magnafloc) synthetic polyelectrolytes were also assessed as was the cationic polymer chitosan (a derivative of the naturally occurring chitin). All coagulants were tested over a broad concentration and pH range. The effects on suspended solids removal were also investigated by combining the metal salts with various polyelectrolytes.

The results demonstrated that coagulation was most effective in the pH range of 7-10. Ferric sulphate was consistently more effective than alum, removing up to 63% suspended solids from the effluent at pH 8 compared to 53% removed by alum at pH 8. The optimum concentration for both these metal salts was 75 mg/L. When Zetag 92 at 7.5 mg/L was used with ferric sulphate (75 mg/L) suspended solids were reduced by about 69%. Chitosan proved to be the most effective of all the polyelectrolytes evaluated, removing about 69% of the suspended solids at 10 mg/L, pH 7. The addition of 5 mg/L alum reduced the suspended solids in the effluent by 78%.

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CHAPTER 1. <u>INTRODUCTION</u>

Industries involved in food processing and manufacturing employ various methods of preparation which include cutting, grinding, washing, etc. Most of these processes require large volumes of water, with the concomitant production of large volumes of effluent. The physical breakdown of the food exposes cell contents and releases proteins, carbohydrates, sugars, fats and enzymes into the water. As a result, the effluent's contain high levels of dissolved and particulate or colloidal organic matter. Because these organic components are readily subject to enzymic and microbial action, the traditional choice of treatment has been biological.

According to the Office of Technology Transfer (1975), in almost all the food processing industries, the effluent problems can be tackled either inside the plant, which involves the production processes, or at the "end of the pipe", which involves treatment of the effluent. Optimising the production process for the purpose of minimising wastes can have a number of advantages: the quantity of saleable products can increase; a decrease in water consumption and a decrease in the volume of waste water generated is obtained; a decrease in energy costs and capital costs to treat the effluent are realised; levies and fines tend to decrease; and the overall burden to the environment is reduced.

Tackling the problem at the "end of the pipe" gives rise to two major aspects of concern on how to best treat the effluent: the volume of the waste water generated and the pollution load of the waste water. The pollution load or quality of the effluent will be largely determined by the type of food produced and the production methods employed. According to Punt and Cook (1965), there are several factors which need to be addressed when designing a system(s) to reclaim/treat food wastes from the waste water, and therefore to clean the water:

- is recovery economically viable?
- how exactly can the waste be reclaimed?
- how will unusable effluent be disposed off?, and
- can the treated effluent be recycled in the process?

The control of the wastes generated by most food industries can be problematic, since these waste components contain very high levels of colloidal and soluble solids, very high levels of BOD and COD, and are biologically unstable. For example, wastes generated from coffee industries in Kenya can have BOD levels as high as 9000 mg/L, and the wastes tend to biodegrade rapidly, creating anaerobic conditions in receiving water bodies (Gathuo et al, 1991).

In most cases, food wastes are treated in a combination of aerobic and anaerobic biological processes which convert the bulk of the organic matter to simpler compounds such as methane, carbon dioxide, water, etc. In reducing the organic levels of the effluent, it avoids the pollution of nearby land areas and receiving water bodies.

Unfortunately, for most biological treatments, these food waste components are completely degraded. According to Birch et al, (1976), wastes generated may well provide food for human or animal consumption. This is because these wastes have organic and nutrient rich components which can be utilised for use in value added products. For these components to be of any use, they need to be reclaimed and purified from the large volumes of water, usually in relatively confined spaces.

The objectives of this project were to evaluate the efficiency of a bench scale physico-chemical methods of chemical coagulation and flocculation with sedimentation in the treatment of effluent generated from a starch factory. This high strength organic waste originated from a process of dough, starch and protein separations, washings and extractions

Chemical coagulation with sedimentation was chosen for a number of reasons:

- the test is a relatively rapid test, with relatively short retention times, compared to biological treatment methods. Shorter retention times are favourable especially for factory sites with limited space;

- the method of coagulation-flocculation with sedimentation is a widely used method, especially for the treatment of municipal water, therefore extensive

work has been done and there is a wide body of literature on the area, although literature is scant regarding the use of coagulation on food wastewaters;

- chemicals are accessible and affordable in developed countries. Although the price of polymers may be expensive on a per unit basis, there is a wide range to meet the cost and needs for most processes;

- coagulation and sedimentation is not a destructive process, therefore the sludge generated has potential to be used as animal feed (if from a food factory), or as part of a value added product;

- in many cases, the supernatant quality is such that it can be used as recycle water or discharged directly into the sewage treatment works if it meets statuatory requirements.

It should be stressed that chemical coagulation is not being viewed as the only treatment process alternative for this starch waste, but potentially, as part of an integrated treatment process. According to the Office of Technology Transfer (1975), chemical coagulation and flocculation of food wastes does not always remove the bulk of the BOD, simply because of the high levels of soluble solids which contribute to the BOD, and which do not readily get removed in coagulation. In such cases the chemically treated effluent would get further treated in a polishing step, such as biological treatment.

As part of the major aim of evaluating chemical coagulation as a potential treatment method, the objectives included the evaluation of the metal salts ferric sulphate and aluminium sulphate and various polyelectrolytes in their efficacy as coagulating agents. Further aims included the evaluation of various physical parameters such as mixing intensity and mixing duration; dosage rates and chemical injection/addition points of the jar test-which served as the test method throughout this research project.

CHAPTER 2. <u>LITERATURE REVIEW</u>

<u>2.1 INTRODUCTION</u>

Effluents from many industrial and domestic sources, as well as many natural water supplies, contain a wide variety of particles suspended or dissolved in water. These particles display large variations in their physical and chemical characteristics, including their size, which can vary over many orders of magnitude.

Dissolved or soluble molecules or ions generally lie in the range of 2×10^{-4} mm to 10×10^{-4} mm, while suspended particles lie in the range from about 0.45 mm to the macroscopic (Eckenfelder, 1980). In between these two levels of soluble and suspended fractions are ther colloids, particles which range in size from 1×10^{-3} mm to 1 mm (Edzwald, 1979 and Barnes et al, 1981).

As indicated in Figure 2.1, there are no distinct boundaries between the various classes of molecules, the diameters are given as broad indicators of the sizes involved.

The separation of these particles from water or wastewater is desirable and necessary for a number of reasons. In the case of natural water supplies, the removal of potentially dangerous microorganisms as well as colour, turbidity and suspended solids will render the water supply safe and aesthetically pleasing for municipal use.

In the case of effluents generated from industries and domestic sources, removal of contaminants helps to meet the strict requirements imposed by regulatory authorities; it decreases the overall impact on treatment works; and in the case of many food industries these "contaminants" may be recycled as part of a value added product. This recovery and utilisation of waste materials from the effluent streams yields cleaner effluents, which may be re-used in the manufacturing process, creating an overall cost-reduction.
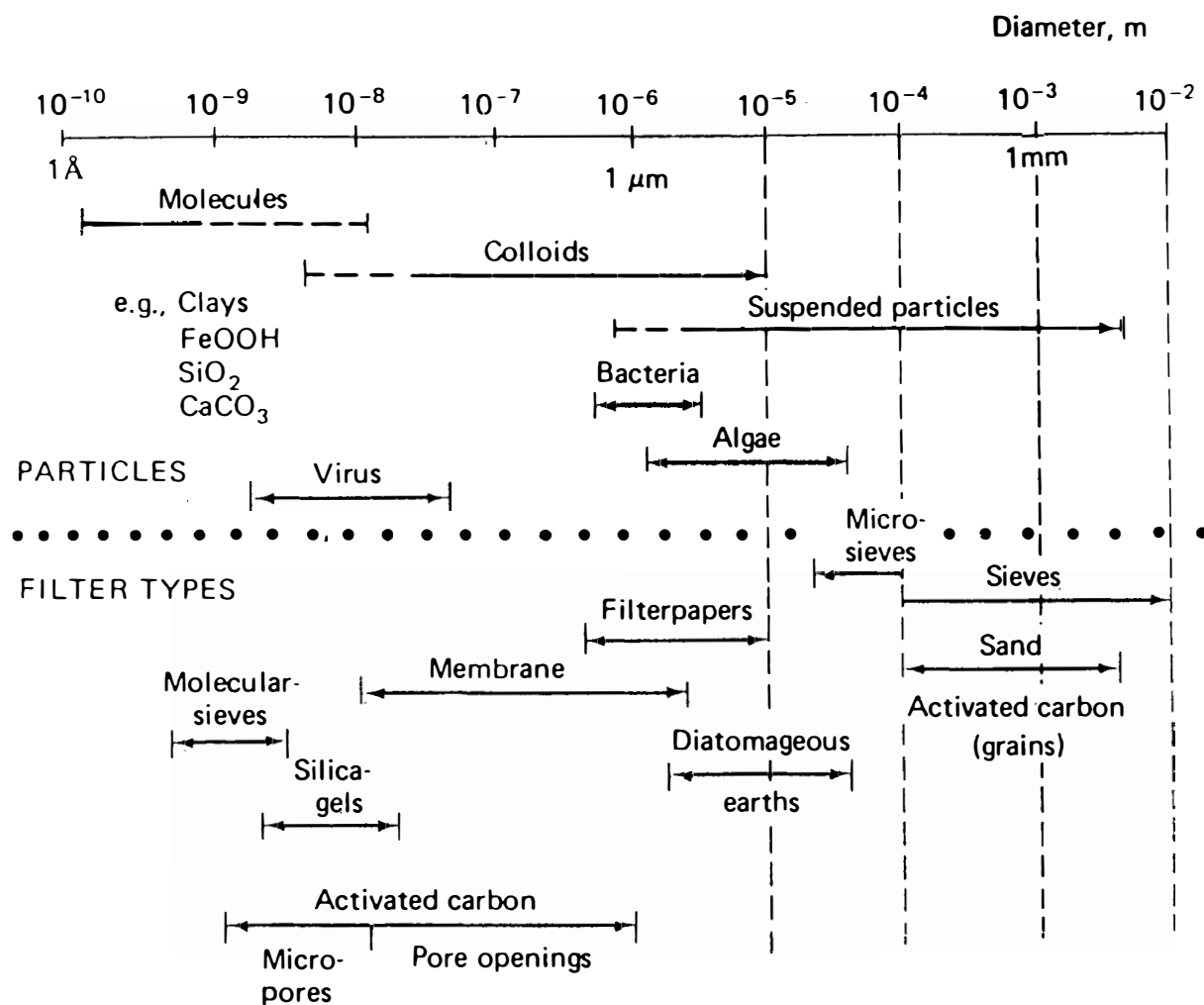
Figure 2.1 The size spectrum of waterborne particles and filter pores (from Benefield et al, 1982, p192)

PARTICLE SIZE mm	CLASSIFICATION	EXAMPLES	TOTAL SURFACE AREA m^2/cm^3	TIME REQUIRED TO SETTLE 100mm IF SP. GRAVITY=2.65
10	COARSE DISPERSION (visible to naked eye)	Gravel, coarse sand, fine sand, mineral substances, precipitated and flocculated particles, silt and macroplankton	6.10^{-4}	0.1 second
1			6.10^{-3}	1 second
0.1			6.10^{-2}	13 seconds
0.01	FINE PARTICULATE DISPERSION (visible under microscope)	Mineral substances, precipitated and flocculated particles, silt, bacteria, plankton and other organisms	0.6	11 minutes
0.001			6	20 hours
0.0001			60	80 days
0.00001	COLLOIDAL DISPERSION (submicroscopic)	Mineral substances, hydrolysis and precipitation products, macromolecules, bio-polymers, viruses	600	2 years
0.000001			6000	20 years
$< 10^{-6}$ mm	SOLUTION	Inorganic simple and complex ions, molecules and polymeric species, polyelectrolytes, organic molecules, undissociated solutes, small aggregates		

Table 2.1 Classification of particle sizes (from Bratby, 1980)

According to Bhatia and Cheremisinoff (1979), those particles with dimensions greater than 10^{-2} mm tend to settle out of the aqueous phase in a relatively short time frame and can be subsequently removed by conventional sedimentation and filtration.

In the treatment of water or wastewater, problems may be encountered when applying separation techniques such as sedimentation and filtration if colloids are a significant component in the system to be treated. This is due to the fact that colloids take an extremely long time to settle.

According to Eckenfelder (1980), all particulate suspensions are thermodynamically unstable and given enough time, the particles and colloids will settle. In most cases, this process is not economically feasible. Shaw (1983), makes a point on the relativity of stable and unstable colloids, colloids which maintain their characteristics for a few days may be stable in one application, while other applications would require that a minimum of 2 years pass for a change in colloidal characteristics.

Table 2.1 indicates colloids can take anywhere from a few days to a number of years to settle! Consequently it is necessary to physically and chemically induce these colloids to agglomerate into larger particles which will aid in their sedimentation or filtration.

Since particles which have colloidal dimensions or smaller remain dispersed and take a relatively long time to settle, they are known as stable particles, or stable suspensions (Bratby, 1980). The stability of the colloids is due to interfacial factors such as the surface charge of the colloid which will be discussed further in detail.

The process used to combine colloids into larger particles is that of coagulation-flocculation. This process changes the surface properties of the particles, creating a tendency for the small particles to aggregate and form into larger particles. This is a destabilisation process, aiming to settle out the colloids and aid in the subsequent sedimentation or filtration stages of treatment processes (Akers, 1975).

In this review, the nature of colloidal and suspended particles in aqueous phase has been presented, as well as the mechanisms of coagulation-flocculation. The knowledge of their physico-chemical structure as well as their behaviour and interactions must be characterised and understood as an essential first step in the development of any treatment regime.

Chemical coagulation and flocculation with sedimentation are studied and reviewed, especially regarding food industry wastes, namely starch manufacturing effluents.

2.2 CHARACTERISTICS OF COLLOIDS AND SUSPENDED PARTICLES

For the purpose of this review, "colloidal systems" and "colloidal dispersions" will be those which occur as dispersions in water, unless otherwise stated. Colloidal dispersions are different from true solutions in that they have weak effects on depressing freezing points and elevating the boiling points of the solvent in which they are dispersed (Barnes and Wilson, 1983).

The word "colloid" was coined to distinguish materials like gelatin which were retained by parchment membranes from smaller molecules such as sucrose (Dickinson and Stainsby, 1982). According to Eilbeck and Mattock (1987), there are two broad categories of colloidal dispersions: hydrophobic colloids which are dispersions of an essentially insoluble phase in water; and the hydrophilic colloids, which are true solutions of molecules having colloidal dimensions.

Hydrophobic colloids include colloids such as clay and non-hydrated metal oxides (Bratby, 1980). They are stable as a result of mutual electrostatic repulsions which arise from ions that are attracted to the surface of the colloids from the bulk solution (Benefield et al 1982). According to Eilbeck and Mattock (1987), these hydrophobic colloids must have a slight affinity for their dispersion medium, otherwise wetting of the particles would not occur and no dispersion would be possible to begin with. In fact Bratby (1980), states that many hydrophobic colloids have a very fine, tightly bound layer of water on their surface, which is one or a few molecules thick and this water layer moves with the particle. In this way, there are many inorganic hydrophobic particulates in natural waters which display hydrophilic properties, such as silica and hydrated iron or aluminium oxides (Eckenfelder, 1980).

The other category of colloidal dispersions, the hydrophilic colloids, generally include solutions of biological polymers or macromolecules such as proteins, lipoproteins, lipopolysaccarides, carbohydrates as well as humic acids, viruses and bacteria. They lack a clear phase boundary and are conferred stability by electrical charges arising from the dissociation of inorganic groups on particles such as carboxyl or amine groups (Faust and Aly, 1983). Also, the formation of adherent, thick, layers of orientated water molecules around the surface provide a liquid barrier to particle collisions. Again, the various ionisable side groups of the macromolecules lead to the formation of the water molecule layer surrounding the colloid.

Hydrophilic colloids, when aggregated and removed from their solvent by coagulation, evaporation, or other means, will return to their former colloidal state when re-introduced into their initial dispersion media, that is, hydrophilic colloids are reversible. On the other hand, hydrophobic colloids are irreversible once they have aggregated, they do not return to their initial, colloidal state on simple mixing with the solvent (Dickinson and Stainsby, 1982).

According to Barnes et al (1981), the stable dispersions of hydrophilic colloids are more difficult to destabilise than hydrophobic colloids. It should be noted though, that in water or wastewater, it is likely that both types of colloids will be present or co-exist, so a certain water sample can not be always classified as a hydrophobic or hydrophilic dispersion.

Some colloids have both hydrophobic and hydrophilic segments. An example would be a lipoprotein or lipopolysaccharide, where one segment of the macromolecule contains a lipid, or hydrophobic, component and the other segment contains the hydrophilic component-the carboxyl, hydroxyl and amine groups of the protein or polysaccharide (Bratby 1980).

2.3 COLLOIDAL AND PARTICLE STABILITY

As was stated earlier, colloids and other associated particles are conferred stability due to interfacial forces which arise from the presence of surface charges and the hydration of the surfaces of the colloids. These factors become significant with colloids because they have enormous surface areas to mass ratios. Referring to Table 2.1, it becomes apparent that as a particle decreases in size, its surface area increases dramatically, and so the greater the influence of the surface asssociated phenomena will be (Peters et al, 1974).

According to Barnes et al (1981), colloids have a natural tendency toward coagulating and precipitating from the medium, but are countered by either mutual repulsion of the particles (hydrophobic) or by a strong attraction of the colloids to the aqueous phase (hydrophilic). All particles exert mutual, attractive forces which are effective only at short distances of separation. When the surfaces of these particles are brought close together, these attractive Van der Waals forces predominate, resulting in an overall attraction between the particles, and natural coagulation can take place (Shaw, 1983).

Stability arises when electrostatic repulsion between similarly charged particles is stronger than the natural coagulating forces. The constant repulsion of particles not only prevents the particles from coagulating, but it retards their settlement by keeping the particles in constant Brownian motion (Benefield et al, 1982).

These forces of mutual attraction, mutual repulsion and the attraction of the colloidal surface for the dispersion medium are all dependant on both the structure and type of colloid as well as the chemical and physical characteristics of the dispersion medium.

In order to understand how colloids and other fine particles can be destabilised, we need to understand the concepts of the charges on the surfaces of these colloids and particles, and how they interact in their medium.

Shaw (1983), gives two important effects of the liquid-solid interactions which confer stability. Firstly, there is a tendency for substances to adsorb or concentrate on the surfaces of the colloid. For example, water is adsorbed from the surroundings onto the colloid surface and forms solvation layers on their surfaces, which tend to make the particles repel. Secondly, the surfaces of colloids and other particles tend to acquire and carry electrical charges, the sign and intensity of which will depend on the nature of the surface and of the aqueous medium.

Many organic, including biological, and inorganic colloidal dispersions on both natural waters and wastewaters tend to carry an overall negative primary charge which arise from ions adsorbing on the particle surface and lattice imperfections in the crystal structures of particles (Clark, 1992). Sundstrom and Klei (1979), add that the negative charges also arise from the various ionisable groups on the colloids such as amines, carboxyl and hydroxyl groups.

Generally there are three important and widely held mechanisms which explain the origin of the surface electrical charge, be it positive or negative. These are outlined below.

2.3.1. Isomorphous Replacement or Substitution

This is a common mechanism of surface charge origin for naturally occurring minerals. Under certain geologic conditions, certain atoms in crystalline materials can be replaced by other atoms, For example, in silicates, silicon can be replaced by an aluminium ion which can give rise to an excess of negative charges to the crystalline material, even though the crystal structures do not change (Eilbeck and Mattock, 1987). It is this process which produces the negative charges on the surface of clay particles (Benefield et al, 1982).

2.3.2. Adsorption of Ions

A wide variety of ions or surface-active substances may adsorb onto the surfaces of colloids and related particles. The adsorption may be of two types: if it is weak, non specific and reversible, then the adsorbed ions can be considered as being part of the diffuse layer (discussed in detail in a latter

section). If the adsorption is strong, via specific chemical bonding, then the adsorbed ion is considered to be an integral part of the colloid's surface and probably contributes to the primary surface charge.

According to Shaw (1983), surfaces in aqueous media tend to be more negatively charged than positive because cations are usually more hydrated than anions and so have a greater tendency to reside in the bulk aqueous medium. This "shield of water" forms a barrier to other colloids (Benefield et al, 1982). On the other hand the smaller, less hydrated and more polarising anions have a greater tendency to be specifically bound.

2.3.3 Ionisation or Chemical Reactions of Surface Sites

Many chemical reactions occur on the surfaces of the colloids, which is not surprising given their large surface area and the fact that their surfaces contain many functional groups exposed to the liquid phase. Particles with ionogenic groups such as carboxylic acids and hydroxide groups dissociate in water and produce a surface charge which is pH dependant. Benefield et al (1982), give an example of proteins or amino acids, which acquire charges through ionisation of various functional groups such as carboxyl and amino groups. The ionisation of these groups and the net charge is pH dependant and so particles may exhibit a net positive charge at low pH and a net negative charge at high pH, and a zero charge at an intermediate pH, the isolectric point (Figure 2.3).

Even though colloids have an electrical charge, colloidal dispersions tend to be neutral. Therefore, according to Benefield et al (1982), the charge on the colloids must be counter balanced by ions of opposite charge, which are contained in the dispersing phase. These ions are arranged in a way to constitute the electrical double layer (EDL).

2.4 THE ELECTRICAL DOUBLE LAYER

When particles are dispersed in water, ions with an opposite charge are attracted to the surface of the particles. As these ions concentrate on the surface of the particles, there is an opposing tendency for them to diffuse in the direction of decreasing concentration. This attraction and diffusion tend to
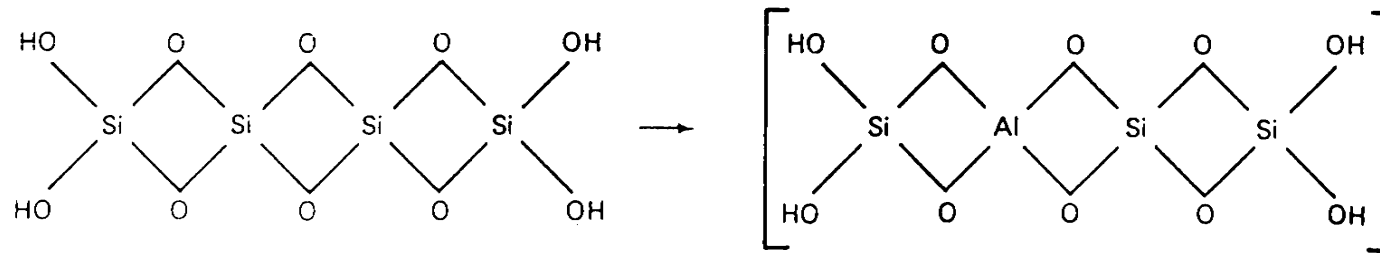
Figure 2.2 Charge acquisition through isomorphic replacement of Al for Si (from Benefield et al, 1982, p194)
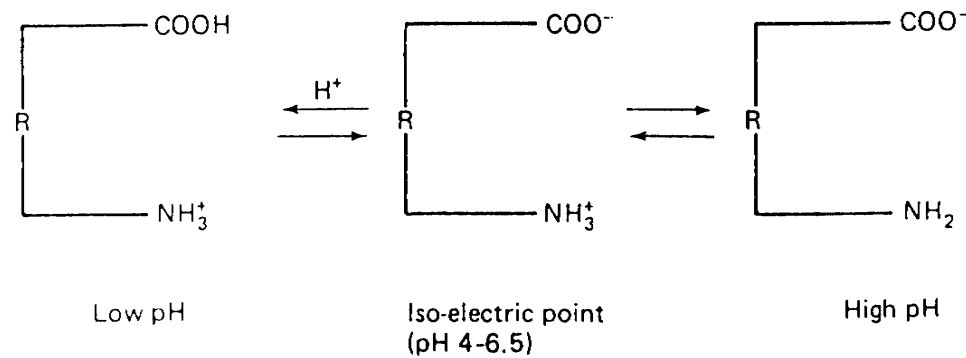
Figure 2.3 Effect of pH on the ionisation of a protein particle (from Benefield et al, 1982, p194)
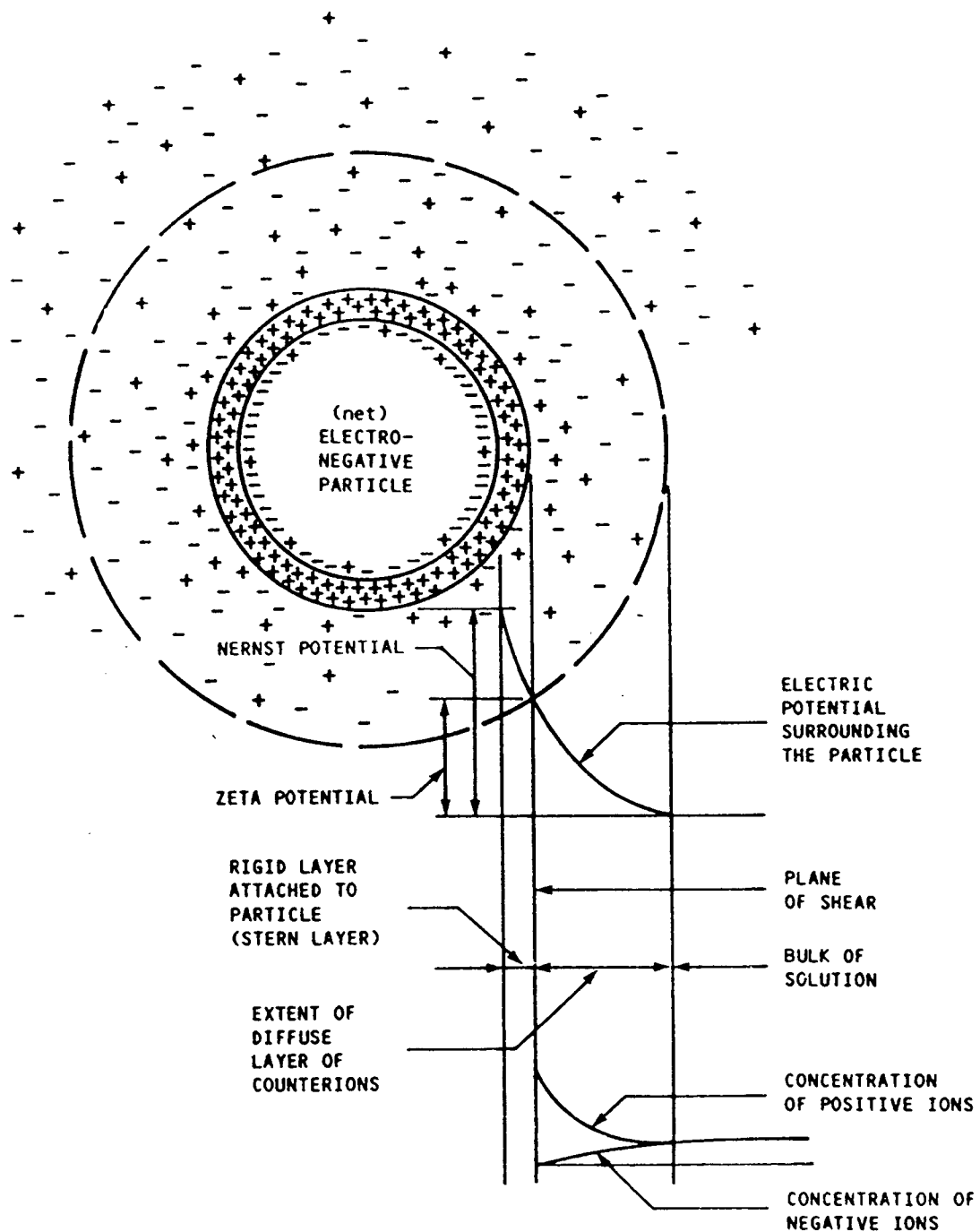
Figure 2.4 The Electrical Double Layer surrounding a colloid particle (from Schowyer, 1986, p216).

produce a diffuse cloud of ions or counterions adjacent to the surface of the particle (Dobias, 1993). As a result of the accumulation of these counterions, the high electrical potential on the surface of the colloid will decrease to zero at a certain distance from the particle surface (Benefield et al, 1982).

The EDL model proposed by Stern and shown schematically in Figure 2.4 contains two major regions: an inner region, the Stern Layer which includes the adsorbed ions and a diffuse region, the Guoy Layer in which the ions are distributed according to the influence of electrical forces and thermal motion (Schwoyer, 1986).

The ions in the Guoy Layer tend to be subject only to non-specific Coulombic forces of attraction or repulsion, but ions in the Stern Layer are under the influence of electrostatic forces and may also be intimately involved in specific chemical interactions with the surface, such as hydrogen bonding and covalent bonding. It is also possible that specific chemical forces can be stronger than electrostatic forces. For example, anions can be adsorbed onto a surface which is already negatively charged (Shaw, 1983).

According to Eilbeck and Mattock (1987), when there is an equivalent number of counterions to the primary charge on the particle surface, then the combination of particles and counterions becomes electrically neutral, so at large distances there are no nett repulsive forces between the particles. If the two particles approach eachother closely enough so that their double layers overlap, then Coulombic repulsion may occur, since their EDLs are of the same sign. This leads to an increase in the electrical potential between the two particles. It is such interactions which represent and energy barrier to the approach of the two particles, and so they will not aggregate (Shaw, 1983).

The concentration and valency of electrolyte in the system determines the energy barrier imposed by the EDLs. Increasing the electrolyte concentration decreases the repulsive electrostatic interactions, which decreases the energy barrier and facilitates effective particle collisions, thus destabilising the system, as attractive Van der Waals forces may be experienced (Barnes and Wilson, 1983).

2.4.1 The Deryagin-Landau and Verwey-Overbeek Theory (DLVO Theory)

The DLVO Theory was developed to quantify particle stability in terms of energy changes, especially in relation to the addition of electrolytes, when particles approach eachother (Dobias 1993). The theory involves making estimations of the energies involved when EDLs are involved in both repulsion and attraction, in terms of interparticle distances. The total interaction energy is determined by the summation of the attractive and repulsive forces (Shaw, 1983). The stability of the colloidal system can then be interpreted or determined by the interaction energy-distance curve, Figure 2.5.

The energy distance curve indicates that repulsion forces predominate at certain distances of separation and Van der Waals forces predominate at small distances of separation. Therefore, if the particles can be brought together closely enough, to overcome the repulsive forces, then the attractive forces will predominate, allowing the particles to attach and subsequently destabilise (Benefield et al, 1982). In order to overcome the repulsive forces, the particles must have enough kinetic energy to overcome the energy hill, as shown in Figure 2.5.

2.4.2 Zeta Potential

According to (Eckenfelder, 1980), the Zeta Potential, commonly used as an indicator of efficiency of coagulation processes, measures the stability of a particle and indicates the potential energy that would be required to penetrate the layer of ions surrounding the particle for destabilisation. The higher the value of the Zeta Potential, the more stable the particle or, the greater the mutual repulsion between particles will be.

The Zeta Potential measures the electrostatic charge on the surfaces of the particles in the liquid, the magnitude being expressed as millivolts (Schwoyer, 1986). One method of estimating the Zeta Potential is by electrophoresis (Baumann et al, 1979). In this method an external electrical force is applied across a suspension whereby particles undergo electrophoretic mobility, migrating toward an electrode of opposite charge. The rate and direction of this mobility, together with the specifically applied voltage detremines the Zeta potential.
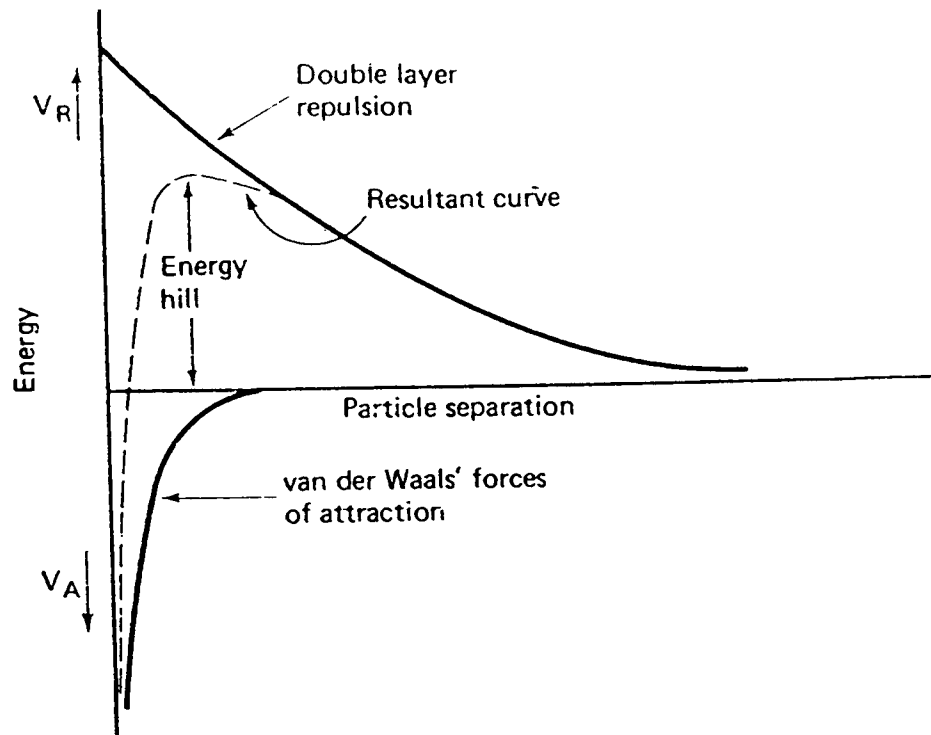
Figure 2.5 Repulsive and attractive energies as a function of particle separation (from Benefield et al, 1982, p197)

2.5 COAGULANTS

Previously, the properties of colloids and related particles were dealt with regarding their surface charges and how these charges confer stability to a suspension. There are a number of mechanisms which can destabilise such suspensions, but these will be discussed in detail in proceeding sections. In this section, we will concentrate on the various coagulants and coagulant aids which are currently used as destabilising agents. Knowledge of these coagulants and of their behaviour in aqueous systems is essential in elucidating destabilisation mechanisms and in ultimately developing an efficient coagulation process protocol. It should be stressed that other than destabilising particulates, a coagulant or coagulant aid also serve to strengthen the flocs formed and hence decrease floc break-up.

According to Adams et al, (1981) and Bourke (1993) there are a number of significant factors that govern effective coagulation. Firstly there is a need to know the relative proportions of soluble and particulate fractions in the water to be treated, as well as the characteristics of these particles. Secondly, an appropriate coagulant needs to be selected, and this must be cost effective, as coagulants may represent about 55% of the total running costs of some plants (Masides et al, 1988). Thirdly, of what standard is the effluent quality once the coagulant has been used?. Fourth, how much sludge is produced and what are the dewatering characteristics?, and fifth, how feasible is coagulant recovery?.

The third and fourth points clearly show that the coagulant of choice has a great bearing on the outcome of the coagulation process as it affects the supernatant quality and the sludge produced. This is why the choice of coagulant is a vital parameter in the coagulation process.

2.5.1 Inorganic or Metal coagulants

According to (Ekenfelder, 1980), metal coagulants need three key properties to be effective: they must be a trivalent cation as this valence has proven to be the most effective in coagulation; they must form insoluble complexes at a near neutral pH range and, ideally, most of the coagulant added to the suspension should precipitate out off solution so that the metal ion does not remain in the effluent; and, the metal ion chosen should be non-toxic.

There are two main classes of metal coagulants currently in use, based on aluminium and ferric salts. On a world-wide scale, mainly aluminium salts, followed by ferric salts are used as cogulants (Alaerts and Van Haute, 1982). These salts include aluminium sulphate, or alum, aluminium chloride and polyaluminium chloride. Ferric salts include ferric sulphate and ferric chloride. Other coagulants include calcium hydroxide (lime) and magnesium carbonate, as well as a wide range of organic polymers.

Aluminium and ferric ions, both trivalent cations, have proven to be popular because they are very effective coagulants. In fact the popularity and widespread use of alum has made it the primary standard in assessing other coagulants (Carnduff, 1976). The chemistries of both the ferric and aluminium salts are comparable in that they both have the ability to form multi-charged polyhydroxy-polymetal complexes in solution, under certain pH levels with enhanced adsorption characteristics (Alaerts and Van Haute, 1982; Bratby, 1980).

Both metals are relatively non-toxic, although aluminium has been the subject of recent debate, due to its possible link with Alzheimer's Disease (Packham and Ratnayaka, 1992; Helfrich et al, 1992; and Selvapathy and Reddy, 1992). Both metals are relatively cheap and readily available. In Australia for instance, alum is traditionally the first choice because it is so relatively available and is produced in most states (Bourke, 1992). These metal salts can be conveniantly stored as either solids or liquids, giving a range of options for treatments with specific needs. These aspects are important as reagent choice is largely governed by economic considerations and conveniance of storage and application.

The general chemistry and physical properties of these coagulants will be presented. All the equations are a general reaction outline as the hydrolysis reactions are believed to be a lot more complex. According to Bratby (1980), destabilisation probably takes place before these reactions reach equilibrium.

This review will concentrate mainly on aluminium sulphate and ferric sulphate, as these were the coagulants chosen in this study. For comprehensive reviews on aluminium and ferric salts refer to Montgomery, 1985; Eilbeck and Mattock, 1987; Bratby, 1980; Benefield et al, 1982; and Tchobanoglous and Burton, 1991)

2.5.2 Aluminium and Ferric Salts

Aluminium and ferric iron coagulants are available in salts of sulphate or chloride and are available in both liquid and solid forms. According to Montgomery (1985), aluminium and ferric ions rarely exist in solution as their trivalent equivalents, preferring to be bound up with various ligands.

The chemistry of both aluminium and ferric iron is very similar and for both, the chemistry is very complex (Montgomery, 1985 and Eilbeck and Mattock, 1987,). When an aluminium or ferric salt is added to an aqueous suspension, it immediately dissociates yielding the trivalent cations Al^{+3} and Fe^{+3} respectively. These cations can react with various ligands in solution, such as hydroxide ions, phosphate ions, and sulphate ions. The metal ions hydrate, forming co-ordination complexes with water, such as $Al(H_2O)_6^{3+}$ and $Fe(H_2O)_6^{3+}$. For conveniance these complexes are often referred to as Al^{3+} and Fe^{3+} respectively.

These complexes undergo a series of stepwise substitutions which are hydrolytic reactions, where the water molecules are replaced by hydroxide ions. These hydroxide ligands can arise either from the dissociation of bound water molecule ligands or by the replacement of water by hydroxide ligands (Bratby, 1980). The pH of the suspension, or the concentration of the hydroxide ions, largely determines the extent to which hydroxide ions are bound to the metal complexes.

With these hydrolytic and substitution reactions, numerous soluble and insoluble products are formed. They include a wide range of mononuclear species such as $Al(OH)^{2+}$ or $Fe(OH)^{2+}$ and polynuclear complexes such as $Fe_8(OH)_{20}^{4+}$ and $Al_8(OH)_{20}^{4+}$. These polymeric aluminium hydroxide and polymeric ferric hydroxide species may be very important in the overall coagulation process (Parthasarathy and Buffle, 1985). Many of these complexes can act as weak acids in water:

$$Al(H_2O)_6^{3+} \rightleftharpoons [\, Al(H_2O)_5OH \,]^{2+} + H^+ \quad (1)$$

The donor capacity of the ligand is not fully bound up to the metal, allowing it to form a co-ordination bond with another metal ion. In this way, the ligand can act as a bridge between two central ions. Metal hydroxy complexes can then

form polynuclear complexes such as:

$$2\{ Fe(H_2O)_5OH \}^{2+} \rightleftharpoons \{ Fe_2(H_2O)_8(OH)_2 \}^{4+} + 2H_2O \qquad (2)$$

These two metal ions are believed to be bound by two hydroxy bridges, as shown below:

$$
\begin{array}{c}
\hspace{4em} H \\
\hspace{4em} O \\
(H_2O)_4Fe \diagdown\diagup Fe(H_2O)_4 \\
\hspace{4em} O \\
\hspace{4em} H
\end{array}
\qquad (3)
$$
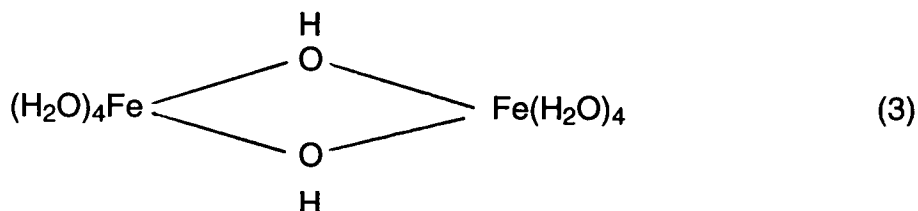
These substitution reactions eventually lead to the formation of metal hydroxide polymers and metal hydroxide precipitates, depending on the concentration of the coagulant used.

According to Qureshi and Malmberg (1985), the pH determines which species predominates at equilibrium, and some species formed at certain pH levels will be more effective than others. For instance, in the pH range of most water and wastewater treatments (pH 6-8) the predominant metal species tends to be insoluble hydroxides such as $Al(OH)_3$. At high pH, this forms $Al(OH)_4^{1-}$ and this leads to a decrease in coagulation efficiency, as the precipitate becomes soluble, which ultimately results in increased residual aluminium in the effluent.

At both low and high pH values, the aluminium hydroxide precipitate becomes very soluble, but at about pH 7, the precipitate is near neutral (and thus least soluble). The aluminium hydroxide precipitate has infact a narrow pH band where it is least soluble; above pH 8.2 it acquires a negative charge and below pH 7.6 the precipitate takes on a positive charge (Eckenfelder, 1980).

Ferric species are more insoluble over a wider pH range. The equilibrium diagrams (Figure 2.6) also show the importance of pH in the control of soluble metal species that will potentially pass through the treatment process. The importance of pH extends to polymeric metal hydroxides as well, not only the
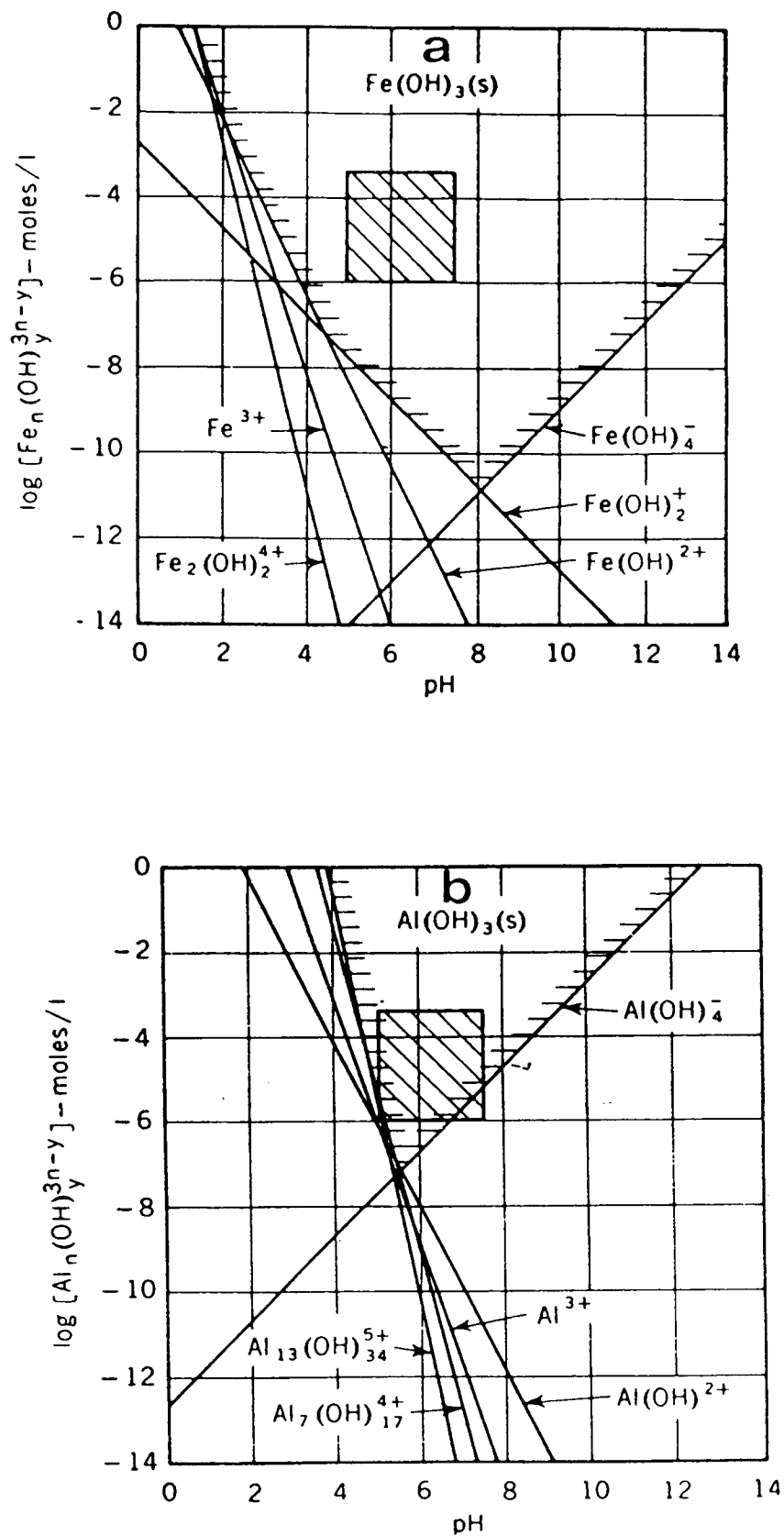
Figure 2.6 Equilibrium diagrams for ferric hydroxide and aluminium hydroxide species at various pH levels (from Benefield et al, 1982)

metal hydroxide precipitates; at the lower to middle end of the pH scale, these products tend to become positively charged and can adsorb very strongly to many negatively charged particles, and hence destabilise them.

The polymerisation action of the metal coagulants is important in destabilisation, as it promotes enhanced adsorption capacity of the metal coagulant species to colloids and associated particles. The hydrolysis reaction sequences of both aluminium and ferric ions is shown in Figure 2.7.

The simplified stoichiometric hydrolysis reactions of aluminium and ferric salts provides useful estimations of coagulant quantities needed:

$$Al^{3+} + 3H_2O \rightleftharpoons Al(OH)_3 + 3H^+ \qquad (4)$$

$$Fe^{3+} + 3H_2O \rightleftharpoons Fe(OH)_3 + 3H^+ \qquad (5)$$
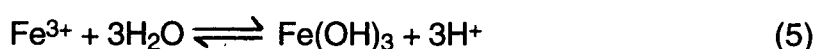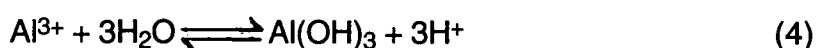
From these equations, one mole of the trivalent cation will yield one mole of the metal hydroxide and three moles of hydrogen ions. The amount of precipitate and acidity formed is not only dependant on the concentration of the metal salt but with the pH, buffering capacity and the overall system chemistry.

According to Smethurst (1979), if there is insufficient alkali the pH will drop substantially due to the formation of sulphuric acid when the sulphates hydrolyse (ferric or aluminium sulphates). This is why alkali is added in the form of soda ash or lime, if the water or wastewater has insufficient alkali or buffering capacity.

2.5.3 Aluminium Sulphate (Alum)

The formula for aluminium sulphate or, alum is:

$$Al_2(SO_4)_3.18H_2O \qquad (6)$$

Assuming the reactions of alum in water go to completion, with the formation of the precipitate aluminium hydroxide, $Al(OH)_3$, then a general equation representing the overall alum reaction in water is shown below:

$$Al_2(SO_4)_3 \cdot 18H_2O + 6OH^- \rightleftharpoons 2Al(OH)_3 + 3SO_4^{2-} + 18H_2O \quad (7)$$
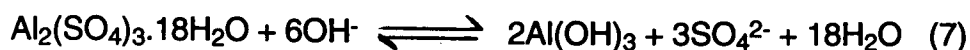
An important function of the aluminium hydroxide precipitate is to improve coagulation reaction kinetics by introducing a larger number of particles into the suspension (Packham and Ratnayaka, 1992). This floc is both voluminous and gelatinous, allowing it to enmesh and adsorb colloids and associated particulates from suspension, this being a major destabilisation mechanism (Culp and Culp, 1971; and Ng et al, 1995).

Lee (1988) sprayed alum on a lake which had received agricultural and urban runoff, in order to minimise eutrophication. The sprayed alum formed both $Al(OH)_3$ and $Al(PO)_3$, both which were very gelatinous and voluminous. As they settled to the bottom of the lake, they removed a substantial level of phosphorus from the water as well as turbidity. As a result algal populations were reduced by 75%.

The use of alum can also have its drawbacks: alum reduces the alkalinity markedly, and it also works in a narrow pH band so its coagulation effectiveness is strongly pH limited. At low temperatures, the ability for alum to remove turbidity is significantly lost (Haarhoff and Cleasby, 1988). Edzwald, Haff and Boak (1977), reported increases in levels of turbidity when using alum as a coagulant, although it is likely to have been a problem due to overdosing or insufficient pH control.

The use of alum recently has led to concerns of the possibility of adverse effects, such as links with Alzheimer's Disease on human health, associated with residual aluminium in drinking water (Packham and Ratnayaka, 1992; Selvapathy and Reddy, 1992). Letterman and Driscoll (1988), feel that greater pH control needs to be exercised in order to minimise residual aluminium in treated effluents or drinking water. Other factors which impact on residual aluminium in treated waters include temperature and complexing ligands in humic acids which occur in natural waters (Van Benschoten et al, 1992).

Helfrich et al, (1992) and Eilbeck and Mattock (1987) include another major problem associated with using alum: that of sludge disposal. Alum flocs create

$$Fe(H_2O)_6^{3+} \rightleftharpoons [Fe(H_2O)_5OH]^{2+} \rightleftharpoons [Fe(H_2O)_4(OH)_2]^+ \rightleftharpoons [Fe(H_2O)_3(OH)_3]_{(s)} \rightleftharpoons [Fe(H_2O)_2(OH)_4]$$

$$Fe_2(OH)_2^{4+}$$

(Other polynuclear species)
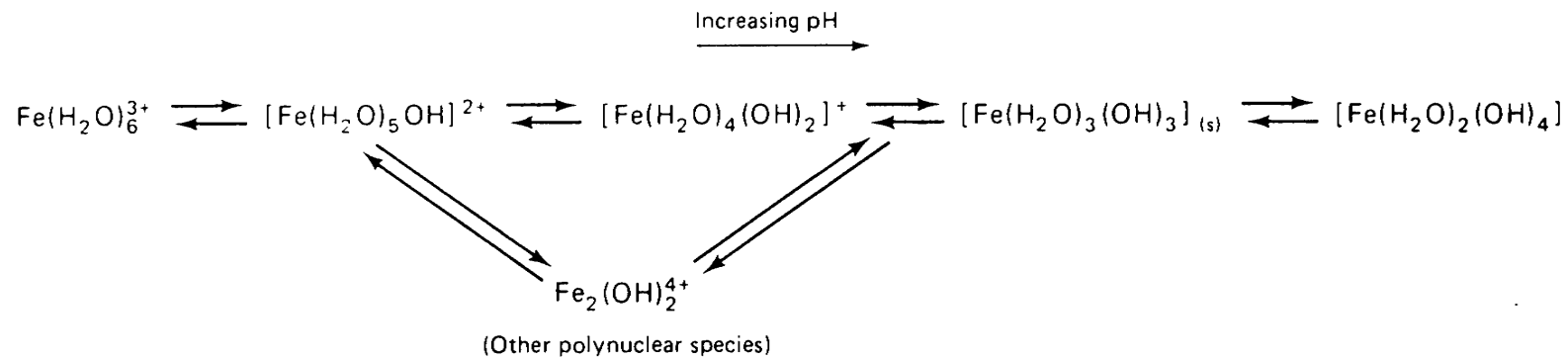
(a) Hydrolysis scheme for iron (III)

Increasing pH →

$$Al(H_2O)_6^{3+} \rightleftharpoons [Al(H_2O)_5OH]^{2+} \rightleftharpoons [Al(H_2O)_3(OH)_3]_{(s)} \rightleftharpoons [Al(H_2O)_2(OH)_4]$$

$$Al_8(OH)_{20}^{4+} \qquad\qquad AlO(OH)_{(s)} \rightleftharpoons Al_2O_{3(s)}$$

(Other polynuclear species)   (Crystalline phases)
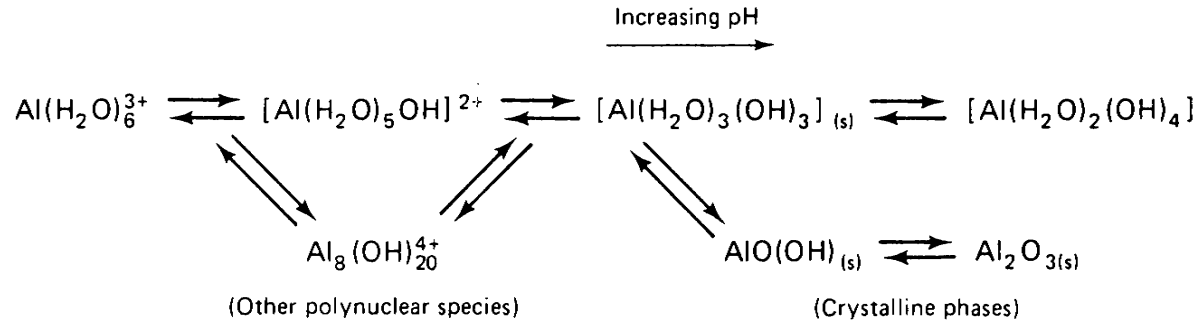
(b) Hydrolysis scheme for aluminum (III)

Figure 2.7 Hydrolysis schemes for iron (III) and aluminium (III)
(from Benefield et al, 1982, p220)

voluminous sludges which can be difficult to dewater. The problem of sludges produced, and their disposal, in the coagulation-flocculation process will be discussed in detail in section 2.11.

The above problems associated with the use of alum has led to the increasing use and popularity of other inorganic substitutes such as polyaluminium chloride and ferric salts, given below.

2.5.4 Polyaluminium Chloride (PAC)

According to Bratby (1980), using PAC provides results likened to that of using alum and a coagulant aid. The formula for PAC, a polynuclear complex of aquoaluminium ions, is given below, Bratby (1980).

$$[Al(OH)_{1.5} (SO_4)_{0.125} Cl_{1.25}]_n \qquad (8)$$

PAC is a generic term for coagulants produced by partial neutralisation of an aluminium and chloride solution by some form of basic solution. This partial neutralisation forms hydroxyaluminium species, and depending on the extent of neutralisation, can be mono, di, or polymeric (Kaeding et al, 1992). Dempsey et al, (1985) treated clay-fulvic acid suspensions with PAC and alum and found PAC produced stronger flocs with increased settling velocities and a decreased sludge volume, compared to the use of alum.

Kaeding et al, (1992), conducted trials at the Anstey Hill Water filtration plant in Adelaide, comparing PAC to alum as primary coagulants. The stronger flocs formed by PAC led to less aluminium residuals in the effluent. The sludge volume produced was greater than that of alum, but on a dry weight basis, PAC produced smaller sludge quantities as it allowed for better dewatering. They also found that PAC could treat waters over a wider pH range, compared to alum. This ability to work over a wider pH range has led to the increasing popularity of PAC as a primary coagulant. The fact that PAC gives stronger flocs which settle out more readily than alum flocs, thus minimising the need for coagulant aids has led to the inceased use of PAC (Packham and Ratnayaka, 1992).

2.5.5 Ferric Salts

According to Bratby (1980) and (Eckenfelder, 1980), all ferric coagulants can be used over the wide pH range of 4-11, compared to that of alums 5-8. The ferric hydroxide floc formed in the reactions of ferric chloride or ferric sulphate in water are very insoluble, even at high pH levels and this is one characteristic that makes ferric iron coagulants preferable to alum.

The generalised reactions of ferric sulphate and ferric chloride in water are given below. In both cases, it is assumed the reactions have gone to completion.

$$Fe(SO_4)_3.8H_2O + 6OH^- \rightleftharpoons 2Fe(OH)_3 + 3SO_4^{2-} + 8H_2O \qquad (9)$$

$$FeCl_3 + 3OH^- \rightleftharpoons Fe(OH)_3 + 3Cl^- \qquad (10)$$
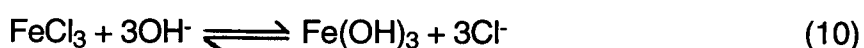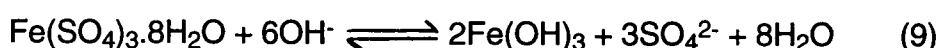
At acid pH, the floc takes on a positive charge and at alkaline pH the floc takes on a negative charge. In the pH range of 6.5-8.0, the floc is a mixture of positive and negative charges (Bratby, 1980).

Ferric salts have been compared to aluminium salts, especially alum in various trials involving mainly raw or drinking water. In most cases the ferric salts were found to be superior. Ferric sulphate, in the commercial solution "Ferriclear" was compared to alum as a primary coagulant in a wastewater treatment plant. Hutchinson, Freeman and Healy (1983), found that ferric sulphate worked over a wider pH range and much less pH control was required, when compared to alum. Even with a substantial drop in pH the plant could still operate when using ferric sulphate. The ferric hydroxide flocs produced a much higher floc density compared to alum floc and the sludge was a lot more cohesive, which subsequently led to increased clarifier capacity.

According to Bourke (1993), ferric salts do not form anionic hydrolysis products until a much higher pH, compared to alum which forms anionic products at a relatively lower pH. Because of faster reaction kinetics, ferric salts hydrolyse much more rapidly than aluminium salts and ferric flocs form faster. Flocs are also heavier than aluminium flocs, and settle much more rapidly. The result is less floc carryover and therefore less coagulant residuals

in the effluents when using ferric salts.

Because of faster reaction kinetics, ferric chloride is more effective at lower temperatures compared to alum. This was found by Helfrich et al, (1992), where at lower temperatures, ferric chloride was more effective than alum and produced much less sludge. Narasiah et al, (1991) on the other hand, found ferric chloride much less effective than alum in comparing their abilities to remove total phosphorus from wastewater effluents.

Ferric coagulants have also been compared to alum in the treatment of drinking water and natural waters. Of interest in natural waters is the presence of humic acids, which have the potential to act as trihalomethane (THM) precursors. This is a major concern to both water and wastewater treatment processes as THM precursors, and hence THMs are potentially carcinogenic (Jodellah and Weber, 1985). Chadik and Amy (1983) conducted trials to coagulate THM precursors from natural waters. Alum was found to be more effective than ferric chloride in reducing the total organic carbon. Haarhoff and Cleasby (1988) found ferric sulphate effectively coagulated natural waters which were low in temperature and had low turbidities. Much less ferric sulphate was required compared to alum, to attain similar removal efficiencies.

2.6 MECHANISMS OF DESTABILISATION USING METAL SALTS

According to Montgomery (1985), there are four major mechanisms of destabilisation by inorganic and organic coagulants:

1. Double layer compression;
2. Electrostatic attraction or charge neutralisation by adsorption;
3. Enmeshment or sweep-floc coagulation; and
4. Interparticle bridging.

Ferric and aluminium ions can destabilise by the two mechanisms of charge neutralisation and enmeshment (Benefield et al, 1982 and Donati, 1992). Interactions of positively charged species such as $Fe(OH)^{2+}$ with the negatively charged colloids or particles brings about destabilisation via charge neutralisation, which relies on destablisation when two surfaces are oppositely charged. Adsorption and charge neutralisation can be obtained when the metal salt is added to water in concentrations less than the solublility

limit of the metal hydroxide (Dentel, 1988).

If the concentration of the aluminium or ferric salt added to water exceeds the solubility of the metal hydroxide precipitate, the hydrolysis products formed are intermediates for the formation of the metal hydroxide precipitate. This precipitate or floc matrix is a voluminous three-dimensional structure which destabilises by entrapment or enmeshment of the particulates into its matrix (Hong-Xiao and Stumm, 1987 b).

Charge neutralisation can also occur during sweep-floc coagulation, as the intermediates generated can adsorb onto the particle surfaces quite well (Benefield et al 1982). The mechanism of sweep-floc can result in the generation of large amounts of wet sludges which can be difficult and expensive to dewater (Schwoyer, 1986).

In both these mechanisms, the pH plays an important role as it controls the charge on the hydrolysis products and the charge on the precipitate of the metal hydroxide. For instance at pH values below the isoelectric point of the metal hydroxide precipitate, the hydrolysis products have a positive charge. For example, below pH 6 for aluminium and below pH 4 for ferric iron, the species formed are positively charged and these can interact with the negatively charged particles and destabilise them by adsorption and charge neutralisation. At their isoelectric points, they form amorphous precipitates and can destabilise by entrapment of particles. Negatively charged species such as $Fe(OH)_4^{1-}$ and $Al(OH)_4^{1-}$, which predominate at higher pH values, are ineffective at destabilising negatively charged particles.

The control of pH is therefore essential to establish optimum conditions for coagulation. This control can be difficult because the ferric and aluminium salts liberate hydrogen when added to water (see Equations 4 & 5) and this acidity reacts with the alkalinity, reducing the pH. If the existing alkalinity or buffering capacity is insufficient, it can be added in the form of soda ash or lime, otherwise the pH may drop out off the optimum pH coagulation range (Swiderska-Broz, 1991).

The concentration of colloids or particulates can also affect the mechanisms and efficiency of coagulation, as well as the concentration of coagulant added. According to Bratby (1980), when the concentration of colloids is low, the

contact between hydrolysis products and surface of the particles is limited. The addition of coagulant results in uneven adsorption, with some particles becoming destabilised, some remaining stable, and others undergoing charge reversal. This problem is mainly due to inadequate dispersion. Charge reversal tends to result from overdosing a system with adsorbable species. The result of charge reversal can sometimes lead to re-stabilisation. Chadik and Amy (1983) found high coagulant doses led to re-stabilisation when coagulating river water solids with alum or ferric chloride.

Higher colloid concentration allows for more oppurtunity to interact and have contact with the coagulant and with other particles, so destabilisation occurs via adsorption and charge neutralisation, but at relatively lower doses of coagulant. Increasing the level of coagulant continues destabilisation by charge neutralisation, but again, charge reversal can occur. When solubility of the metal hydroxide is exceeded, then destabilisation by sweep-floc coagulation occurs, because the high level of colloids serve as a nuclei to enhance precipitate formation.

(Odergaard, 1979; Bratby, 1980; Schowyer, 1986) advance the mechanism of interparticle bridging as another mechanism of colloidal destabilisation. As the hydrolysis of the metal salts progresses, the higher polynuclear species form. These can adsorb to a particle surface and a coagulant bridge forms, which can span across adjacent particles and promote destabilisation via floc formation. It is believed this mechanism dominates for primary particle aggregation in the precipitation of proteins (Fisher and Glatz, 1988).

It becomes apparent that destabilisation with metal coagulants is not exclusively attributed to any one mechanism, but can infact be due to an interplay of all the mechanisms.

These mechanisms can be applied to both hydrophobic and hydrophilic colloids, although the latter do not contain an EDL but are stabilised by hydration layers. Sweep-floc coagulation appears to predominate with hydrophilic colloids or particulates (Bratby, 1980).

The surface charges on hydrophilic colloids is often due to the ionisation of ionogenic groups which include carboxylic groups, aromatic and aliphatic hydroxl groups, amino groups etc. The metal ions can form complexes readily
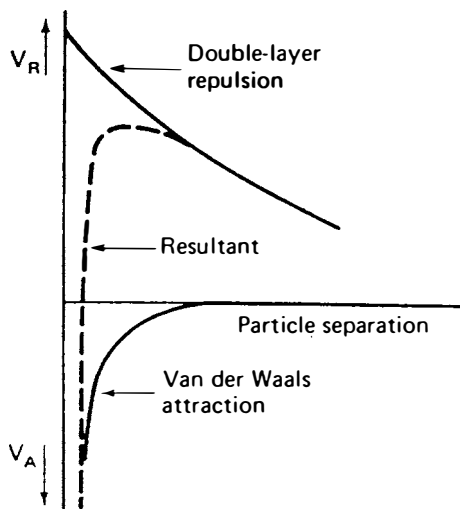
(a) Low electrolyte concentration with normal double-layer thickness. System is stable and agglomeration is imperceptable.
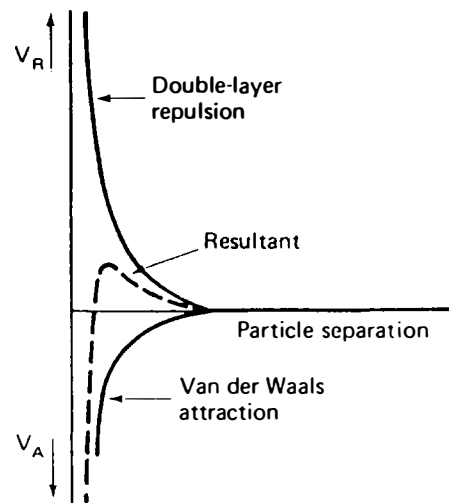
(b) Intermediate electrolyte concentration causes some double-layer compression. Slow agglomeration can occur.
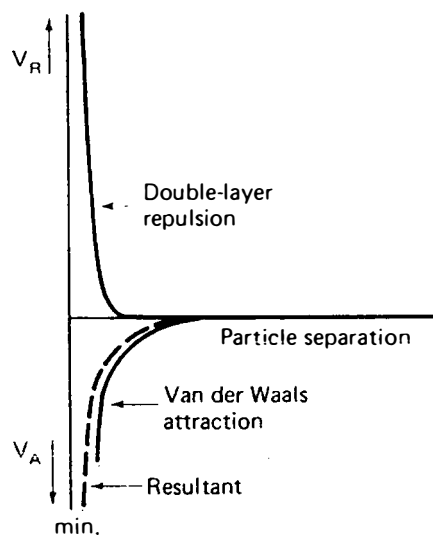
(c) High electrolyte concentration causes severe double-layer compression. Rapid agglomeration can occur.

Figure 2.8 The effect of electrolyte concentration on double layer compression (from Benefield et al, 1982, p 214).

with these ligands, together with the hydroxide groups, so the destabilisation mechanism appears to be due to the precipitation of the hydroxo-metal-ligand complex. The removal of humic acids and microorganisms such as viruses and bacteria with metal salts has been attributed to this mechanism (Bratby, 1980).

Another mechanism of destabilisation is that of double layer compression. The EDL has a quantity of counterions which is sufficient to balance the electrical charge on the particle. The addition of electrolytes increases the ionic strength of the solution which increases the charge in the diffuse layer, and this compresses the EDL, in that its thickness is reduced because less volume of the diffuse layer is required to neutralise the surface charge (Montgomery, 1985).

The compression changes the distribution of the EDL repulsion forces resulting in a decrease in surface potential as electrolyte concentrations increase. This allows Van der Waals forces to predominate and so enhances particle aggregation. Figure 2.8 shows the effect of increased electrolyte concentration on particle stability: when the electrolytes are in low concentration, a high level of energy is needed to overcome repulsion; but high levels of electrolyte compress the EDL and greatly reduces repulsion, allowing agglomeration to occur.

2.7 POLYELECTROLYTES

Polymers or polyelectrolytes are long molecules, usually hydrophilic, with repeating chemical units which have an ionic nature that gives the polymer an electrical charge (Eilbeck and Mattock, 1987).

2.7.1 General Characteristics of Polyelectrolytes

The use of polyelectrolytes as primary coagulants or coagulant aids in water or wastewater treatment has a number of advantages over the use of trivalent cations: in most cases sludge volumes are reduced significantly (Letterman et al,1979); pH adjustment or control is minimised; and metal carryover from the sedimentation basins into the effluent via soluble species or light-flocs is prevented (Benefield et al, 1982).

The two main objectives of using polyelectrolytes is to destabilise stable suspensions of colloids and related particulates and to improve floc forming properties by formation of larger and more shear resistant flocs.

According to Eilbeck and Mattock (1987), polyelectrolytes can be classified by the nature of the functional groups present in the chain, and can be either cationic, anionic or non-ionic.

Polyelectrolytes can be naturally occurring polymers (or derivatives thereof) which are also known as biopolymers, or synthetic polymers based on acrylamide/polyacrylamide. Typical examples of biopolymers, derived from exoskeletons of crustaceans or cellulosic material from plants, includes starches, galactomannans and chitin/chitosan. An advantage of using biopolymers over synthetic polymers is they are virtually toxin free and so are safe for potable water and also in applications where the water or sludge generated can be recycled or used as part of a value added product.

Synthetic polymers have become more widespread in their use compared to the naturally occurring polymers because they are more effective as flocculants due to the fact their chemical and physical characteristics can be controlled during manufacture, thus having the potential to be applied in a tailor made way (Schwoyer, 1986).

According to Schwoyer (1986), the more popular synthetic polymers are composed of polyacrylamide and its various derivatives. Polyacrylamide is synthesised by catalytic polymerisation of the acrylamide monomer. The degree of polymerisation can be controlled by altering reaction conditions. Further chemical modifications yield various functional groups, with either cationic, anionic, or non-ionic charges. Polyacrylamide has a unique structure in that its ionic characteristics can be easily obtained through co-polymerisation or other chemical reactions on its amide functional group (Mishra, 1989). An extensive review covering the processes used to manufacture various polymers can be found in Schwoyer (1986).

One problem in using synthetic polyelectrolytes is their potential toxicity, as many are believed to be genetically toxic (Hasegawa et al, 1991). This may pose a problem, according to Letterman and Pero (1990) in using relatively high concentrations of synthetic polymers in operations where their residue

may end up contaminating value added products, as in the treatment of food wastes. Their use in the treatment of potable water can also pose public health risks. Amy and Chadik (1983) reported that polymers are capable of forming trihalomethanes during the chlorination stages of water treatment, if any polymer residuals remain in the treated water.

According to Clark (1992), cationic polymers neutralise the particle surface charge. The concentration of polymer needed for optimum flocculation is proportional to the number of particles present. Cationic polymers tend to have lower molecular weight than their anionic and non-ionic counterparts. Anionic and non-ionic polymers tend to have very high molecular weights and are mainly used as coagulant aids, usually after the addition of the primary coagulant. They increase the floc size and floc strength.

The mechanisms used for destabilisation are (Eckenfelder, 1980):

cationic: adsorption onto negatively charged species on particle surface;
anionic: replace anionic groups on a particle surface and permit hydrogen bonding between the colloids and the polymer;
non-ionic: Adsord and flocculate by hydrogen bonding between solid surfaces and polar groups in the polymer.

The mechanisms of polymer destabilisation will be discussed in detail in section 2.9.

Besides classifying polymers according to their charge, other factors regarding polymers includes their molecular weight and charge density, also known as percentage hydrolysis (Schwoyer, 1986).

The charge density strongly influences the configuration of the polymer in solution. For a given molecular weight, an increase in the charge density will result with a higher polymer viscosity in solution. An increase in molecular weight will also increase the viscosity of a dissolved polymer. The higher the charge density, then then the higher the number of charged units and when the polymer dissolves in water, it gets stretched by the electrostatic repulsion between the units. It follows that for a given polymer, the higher the number of charged units, then the larger the electrostatic repulsion will be and so the polymer will extend and unfold more.

Generally, polymers with low charge density are tightly coiled in solution, and those with medium charge density (about 15% hydrolysis) are randomly kinked or flexed coils. Polymers with high levels of charge density become fully flexed or extended filamentous rods in solution (Eilbeck and Mattock, 1987).

Bilanovic et al, (1988) found that in solutions of high ionic strength, even polymers with high charge densities lost their extended configuration and became randomly coiled. This was attributed to the fact that the ions in the solution masked the charged groups and this decreased the repulsion forces between them. It is apparent that the ionic strength of the suspension to be treated is an important factor in polymer flocculation efficiency. According to Gadiel (1978), in an acidic pH, a cationic polymer tends to uncoil and extend due to charge repulsion along the chain. Under alkaline conditions, however, the positively charged functional groups become neutralised and the polymer tends to become randomly coiled.

The molecular weight of the polymer can also influence the efficiency of a coagulation/floccualtion system, as it determines the number of configurations which can be acheived on the surface of an adsorbed particle. Lee and Fuller (1985), found that polymers with higher molecular weights were preferentially adsorbed and tended to be adsorbed faster, as they had a higher number of functional units per segment. This aspect of configuration will be discussed further in the section detailing various destabilisation mechanisms.

2.7.2 Preparation and Use of Polyelectrolytes

In the preparation and use of polymers, some information should be kept in mind. Polymers can come either in liquid or solid form. The latter tends to be more economical as liquids occupy large volumes. Also, some liquids become so viscous, they are difficult to handle. For instance, a 1% concentration of a high molecular weight polymer is virtually unpumpable (Schowyer, 1986). An advantage with the lower molecular weight polymers in liquid form, is they are easy to use and dilute, as their viscosity tends to be lower. The solid forms come in granular, flake or bead form, and they need to be made up as liquids. In preparing polymers, they need to be discretely dispersed and stirred to ensure all the particles are wetted and complete dissolution takes place. Care must be taken to avoid high turbulence exposure, as the polymer molecules

are easily ruptured by excessive hydrodynamic forces (Kim, 1988).

Polymer solutions should also be made fresh, as ageing causes a decrease in polymer performance (Carns and Parker 1985), as well as becoming susceptible to microbial degradation (Dentel et al, 1988).

As previously discussed, polymers may be used as primary coagulants or more commonly as coagulant aids. In this latter capacity, they increase the efficiency of flocculation as well as the settling rate of the solids, by increasing the floc density, and this ultimately increases the output of settling tanks (Griffiths, 1964).

A study by Letterman and Pero (1990) suggest using coagulant aids after the primary coagulant has been added, as this allows the primary floc particles to form on the destabilised particles, and time for the flocs to grow to an appreciable size, thereby resulting in lower coagulant aid requirements. Because of their role in creating bigger and stronger flocs, coagulant aids tend to have relatively higher molecular weights than the other polymers. Robinson (1979), trialled a cationic polymer as a primary coagulant replacement for alum-lime treatment at a Kansas water treatment plant and found a large reduction in the level of organic contaminants treated, compared to alum. Carns and Parker (1985) replaced alum with a polymer as a primary coagulant for the treatment of drinking water and found that with the addition of clay as a coagulant aid, excellent removal of turbidity was obtained. The sludge generated was 55% less than the alum sludge.

Harvesting algal biomass from marine water and fresh water was successfully carried out with synthetic cationic polyelectrolytes, as well as the biopolymer chitosan (Bilanovic et al, 1988). Kracman et al, (1989), trialled a low toxicity cationic polymer, LT35, over alum at the Morgan water treatment plant on the Murray River in South Australia. They found that: sludge production and associated costs decreased; pH adjustment was unnecessary; and because the polymer came in a liquid form no capital expenditure was needed as existing equipment could be used. Even though this polyelectrolyte was more expensive than alum, it reduced overall costs of treatment by about 45% making it a viable alternative.

Ho and Tan (1989), treated palm oil mill effluent with both alum and a polymer. When alum was used only, restabilisation tended to occur when the pH dropped out of the narrow range of 5.9-6.0. Restabilisation was evident by the turbid supernatant, and poorly formed flocs which never quite sedimented. The addition of polymer enhanced flocculation and settling rates, and decreased the levels of alum addition by 40%.

Selvapathy and Reddy (1992), trialled drinking water with alum and a cationic polyelectrolyte, Indfloc 238. They found coagulation was most effective if alum was added first, followed by Indfloc 238, as a this sequence produced the lowest residual turbidity. With this sequence, both the flash mix time and flocculation time were reduced and chemical costs were reduced by 35% compared to alum alone.

Even though the above cases are based on drinking water, they serve to illustrate how the use of polymers as either primary coagulants or coagulant aids can successfully replace metal salts. This use of polyelectrolytes is widely applied in the treatment of drinking water and wastewaters, but the literature is scant on the applications toward food industry wastewaters. Later sections, though, will detail cases of the use of metal coagulants, with and without polyelectrolytes in the treatment of food processing wastes.

2.8 CHITOSAN

The use of metal salts as coagulants can have some drawbacks: their effectiveness is highly pH dependant; their flocs tend to be weaker than polymer-particle flocs; residual metals can be sometimes found in the treated effluents and the sludges produced tend to be voluminous. The use of synthetic polymers, whether in combination with metal salt or not, can be highly efffective coagulants or coagulant aids, but the unit costs of polymers tend to be much higher compared to the cost of the metal salts (Kawamura, 1991). Also, synthetic polymers pose the possibility of long range toxic, carcinogenic and mutagenic effects on higher organisms.

Natural polymers, or biopolymers, may be a workable alternative to synthetic polymers. Besides their potential for being equally effective coagulant agents as their synthetic counterpart, most biopolymers have the added advantage of being safe to human and animal health and are biodegradable, abundant,

and a naturally regenerating resource.

One biopolymer in particular, which has been effectively used by various researchers, and which has also been used in this thesis project is chitosan, a derivative of chitin. Muzzarelli (1973) reports that chitosan presents very little toxicity even at 18g/Kg body weight.

According to Muzzarelli (1973), invertabrate and shellfish contain chitin as a structural component. Gooday (1990) adds that chitin serves as a carbohydrate and nitrogen reserve. Currently, the bulk of this material is discharged to sea as wastes from marine canning operations.

2.8.1 General Chitosan Chemistry

Chitin is insoluble in water and organic solvents, but under harsh heat and alkaline conditions, deacetylation of chitin can yield chitosan (Pelletier et al, 1990). An excellent review can be found in Muzzarelli (1973) and Li et al, (1992).

Chitosan, the polysaccharide obtained by deacetylating chitin, is a high molecular weight, linear, cationic, non-toxic, biodegradable polymer. It contains β-1,4 linkages of D-glucosamine and N-acetylglucosamines. According to Kawamura (1991) and Li et al (1992), chitosan is unique in that its polyamine character makes it insoluble in water, but soluble in most organic acids at pH below 6. Even though chitosan is insoluble in water when it is used, it is dissolved in a acetic acid (Bough 1975 a).

Chitosan is widely used in producing high value products such as cosmetics, drug carriers feed additives, pharmaceuticals, etc. An extensive review of the various uses and functions of chitosan can be found in Li et al (1992).

One of the very useful properties of chitosan is that of chelation or coagulation, as it can selectively bind desired materials such as fats, proteins, cholesterol, metal ions, dyes, etc. This is due to the high density of amino groups on the polymer chain, which can interact with negatively charged substances.

Dyestuffs and pigments are released into effluents from textile processing factories. In the past, they were usually treated by activated sludge processes, but biological oxidation tended to be slow. As a result, a high level of these contaminants were discharged into rivers, resulting in toxicity to aquatic life. The removal of these pigments and dyes from the effluent would not only minimise their toxicity, but allow for product and cost recovery. Yoshida et al, (1991) used chitosan fibres and activated carbon fibres in an attempt to adsorb and therefore recover these pigments from textile factory effluents. Both were effective at removing or adsorbing the dyes and pigments, but chitosan could bind alot more of the pigments, compared to the activated carbon, and costing showed chitosan was cheaper to use. Smith et al, (1993) used chitosan and chitin to successfuly decolourise effluents from textile processing factories. The performance of chitosan was higher than chitin, and this was attributed to chitosan containg higher numbers of sorption sites than chitin.

Thome et al, (1993) used chitosan and cross-linked derivatives (gluteraldehyde cross-linked to chitosan and benzoquinone cross-linked to chitosan) to remove polychlorinated biphenyl (PCB) from water. The derivatives of chitosan were chemical modifications of the amino groups on the polymer. Both chitosan and its derivatives exhibited high rates of PCB adsorption, and it was concluded that the amino groups on the chitosan chain were essential for chitosan to preserve its adsorptive properties for PCBs.

Lee et al, (1992) compared alum to chitosan in trials involving floatation as a method for algal harvesting. The algal solids concentration was monitored as a performance parameter. They found that chitosan produced the highest algal sludge levels, at about 20g/L, compared to alum which contributed to only 11.3g/L algal solids.

Whey, the liquid remaining after casein is removed from milk is composed of fat lactose and the valuable proteins b-lactoglobulin and serum albumin. The extraction and recovery of these proteins from the waste streams is considered a viable step. Kennedy et al, (1994) aimed at testing the effectiveness of chitosan as a coagulant in the removal of proteins from whey. They found that at about 200 mg/L, chitosan could coagulate and extract most of the protein from the whey, at neutral pH.

Bough (1975a) treated vegetable canning wastes with chitosan as well as other synthetic polyelectrolytes and found chitosan was an effective coagulating agent, without the aid of other polyelectrolytes or inorganic salts.

When treating egg-breaking effluents in order to reclaim the components for addition to animal feed, Bough (1975b) again found that chitosan was a superior polymer compared to a range of synthetic polyelectrolytes tested, although the addition of an anionic polymer as a flocculation aid after coagulation with chitosan proved very effective.

Poultry processing wastewaters were treated by dissolved air floatation and chitosan to recover coagulated by-products (Bough et al, 1975). Compared to synthetic polyelectrolytes, chitosan was able to outperform them in its ability to coagulate and produce the highest yields of protein and fat in the dried sludge. The resultant effluent was acceptable for recycling into the poultry slaughterhouse.

One case where chitosan failed as an effective coaguant was in the treatment of swine wastewaters. Gao et al, (1993) treated swine wastewaters with a combination of biological and physico-chemical treatments. Using BOD, COD and suspended solids as parameters of coagulation efficiency, lime ferric chloride, alum, chitosan and a synthetic polymer were used. Chitosan failed in treating low, medium and high strength wastewaters.

Biopolymers can also be used to clean certain wastewaters such as those containing coal fines and heavy metals, as well as toxic organics such as polyaromatic hydrocarbons. Bacteria can secrete extracellular polymers which could be used in the uptake of heavy metals (Rudd et al, 1984) but most biopolymers used are made commercially on a large scale, and used in the same way synthetic polymers are used.

Venkatadri et al, (1989) successfuly trialled a biopolymer, a-emulsan, in flocculating iron pyrite, silica and kaolin from coal fines. Jang et al, (1991) worked on using various biopolymers as adsorbents for hazardous materials. They found that biopolymers were feasible as they had excellent selectivity for certain metals. The capacity of these polymers to bind to specific metals is dependant on a number of factors which include the size and valence of the

metal ion; the morphology of the metal binding groups, or metal binding cavities on the biopolymer; the pH, temperature and ionic strength of the solution; and the charge density and chain structure of the polymer.

Deans and Dixon (1992) also conducted work with metal ion uptake using biopolymers. They found that chitosan removed high levels of copper ions and chitin was able to remove high levels of lead. Findon et al (1993) also used chitosan to successfully adsorb and remove copper from solution.

2.9 MECHANISMS OF DESTABILISATION USING POLYELECTROLYTES

According to Bratby (1980); Odergaard (1979) and Ho and Tan (1989), there are two main mechanisms of destabilisation by polyelectrolytes and these mechanisms either work together or one may predominate over another:

1. The bridging model mechanism, which proposes that polyelectrolytes will attach to particles at one or more sites, forming a large aggregate of polymer-particle complexes, and

2. The electrostatic patch model, in which an ionic polymer with a sign opposite to the particles adsorbs and thereby decreases the potential energy of repulsion between adjacent colloids. A summary of the characteristics of the mechanisms can be found in Table 2.2.

2.9.1 The Bridging Mechanism

The bridging mechanism has several stages:-
1. The dispersion of the polymer in suspension;
2. Adsorption of polymer at the solid-liquid interface;
3. Compression or settling down of the adsorbed polymer; and
4. Collisions of adjacent polymer coated particles to form bridges and thereby increasingly larger flocs.

Polyelectrolytes with high molecular weights tend to be viscous in solution and have low diffusion rates. The adsorption of polymers onto a surface of a particle tends to be irreversible, so the polymer must be dispersed throughout the suspension as quickly and efficiently as possible, normally done in the rapid mix phase. The adsorption of polymer onto the surface of a particle is a very fast process: as soon as the polymer has diffused through the solid-liquid

interface, adsorption begins, whereby functional groups attach to the surface. As time proceeds, along with Brownian motion and induced flocculation/turbulence, the polymer chain becomes successively attached along its length.

Hawkes (1970) and Ghosh et al, (1985) found that rapid mixing and its control, was a crucial stage, when using polymers, as it served more to just disperse the polymer. Their work found a significant amount of flocculation takes place between the point of polymer addition and the end of the rapid mix phase. The intensity of the rapid mix should be controlled. According to Kim (1988), polymer molecules are easily disrupted by excessive hydrodynamic forces, especially once they have been extended. When an extended polymer chain is broken, it loses its ability to serve effectively as a bridge between particles. Gadiel (1978) adds that the polymer should be dosed through multiple dosing points to acheive homogeneous dispersion. Also, adding a polyelectrolyte solution in a very dilute form where practicable could be an advantage.

If the polymer is of opposite charge to the particles, attachment may be by coulombic forces of attraction. Polymers and particles which have like charges tend to get attached by ion-exchange, hydrogen bonding or Van der Waals forces of attraction (Benefield et al, 1982).

As the chain successively attaches to the particle, eventually the polymer tail (the end of the chain) will extend into the bulk solution, as well as a number of loops. These loops and tails, according to Ghosh et al, (1985) form the bridges between colloidal particles, by adsorbing onto available sites of other particles. This polymer conformation on the surface of the particle is the basis for the bridging mechanism of flocculation. If the extended loops and tails fail to come into contact with another particle, then no bridging occurs. They may then fold back and attach to other sites on the same surface, and this, according to Benefield et al (1982) can cause restabilisation.

According to Bratby (1980), the optimum rate of flocculation will occur when approximately 50% of the particle surface is covered with polymer. At this level, there will probably be enough surface area not coated so that the polymer already attached to one suspended solid particle surface can form a bridge to an available space on another particle surface. When there is less

than 50% coverage, it is not optimum for bridge formation, and over 50% coverage leaves insufficient free surface sites for bridging and restabilisation may occur.

Dobias (1993) reports that if a high concentration of polymer is used, then all the particles will get coated by a relatively dense layer of polymer. When two such particles approach one another, the interpenetration of these polymer layers leads to steric stabilisation. The optimum dose of polymer, therefore becomes proportional to the concentration of particulates to be treated.

In order for attachment to occur, the loops and chains of the attached polymer need to be able to overlap the diffuse double layers of the neighbouring particles. According to Eilbeck and Mattock (1987), if the polymer chain or loop can not reach over the EDL of the approaching particle, Coulombic repulsions between the diffuse layers will not allow the second particle to approach within the capture distance of the loops or tails of the polymer to the first particle. This can be seen in Figure 2.9. Therefore, polymers with higher molecular weights have an advantage because their larger size increases the potential for bridging, as the loops and chains will extend further into solution (Benefield et al, 1982). This is why bridging is more efficient with anionic polymers, as they tend to be the higher molecular weight polyelectrolytes (Eckenfelder, 1980).

The size of the loops is also influenced by flexibility of the segments of the loop. A relatively inflexible polymer chain will tend to be attached by only a few segments, giving rise to shorter or flatter loops. On the other hand, a relatively flexible chain will give rise to larger loops (Shaw, 1983).

The adsorption mechanisms depend on both the chemical characteristics of the polymer and the particle surface, and includes factors such as hydrogen bonding, Van der Waals forces, ionic bonding and electrostatic interactions between the ionised polymer and the charged particle surface (Leu and Ghosh, 1988). Solution properties such as pH and ionic strength may also affect polymer adsorption/configuration in both the solution and at the interface. According to Mishra (1989), ionic strength may be important in polymer bridging, especially when the polymer and particle have like charges.

Certain cations such as Ca^{2+} may complex with functional groups on the polymer and aid adsorption of an anionic polymer to the colloid by a cation

bridging effect. Added electrolytes may also compress the EDL, thus decreasing the repulsion between particles and the polymer segments, enhancing the bridging effect as particles approach each other more closely, before extensive compression of the polymer segments occur on the surface (Eilbeck and Mattock, 1987). Montgomery, (1985) stresses, though, that if the ionic strength is too high, bridging can be impeded because the polymer loses its ability to extend and takes on a random coil configuration.

As was stated earlier, if a polymer chain has functional groups with like charges, then repulsion between these groups will cause them to maximise their separation and hence extend the conformation of the polymer. Clearly then, the configuration of a polyelectrolyte in solution and at the solid-liquid interface can be controlled by the nature of the functional groups and the charge density, or, the concentration of the functional groups on the polymer. As the charge density increases, so does the repulsion between the adjacent segments, and so the more extended is the polymer chain for a given molecular weight, and so further the loops will extend into solution and more effective will be the bridging mechanism. If the charge density is too high, then the repulsion between the functional groups will be so great, that it will effectively retard adsorption onto particle surfaces (Schwoyer, 1986).

As the polymer segments progressively adsorb on the colloidal or particle surface, the chains become compressed to the surface. This allows the bridging to become stronger and hence more effective, allowing stronger flocs to form.

From the above section, it becomes evident that the configuration of the adsorbed polymer, and ultimately the efficacy of the flocculation process (where polyelectrolytes are concerned), depends on an interplay of various factors: the molecular weight of the polymer, structure, flexibility, charge density, sign of charge, interaction energies between the polymer and the particle, the chemical nature and the physical spacing of the adsorption sites and the competition between the polymer and the other adsorbing molecules in solution. A scheme of the various reactions of colloids with polymers can be seen in Figure 2.10.

2.9.2 The Electrostatic Patch Mechanism

These above factors are also applicable for the electrostatic patch mechanism of particulate destabilisation. According to Goossens and Luner (1976), the electrostatic patch mechanism was developed to clarify cetain mechanisms of polymer flocculation, in systems where strong electrostatic attraction between the polymers and particles exists.

The model is applied for dispersions with particles carrying surface charges of opposite sign to the polymer. This model is applied more to cationic polymers in negative colloidal dispersions, but it can include anionic polyelectolytes applied to dispersions destabilised with metal coagulants, that is, as flocculant aids to particle-metal hydrolysis products, which may be positively charged (Bratby, 1980).

In this model, the partial neutralisation of the particle takes place and flocculation occurs at the attraction between negative and positive sites on different particles. Ghosh et al, (1985), explain that the cationic polymer, when adsorbed onto the negative surface, forms small patches of high density positive charges, surrounded by a sea of lower density negative charges. Here, there are no remainder chains or loops which extend in the bulk solution phase, but complete adsorption of the polymer onto the particle surface takes place.

A charge mosaic is formed, with alternating regions of positive and negative charges, Figure 2.11. When the positive charges bond to negative sites of neighbouring colloids, flocculation then occurs. According to Benefield et al (1982), the effectiveness of this mechanism depends on the ability of the polymer to form uneven charge distributions on the particle surface.

Montgomery (1985), explains that this mechanism is similar to the charge neutralisation mechanisms for the metal salts explained in an earlier section: when the cationic polymer attaches to the particle in a
patch-like pattern, it decreases the electrostatic repulsion, so particles may attach to eachother following induced collisions.

PARAMETER	INFLUENCE OF INDICATED PARAMETER ACCORDING TO MODEL	
	BRIDGING	ELECTROSTATIC PATCH MODEL
MOLECULAR WEIGHT OF POLYELECTROLYTE	The higher the M.W., the more effective the bridging. Upper limit dictated by dissolution	In strict accordance with model M.W should have no effect. Improved performance evident in some cases with higher M.W. due to onset of bridging
CHARGE DENSITY, OR PERCENTAGE HYDROLYSIS	The higher the charge density, the more effective is bridging due to larger loops. Upper limit dictated by electrostatic repulsion between polyelectrolyte and surface of particle, where respective charges similar, during adsorption	The higher the charge density, the more pronounced will be the charge mosaic and the more effective will be destabilization
POLYELECTROLYTE OF SIMILAR CHARGE TO PARTICLE SURFACE	Destabilization by bridging mechanism	Electrostatic patch mechanism inoperative
POLYELECTROLYTE OF OPPOSITE CHARGE TO PARTICLE SURFACE	Destabilization by bridging mechanism possible	Destabilization by electrostatic patch mechanism possible
NON-IONIC POLYELECTROLYTE	Destabilization by bridging mechanism	Electrostatic patch mechanism inoperative
AMPHOLYTIC POLYELECTROLYTE	Destabilization by bridging mechanism possible	Destabilization by electrostatic patch mechanism possible
EFFECTIVE SURFACE CHARGE AT OPTIMUM CONDITIONS FOR DESTABILIZATION	Usually not zero	Not necessarily zero
EFFECTIVE SURFACE CHARGE WITH EXCESS POLYELECTROLYTE ADSORPTION	Possible reversal of charge by excessive adsorption. Destabilization still possible	As for bridging
POLYELECTROLYTE CONCENTRATION	Optimum destabilization when surface site coverage $\theta = 0.5$	Optimum destabilization not limited by residual adsorption sites. However θ probably < 0.5
EXCESSIVE POLYELECTROLYTE ADDITION	At excessive dosages particles restabilized by complete site coverage or charge reversal	At excessive dosages possible restabilization from charge reversal
MIXING	Important for polyelectrolyte to be adsorbed evenly on particles. Mixing should be short and vigorous at the time of polyelectrolyte addition. If mixing too violent or for too long a period desorption and/or rearrangement of adsorbed chains could give rise to restabilization	Essentially as for bridging
PARTICLE CONCENTRATION	Bridging most effective the higher the particle concentration. At low concentrations a longer time is available for compression	At high concentrations, because of kinetic effects, bridging mechanism likely. At low concentrations ($\sim 10^4/l$) electrostatic patch mechanism possible
IONIC STRENGTH	High ionic strength aids adsorption by reducing electrostatic repulsion in the case of poly-electrolytes and particles of opposite charge, and reducing size of polyelectrolyte coil thus permitting adsorption of more segments; aids destabilization by reducing electrostatic repulsion between particles of like charge; can impede bridging by constricting polyelectrolyte loops	High ionic strength probably aids destabilization by reducing repulsive interaction energy between polyelectrolyte adsorbed particles

Table 2.2 Summary of the characteristics of the mechanisms of destabilisation with polyelectrolytes (from Bratby, 1980, p 148)
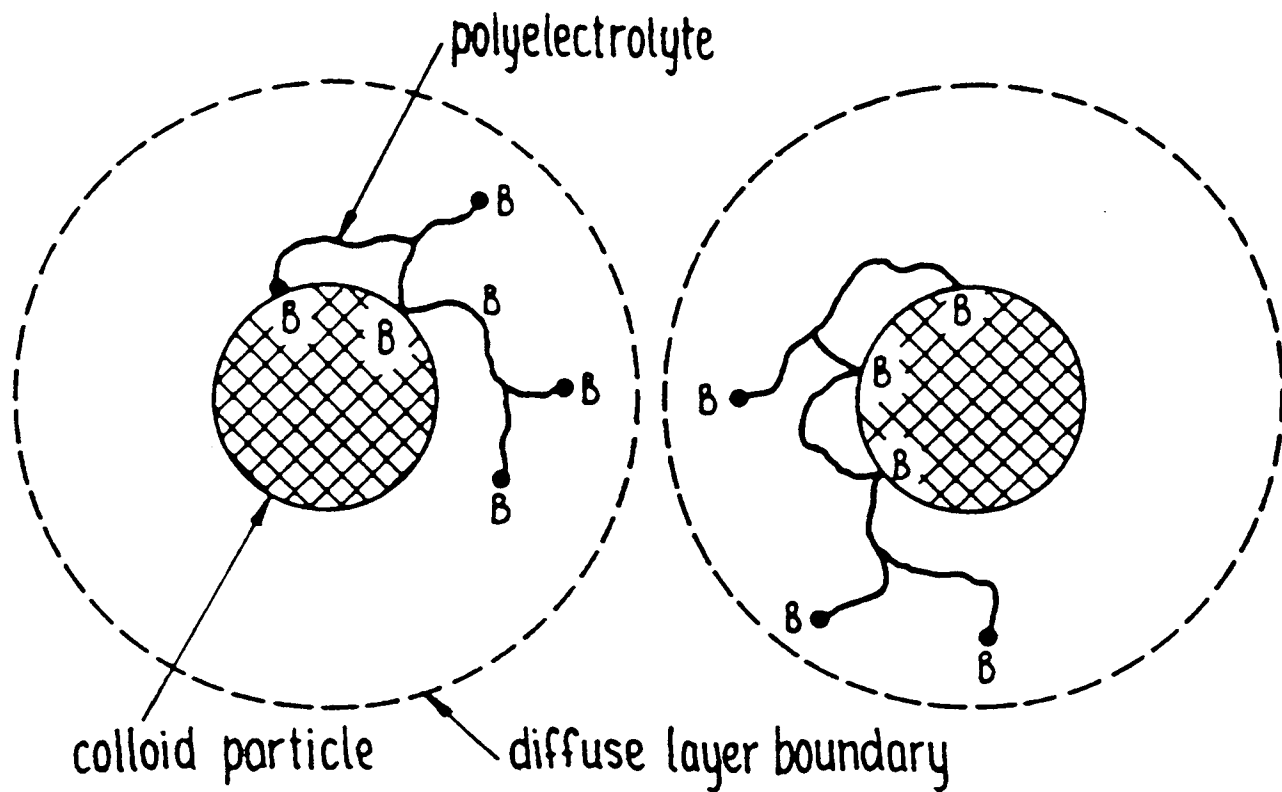
Figure 2.9 Representation of the failure of adsorbed polyelectrolytes to bridge across colloid particles because binding sites are inadequate to extend beyond the double layer
(from Eilbeck and Mattock, 1987, p248)
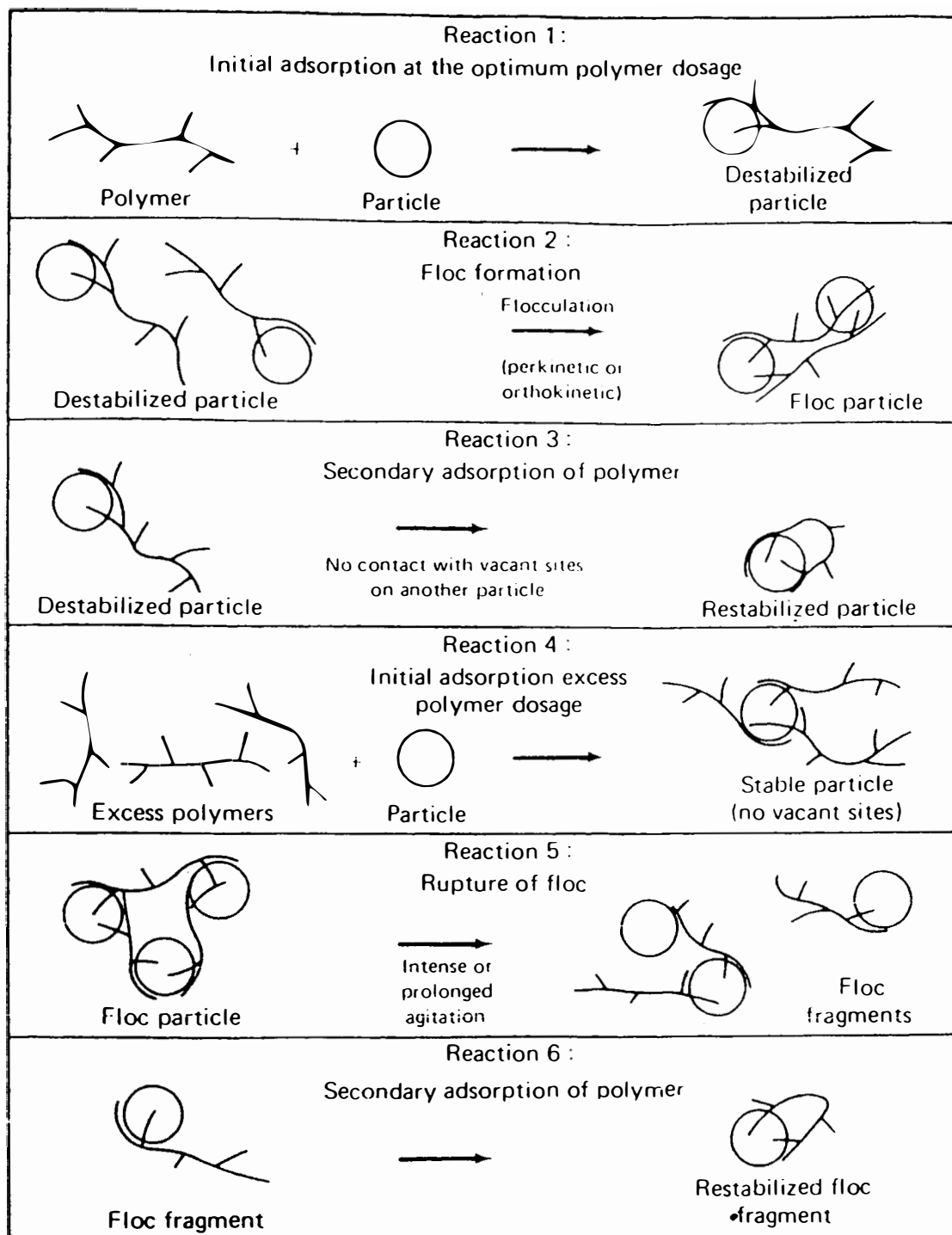
Figure 2.10 Schematic of reactions between colloidal particles and polyelectrolytes (from Benefield et al, 1982, p 218)
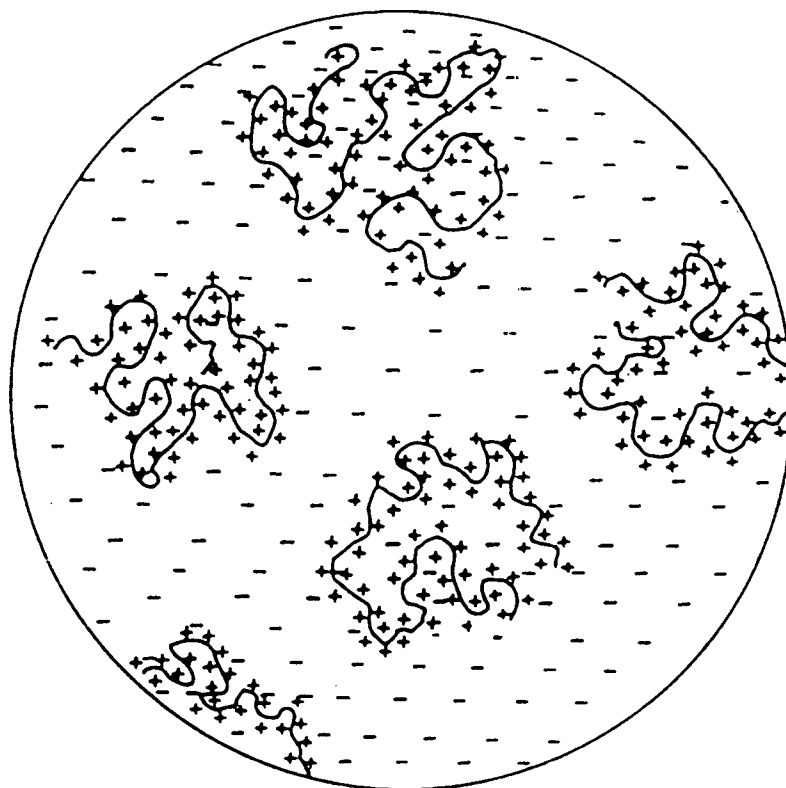
Figure 2.11 Possible arrangement of adsorbed polycations on a particle with low negative surface charge density (from Bratby, 1980)

2.10 OTHER USES OF POLYELECTROLYTES

Polyelectrolytes are also used as filter conditioning agents to treat potable water in filtration processes, particularly direct filtration. Here they function as a type of adhesive which aids in attaching suspended particles and other particulates firmly to the grains of the filter medium (Oulman et al, 1979).

Polymers are widely used as sludge conditioning agents, as they are effective in dewatering and producing tougher flocs (Bratby, 1980). A sewage treatment plant in Blackburn England, conditioned its sludge with polymers and found it was able to dewater a lot more effectively, producing a higher throughput per filter pressing. Immediate transport cost savings were made (Paine and Thompson, 1983).

2.11 SLUDGE

The problem of sludge generation is becoming an increasingly important issue because the growing emphasis on environmental protection has led to increased restrictions on waste disposal practices. This has forced industries to move toward minimising wastes and maximising recovery of useful products from these wastes. Not only will this become a long-term cost effective operation, but there will also be an increase in water re-use. This is important in many areas of Western Australia which tend to have shortages of available water (Gale and Carpenter, 1989).

According to Bhamidimarri (1991), technologies for sludge disposal and/or treatment must be developed which take into account the socio-economic and environmental conditions. One problem of using physico-chemical methods to treat wastewater is the large volumes of sludge produced making it expensive to dispose of (Green and Sokol, 1985). The volumes of sludge to be processed largely govern the capital costs of the sludge treatment processes (Hilson, 1971) and reducing or dewatering even a small volume of water in the sludge offers great economic advantages, such as reductions in costs of handling and transportation (Kayode and Gregory, 1988).

Ferric and aluminium salts have traditionally been used as conditioning or dewatering agents, although organic polymers are fast becoming the primary choice of conditioning agents for sludge dewatering operations (Crawford,

1990). According to Tiravanti et al, (1985) the selection of polymer and the control of its optimum dosage is important since the sludge conditioning costs can be up to about 25% of the total sludge treatment costs.

Sludges result from coagulation with metal salts, in particular alum sludges, and tend to be bulky and gelatinous, which is why polymers are used to flocculate wastes as their sludges are higher in solids, more compact, and dewater relatively easily. In treating wastewater from a pulp and paper manufacturer, alum produced about 23 metric tons of sludge per day, compared to 18 metric tons produced by using a polyamine polymer. Sludge solids also increased from about 22% to 28% (Smith and Malloy, 1990).

Alum sludges have two main characteristics which makes them difficult to handle: they have a very high water to solids ratio, which results in large volumes of sludge, and they are very difficult to dewater. The disposal of alum sludges to low velocity streams or quiescent water bodies has been ruled out, according to a Disposal Committee Report (1987), as sludge blankets may form over the sediments. This can lead to many adverse environmental effects including the onset of anaerobic conditions, decreasing the pH and increasing the solubilisation of certain metals from the sediment and sludge residue.

Grabarek and Krug (1987) sought alternatives to the disposal of alum sludges. Landfill disposal was ruled out as high levels of dewatering were required which in turn would require high levels of preconditioning and this could prove costly in the long run. Soil applications were considered viable because trials showed there was little aluminium toxicity to the plants in either alum-soil mixtures or alum-sludge-soil mixtures. Smith and Malloy (1990) applied alum sludges produced from a pulp and paper mill waste to crops and no problems with the crop were reported.

Masschelein et al, (1985), looked at the feasibility of recycling aluminium from alum sludges generated from a wastewater treatment plant (actually, aluminium hydroxide sludges) in order to decrease the costs associated with conventional sludge handling. They found that alkaline treatment of the sludge allowed for about 80% recovery of the aluminium. This use of the recovered aluminium decreased the expenses of chemicals as well as the sludge solids. Because alum sludges have an amphoteric nature, it is possible to recover the

alum via acidic reactions (with sulphuric acid) or basic reactions, with caustic soda (Masides et al, 1988).

Bishop et al, (1987) compared the coagulation efficiencies of recovered and re-used alum to commercial alum. They also tested the dewaterability of the sludge remaining after alum recovery. Alum was recovered by reacting the sludge with sulphuric acid. They found that the remaining sludge was less voluminous and could be easily dewatered. Two major costs were also realised: less alum was purchased and the size of the sludge handling facilities could be decreased. Comparing the two types of alum for coagulation efficiency, the commercial preparation was only slightly better, as the recovered alum led to slightly higher TOC and turbidity residuals.

Schultz et al, (1989) used waste alum sludge from a water treatment plant to precipitate phosphate at a wastewater treatment plant. When the sludge was reacted with sulphuric acid, about 80% of the alum was recovered, and the volume of the sludge dropped to 20% of the original volume. It was found that the recovered aluminium was as effective as commercial aluminium in removing phosphates.

Once the sludges (alum or otherwise) have been conditioned for dewatering, other mechanical methods such as vacuum filtration or continuous belt filter presses which are used to dewater the sludges (Paschke, 1980). There are also natural methods of sludge dewatering used in countries such as Scandanavia, where sand beds are used for the freeze-thawing of sludge in winter, and as a sludge drying bed in summer (Marklund, 1990).

Once the sludge has been dewatered to its maximum capacity it needs to be disposed of. Earlier in this section, some ideas were given for alum sludge disposal. This thesis does not intend to give much scope to this area, suffice to point out that novel uses of sludge are being found. Grau (1991) states that sludge produced by treating textile wastewaters was used as an additive in cement production!

2.12 RAPID MIXING AND FLOCCULATION

According to Hudson and Wolfner (1967) and Adams, Ford and Eckenfelder (1981), effective hydraulics, effective mixing and effective flocculation are of

critical importance because the overall coagulation process effectiveness is very dependant upon them. It becomes apparent that effective mixing ultimately determines how effective the coagulation-flocculation process becomes. The intensity and duration of agitation are two major criteria which govern the processes of mixing and flocculation.

Coagulants on their own cannot cause efficient destabilisation unless they are properly distributed and can come into maximum contact with colloids. Likewise, colloids, once destabilised, need to be given the opportunity to come into sufficient contact with eachother in order to aggregate. It is for these two factors, efficient coagulant dispersion and particle transport, that rapid mixing processes and flocculation processes have been designed and developed. These process units are crucial to optimum coagulation-flocculation and so need to be properly designed and operated.

2.12.1 Rapid Mixing

The rapid mixing unit process is there to provide complete and thorough dispersion of the coagulant with the water to be treated. The mixing needs to be as efficient as possible, because it is here that destabilisation reactions occur and where primary floc particles are formed. These particles formed markedy influence the subsequent flocculation kinetics.

According to Eckenfelder (1980), destabilisation reactions occur extremely rapidly-within microseconds. Bratby (1980) explains that mononuclear complexes such as $AlOH^{2+}$ and $FeOH^{2+}$ can form in the order of 10^{-10} seconds while polynuclear complexes can form anywhere between 10^{-2} second and 1 second. The adsorption of these species onto particulate surfaces can be in the order of 10^{-10} seconds. This is why proper dispersion of the coagulant during the rapid mix phase is crucial, so that the hydrolysis products have the opportunity to come into contact with the particles. Work done by some researchers investigating the effects of rapid mixing found that if rapid mixing was excluded, or greatly diminished, very poor floc formation resulted and the flocculation processes became very inefficient (Francois and Van Haute, 1984).

Proper design of rapid mixing units therefore, is essential as they optimise these processes, which can ultmately result in a decrease in coagulant

demand and improve aggregation in subsequent flocculation. Hudson (1981) recommends that the coagulant be added at the agitator blade and not the surface, as a significant level of the metal salt will hydrolyse immediately, and if on the surface of the liquid, will not have the oppurtunity to come into sufficient contact with the particles. Poor mixing can also lead to wasted coagulants because it gives rise to high dosages and uncertain pHs in localised regions of the bulk liquid (Qureshi and Malmberg, 1985).

Reviews of the various rapid mixing units most commonly installed and used, as well as the evaluations of their design and performance considerations, can be found in Bratby (1980), Benefield et al (1982), Montgomery, (1985), Hudson (1981) and Bhatia and Cheremisinoff (1979). Rapid mixing units included are back-mix reactors, partial back-mix reactors and plug flow reactors.

The velocity gradient is a measure of the mixing intensity in mixing and flocculation procedures (Hudson 1981), and is calculated taking into account the impeller rotational speed, the net torque of the liquid in rotation or mixing, the volume of the water and the kinematic viscosity of the liquid (Stump and Novak, 1979). According to Bratby (1980), a high velocity gradient alone is not sufficient guarantee that efficient rapid mixing is achieved.

The scale and intensity of the turbulence at the point of coagulant addition is very significant. Very high velocity gradients and prolonged agitation may give rise to primary floc break-up. Sylvester and Toure (1978), aimed at evaluating the effect of prolonged shearing or agitation of polymer solutions on their flocculating ability. They studied the coagulating behaviour of various polymers on the clay kaolin. They found that shearing caused marked decreases in polymer viscosity, demonstrating polymer degradation via chain disruption. Increasing the duration of shearing led to decreasing viscosities. Increasing the shearing also led to reduced flocculation effectiveness, with a 50% drop in efficiency of turbidity removal, for most of the polymers. Keys and Hogg (1979) also found that by increasing the mixing intensity, floc rupture and poor sedimentation rates resulted. Floc rupture also occurred by prolonged agitation.

Francois and Van Haute (1984) studied the influence of the duration of rapid mixing on a flocculation process by investigating the characteristics of the

hydroxide flocs formed. They found there was a critical time frame for rapid mixing, and if the mixing continued beyond this critical time, it led to a disturbance of the floc growth and subsequent breakdown in floc sizes.

Bratby (1980) recommends a dilute coagulant or polymer solution be used as it tends to disperse more rapidly and uniformly than concentrated solutions. Solutions which are too concentrated may result in poor water quality after flocculation due to poor solid-liquid separation.

Keys and Hogg (1979) found that polymers were best used dilute, as this gave overall larger flocs, as long as the mixing was sufficient to disperse the polymer. Stratton (1983) also found that the polymer feed concentration was best kept as low as practicable. To ensure uniform distribution, Stratton found that two or more injection points should be added. Multiple point addition is often recommended when dosing with polymers, as it probably assists in the bridging mechanisms of destabilisation by retaining loops for a sufficient period.

The point of polymer or metal salt addition, whether it be into a certain area of the rapid mix, or in certain sequence of chemical addtions, can also have a strong bearing on floc formation. Lewis (1980) for instance, found that if a polymer solution was injected after the primary coagulant, sufficient time spacing was required to allow the metal salts to hydrolyse fully and to perform the initial coagulation and formation of micro-flocs prior to the introduction of the polymer. Stratton (1983), using polymers in a papermaking process, found it was essential to inject the polymer feed solution into the region where mixing was maximised.

Other factors which determine the effectiveness of the rapid mix phase include the rate of polymer addition or coagulant addition. Keys and Hogg (1979) found that polymer added by a slow continuous rate yielded bigger and stronger flocs and higher turbidity removal.

Fisher and Glatz (1988) recovered lysozyme from egg white using a continuous precipitation process with polyacrylic acid. They found agitation levels influenced the precipitation process, and the rapid dispersion of the polymer was important in the formation of protein-polymer complexes. Agitation also affected the shearing rate of the polymer and this in turn

affected the polymer interactions with the proteins and ultimately, floc size. The length and intensity of agitation also affected the dosage requirements. A fine balance was needed with impeller speeds: slow speeds led to incompete mixing, while high speeds led to floc disrupture, polymer shear and denatured albumin. The authors also found that optimum flocculation was achieved with incremental or continuous polymer addition.

The adjustement of pH is another important factor. According to Bratby (1980), the pH should be adjusted before coagulant addition, or in some cases at the same point of coagulant addition but not after, since destabilisation reactions which commence at the point of coagulant addition tend to be irreversible.

The change over from the rapid mix phase to the flocculation phase is a step which needs to be done carefully. Sharp declines in turbulence, or velocity gradients, during the phase change over can shear the flocs. Tapered changes in velocity gradients are required, when changing from rapid mix to slow mix, so that the gradual decrease in velocity gradient will leave the flocs intact, and allow them to grow bigger and stronger to practicable limits.

2.12.2 Flocculation

The rapid mix stage is where destabilisation occurs and where primary pin-point flocs, or micro-flocs form and commence growth. It is then necessary to induce these primary floc particles to approach each other closely enough to make contact with each other and to promote agglomeration which is the very aim of flocculation. The only way for appreciable contact between the particles to be promoted is to induce shear motion in the liquid. This agitation allows the colliding particles to produce larger flocs, which tend to settle out more readily (Eckenfelder, 1980). The degree or extent of flocculation is governed by both the velocity gradient and the period of mixing. These two factors can largely influence the extent of particle aggregation and the extent of partcle breakdown.

According to Benefield et al (1982) flocculation includes two mechanisms of contact between the particles. The two mechanisms are bulk liquid motion (fluid shear) and differential settling (particle sedimentation). Both these mechanisms are termed orthokinetic flocculation. With this type of flocculation, mechanical mixing is needed to accelarate particle aggregation. By

maintaining velocity gradients within the bulk liquid, it provides particles the oppurtunity to collide and agglomerate. Sanks (1978) Bratby (1980) and Montgomery (1985) give mathematical models which describe the perikinetic and orthokinetic flocculation of colloidal particles.

The velocity gradients in the flocculation stage must be high enough to maintain the particle contacts needed to promote and enhance aggregation. High velocity gradients may also shear the flocs apart. This is because the local shear stresses exceed the binding forces of the aggregates, causing them to spilt or erode (Montgomery 1985). According to Schowyer (1986) flocs produced by organic polyelectrolytes tend to be stronger than those produced by metal salts and tend to withstand higher shearing forces/stresses.

It becomes apparent, then, that the effectiveness of the flocculation process is a combination of the velocity gradients or the hydrodynamic forces and the retention times of the destabilised coloids. These hydrodynamic forces affect the final size distributions of the flocs, their density and strength. Flocculation facilities should therfore be designed so as to maximise contact between the particles, while minimising their shearing.

Flocculation basins employ various methods to induce velocity gradients, the popular of which are the mechanically mixed tanks. Methods include the use of baffled chambers, diffused air, spiral flow, reciprocating blades and rotating blades. Bratby (1980), Montgomery (1985), and Hudson (1981) provide extensive reviews and evaluations on the various methods used for flocculation facilities.

Because of difficulties arising when designing the transfer of a flocculated suspension to a sedimentation unit, such as floc break-up due to the shear forces generated, one solution is to combine flocculation and sedimentation into the one unit. Such units include Solids Contact Blanket Clarifiers, and an extensive review and survey of these units can be found in Montgomery (1985) and Hudson (1983).

2.13 THE JAR TEST

As was just seen in the preceeding sections, for any type of water treated, the coagulation-flocculation process is dependant on a large number of variables:

* the coagulant type, dosage and feed concentration,
* the flocculant/coagulant aid type, dosage and feed concentration,
* the sequence of chemical additions and the time lag between dosing points,
* the initial and final pH,
* the intensity and duration of mixing during the rapid mix stages and the flocculation stages,
* the type of rapid mix device,
* the flocculator geometry.

A testing procedure is required which can sequentially maintain most of these parameters constant, while the optimum value of a selected parameter is determined. This allows the assessment of the optimum conditions for coagulation-flocculation.

The jar test is a laboratory procedure which allows for a rapid means of assessing the effects of these above-mentioned parameters (Culp and Culp, 1971). According to Brink et al, (1988), Langelier introduced the conventional jar test in the early 1920s. The general principle of the jar test is to reproduce, as closely as possible, existing or anticipated conditions of a treatment plant or process. In this way, it is a scaled down attempt of a treatment plant, or potential treatment plant. The jar test allows us to determine what effect changing a certain variable will have on the overall process, if all the other variables are held constant, and at levels which are representative of a full-scale process.

The jar test is favourable because it can simulate rapid mixing, flocculation and sedimentation all in the one unit. The operation of the jar test is also flexible, allowing for many variations to be carried out.

The jar test is essentially a series of identical water samples (usually 4 or 6), all of equal volumes, placed in a rack of stirring paddles which are driven by a motor. The paddles can be rotated at a variable speed to mix the contents of the beakers which are placed under the stirrer. In this way, the simultaneous

treatment and observation of a number of samples under identical mixing conditions can be observed. The rotational speed of the stirrers is variable, and this is required as coagulant addition occurs at the rapid mix or fast speed stage, and the flocculation stage is with slow speed.

Because the jar test apparatus is such a simple device which can assess many variables, it has become the most widely used method employed to evaluate the coagulation-flocculation process (Dentel, Resta, Shetty and Bober, 1988).

Prior to evaluating all the parameters of interest, the water sample needs to be characterised in order to determine which components will require most consideration for treatment. Also important is the quality of the final effluent.

According to Baumann et al, (1979) and Hudson (1981), if the jar test is performed under controlled conditions, very useful information can be derived which can aid engineers in designing new treatment plants or revise existing ones or allow for the full-scale coagulation-flocculation process to be optimised.

The following variables need to be controlled carefully when performing jar tests in order to assess which variable are relatively important:

* the temperature of the jar contents,
* the pH of the sample/jar contents,
* coagulant solution strength,
* coagulant dosage quantity,
* method of coagulant addition (including feed rate),
* flocculant/coagulant aid dosage and solution strength,
* method and feed rate of flocculant/coagulant aid addition,
* sequence and timing of addition of the reactants,
* the duration and intensity of the rapid mix phase,
* the duration and intensity of flocculation,
* sample withdrawl method (for analysis)
* the equipment used for jar testing, and
* the laboratory analysis required, in order to assess the degree of clarification.

According to Lai et al, (1975), the velocity gradient (G) or mixing intensity is a very important variable in the jar testing procedure, as it can determine the effectivness of the destabilisation of the particles and hence their removal. This is why it is essential that standardised procedures be used in the jar tests, especially with the mixing devices and the dimensions of the jars and beakers. Keys and Hogg (1979) found that slightly changing the geometry of the mixing tank led to significant changes in the intensity and frequencies of turbulent eddies, and therefore the overall velocity gradient. They also found that slight changes to the dimensions of the mixing device led to significant changes in fluid dynamic behaviour.

Since destabilisation reactions occur at such extremely rapid rates (see Section 2.12) proper dispersion of the coagulant for maximum contact with the particles is essential. This is significant when using organic polymers as poor dispersion may result in adhesion to the container walls, resulting in substantial loss of the polymer. Ideally the jar testing unit should incorporate a system of applying the coagulant dose at a fixed point, located at or close to the hub of the impeller during the brief rapid mix period.

According to Bhatia and Cheremisinoff (1979), the jar test experiments should be designed to vary the intensity of mixing and mixing time, in order to reach the optimum velocity gradient. Griffith and Williams (1972) conducted a range of tests to evaluate certain aspects of the jar test. They treated drinking water in Phoenix, Arizona. They found that rapid mixing time optimum was 5 seconds (their tests varied from 5-60 seconds) and anything over 5 seconds did not improve turbidity removal. Flocculation time was found to be optimum at 30-35 minutes. Flocculation under 20 minutes and over 40 minutes resulted in very poor turbidity removal. They also found that if the optimum alum dosage was used dilute, it actually improved the overall coagulation process as well.

According to Schwoyer (1986), if the jar tests are carefully planned and executed, they may allow us to :

* determine the types of coagulant and/or flocculant that will effectively remove supended solids from water or wastewater,
* establish the effective concentration ranges of the coagulants and flocculants,

* establish the optimum dosages of the coagulants,

* establish order and time of coagulant/chemical additions,

* establish optimum rapid mix time and intensity as well as flocculation time and intensity,

* estimate the treated effluent quality,

* estimate the settling rates of the flocculated particles, and

* estimate the sludge volume produced as a result of each treatment parameter variation.

According to Bratby (1980), the first aspect of testing toward the optimisation of coagulation-flocculation for a given water sample is to determine the appropriate primary coagulant, its dose, and its optimum pH. Adams et al (1981) feel that once variables such as optimum coagulant dose and other operating conditions are determined, then the other design parameters such as settling properties and sludge production should be determined.

The rate of the coagulant or chemical addition is very important, as is their sequence of addition. They should also be added in a controlled flow rate, so as to maximise destabilisation and hence flocculation. Flocculants/coagulant aids are normally added one minute into the flocculation stage, that is, one minute after the rapid mix stage, but this can be varied to determine the optimum time of addition (Hudson, 1981). Vik et al, (1985) looked at trialling various sequences of coagulant (alum) and alkalinity (sodium hydroxide) to samples of river water in an attempt to coagulate humic substances. They found that adding the alkalinity before or after alum addition yielded silmilar results. When alum and sodium hydroxide were added simultaneously, poor results were obtained, probably due to the immediate formation of $Al(OH)_3$, and therefore not allowing time for alum to become properly hydrolysed.

Since one of the aims of jar testing is to simulate as nearly as possible the plant conditions (existing or anticipated), the jar tests should be conducted to simulate mixing intensities. Hudson (1981) feels that full turbulence needs to be attained and maintained in the jars during the rapid mix, and this is not necessarily obtained by very high shear motion of the liquids. This tends to be the case with using cylindrical jars or beakers. Incorporation of stators may provide full turbulence with very low speeds. Using square jars may also produce higher velocity gradients.

Even though beakers or cylindrical jars have been a conveniant and therefore popular choice as containers, square jars have also been used. Finch and Smith (1986) and Brink et al (1988) found that square jars increased the turbulence during mixing with minimal bulk rotation of the water. The corners of the square jars were found to prevent this rotational flow, typical in beakers. Dentel et al (1988) compared both cylindrical and square jars in regard to polymer coagulation and flocculation. When the same polymer levels were added to both jars, the former yielded higher suspended solids removals, suggesting that the optimum polymer dose (or coagulant dose) may be overpredicted when using a cylindrical jar. These workers also found that at a given speed, square jars were able to attain a higher velocity gradient and therefore more efficient mixing compared to beakers. It was also easier to drill in the sides of square jars to add sampling ports.

2.13.1 Flocs

The effectiveness of the coagulation-flocculation method is ultimately reflected in the level of particles removed. Floc growth is therefore important as flocs incorporate the particulate matter into their matrix, thus removing the particles from the bulk of the solution. As the flocs grow, they become heavier and eventually settle. It is important to control and optimise flocculation, as optimum fluid motion will induce maximum floc growth and hence particle removal.

The floc size distribution, floc density and floc shape are important characteristics which influence the settling velocity of the flocs and this influences the clarification rate of the sludge (Reed and Mery, 1986). According to Dobias (1993) various methods exist for measuring the average size or size distribution of aggregates and it includes microscopy, photography, individual particle sensors, sedimentation rates and light scattering, as well as direct observation.

2.14 APPLICATIONS OF COAGULATION - FLOCCULATION

The unit process of coagulation-flocculation is widely used in the treatment of various wastewaters. The aim of this section is only to give a very brief overview of the various waste streams which can be treated by coagulation-flocculation.

A comprehensive survey/review can be found in Eilbeck and Mattock (1987).

Sewage treatment works incorporate coagulation as a major stage in the process chain, as coagulation caters for overloaded biological treatment plants and is also helpful in cases where there are specific requirements such as the control of outgoing phosphorus levels (Diamadopoulos and Benedek, 1984). Bell et al, (1983) also used coagulation-flocculation as a means of successfully reducing the phosphorus levels, while Booker et al, (1991) used the magnetite process, a slight variation of coagulation, to successfully treat sewage.

Coagulation is also used extensively in water treatment. Carnduff (1976) replaced alum with ferric chloride to treat a water supply for Toronto. Anderson et al, (1983) used the magnetite process to treat natural waters, with very good removals of colour and turbidity. Amy and Chadick (1983) successfully removed high levels of total organic carbon from treated waters, in an attempt to remove trihalomethane precursors, and Brink et al, (1988) evaluated a range of coagulants to remove turbidity and colour from low turbidity waters.

Grau (1991) was able to successfully treat textile and wool wastes with coagulation, as was Nicolaou and Hadjivassilis (1992). In both cases, colour, which is typically produced from the dyes in textile processing, was also removed.

The treatment of pulp and paper wastes also relies heavily on the use of coagulation-flocculation. Ackel (1988) and Smith and Malloy (1990) used coagulation to treat pulp and paper wastes, in order to remove high levels of colour and suspended solids.

The electroplating industry, as well as other metal finishing facilities uses coagulation to remove heavy metals from waste streams. The bulk of the literature indicates the use of biopolymers is increasing, regarding the coagulation and adsorption of heavy metals. Jang et al (1991) used various biopolymer gels to recover copper and cobalt ions. Deans and Dixon (1992) also used biopolymer gels to recover copper and lead, while Findon et al, (1993) recovered copper using chitosan.

2.14.1 The Application of Coagulation-Flocculation in the Treatment of Food Industry wastes

The bulk of the liquid effluent from most food processing industries is biodegradable, which explains why biological treatment methods are commonly used. Since there is always the possibility of recovering by-products from the effluent, coagulation-flocculation becomes an attractive alternative.

If the recovered sludge is to be used as animal or human food products, the use of synthetic polymers and metal salts becomes restricted in both the quantity and the types that can be used, although further processing of the sludge may reduce them. The use of biopolymers is therefore considered a viable alternative to synthetic polymers, as the former are non-toxic and biodegradable.

Bough (1974) successfully treated effluents from a leafy green cannery factory with a range of synthetic polyelectrolytes. Vegetable canning wastes (Bough 1975a) and egg-breaking wastes Bough (1975b) were also treated with synthetic polyelectrolytes and the biopolymer chitosan. Very high removals of suspended solids and turbidity were recorded with chitosan. Poultry slaughter-house processing wastewaters were also treated with chitosan, again with excellent removals of suspended solids (Bough, Shewfelt and Salter, 1975).

Karim and Sistrunk (1985) treated peeled potato wastewater successfully with ferric chloride and anionic polymers, reporting very large reductions in the total suspended solids and COD. When steam peeled potato wastewater was treated, high levels of total suspended solid were removed, but the COD remained high, possibly due to the formation of soluble starch products by high temperature steaming.

Moore et al, (1987) treated snap bean and dry bean wastewaters with two cationic polymers, a synthetic polymer (Floculite 250) and chitosan. Coagulation with both proved effective, but less chitosan was required than the synthetic polymer.

Fisher and Glatz (1988a, 1988b) coagulated proteins from egg albumin using a synthetic polymer. Rusten and Sandberg (1991) used coagulation as a means of pretreating yeast industry wastes. For both effluent types, alum and ferric chloride were able to remove significant levels of suspended solids.

2.15 THE TREATMENT OF STARCH MANUFACTURING EFFLUENTS

2.15.1 Introduction

Of the many and varied food processing industries, the starch industry in particular generates huge volumes of wastewater which contains high levels of both soluble and colloidal impurities (Radley, 1976a). This results in high total solids and high BOD/COD loadings. According to Zeevalkink and Jans (1986), in the majority of food processing industries, the volume of wastewater produced can often be controlled and reduced by taking preventative measures in the production process itself.

2.15.2 The Importance of Starch

As a natural organic polymer starch is found abundantly in nature and this is due to its function, which is the principal food storage in most forms of green leafed plants (Rehwald, 1926). Starch is commonly laid down in the form of minute cells or granules and is located in the seeds, fruits, stems and roots, and serves the plants as food for the energy required during the dormant and germinating stages (Curtis, 1983). Whilst starch is also a principal energy source for humans, it has found increasing applications in a wide variety of food and non-food industries. As a result, the starch industry has, and is, expanding at a rapid rate.

The non-food industries utilise starch in a number of areas which include adhesives, explosives, laundry, medicine (tablet mixtures), printing (printing pastes), and textiles. The function of starch in the food industry is also wide and varied: it is used in bread products for adhesion and moisture retention; as a binder in meat products; as a gelling and moulding agent in gums and jellies; as a thickener in pie fillings, soups and baby foods; and as a dusting agent in bread and gum (Marotta, 1966).

2.15.3 The Structure of Starch

In its pure isolated state, starch is a white amorphous solid which is insoluble in cold water (Radley 1976a). When starch is produced in plants it is laid down in minute granules or cells. The starch molecule is arranged within the granules in a highly organised and radial fashion (Knight, 1969).

Despite the very diverse sources of starch, once purified, it is chemically identical. It is a polymer of anhydroglucose units linked together through alphaglucosidic bonds (Curtis, 1983). As illustrated in Figure 2.12, there are two major types of starch polymers, amylose and amylopectin. Amylose is the linear polymer, with each chain containing between 200-4000 monomer units, and each of these units having two secondary and one primary hydroxyl group. Because amylose has an affinity for large molecules containing both hydrophobic and hydrophilic groups such as fatty acids and fatty alcohols, it would be expected that the amylose hydroxyl groups would impart hydrophilic properties to the amylose polymer when dispersed in water, due to their affinity for moisture and dispersibility, but in water, amylose behaves hydrophobically (Harbitz, 1983). This is because the linear polymers tend to attract to each other and align themselves by hydrogen bonding through the hydroxyl groups on the neighbouring polymers. This reduces the affinity for water, forcing the amylose polymers to come out of solution and form precipitates (Radley, 1976b).

Amylopectin is similar to amylose in that it has a predominance of a-1,4 linked anhydroglucose units, but with every fifteenth unit, there is a branched point, as seen in Figure 2.12. Amylopectin is a much larger molecule. It has up to 1,450,000 glucose units and its highly branched structure interferes with molecular mobility and prevents molecules from approaching the polymer closely (Marotta, 1966).

Wheat starch is composed of both amylose and amylopectin at 30% and 70% respectively. Besides starch, another major component of the wheat grain is protein. This protein can be divided into two major fractions: the soluble and insoluble. The former can be obtained by extraction of the flour or by manipulating dough with water. The insoluble fraction, known as gluten, is very elastic and rubbery in the hydrated state (Knight, 1969).
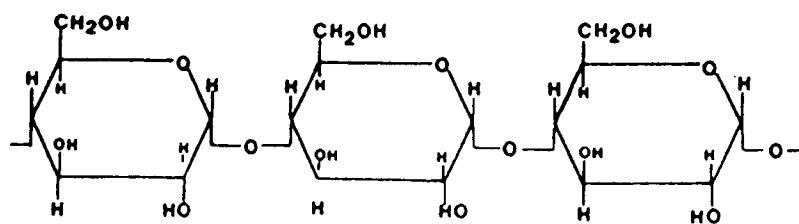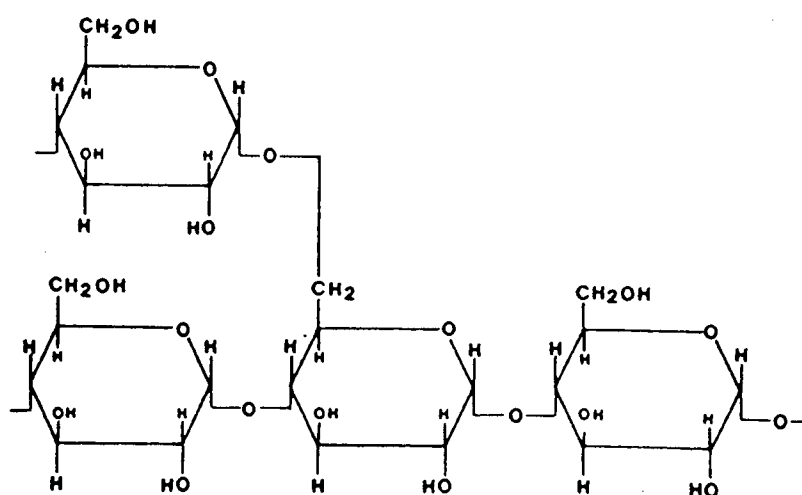
Figure 2.12 Structure of (A) amylose and (B) amylopectin
(from Schwoyer, 1986, p 49)

This physical characteristic of the hydrated gluten is due to the coiled and folded protein molecules (Simmonds, 1989). Gluten plays a vital role in breadmaking. It forms the structural support for the dough during baking and it increases the final protein content of bread and other yeast-raised goods.

2.15.4 Starch and Gluten Production

In the production of starch, the major objective is not only to separate the starch granules from the wheat kernel (or corn kernel, or potato, depending on the source), but to obtain the maximum possible yield of starch and gluten. As we shall see further on, in the production of starch and gluten there is the inevitable generation of waste streams. So the challenge becomes that of limiting these streams and obtaining the highest possible yield of products.

The effluent used in this study originated from a wheat starch and gluten production factory. The production process description given in this section pertains only to wheat flour. Further information on other starch making processes for corn, rice, potato, and so on can be found in Radley (1976a) and Knight (1969). In order to discuss and appreciate the sources, characteristics and problems associated with the effluents generated from the extraction of starch and gluten from wheat flour, an overview of the extraction process needs to be examined.

This brief overview is described below and has been obtained from the descriptions of Radley (1976a), Knight (1969) and Simmonds (1989). Since the extraction process begins from the wheat grain, the process overview will also begin with the wheat grain.

The wheat grain has three major components: the germ, which is high in fat and protein; the husk, which has no fat, protein or starch but is composed of pericarp and scutellum (bran) and protects the endosperm; and the endosperm which contains all the starch and most of the protein. It is this endosperm which is of prime importance to the gluten and starch manufacturer. The typical composition of endosperm is 70% starch and 12% gluten, the rest being minerals, enzymes, cellulose, sugars and lipoproteins. Since the endosperm is the obvious starting material in the starch and gluten extraction process, the husks and germ must first be separated.

The grains are initially soaked in water to soften the endosperm. The grains are then broken in corrugated rolls which remove the husks and germ. The remaining endosperm is then milled or ground to produce flour which the material from which the starch and gluten will be extracted.

The production of flour and the subsequent extraction of gluten and starch will inevitably damage starch granules, therfore the extent of the damage needs to be controlled. Damaged starch granules can be problematic in two major ways: since they absorb their own weight in water, such swelling tends to block filters and centrifuges and this impedes the overall extaction process; and high levels of damaged granules are undesirable from a processing viewpoint as they give rise to foaming and large amounts of impure tailings. Also, high levels of soluble solids leak out of these granules, resulting in increased levels of BOD.

The first step then, in controlling the level of damaged starch granules, is in the grinding of the endosperm. During this milling process, the breaking of the endosperm releases broken and intact starch granules, proteins, minerals and cellulosic material which is composed of cellulose, hemicelluloses, glucomannans and gums-collectively known as pentosans. These pentosans also prove problematic for the extraction process because they become viscous and build up on filters and centrifuges which cause blockages and slow down the throughput of the process (Radley, 1976b). Once the endosperm has been milled, the resultant flour is ready to be processed for starch and gluten extraction. This process is known as the Martin Process.

2.15.5 The Martin Process

The Martin Process, one of the older and popular methods of starch and gluten extaction, is employed by many industries including the one in this study. This process leads to high yields and high quality of gluten and starch (Knight, 1969). It involves a system of dough washing to separate the gluten from the starch. The success of this process is depedant upon the unique characteristics of gluten, which on hydration with water, forms an insoluble, elastic, cohesive mass (dough) from which the starch granules may be washed off with water.

Flour   Water

Continuous
dough
mixer

Rester
hopper

Cold water

Cooling
mixer

Concentrated effluent

Evaporator

Clear
liquor

Extractor   Starch
(Gluten)

Shaker
sieve

Gluten

Gluten
lumps

Rolls

Water

Gluten

Dryer

Wheat gluten

Protein

Continuous
centrifugal
separator

D.S.M.
screens

Water

Continuous
centrifugal
separator

Starch
slurry

Continuous
centrifugal
separator

Conc.
starch   slurry

Dewater   Overflow

Starch
cake

Dryer

Prime starch

Sterilised
liquor

Heat
exchanger

Clear
filtrate

High
speed
centrifugal   Liquor

Filter

Solids

Starch   sludge

Steam
rolls

Second grade starch

Figúre 2.13 The Martin process for wheat starch and gluten
production (from Knight, 1969, p 47)

The dough is formed when the flour is blended with about 60% of its weight in water. Once the dough has been formed, it is fed into an extractor vessel which washes and separates the starch from the gluten mass. The gluten is then compressed through a series of rolls, chopped into pellets, flash-dried, ground and bagged off as gluten powder. Details of this process are found in Knight (1969). Also, refer to Figure 2.13 for a detailed flow diagram of the Martin Process.

The washed strach and starch liquor from the extraction vessel is passed through a sieve to remove any small amounts of gluten (which is taken along with the bulk gluten to the dryer). The starch liquor is then screened through a very fine mesh. This retains the coarse starch fraction which is high in pentosans and proteins, and is discharged to the secondary recovery system (Knight, 1969).

The main body of starch, high in fine grade particles, is processed in a series of continuous centrifugal separators which wash, concentrate and separate the starch from the remaining protein. In the cleaning stages, effluent streams are produced and they contain protein, fibre, and small amounts of high grade starch. This effluent is also discharged to the secondary recovery system.

The purified and concentrated main starch stream is further dewatered and washed (Radley, 1976a), further producing waste streams which go to the secondary recovery system. After the final dewatering stage, the starch is fed into a flash dryer and dried, ground and bagged off.

2.15.6 Effluent Treatment and Disposal

The biggest problem in the manufacture of wheat starch and gluten is the volume of water used and effluent generated, as well as the loss of yield into the effluent. This problem must be dealt with not only because of the large volumes of water used, but of the nature of the material lost in the effluent. According to Radley (1976b), normal wheat starch effluent tends to be at least ten times stronger than domestic sewage.

Discharge to sewage treatment works is becoming increasingly unattractive because not only is potentially valuable material lost, but levies and fines are

MATERIAL BALANCE MARTIN PROCESS
DRY SOLIDS BASIS

100 parts flour	
Starch	79·6
Insoluble protein	11·9
Fat	1·2
Remainder	7·3

Gluten separation → 14·3 parts gluten   80% protein

Protein	11·44
Starch	2·0
Fat	0·86

85·7 parts

14·61 parts effluent		Starch Purification → 5·89 parts secondary starch 1·1% protein

Starch	6·9
Protein	0·2
Fat	0·25
Remainder	7·26

Protein	0·06
Starch	5·79
Remainder	0·04

65·2 parts primary starch		0·3% protein

Protein	0·2
Starch	64·91
Fat	0·09

Figure 2.14 material balance for the Martin process, on a dry solids basis (from Knight, 1969, p53)

being increased and requirements are becoming stricter on the quantity and quality of the discharged effluent. In this way, many factories are faced with the need to develop or implement systems to minimise their losses via cost effective processes.

The losses accompanied with the production of starch and gluten are substantial. Radley (1976b) has approximated that for every 100 tonnes of flour processed, anywhere between 16-21 tonnes of solids are lost in the effluent-depending upon the type of flour used and the degree of purity required in the final products. Figure 2.14 illustrates a typical material balance of the Martin Process. Of the losses to the effluent, about half are from the solid fractions of protein, starch and fat. The other half, or "remainder" are in fact the water soluble organic compounds. This was also confirmed by Simmonds (1989) who found that of the overall solids content in the effluent, roughly half is in solution (or soluble) and the other half is in suspension.

As stated earlier, the Martin Process generates a number of effluents or overflow liquors which originate from the washing, centrifuging and dewatering stages. As these liquors are high in insoluble bran particles and gel-like pentosan-protein-starch complexes, they tend to block filter runs and centrifuges, which impede the process and cause spillages and overflows-all of which are washed down the drain. These liquors are also high in the soluble fractions of the starches, proteins and minerals.

In order to produce a commercial starch of acceptable standards, the maximum possible removal of these impurities must be attained. The two-fold objective of this is that a reduction in these impurities allows the process to continue at its maximum throughput and the customers receive their desired product.

The system for dealing with these inferior, impure liquors is shown in Figure 2.13. This system is essentially an extension of the Martin Process itself. Here the waste liquors are collected, sieved and separated on centrifugal separators. The waste streams are then treated with enzymes which breakdown the pentosans. The resultant liquor is then dewatered and dried to yield second grade starch. While this treatment system ensures there is no return liquor to the main starch extraction process, the production of second grade starch produces an effluent problem of its own.

Many factories have limited capital and space. With the effluent generated in the production of second grade starch, many factory sites dispose of it as conveniantly as possible. Some factories do not even have a system for the collection of waste streams for the secondary recovery systems, and as such, do not have a secondary recovery system so the washings and overflows of the initial extractions go down the drain!

For factories which employ the recovery system, some use a recirculation step with the waste liquor generated. This is shown in Figure 2.13. In this system the secondary liquors are processed in a suitable high speed centrifuge separator which removes the solids as a very concentrated slurry-which can be used as cattle feed (Radley, 1976a). The clarified secondary liquors produced are then filtered and coagulated by heating. The coagulated particles are actually coagulated soluble proteins. These coagulated proteins are then removed and the remaining liquor is mixed with clean water and reused for the starch extraction stage.

2.15.7 Treatment Options for Starch Wastewaters

While the secondary recovery process is ideal, many existing factories are limited not only by the economic aspect of implementing an additional process, but by the lack of space or area to physically set up the process. This is especially so for sites in dense urban/industrial zones. According to Radley (1976a), the following areas have been investigated, or have potential for future development in the treatment of wheat starch wastes, as well as other starch wastes:

1. the conservation of water and therefore the reduction of the volumes generated;

2. acceleration of oxidation of effluents (biological, for example, fermentation or antibiotic production);

3. solvent extraction to replace water extraction;

4. use of effluent for land irrigation, particularly in the case of grass crops; and

5. recovery of solids by physico-chemical means such as precipitation, coagulation-flocculation combined with sedimentation, dissolved air flotation, membrane filtration, and so on.

While some of these proposals appear ideal, they may be difficult to implement. It is difficult to reduce the volume of water used in the wheat starch extraction process. The minimum volumes are needed to form the dough and to subsequently extract the starch and gluten. Additional water, as wash water, is always used to ensure the maximum removal of starch and gluten from centrifuges, filters and so on. Also all machines, pumps, pipes and extractors must be constantly washed and cleaned to avoid a build-up of contaminants or deposits of starch and gluten. Because starch is an excellent nutrient and there is always a high moisture content in the process, microbial spoilage will readily occur, so constant washing and cleaning is essential (Poock, 1985). Due to the present high costs and penalties involved in the use and discharge of excess water, most factories will tend to use the minimum volumes of water necessary.

The use of solvent extraction to replace the water extraction process(es) would require large changes to the process design of most plants. For many of the existing industries, this means a big disruption and may prove economically unfeasible. This would be especially so with a continuous running plant.

Using the effluent for land irrigation is only useful if the factory site is located near a remote area or close to agricultural land. In this study, the factory is located in the inner Western suburbs of Sydney, about 20 kiometres from the central business district. It is located amongst many other industries, factories and residential zones. Transport costs to deliver the effluent to irrigation areas would be high, and coupled with an increase in traffic makes this type of proposal for this factory site non cost-effective.

Radley (1976a) reported the successful application of wheat starch effluent for land irrigation on farmlands where the use of the effluent resulted in excellent growth of grass and a soil structure which remained in good condition over a five year period of irrigation. Knight (1969) cited a factory which piped its effluent two kilometres away to a farm, where it was allowed to flow into holding tanks and settle out. The supernatant was then used for spray irrigation and the sludge as pig and cattle feeds. In both these above examples, the starch factories were located in sparsely populated areas, near farmlands, where it was conveniant to dispose the effluent as an irrigant.

When very stringent restrictions are placed on effluent discharge levels, some factories need to undergo extensive modifications to their process designs in order to comply with the ordinances. For the factory which has both capital and land there exists choices of (or combinations of) physical, chemical or biological treatment methods which can effectively reduce the organic loads and accomodate the overall process.

Before any process is implemented, extensive small scale tests and larger scale trials need to be carried out on the selected treatment method(s) to determine optimum conditions and cost effective strategies.

In a report by Jost et al, (1981), a corn wet milling plant in Indianna, USA, had restrictions placed on its BOD and suspended solids concentrations discharged to the city sewer system. An extended aeration system had been in operation for sixteen years and performed well enough to remove about 50% of the BOD most of the time. As it was, though, it could not meet the new requirements and the entire process was upgraded with the addition of clarifiers, new aeration equipment and waste sludge thickening equipment, as well as holding tanks. The final effluent then was able to meet the new criteria and therefore attained direct discharge quality. All this was possible because this plant had access to land area as well as capital to carry the project through.

Biological treatment processes are commonly used in the treatment of starch plant wastes, and these processes are constantly being improved, so as to meet the stringent requirements for effluent quality and to minimise energy and capital costs.

One corn wet milling plant tackled its problem by using an aerobic treatment method which employed the fluid bed reactor concept. Stavenger (1979), used such a system to treat corn starch wastewaters. Bacteria were fixed on sand and fluidised with high upflow rates and high organic feed rates. Very high BOD levels were removed and it was expected that the process would show capital cost reductions of about 30% and reduction of land area requirement by at least 80%.

Another treatment, anaerobic biological treatment, is considered a favourable option for wastewaters high in biodegradable dissolved material. Anaerobic treatment has found many applications for starch wastes (Zeevalkink and

Jans, 1986). A potato starch factory used a combination of two different biological treatments (Frostell, 1983). An anaerobic contact reactor followed by an aerobic activated sludge process was used. The former treatment removed about 82% of the BOD, and the latter gave a total reduction of about 97% BOD.

The same trials were conducted with wheat starch wastewaters and similar results were obtained. These results were obtained after trials lasting over 140 days. This is an extremely long period so unless a site has the tank space and land space to set up such a process, it would have to adopt a procedure which would work in a much shorter time frame. Also a continuous running plant such as the one in this study processes large amounts of flour and therefore discharges large volumes of effluent on a daily basis and would therefore need sufficient space to set up such a process.

It should be stressed that long retention times are normal in biological treatment processes. Barnes et al, (1986, 1987) conducted trials to investigate the aerobic and anaerobic treatment of starch wastewaters. They found that to ensure at least 80% reduction of BOD in the waste, at least 60 days hydraulic residence time would be required for an anaerobic/aerobic lagoon system to treat the wastes effectively. They also found that the system needed considerable pH control-a factor of prime importance as most biological reactions take place in narrow pH ranges. Slight changes to the optimum pH can severly impede the biological reactions.

Zeevalkink and Maaskant (1984) used an anaerobic upflow sludge blanket process to successfully treat corn starch wastes. Their treatment reduced COD levels by 90% and produced high levels of biogas which was fed to the boilers in the factory. This same process was employed by Zeevalkink and Jans (1986) in the treatment of wheat starch wastewaters, again with very high COD reductions.

Anaerobic digestion was also investigated by Herzka and Booth (1981), with the effluent of a wheat starch producing factory. Due to the nature of the wheat starch/gluten in the effluent, recycling of the wastestreams was not considered practical, so an anaerobic digestion process was implemented to treat the effluent. Trials with this process gave a 96% reduction in BOD over a 25 day period. An interesting aspect about this work was that in the initial treatment

proposals, the use of the effluent for the purpose of harvesting yeast cells was considered, but subsequently abandoned, as it was felt that sterilising the effluent would be costly.

Knight (1969) on the other hand felt that starch effluents would be ideal for the cultivation of yeast cells without the need for sterilising the effluent. Torulospsis utilis or Torula Yeast was proposed as a very suitable material for the production of a food yeast from most starch liquors. Torula Yeast grows at an optimum of about pH 4-5. Because it has a resistance to contamination by other organisms when the pH is held to 5 or below, the waste liquor does not really need any sterilisation as the acidity inhibits most bacterial growth. The yeast produced is a palatable product, high in vitamin B and protein and can be grown aerobically and in acidic conditions at 30°C.

Reiser (1954) was able to grow T. utilis on potato starch wastewaters to yield a product with 55% protein. With optimum conditions, the mean propogation time was 7 hours. BOD loads were reduced by up to 60% and a costing analysis showed this process was economically favourable. An eight hour residence time was optimum and allowed high throughputs of effluent, so a plant discharging large volumes with limited space could consider a similar process.

Eldin et al, (1981) also used starch plant effluent as a substrate for profitable biomass production. Aspergillus terreus, a fungus, was grown for the production of protein biomass and the reduction of BOD in the effluent. The corn starch plant in Egypt produced 300m³ of effluent daily, which was disposed of in the Nile River. The BOD was extremely high at 30,000 mg/L and had a protein content of about 10,200 mg/L. At a hydraulic residence time of 48 hours, about 72% of the BOD was removed and 29% of the protein recovered. At 144 hours (6 days) the BOD was reduced by 86% and protein recovery was up to about 43%. The optimum retention time was infact 6 days. It became evident that fungal growth for biomass production would give considerable reduction of BOD of the generated waste, as well as provide mycelium biomass as feed supplement.

Besides biological treatment methods, some researchers have used physico-chemical methods in the treament of starch wastes. The advantages of the latter are that besides the ease of control of the process, many of the reactions

are carried out in a relatively small time frame, so that the treatment of large volumes can be carried out in minimal space. The use of one such process, ion exchange, to treat potato starch wastes has proved successful.

One of the major problems in potato starch manufacturing is that very large volumes of water are used, which contain extremely high BOD loads. Radley (1976a), acknowledges that purification of this effluent is difficult. Anerobic sludge processes are not feasible because of their long-take off time: in the first six weeks virtually no organic matter is destroyed.

According to Knight (1969), potato wastewaters contain about 15% protein, 30% amino complexes, 30% organic acids, 15% reducing sugars and 10% potassium. Rather than use a biological process to degrade these compounds, Heisler et al, (1972) decided to use the cation-exchange process to reclaim as much as possible from the wastewater and simultaneously reduce the BOD level. They were able to recover amino acid mixtures, potato protein and a liquid fertiliser containing about 8% ammonium sulphate. The BOD was also reduced by 60%. The authors felt that besides the obvious benefits of a big reduction in BOD, the sale of the recovered materials may provide a means of recovering some costs of the treatment.

This above mentioned work was slightly modified by Schwartz et al, (1972). They found that despite the work of Heisler et al (1972), the resultant effluent was still high in organic sugars and organic acids and thus, the potential for high BOD and eutophication existed. Schwartz et al (1972) added an additional stage, a polishing step which included anion exchange resins, to treat the water discharged from the cation exchange step. The anion resins were able to remove 99% of the incoming organic acids and sugars and the market potential for food uses of these acids and sugars was considered.

Reverse osmosis was another process used to treat potato starch effluent. Pepper and Orchid (1981), set up a reverse osmosis process at Avebe, Holland. Avebe processes about 2.7 million tonnes of potatoes per year and the resultant high pollution levels make the effluent unsuitable for discharge into open bodies of water. The protein fraction in the effluent was desirable to recover, but because of the large volumes of water, vast amounts of energy were required to heat and thereby coagulate the proteins to facilitate their removal and to evaporate the water.

The use of reverse osmosis was able to produce a much cleaner effluent and reduce the cost of protein extraction from the potato fruit wash water. This is because the reverse osmosis process concentrated the diluted protein in the water and saved energy at the heating and evaporating stages. When the trial reverse osmosis recirculation tank was set up, the effluent which was treated yielded two major streams: a concentrated protein stream and a water stream with substantially lower total solids. This latter stream was re-used as potato wash water, so an immediate saving was realised with recirculation of the water. The former stream was steam injected and this coagulated the proteins which were then dried. The water remaining when this protein was removed contained 18% total solids, which were high in nitrates, ammonia, sugars and organic acids. This was applied as part of cattle fodder.

Meindersma (1980) felt that reverse osmosis was too expensive a procedure, but Pepper and Orchid (1981) claim it is a cost effective treatment because 50% of the water volume is recovered and recycled. Also, 50% reductions were met in the heat coagulation and evaporator stages. This represented a substantial capital cost reduction and energy cost reduction, making the process economically feasible.

Another physico-chemical treatment process is that of coagulation-flocculation. The literature shows that some work has been done using this process to treat general food wastes (see section 2.14.1) but little has been done regarding the treatment of starch wastes. Most food industry wastes are treated by biological means.

Salem et al, (1972) conducted a simple coagulation process on corn starch wastewaters to reclaim the protein as a cheap source of vegetable protein. These authors used ferric chloride and calcium hydroxide as coagulating agents and obtained protein at 26% and 16% respectively. Using NaOH alone they were able to obtain about 44% of the protein in the effluent at pH 7 (compared to obtaining about 32% at pH 6). Unfortunately the authors did not describe how the actual coagulation-flocculation tests were conducted. intensity and duration of the mixing is not even given, except that 6N NaOH was added gradually to the waste liquor with continuous stirring to the desired pH (pH 7). Their system not only provided a simple wastewater treatment process, but a cheap and feasible method for obtaining large amounts of proteinaceous precipitates which were of high quality.

The starch processing industry, along with the majority of food processing industries, produces an effluent which contains a substantial level of material that has the potential to be directly re-used or converted to a different, yet useful, form. With the various potential methods involved in treating the effluent, it appears that a combination of physico-chemical with biological treatments may be the way to go, but would ultimately depend on what purpose the product will have.

2.16 BIOLOGICAL TREATMENT VERSUS PHYSICO-CHEMICAL TREATMENT

2.16.1 Introduction

The preceding section just discussed various treatments used for starch factory wastes, most of them being biological methods. Conventional biological treatments have proved to be popular in the treatment of virtually all food wastes because their organic rich effluents are well suited to microbial degradation. Unfortunately, there are potentially valuable materials which get degraded to useless sludges in many cases, which in itself poses further disposal or treatment considerations. This section will briefly compare biological treatments with physico-chemical treatments and weigh out the advantages and disadvantages of both.

2.16.2 Biological Principles

Biological treatments rely on the ability of certain microorganisms to convert both the organic and inorganic components of waste streams into cellular tissue and simpler substances such as water, carbon dioxide, methane, ammonia, nitrates and so on, depending on whether the treatment system is oxygenated (aerobic) or non-oxygenated (anaerobic). This process of converting wastes to simpler organic compounds and cellular tissue is known as "stabilisation" (Jorgensen and Johnsen, 1981). By this implication, methods or systems can be designed whereby microorganisms can be employed and exploited to degrade and therefore remove the organic matter in effluents. This is infact the major objective of biological treatment.

Biological treatment processes do not and can not degrade everything organic, but by providing favourable conditions certain processes can be

enhanced by allowing for increased rates of breakdown and/or an increase in the variety of compounds that can be degraded. According to Gaudy et al, (1981), favourable conditions should allow for the opportunity that within the heterogeneous biological population there exists a genetic capability, or its potential to evolve exists, which will be phenotypically expressed. Its ultimate expression would be in the formation of complex enzymatic systems designed to degrade these compounds.

The design and control of successful biological systems can only be achieved by a thorough knowledge of the chemical and physical characteristics of the effluent to be treated, and an understanding of the complex biochemical and physiological reactions and interactions affected by the wide variety of microbes in the treatment unit.

There are various microbes or communities of microbes which include bacteria, fungi, algae, protozoa and even viruses. Most biological treatment systems contain very diverse microbial communities made up of populations of the above-mentioned organisms, with a host of complex interactions between them. These interactions influence the overall biological growth and performance of the treatment process.

Comprehensive reviews of these organisms, their characteristics and biochemistries can be found in Grady and Lim (1980), Lynche and Poole (1984) and Brock et al, (1984).

Other factors influence the efficacy of biological treatments: temperature, pH, mixing intensity, and the presence of toxic compounds. Many of the biochemical interactions occur within a narrow pH and temperature range, so slight changes may impair the reactions. The growth rate of the microorganisms needs to be constantly checked, since overgrowth may result in a build-up of products which may inhibit the growth of a certain set of microbes and allow an unfavourable one to flourish. Also, the build-up of certain products, such as organic acids can alter the pH. Mixing intensity is essential to maintain adequate dispersion of nutrients and to avoid localised build-up of waste products.

A balance of requirements is also important. If the balance of nutrients and gases such as oxygen are slightly altered, the delicate network of interactions

in the microbial community may be altered. Because microbes are so sensitive to slight changes in their conditions, research is often conducted in improving strains which will be capable of tolerating such changes, and still be able to perform to their desired levels. This includes the selection of stronger strains (Kyosai et al, 1989), and genetic manipulation (Matsui et al,1991).

2.16.3 Biological Treatment Methods

Barnes et al (1981) and Brock et al (1984) provide detailed descriptions of the major biological processes used for the treatment of wastewaters. Even though significant changes have been in this field to date, the treatment principles and basic processes remain the same. Generally, microbial populations can exist in a treatment process in either suspended growth, for example, activated sludge processes, or attached growth, also known as fixed films or biofilms and these include trickling filters. A combination of suspended growth and biofilm can also be used. Treatment systems can be either aerobic or anaerobic.

Aerobic processes are those which require oxygen. Microorganisms degrade the organic content of the waste and consume the available, dissolved oxygen. The products tend to be carbon dioxide, water, nitrates, sulphates and other oxidised compounds. Efficient aeration and mixing are essential in aerobic treatments (Forster, 1985). The activated sludge process is a common aerobic process (Erikson and Alm, 1991) and has been used to treat food wastes such as pineapple processing effluents (Fuaad et al, 1991).

Anaerobic processes are mainly used to treat organic sludges generated from aerobic processes, especially in sewage treatment. Anaerobic treatment is also used to treat concentrated organic industrial wastes and insoluble organic matter such as lignin and related material. The main products include carbon dioxide, methane, hydrogen and other gases, as well as organic acids. Rintala (1991) feels that anaerobic processes are superior to aerobic processes because the former produce much less sludge, have lower nutrient demands and lower energy requirements. The main drawback with anaerobic processes is they can be very sensitive to toxic compounds and slight changes in process conditions.

2.16.4 Disadvantages of Biological Systems

From the very brief overview given on biological processes, certain aspects arise which indicate problem areas in biological treatments.

Optimum physical mixing is essential. Insufficient mixing has two negative effects: it reduces contact between the microorganisms and nutrients, thus reducing the efficiency of biodegradation and it creates localised build-up of digestion products. The latter can have an inhibitory effect, especially in anaerobic systems because a build-up in volatile fatty acids will not only cause direct product inhibition but it will drop the pH and therefore alter the microbial environment and thus compromise microbial efficiency (Stronach et al, 1986).

Even if optimal physical mixing is maintained, product inhibition may still arise. As the microbes utilise the substrate, they will produce waste products. Ideally, these products may serve as substrate for other microbes, but if not, their build-up will inhibit the overall process, as previously stated. The formation of all these products will cause changes in the pH and buffering capacity. For this reason the pH needs to be constantly monitored and maintained in the optimum range for the microbial processes. As with pH, most of the reactions occur in narrow temperature ranges, so temperature also needs to be monitored and maintained. Slight changes can significantly reduce the biological activity of the system.

In the case of aerobic processes, optimum oxygen levels are crucial. This is why not only is ambient mixing important to effectively transfer the oxygen throughout the tank, but the level of organic matter is also significant. If the waste streams have a high organic load, the oxygen may not be effectively distributed due to the physical difficulty of transferring sufficient oxygen.

Another problem encountered with biological treatment processes is that of geneartion times. In many cases, for reasons often unknown, the inoculum may never start-up or may take a considerably long time to start-up. If the initial conditions have been affected by extremes of pH or temperature or the presence of toxic compounds, then the inoculum may never take hold.

According to Forster (1985), toxins or toxic substances can be components of the influent waste steams and include material such as heavy metals, organochlorines and antibiotics. Toxins can also be the by-products of the metabolic activities of the digestor bacteria.

Toxic compounds can also adversely affect the biological activity and finally, the nutrient balance is an important aspect of all biological processes. The food to microorganism ratio must always be analysed and maintained to ensure steady-state conditions of the biological reactors.

2.16.5 Physico-chemical Treatments

There exist many physico-chemical methods to treat water and wastewater. A comprehensive review can be found in Eckenfelder (1980), Barnes et al (1981), and Montgomery (1985). The process of coagulation-flocculation, followed by sedimentation has already been covered in this review and so will not be discussed in this section. Another method which uses coagulation, but relies on the sludge to rise and not settle is that of dissolved air floatation (DAF). Many of the pretreatment stages of DAF resemble conventional coagulation treatment regarding rapid mixing and coagulant addition.

According to Meyers (1980a) DAF is a gravity separation process, whereby two phases are separated by the specific gravity difference between them. This change in specific gravity is acheived by attaching air bubbles to one of the phases, done by saturating water with air under pressure and then expanding the water stream through a valve to atmospheric pressure. This pressure drop allows the dissolved air to expand, forming microbubbles, which attach closely to the solid particles, which by now have been destabilised with an appropriate coagulant.

DAF was used to treat slaughter-house wastewaters. A combination of alum and ferric chloride in the DAF process led to an 80% reduction in BOD (Meyers, 1980c). Ackel (1988) used DAF to treat effluent from a pulp and paper mill site. Using an organic polymer, 95% of the colour and 95% of the suspended solids were removed, compared to 25% colour removed with coagulation followed by sedimentation. Ho and Tan (1989) also used DAF to successfully treat anaerobically digested palm oil mill effluent, and found it was more efficient than coagulation with sedimentation.

Templeton (1991) reported of a DAF process used to treat ice-cream wastes, generally high in grease, carbohydrates and soluble proteins. Grease levels were greatly reduced with DAF. Bough et al, (1975) treated food wastes with DAF as well as coagulation followed by sedimentation. Chitosan was the primary coagulant in both treatments. It was actually found that sedimentation gave far better results with 94% reduction in suspended solids compared to 58% reduction in suspended solids using the DAF process.

DAF has a number of features which make it an attractive and viable process. According to Templeton (1991), grease, light and heavy solids and grit can be removed in the one unit; odour is minimised due to the shorter retention periods as well as the presence of dissolved oxygen in the effluent; and the sludge/scum produced in DAF tends to have a higher solids content than the sludge produced by gravity settling.

Edzwald et al, (1992) reviewed the advantages and disadvantages of DAF. The advantages included smaller tanks compared to gravity settling (and therefore lower capital costs); smaller flocculation tanks, as less flocculation time was required; coagulant and flocculant aids tended to be a lower dosage compared to sedimentation treatment; and higher sludge solids were produced in comparison to gravity settling. Also better removal of low density particles, colour and very fine suspended solids were obtained with DAF compared to sedimentation.

The main drawback with DAF was that it had higher power costs due to pumping recycle water, but this could be offset by the decrease in costs of coagulants, sludge treatment and sludge disposal.

Activated carbon is used usually as a polishing step to remove very fine particles or soluble compounds. Semmens et al, (1986) found that higher molecular weight compounds were not easily removed by activated carbon, but the lower molecular weight soluble compounds were.

Suzuki and Chihara (1986) used activated carbon to adsorb soluble fractions in organic rich wastes such as food and pulp and paper mill effluents. Trihalomethanes were removed successfuly from water using activated carbon (Takeuchi et al, 1991). Coagulation was used first to remove the larger

molecular weight compounds, then the supernatant was treated with activated carbon to remove the smaller, soluble organic compounds which were not easily removed by coagulation.

Ozonation is another process used to treat wastewaters and water. It is usually used as a pretreatment step to DAF or coagulation with sedimentation. According to Stone (1989), pre-ozonation tends to significantly increase coagulation efficiency. Saunier et al, (1983) used pre-ozonation to treat drinking water and found significant removals of turbidity, suspended solids and COD. Flocs were much larger and their settling velocities were also higher. Pre-ozonation also decreased the coagulant demand by 70%.

According to Rice and Bollyky (1986) ozonation works by the oxidation of organic compounds in water. The oxidation introduces oxygen into the organic structure, producing oxygen moieties such as carboxyl, carbonyl or hydroxyl groups, all of which are polar and negatively charged. They can therefore bind themselves via hydrogen-bonding (micro-flocculation) or with polyvalent cations such as ferric ions or aluminium ions and form insoluble complexes.

Other physico-chemical treatments used to treat wastewaters especially food wastewaters include ion-exchange processes, successfuly used to remove at least 60% of the BOD from starch wastes (Heisler et al, 1972; Schwartz et al,1972). Meindersma (1980) used membrane filtration as a pretreatment step to coagulation, to treat potato starch wastes, while Pepper and Orchid (1981) trialled reverse osmosis successfuly to remove soluble proteins from starch wastes.

Coarse-media sand filtration, widely practised in Asia was used by Vigneswaran et al, (1991) to treat municipal water. It is a common, cost-effective and popular pre-treatment method of filtering water prior to treating it for recycling. The study found that compared to a biological filter, the sand filter had much higher removal efficiencies for turbidity, organic carbon, total phosphorus and total nitrogen.

Malina and Gromies (1982) compared various physico-chemical methods with coagulation-flocculation regarding the removal of viruses and other enteric organisms from potential water supplies. Ferric and aluminium salts were found to have almost equal capacity in removing the viruses, and were able to

remove about 95%. Sand filtration with diatomaceous earth, coated with polymer, was trialled and found to have removed about 98% of the viruses. Reverse osmosis was able to remove 99% of the viruses, while activated carbon removed between 75% and 99% of the viruses, depending on the type of carbon.

Membrane filtration is another physico-chemical treatment process used, especially for the treatment of potable water. Lahoussine-Turraud et al, (1990) treated Seine River water via membrane filtration. Because the membranes had a problem with fouling, it was decided to use coagulation as a pretreatment step to reduce levels of soluble organics so as to minimise membrane fouling. Powdered activated carbon was used as the coagulant, and it was found that coagulation only slowed down the rate of fouling but did not reduce the extent of fouling. It was found that even when greater levels of organic material were removed by coagulation, the effectiveness of membrane filtration did not necessarily improve. This is because smaller, soluble organic molecules such as polysaccharides remained in solution after coagulation and adsorbed to the membranes during the filtration step. The authors should have considered the possibility of adding an activated carbon treatment method after the coagulation step, to adsorb the soluble organics.

2.16.6 Advantages of Physico-chemical Treatments

Physico-chemical treatment systems have a number of advantages over biological treatment systems.

* Chemical treatment systems are alot more controllable than the biological systems, and are more predictable.

* Masides et al, (1988) point out that chemical plants can be put into operation almost immediately as there is no need for long periods required for acclimitising cultures.

* An important aspect in many treatments is the availability of space. Physico-chemical treatment systems tend to take up much less space than the biological ones, and this is significant in urbanised areas and industrial sites where space may be scarce or expensive.

* Installation costs tend to be cheaper than biological systems (Eilbeck and Mattock, 1987)

* Physico-chemical treatments are relatively tolerant of temperature changes. Obviously, changes in temperature will effect the reaction kinetics, but in a biological system, even slight changes can dramatically alter the overall process.

* Physico-chemical treatment plants are less susceptible to changes in both flow rates and in the composition of the wastewater to be treated, as well as foreign or toxic compounds. This is because most chemical processes can be designed to accomodate intermittent and sometimes complex changes in the wastewater. In all cases the chemical processes must be designed to take into account the compostion of the water to be treated, but slight changes such as the presence of toxic materials, which would render a biological process inoperative (and lead to a shut down or re-acclimatisation), would not dramatically alter the system.

* Modifications can be carried out much more rapidly in a physico-chemical plant, by either changing the chemical processes or adding a further stage or removing an existing one.

* By-products can often be useful, as most physico-chemical treatments do not degrade them as do biological systems, but can change their form or incorporate them into another substance. This is particularly useful in the food industries where waste carbohydrates, protein and grease are recovered from the effluent and re-used as part of a value added product.

* While it may be argued that the running costs for chemicals are much higher for physico-chemical systems, the revenue from the sale or the re-use of the recovered materials may re-coup at least in part the capital and running costs of the plant (Birch et al, 1976).

2.16.7 Disadvantages of Physico-chemical Treatments

Physico-chemical treatment processes also have some inherent disadvantages. In many processes, chemicals are added to remove particles via precipitation/coagulation. This results in significant increases in the weight

and volume of sludges, which tend to be high in precipitated metal hydroxides and carbonates, as well as useful material which can not be reclaimed but is trapped in the sludge.

2.16.8 Combined Biological and Physico-chemical Treatments

There is rarely a treatment process which operates as a single entity. Processes are usually integrations of various biological treatment units, or various physico-chemical treatment units, or in most cases, an integration of biological and physico-chemical treatment units. The following examples demonstrate how in most cases, physico-chemical treatments can be used first as a pretreatment stage to remove the bulk of the waste as well as that part which may contain toxins, followed by biological treatment systems which can be used to further reduce the remaining wastes.

Li et al, (1986) treated slaughter-house wastewaters with coagulation-flocculation with sedimentation followed by a fluidised-bed biofilm reactor. The coagulation-flocculation step was done first in order to remove grease, as it had adverse effects on the oxygenation capacity of the biofilm reactor. The use of alum or ferric sulphate in the first step removed about 65% grease, and with a polymer, 92% grease was removed. Subsequent biological treatment proved very effective.

Trials were conducted to test the efficacy of a treatment system as to whether biological treatment applied before or after a coagulation-flocculation-sedimentation treatment gave better performance. Activated sludge treatment, followed by coagulation, proved effective in removing high colour content from pulp and paper mill effluents (Smith and Malloy, 1990).

Galil and Rebhun (1990) found that in treating municipal wastes, the chemical treatment unit was best placed as the primary stage, because the organic load was reduced by about 45%. This reduction in organic load led to subsequent increases in kinetics of the biological treatment stages (by up to 20%) and decreases by 23% of the energy requirement.

Milstein et al, (1991) treated spent bleaching effluents from pulp and paper mills by first coagulating the effluent with cationic starches as polymers. This removed 75% of the adsorbable organic chloride and 59% of the COD. The

resultant supernatant was then supplemented with glucose and it supported the growth of <u>Aspergillus spp</u> and <u>Penicillium spp.</u> These fungi were able to degrade the remaining adsorbable organic chloride by 21%, resulting in a total removal of about 80% of the chlorinated organics.

Grau (1991) treated textile wastewaters to remove colour. Coagulation-flocculation was used initially, removing about 85% of the colour and 38% of the COD. Subsequent biological treatment further removed the COD and colour.

El-Gohary et al, (1991) aimed at reclaiming and reusing municipal effluents for beneficial purposes, if any. Using ferric chloride, with lime, about 92% of the turbidity and 82% of the COD was reduced. Biological treatment involved a sand bed which further removed soluble COD and soluble BOD components. It was felt that the effluent resulting after these treatments was suitable for agricultural purposes.

Hadjivassilis (1992) treated dairy wastewaters by flocculation, followed by high-rate multi-layer granular filtration, followed by activated sludge treatment and obtained excellent reductions in suspended solids, BOD and COD.

Haberl et al, (1991) treated pulp bleaching effluents first with coagulation-flocculation, then oxidation with ozone, followed by biological treatment. This led to a marked increase in performance of the biological stage, because the inhibitory effects were decreased in the pretreatment stages. As a polishing step after biological treatment, activated carbon adsorption was used to remove residual soluble organics.

Boman et al, (1991) treated paper and pulp mill effluents by ultrafiltration followed by treatment in an aeration lagoon. COD removals corresponded to 41% and 72% respectively.

Tebai and Hadjivassilis (1992) used a combination of DAF (to remove grease and fine suspended particles) then alum coagulation followed by an extended aeration process to treat soft drink industry wastewaters. A total of about 84% reduction in COD was achieved.

The main aim of this thesis is to apply a physico-chemical process, in this case, chemical coagulation-flocculation followed by sedimentation, as a possible step toward treating the starch wastewater. Such a process could create a number of advantages (which are not the scope of this thesis): the treated water would be of a higher quality, suitable for direct discharge or recycling into the manufacturing process or even for further polishing steps; and the material reclaimed in the coagulation process may be suitable as a value added product thus making the treatment system cost-effective.

CHAPTER 3. <u>MATERIALS AND METHODS</u>

Most of the analytical methods used in this study were based on those in the Reference Book, Standard Methods for the Examination of Waters and Wastewaters, APHA, 17th ed. (1987).

3.1 Effluent Sampling

Two types of samples were collected: grab samples and discrete samples (over 24, 48 and 72 hour periods). The purpose of the discrete sampling was to collect data and create profiles of certain characteristics in the effluent such as tubidity, total solids and suspended solids, over a certain period of discharge. This allows observation of any fluctuations in the effluent. The purpose of the grab samples was to collect enough sample volume to carry out the experimental work.

3.1.1 Grab Sampling

Samples were obtained directly from the effluent discharge point, behind the factory. A bucket was tipped into the effluent stream, filled, and decanted into a 25 litre cubic drum. The drum was filled to the 20 litre mark rather than to the very top, to allow sufficient air space to mix the sample thoroughly before using it. The process of filling the drums was done until 6 drums were filled, corresponding to 120 litres of sample.

3.1.2 Discrete sampling

The Portable Discrete Sampler (Model S-4401, Texas Nuclear Corporation, Austin, Texas, 78766) was used for the collection of the discrete samples, collected every hour over a 24, 48 or 72 hour period).

The sampler was housed in a shed, approximately 100 metres from the factory effluent discharge point, and was installed on a firm level floor, about 3 meters above the channel flow, which contains the discharged effluent stream from the factory.

The intake hose was placed in the centre of the channel main flow. The hose, and the strainer attached to the end, were heavy enough so that the rapid flow did not pull the intake hose to the surface of the flow. The hose was placed straight, with no S-bends, to ensure no traps were formed, as this would impede proper uptake of the sample.

The procedure for setting up and programming the sampler and collecting the effluent is detailed in the suppliers publication: Installation and Operation of Texas Nuclear Manning Products, Model S-4401 Portable Discrete Sampler, Publication No. 717662 (05443-000) March, 1989.

The method is given briefly: a programmable single-time interval in the time mode was chosen so that a 500 mL sample was drawn into a 500 mL bottle every hour over a 24 hour period. When all bottles were filled at the end of the 24 hours, the sampling action stopped. The bottles were then removed, and the samples withdrawn for future analysis. The bottles were then washed, rinsed and replaced for the next 24 hour cycle (if any).

3.2 Effluent storage

Effluent was stored in the cold room of the School of Microbiology, at the University of NSW. The cold room was maintained at about $3^{o}C$-$4^{o}C$. Samples were stored in the cold room for a maximum of 3 weeks. Samples were placed in the cold room usually within 90 minutes of collection. A portion of the sample was withdrawn immediately for pH, turbidity, BOD and TOC readings.

3.3 Effluent sample preparation

When the effluent was used in the coagulation-flocculation trials, it was used in an unmodified state, in that it was not altered chemically or physically before it was tested. The effluent was modified only when experiments were conducted to test the effects on coagulant dosage and therefore coagulation efficiency when the solids load of the effluent was significantly altered.

Under quiescent conditions, the bulk of the settleable solids settled out in about 20 minutes, therefore 20 minutes was used as the standard settling time.

Treated samples were allowed to settle for 20 minutes after the slow mix phase of the jar tests, and the supernatant characteristics or quality were compared to the control, which was the supernatant of the untreated effluent mixed in the same manner as the treated effluent. Changes in the treated supernatant compared to the untreated were usually reported as percentage differences.

Tests were also done on the whole effluent sample to determine the overall quality of the effluent to be treated. During this project, it was decided to investigate the effects of modifying the solids content of the effluent, and to determine if these solids changes have any bearing on the effectiveness of the coagulation process.

3.3.1 Effluent, half settleable solids (1/2 E)

Normal effluent was allowed to settle for 20 minutes. The supernatant was carefully decanted and mixed into an effluent sample with the same volume as the first. For instance, 10 litres of effluent were settled out, and the supernatant added to 10 litres of another, same sample lot. This effectively halved the levels of settleable solids, but retained the soluble and suspended solids levels as in the original samples.

3.3.2 Effluent, double settleable solids (2E)

Normal effluent was allowed to settle for 20 minutes. The supernatant was carefully decanted and discarded, while the settled solids were added into an effluent sample of the same lot, with the same volume as the first. Effectively, the content of the settleable solids is doubled, while the levels of the soluble and suspended solids remain the same.

3.3.3 pH

The pH of the effluent was adjusted to the desired level prior to the addition of coagulants or coagulant aids. The pH was adjusted with either 10% sulphuric acid or 10% sodium hydroxide, while the samples were being mixed.

3.4 The Jar Test

The jar test method was that adapted by Bough (1975) and Kawamura (1991). The jar test apparatus used was a PCI Model, shown in Figure 3.1 (Patterson Candy International, Ltd. 21 The Mall Ealing, London, W5, England). The model consists of a set of four vertical paddles which are motor driven, with a variable speed control. The jar test apparatus had a floc illumunator, which was a well diffused light source platform on which the beakers stood. This allowed for uniform observation conditions.

All the jar tests were conducted at room temperature (19°C-23°C). The containers with the effluent were shaken thoroughly and decanted into 4 or 5, 5 litre beakers and allowed to reach room temperature.

The pH of the samples was adjusted to the desired levels with either 10% w/v NaOH or 10% v/v H_2SO_4. 500 mL lots of well mixed sample was measured into 1000 mL beakers (the beakers were all identical in size and shape, and the same ones were used throughout the study).

The jar tests were conducted in batches of four, as the apparatus only had four paddles. The beakers were positioned under the mixing paddles so that the flat blades were placed in the central position of the beakers, about 1 cm above the base of the beaker.

The test reagents were loaded into disposable plastic syringes and placed close to each beaker which was to receive it. When there were several reagents to be added, a number of syringes were used and placed in order of addition sequence. In most cases the reagent volume was less than 10 mL. Once the reagent was added, the syringe was quickly flushed with about 5 mL water, and added into the vortex.

The multiple stirrers were started at 120 rpm, the rapid mix phase. The test reagents were added into the vortex in their appropriate order. The first reagent in the sequence was added to each beaker before the second reagent was added. In all cases, the first reagent was the inorganic coagulant and the second was the polyelectrolyte.

Figure 3.1 The jar test apparatus used in the study

In the course of the project, it was decided to study the effects of feed rates, dosage feed points and the duration and intensity of the rapid mix and flocculation periods. For instance, while the reagents were added into the vortex in the majority of the cases, tests were also done with reagent being added on the surface of the effluent, mid way between the vortex and the beaker wall. Approximate flow rates of the reagent were also monitored, from the rapid squirting of the reagent from the syringe to the controlled release, in which case the graduations on the syringe served as a guide to controlling the flow rates. The syringes were alwayes flushed with a small quantity of water.

The strength of the dose was also monitored, and there were cases where dilute dosage feeds exceeded 10 mL.

Generally, once the coagulant was added, rapid mixing would continue for a maximum of 5 seconds. If the polymer was added after the coagulant, it was added about 5-10 seconds later, and the rapid mix continued for a further 15 seconds.

At the end of the rapid mix period, in the majority of the cases, the speed was gently reduced to 30 rpm for 20 minutes. This slow change in speed was done so as to minimise floc disrupture. Once again, variations in both the intensity and duration of the flocculation period were evaluated for their effects on the overall flocculation process.

The mixing was stopped at the end of the flocculation period, and 20 minutes of quiescent settling followed, after which 200 mL was decanted and measured for various parameters such as turbidity, suspended solids, TOC, etc.

The pH measured at the end of the jar test was measured as the operative pH. This is because destabilisation reactions are normally irreversible, and so the pH will remain at the point where the reactions went to completion (Edwards and Amirtharajah, 1985).

3.4.1 Determination of Floc Sizes

Floc sizes were determined by direct visual comparison to the Floc Comparator, Figure 3.2. The visual comparison gave an estimate of the floc sizes, and this information was used to compare floc size and growth to other parameters, such as supernatant turbidity, suspended solids and TOC, as well as mixing times and intensities.

The floc sizes were recorded at 20 minutes only, ie, at the end of the flocculation period. The sizes recorded were an average of the ranges given in the Floc Comparator, for example, flocs of size 0.75 mm-1.0 mm (B) were recorded as 0.87 mm (the average).

3.5 pH

The pH was measured with the HI 8418 Microprocessor Bench pH Meter (Hanna Instruments, Limena, Italy).

Calibrations were carried out with pH 4, 7, 9 and 10 buffered solutions.

3.6 Turbidity

Turbidity, which is the amount of light scattered and adsorbed by particles, is proportional to the level of particles in a water sample and is therefore an excellent indicator of water quality (Barnes et al, 1981).

Turbidity was measured as Nephelometric Turbidity Units (NTU) using a Hach Turbidimeter Model 2100A. The procedure(s) used was that followed in the "Instrument Manual for the Laboratory Turbidimeter, Model 2100A, Hach Company, 1987".

The Model 2100A is a nephelometer calibrated for measuring turbidity in colourless liquids. Since APHA (1987) allow the use of instruments which measure the intensity of light scattered at $90°$ angles to the incident beam, that is, nephelometers, the use of the Model 2100A is an acceptable standard method for measuring turbidity in a water sample.

Joseph Crosfield & Sons, Limited, Warrington

FLOC COMPARATOR

FOR THE DETERMINATION OF FLOC SIZES IN WATER CLARIFICATION AND SOFTENING PROCESSES

A — 0·3mm. — 0·5mm.

G — 3·0mm. — 4·5mm.

B — 0·5mm. — 0·75mm.

F — 2·25mm. — 3·0mm.

FLOCSIL
BRAND

C — 0·75mm. — 1·0mm.

Crosfield

E — 1·5mm. — 2·25mm.

D — 1·0mm. — 1·5mm.
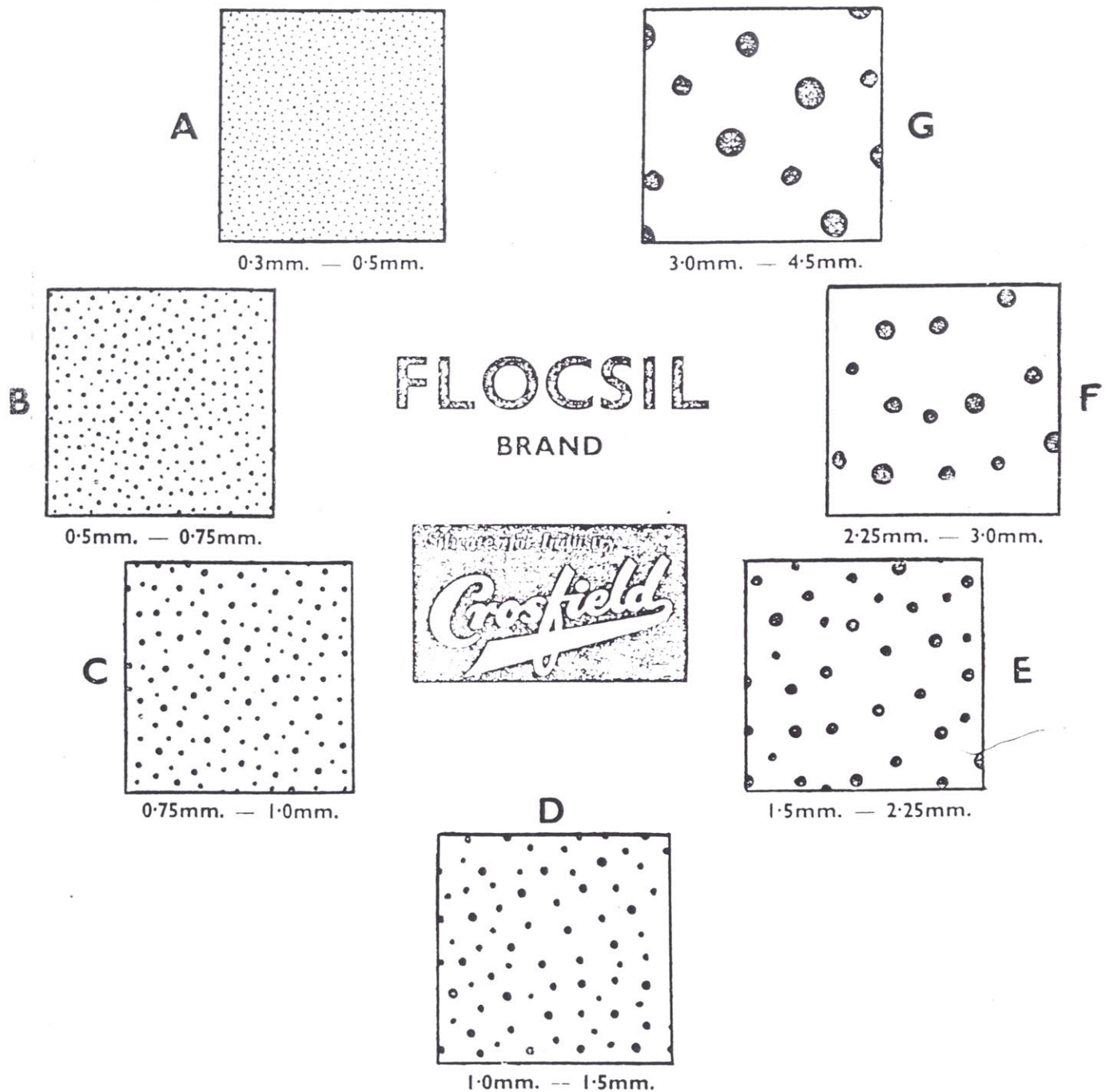
Figure 3.2 Floc Comparator used for estimating floc sizes

The instrument can be used to measure a number of turbidity ranges from 0-0.2, 0-1.0, 0-10, 0-100 and 0-1000 NTU. The turbidity is measured on a turbidity scale of units established for a standard system.

The principle of turbidity measurements, by nephelometry, is as follows: a light beam is directed up through the bottom of a glass cell containing the test sample. Light scattered by particles in the sample is detected by a photomultiplier tube, which is located at a 90° angle from the incident light beam. The intensity of the light which reaches the photomultiplier is proportional to the turbidity of the sample. The photomultiplier converts this light energy into an electrical signal which is then displayed on the instrument meter.

Calibration is done with a formazin primary standard solution and secondary standards, which are turbidity solutions with known turbidity values. These solutions were provided with the instument. Standardisation was always carried out before each series of measurements were done.

Once calibrated, the sample cells were rinsed three times with small volumes of the sample to be measured, and in between every sample. The cells were filled with 25 mls of sample carefully, so as to avoid the entrainment of air bubbles. The outside and bottom of the cell were cleaned thoroughly and dried.

Cells were then inserted into the sample compartment. To ensure the cells were always in the same position, they were marked on the rim with a felt pen, and this mark always faced the same position.

When samples were higher than 100 NTU, a cell riser was added to the bottom of the compartment, as advised in the manual.

Samples were always stirred well to ensure a uniform suspension before examination. Care was taken not to stir so much that air would mix into the sample.

When samples were brought in from the field, or generated during an experiment, analysis was done within 2-4 hours. On rare occassions where

samples were not analysed until 12 or more hours (maximum of 24 hours) they were put into plastic bottles and stored in the dark at 4°C. Samples were allowed to reach room temperature before analysis.

Samples greater than 1000 NTU were diluted within the range of the machine and measured. The result was then multiplied by the appropriate dilution factor.

3.7 Oil and Grease

Oil and grease were determined by the standard method of partition gravimetry (APHA, 1987). Trichlorotrifluoroethane (1,1,2-trichloro-1,2,2-trifluoroethane) was used as the solvent, and is commonly known as Freon (Lovelock Luke, Pty Ltd, Australian distributors). Because Freon has the ability to dissolve organic substances other than oil and grease, an absolute quantity of oil and grease is not actually measured and the actual mass will include soluble proteins, starches, as well as oils and greases.

The partition-gravimetric method relies on dissolved or emulsified oil and grease to be extracted by intimate contact with Freon. This standard method can be found in APHA (1987) and is briefly outlined below.

Samples were collected in a wide-mouthed glass bottle or beaker which had been rinsed with the solvent to remove any traces of detergent. The standard methods recommend one litre of sample, although in most cases 400 mL of experimental samples and 500 mL untreated grab samples were used. In these cases, the samples were made up to 1L with distilled water and the dilution factor was taken into account when determining the levels of oil and grease.

The sample was acidified to about pH 2 with concentrated HCl. The sample was then transferred to a 1L separatory funnel, and the bottle/beaker was then rinsed with 30 mL Freon. This was added to the separatory funnel. The contents of the funnel were then shaken vigorously for 2 minutes. The contents were then placed upright and the two layers allowed to seperate.

The solvent layer was then drained through a funnel with a solvent moistened filter paper, into a clean, tared, quickfit distilling flask.

Extractions were carried out twice more with 30 mL freon. The sample bottle was first rinsed with the solvent, and this was added to the aqueous layer in the separatory funnel, followed by 2 minutes of vigorous shaking, followed by separation. The solvent layer was filtered through the same filter paper. In this way, the three extracts were combined in the tared distilling quickfit flask. The filter paper was then washed with 20 mL solvent. The flask was then immersed in a water bath and attached to a reflux condenser. The solvent was distilled at 70°C (Freon has a boiling point of 47°C). When all the solvent was distilled, the flask was placed in a dessicator for 30 minutes then weighed.

Assuming the solvent is residue free, the increase in the weight of the flask is due to oil and grease.

Calculations were done according to standard methods (APHA, 1987) and they took into account the dilution factors.

3.8 Biological Oxygen Demand (B.O.D)/Dissolved Oxygen (D.O)

The BOD test measures the concentration of oxygen consumed by microorganisms to assimilate and oxidise the organic matter during their metabolism (Eckenfelder, 1980).

The BOD is an empirical test employing standard practices to determine the relative oxygen requirements of certain waters tested. This test is determined by measuring oxygen levels before and after a five day incubation period at 20°C-21°C. The difference between the initial dissolved oxygen concentration and the final concentration is the BOD, measured in mg/L.

Dissolved oxygen can be measured by use of a membrane electrode or by iodometric titration. According to APHA (1987), membrane electrodes "provide an excellent method of dissolved oxygen analysis in polluted waters...and strong effluents". Therefore, electrometric measurements of dissolved oxygen are acceptable as a standard method, which is based on the diffusion rate of molecular oxygen across a membrane.

The membrane electrode method was chosen over the iodometric method because it involves conveniant, easy-to-operate instruments with minimal maintanence; it eliminates errors caused during the normal experimental

procedures for titration; and it can be used with samples high in suspended solids, as these samples tend to interfere with the visual end-point detection of the titration procedures.

Various factors may affect the BOD result: some compounds or conditions may be toxic or detrimental to the inoculum, including heavy metals, pH variations and temperature variations, as well as salinity. The inoculum may be sensitive to these factors and as a result, the organisms may need to be adapted to the water or wastewater under investigation. Also the standard five days used may not be sufficient for some samples, which may require more time for appreciable degradation.

The effluent used in this study was mainly composed of soluble and insoluble starch wastes, which are infact a very rich food source for many types of microbes. Hence, this effluent contained a naturally occuring inoculum.

The materials used and the methods followed were standard methods, as set out in APHA (1987). All chemicals used were analytical grade reagents and were made up as directed. The procedures are briefly described below.

Samples were collected as grab samples and used within three hours of collection. Because the BOD generated from this factory site was approximately between 4000-5000 mg/L (personal communication) the sample was diluted with the aerated water at 1 part per thousand in order to bring the oxygen demand and supply into the appropriate balance.

Dilution water was prepared by adding $MgSO_4$, $CaCl_2$ and $FeCl_3$ solutions at the equivalent of 1 mL/L to distilled water. The water was sparged with air for at least 12 hours before applying the test. The dissolved oxygen concentration in the water was always determined prior to use.

The effluent samples were diluted appropriately in the aerated dilution water and the diluted samples were carefully added into narrow-mouthed glass stoppered BOD bottles of 300 mL capacity. Samples were added until they overflowed, to ensure the very top of the bottle was filled. The BOD bottles had flared mouths and came with tapered and pointed ground-glass stoppers. Care was taken to avoid entraining air bubbles into the sample. The dissolved

oxygen was determined immediately after filling the BOD bottle with the diluted sample. The bottle was then capped with the lid, carefully, to ensure no air entered the bottle.

A water seal was used to ensure no air was drawn into the bottle during incubation and this was done by adding water to the joint of the flared mouth and the lid. To minimise evaporation during incubation, a small plastic cup was placed over the flared mouth of the bottle. Bottles were placed in a fan forced incubator set at 20°C-21°C, in the dark, for five days. Triplicates of the 1:1000 diluted samples were run each time.

At the competion of the five day period, the bottles were removed from the incubator, and the lids very carefully removed. The membrane electrode (calibrated) was inserted carefully, ensuring no air bubbles became trapped inside, and the dissolved oxygen was measured. The BOD was expressed in mg/L (APHA, 1987).

Three bottles containing dilution water only were also run, along with the test bottles. The blanks were run to ensure the quality of the unseeded (uninoculated) dilution water and the cleanliness of the incubation bottles.

Also run in triplicate were bottles containing about 1 mL of starch effluent as the inoculum in a standard glucose-glutamic acid mixture. This test was done to check the quality of the seed. The test was carried out as in APHA (1987).

3.8.1 Measuring D.O.

The dissolved oxygen meter used was the YSI Model 50 Dissolved Oxygen Meter, with the YSI 5700 Series Dissolved Oxygen Probe (YSI Incorporated, Yellow Springs, Ohio, 45387, USA).

The probe and meter were calibrated and used according to the operating instructions supplied. The YSI 5700 Series D.O. probe was the YSI 5720A BOD bottle probe, designed to measure DO in standard BOD bottles. This probe had a stirrer attached to it.

3.9 Total Organic Carbon (T.O.C) Analysis

In order to determine the effectiveness of the coagulation-flocculation testing system on the effluent samples, the method of TOC analysis was used as an indicator of supernatant quality.

According to Eckenfelder (1980), the TOC analyser combusts all the organic matter to carbon dioxide and water. The carbon dioxide gas is then passed through an infra-red analyser which has been sensitised for carbon dioxide, and the response is then recorded.

The TOC method was chosen for a number of reasons. The test is relatively simple to use, producing results almost immediately (about 2-3 minutes per sample) which are reproducable. This allows for large numbers of samples to be tested in relatively short time frames. Also, the use of wet chemicals is greatly reduced, making the overall method more time efficient.

The analysis of TOC was conducted using the Beckman 915B TOC Analyser with the Beckman ten-inch recorder. The machine was operated according to the manufacturers instructions, detailed in the instruction manual provided (Beckman Instructions for the Model 915 B Total Organic Carbon Analyser, Beckman Instruments Inc. Fullerton, CA, USA).

When the sample is injected into the toatal carbon channel (TC) it is combusted at 950°C. All carbon based material is converted to carbon dioxide and water (that is, both organic and inorganic carbon based compounds). Since in most cases only the organic component is of interest, the level of inorganic carbon (IC) needs to be detremined and subtracted from the TC. This is usually done by injecting the same sample into an IC channel, which combusts the sample at 155°C, converting only the inorganic carbon compounds to water and carbon dioxide. The total organic content, then, is the total carbon minus the inorganic carbon, or,

$$TOC = TC - IC$$

The organic carbon standard was made according to APHA (1987). Anhydrous potassium biphthalate was used as the organic carbon standard, with a stock solution made up to 1000 mg carbon/L. This stock was prepared

fresh 1-2 days before it was required and was stored for a maximum of 2 weeks at 4°C. Standards were prepared daily from this stock solution and were made up to 12.5, 25, 50, 75, 100 and 200 mg carbon/L, as required.

All samples appeared homogeneous, with no gross solids or apparent insoluble matter, so there was no need to homogenise or sonicate them. The fresh, unsetled effluent contained fibrous particles, but vigorous mixing broke them up and dipersed them, making the sample homogeneous.

Throughout the study only the TC channel was operational, so a specific method of sample preparation was needed to measure the organic content of the sample only. This method required the removal of the inorganic carbon in the sample.

The method of removing organic carbon was that adapted by APHA (1987). Briefly, 20 mL of sample was acidified with about 1 mL of concentrated sulphuric acid to a pH of about 1-2 in a 50 mL beaker. The sample was then stirred for three minutes with a magnetic stirrer, while a carbon dioxide free nitrogen gas stream was directed into the stirred sample. This was done for both treated and untreated samples.

Samples were then serially diluted, 1:10 and 1:100 to give carbon concentrations which would fall between the range of 25 and 100 mg/L. Samples were diluted in distilled water which had nitrogen gas (carbon dioxide free) bubbled through it for one hour to ensure carbon dioxide was removed. 100 ml of sample was injected into the TC channel and triplicate samples were run. Standards of 12.5, 25, 50, 75, 100 and 200 mg carbon/L were injected in triplicate, before the samples were run, as well as at the completion of the run and once during the running of the samples. This was done to ensure the calibrations were not drifting in the duration of the analysis. A distilled water blank was run to establish any background inorganic carbon levels. For both standard injections and sample injections, peak height and peak area reproducibility was very good.

The standard peak areas were averaged and plotted against the concentration of carbon, mg/L, to give a straight line of best fit, which was used as the standard curve.

Once the areas for the test sample peaks were determined, they were averaged and this value was read off the standard curve and multiplied by the dilution factor, to give the carbon concentration in the original sample, recorded as mg/L.

3.10 Capillary Suction Time (C.S.T.)

CST measurements were made using a CST measuring unit, The Multipurpose Filtration Unit TW 166 (Triton Electronics Ud, Essex, UK). The apparatus and set up are shown in Figure 3.3.

The equipment consists of two plastic rectangular blocks, in which porous filter or chromatography paper is placed in between the two blocks. The upper block has electronic contacts at different radii which are fixed into the block and which are connected to an electrical timer. The upper block also has an opening or hole which accomodates a cylidrical, stainless steel sludge reservoir, in which 20 mL of sample is poured.

The test is started when the sample of sludge is poured into the cylinder. Filtration begins almost immediately and progressively, water begins to flow through the filter paper forming a circular, wet blot. When this expanding blot hits the first electrical contact, an electrical signal starts the timer. As the blot continues expanding, it hits the second contact and stops the timer. The CST is then recorded directly from the timer and reported in units of seconds. In this way, the time interval recorded is a measurement of the filtrability of the sludge.

Because the results of the CST test are sensitive to the type of filter paper used, Whatman No.17 chromatography paper was used only and it was dried at 105°C and stored in a desiccator before use.

Samples of CST were obtained in the following manner: sludges were generated in the usual way in the beakers in the jar tests. A portion of the supernatant was collected after 20 minutes quiescent settling. The beaker was then tilted about 45° and 20 mL of sludge was gently collected with a pipette, with very gentle sucking action to avoid floc/sludge shear. The sample was then carefully dispensed into the metal cylinder of the CST apparatus.
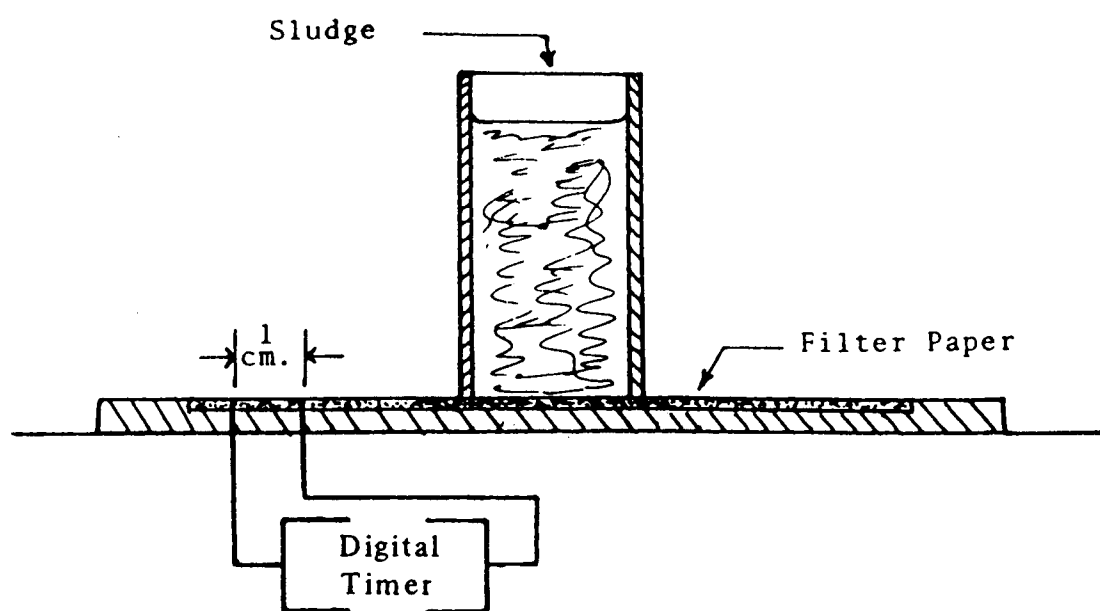
Figure 3.3 Schematic diagram of CST test apparatus
(from Schowyer, 1986, p187)

3.11 Total Solids (TS)

The TS were determined by standard methods (APHA, 1987). The drying dishes (porcelain evaporating dishes) were cleaned and scrubbed thoroughly in Decon 90 or other detergent and rinsed in tap water then twice rinsed in distilled water. The dishes were then dried in a fan-forced oven set at 105°C for 2-3 hours and were then placed in a dessicator until needed. They were weighed immediately before use.

Since it was generally known (private communication and results of very initial work) that the TS content was anywhere up to 4000 mg/L (or 40 mg/10 mL), we limited our sample volume to 20 mL, as APHA (1987) suggest the sample size is limited to a volume which will yield no more than 200 mg residue (in the case of this work, 80 mg residue maximum, if a 20 mL sample is used). This is done to ensure a water trapping crust is not formed, which tends to happen if there is excessive residue in the dish. 20 mL of well mixed sample was placed in the pre-weighed dish and very carefully evaporated on a hot plate to avoid splattering and hence sample loss. The heating was also done gently to avoid the formation of a water trapping crust.

The evaporated sample was then placed in the oven at 105°C for one hour, then cooled slightly and placed in the dessicator until the temperature was balanced. The residue and the dish were then weighed.

The cycle of drying, cooling dessicating and weighing was repeated until constant weight was attained. This increase in the weight of the dish represents the total solids.
To calculate TS,

mg TS/L = (A-B) x 1000/volume (20 mL)

where A= weight of dried residue + dish (mg), and
B= weight of dish (mg).

3.12 Total Suspended Solids (T.S.S)

The TSS was determined by standard methods (APHA, 1987). Briefly Whatman GF/C glass fibre filter papers with pore sizes of 0.45mm were washed with 3 x 20 mL lots of distilled water, dried in a fan-forced oven at 105°C and cooled to a constant weight in the dessicator.

A well mixed sample volume of 20 or 50 mL was filtered through the paper and the paper was again washed with 3 x 10 mL lots of distilled water. The paper was then carefully placed in the 105°C oven for one hour, cooled in a dessicator for one hour to balance temperature and weighed. The cycle of drying, cooling, dessicating and weighing was repeated until constant weight was attained.

The increase in the weight of the filter paper represents the total suspended solids.

To calculate TSS,

$$mg\ TSS/L = (A-B) \times 1000/volume\ (20\ or\ 50\ mL)$$

where A= weight of dried residue + filter paper (mg), and
B= weight of filter paper (mg).

3.13 Total Dissolved Solids (T.D.S)

TDS was only performed on the experiment which assessed changes in the pH and the solids levels of starch effluent at ambient and cold temperatures.

TDS was not conducted for the other experiments because there was no other oven which could be used at 180°C. The oven used for TDS determinations was the oven normally used for TSS and TS and was normally set at 105°C and was needed for that purpose.

Also, because we were evaluating coagulation efficiency of our trials, TSS was used as one of the main parameters to test this efficiency, and so the use of TDS was abandoned.

TDS was determined by standard methods (APHA, 1987) as follows:

A well mixed sample volume of 50 mL was filtered through a Whatman GF/C (0.45mm) filter disc. The sample on the filter paper was used for suspended solids determination, but the filtrate (approximately 45 mL) was collected (free from the pre-washing and post washing of the filter discs) and stored in the freezer until ready for analysis (8 days maximum), depending on the time of sample collection.

The reason the samples for TDS were stored in the freezer was for conveniance. Because the oven was constantly used at 105°C, a narrow time slot could be borrowed where the oven could be turned up to 180°C. In this way, I could dry all my samples in one run and return the oven to its much needed temperature of 105°C. Samples were stored in the freezer to inhibit any microbial spoilage.

When ready for analysis, samples were allowed to thaw and reach room temperature. Samples were mixed thouroughly and duplicates were added (20 mL x 2) to tared porcelain evaporation dishes. The dishes had been washed well and dried to 180°C for 2 hours in the oven, stored in the dessicator and weighed immediately before use. The samples in the dishes were evaporated gently to dryness on a hot plate and dried in the oven at 180°C for one hour, then cooled in a dessicator to balanced temperature and weighed. This drying, cooloing, dessicating and weighing cycle was repeated until constant weight was attained.

To calculate TDS,

$$\text{mg TDS/L} = (A-B) \times 1000/\text{volume (20 mL)}$$

where A= weight of dried residue + dish (mg), and
B= weight of dish (mg).

3.14. Synthetic Polyelectrolytes

A number of synthetic polyelectrolytes trialled were supplied by Allied Colloids, Pty Ltd, Wyong Australia. They were the cationic polyelectrolytes, Zetag 57, 63, 87 and 92; anionic flocculants, Magnafloc 155, 156, 336, 919

and 1011; and a non-ionic polyelectrolyte, Magnafloc 333. The technical and processing specifications of these polymers can be found in Appendix A. All polymers came in a white-free flowing granulated powder or beads.

The polyelectrolytes were prepared according to the method given in the technical and processing data, Appendix A. Briefly, 0.5 g of powder was added into a 200 mL plastic bottle, followed by 3 mL of absolute ethanol. 97 mL of distilled water was rapidly poured, the bottle capped tightly, and vigorously shaken for 15 seconds. The bottle was allowed to stand for 1 hour, with vigorous hand shaking every 5 minutes. This solution served as the stock solution.

The cationic polymer stock solutions were stored for a maximum of one week, and the non-ionic and anionic stock solutions were stored for 2 days maximum, in the refrigerator at 4°C. For use in the coagulation-flocculation trials, stock solutions were allowed to reach room temperature, and were then diluted to the desired level with distilled water. Stock solutions were taken up with a syringe, slowly and carefully to avoid excessive shear forces which can disrupt the polymer chain. Likewise, when the solution was expelled from the syringe, it was also done steadily. All solutions made from the stock solutions were made fresh daily, in distilled water.

Feed solutions were also added to the test sample with a syringe.

3.15 Chitosan

The chitosan used was a practical grade from crab shells, and was in a granular/flake form, with a grey-brown colouration. It was obtained from the Sigma-Aldrich Corporation, Milwaukee, WI, USA.

Chitosan was prepared for use following Bough (1975a), by dissolving 5g in 1L of 1% acetic acid, and this served as the stock solution.

3.16 Aluminium sulphate (Alum)

Alum was supplied as a technical grade, white, granular solid from Ajax Chemicals. The technical information on alum can be found in Appendix B. A 10g/L stock solution was made by dissolving 10g to 1L of distilled water. The

stock solution was kept for 4 weeks maximum at 4°C in the dark. Appropriate feed solutions were prepared by diluting the stock solutions with distilled water to the desired levels. Feed solutions were always made fresh when required.

3.18 Ferric sulphate

Ferric sulphate was supplied by Tioxide (Tioxide Australia, Pty Ltd, Launceston, Tasmania) in the form of a red-brown viscous liquid, under the product name Ferriclear. The chemical and safety specifications of Ferriclear can be found in Appendix B. The active constituent of Ferriclear, the ferric sulphate component, comprised 45% of Ferriclear, w/w. Therefore 22.2g of Ferriclear (or 13.8 mL) were made up to 1L of distilled water, to give a 10g/L stock solution, and used fresh. The stock solution, when made fresh, was kept refrigerated for 3 days maximum at 4°C in the dark.

Appropriate feed solutions were prepared by diluting the stock solution with distilled water to the desired concentrations.

CHAPTER 4. <u>RESULTS</u>

<u>4.1 STARCH EFFLUENT CHARACTERISTICS</u>

In the course of the project a total of twelve effluent profiles were conducted in order to determine the typical or characteristic levels of organic waste components in the effluent. In the course of the sampling and analysis only three major parameters were looked for: Total Suspended Solids (mg/L), Total Solids (mg/L), and, Turbidity (NTU). These parameters also acted as indicators of the effluent quality. The approximate rate of discharge of the effluent was 30,000 litres per hour.

Tables B1-12 of Appendices section reveal effluent profiles taken over a 14 month period. The profiles taken ranged from 24 hour profiles to 72 hour profiles. Table 1 in this Chapter 4 lists the average of each of the profiled periods along with their corresponding standard deviations and relative standard deviations (which are the standard deviations expressed as a percentage of the average result). The relative standard deviations were calculated in order to judge the size of the standard deviations, both for each profile and in comparing various profiles.

Table B1 is a profile for a 24 hour period in late June, 1990. In this time frame, from start to finish, there is a slight drop in all the parameters tested, with 2280 mg/L suspended solids at the start of the 24 hour run ending with 2030 mg/L suspended solids. Table 4.1 shows this change was in fact slight as there are low standard deviations for all the parameters: 4.6%, 3.5%, and, 10.5% for suspended solids, total solids and turbidity respectively.

Table B2, a 24 hour profile conducted in early July, 1990, is similar to Table B1. Both the total solids and the suspended solids did not deviate highly as this is reflected by their low relative/standard deviations in Table 4.1. The turbidity levels increased between 12 p.m. and 8 p.m. to 1600-1500 NTU and dropped to about 900 NTU at the end of the profile period.

A 48 hour profile was conducted in late July, 1990 (Figure 4.1(A) and Table B3). A very high peak for all parameters was recorded near the completion of the 48 hour period (4 a.m.). The suspended solids levels arose to about 3330 mg/L and the total solids peaked at 4010 mg/L. The background or average

A

B
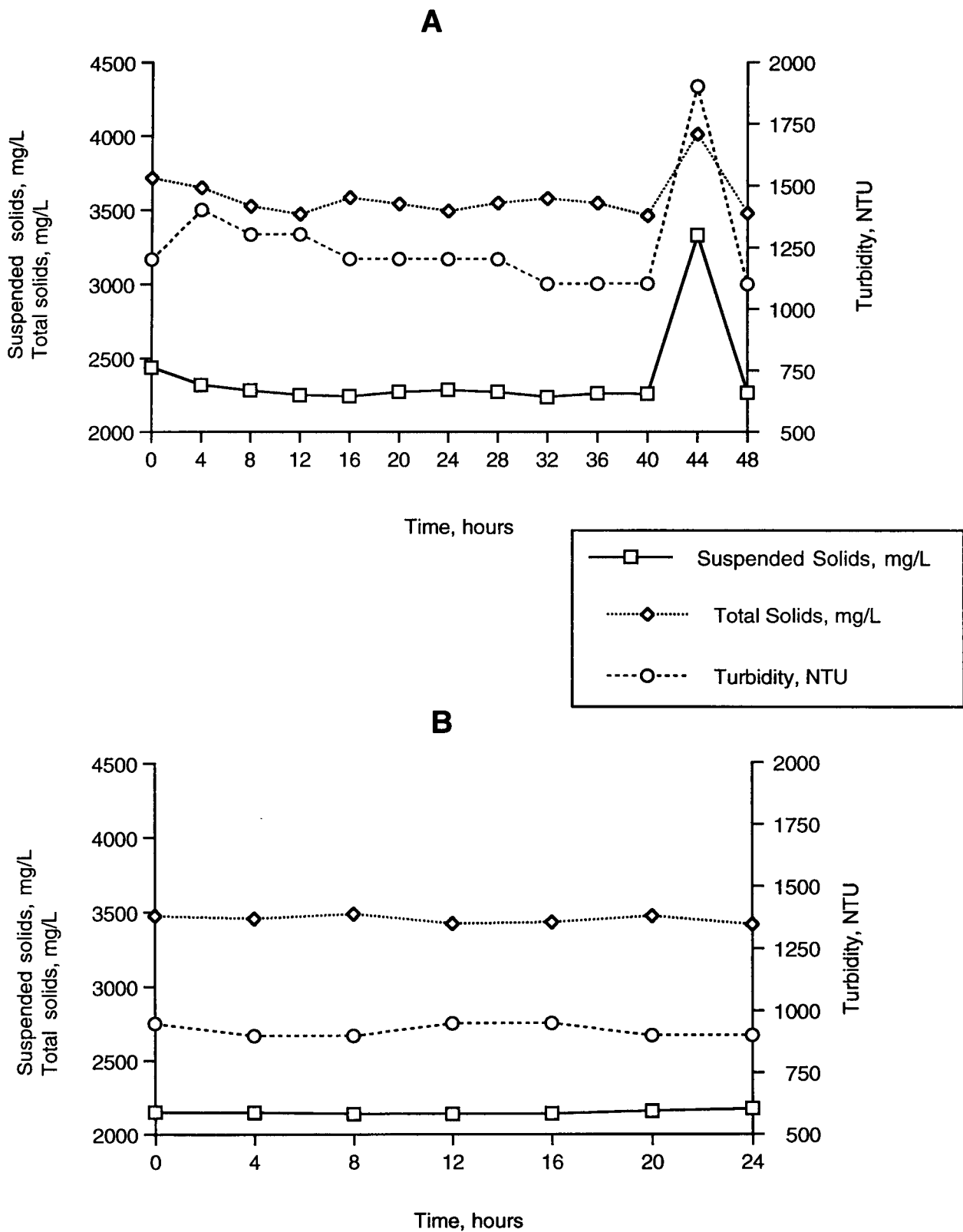
Figure 4.1 (A) 48 hour effluent profile for late July, 1990,

(B)24 hour effluent profile for mid June, 1991.

Note: 0 hour denotes an 8 a.m. start

reading for these two parameters was 2361 mg/L and 3583 mg/L. Turbidity also peaked at 1900 NTU against an average reading of 1254 NTU.

A 72 hour profile was conducted about 3 months later in early November (Table B4). The average and standard deviation results of this profile are comparatively similar to the three previous profiles, as seen in Table 4.1. Near the completion of this cycle, at 12 a.m. the total solids rise sharply to 4235 mg/L (against an average of 3355 mg/L) and the turbidity rises sharply to 2000 NTU (against an average of 1050 NTU). At this point the suspended solids do not rise, remaining at 2203 mg/L (against an average of 2226 mg/L).

Table B5 is a 24 hour effluent profile conducted in early December, 1990. All parameter readings were very low compared to the previous profiles, with averages of 570 mg/L, 1890 mg/L and 200 NTU for suspended and total solids and turbidity, respectively.

Table B6 is the effluent profile carried out in late January, 1991. The average results, shown in Table 4.1, for suspended and total solids, 1931 mg/L and 3013 mg/L are comparable to the earlier profiles, but are slightly lower. Turbidity, though, was much lower at 700 NTU compared to an approximate average of 1100 NTU for the previous profiles.

A 48 hour effluent profile (Table B7) was conducted in early February, 1991 and mid March, 1991 (Table B8). The total averages for each parameter are similar for each profile and comparatively similar to most of the previously conducted profiles (see Table 4.1). During the profile of mid March, at 12 a.m. there is a sharp rise recorded in all parameters with a rise to 2890 mg/L suspended solids (average is 2109 mg/L), 4270 mg/L total solids (average is 3204 mg/L) and 2000 NTU (average is 904 NTU). By 4 a.m. the levels were still relatively high compared to the averages although they had dropped slightly.

The 24 hour effluent profile for late April, 1991 (Table B9) yielded similar suspended solids and total solids results comparable to the other profiles, with average readings of 2162 mg/L and 3178 mg/L respectively. Table 1 indicates low relative standard deviations for these parameters. The turbidity was initially at a comparable level, 1200 NTU, to the typical effluent profiles. By the completion of the sampling cycle the turbidity had decreased to 300 NTU. This

Table 4.1. Total Averages, including standard deviations and % relative standard deviations
for all sampling periods, 1990-1991 (see appendices for complete data on all profiles)

	A	B	C	D	E	F	G	H	I	J
1		AVERAGE	STANDARD	% REL. STD	AVERAGE	STANDARD	% REL. STD	AVERAGE	STANDARD	% REL. STD
2	SAMPLE PERIOD	SUSPENDED SOLIDS mg/L	DEVIATION	DEVIATION	TOTAL SOLIDS mg/L	DEVIATION	DEVIATION	TURBIDITY NTU	DEVIATION	DEVIATION
3	Late June 1990, 24 hours	2173	101	4.6	3359	119	3.5	1093	115	10.5
4	Early July 1990, 24 hours	2179	40	1.8	2944	77	2.6	1271	237	18.6
5	Late July 1990, 48 hours	2361	284	12	3583	141	3.9	1254	206	16.4
6	Early November 1990, 72 hours	2226	50	2.2	3355	235	7	1050	259	24.7
7	Early December 1990, 24 hours	570	116	20.4	1890	187	9.9	200	27	13.5
8	Late January 1991, 24 hours	1931	55	2.8	3013	122	4	700	169	24.1
9	Early February 1991, 48 hours	2109	112	5.3	3627	152	4.2	1108	228	20.6
1 0	Mid-March 1991, 48 hours	2174	256	11.8	3204	504	15.7	904	386	42.7
1 1	Late April 1991, 24 hours	2162	75	3.5	3178	259	8.1	779	328	42.1
1 2	Late May 1991, 24 hours	1176	419	35.6	2065	592	28.7	303	213	76.2
1 3	Mid-June 1991, 24 hours	2149	12	0.6	3448	27	0.8	921	25	2.7
1 4	Early July 1991, 24 hours	2161	64	3	3118	169	5.4	643	203	31.6
1 5										

Figure 4.2 **(A)** Changes in suspended solids, mg/L and soluble solids, mg/L of starch factory effluent over time under cold and ambient storage conditions; **(B)** Changes in pH of starch factory effluent over time under cold and ambient storage conditions

decrease in turbidity corresponded to the decreases in total and suspended solids but the decrease was not as prominent for these latter two parameters.

A profile conducted in late May (Table B10) reveals low levels of all the parameters tested. The sampling cycle begins with low concentrations and low turbidity. By 24 hours both suspended and total solids have reached typical concentrations although turbidity maintains low levels. Near the mid-way point of sampling (8 a.m.-12 p.m.) all parameters reach low concentrations: 703 mg/L suspended solids, 1373 mg/L total solids, and, 95 NTU. The result of these extreme changes in levels is reflected in the very large standard deviations and relative standard deviations (Table 4.1).

Mid June 1991 (Figure 4.1(B) and Table B11) showed typical suspended and total solids concentrations and typical turbidity results. These levels changed very little over the 24 hour period. Table 4.1 reveals very small standard deviations.

The final effluent profile conducted was a 24 hour profile in early July, 1991 (Table B12). This profile was very similar to the other profiles, with averages of 2161 mg/L and 3118 mg/L for the suspended and total solids results, respectively. Over the 24 hour period there were slight drops in the concentrations of these two parameters. The greatest drop was with turbidity which yielded initial readings of 950 NTU and final readings of 250 NTU.

4.2 PRELIMINARY EVALUATION OF POLYELECTROLYTES

A series of jar tests were conducted in order to evaluate the efficiency of a number of cationic polyelectrolytes (Zetag range), anionic polyelectrolytes (Magnafloc range) and one non-ionic polyelectrolyte, Magnafloc 333. The efficiency of these polymers was indicated by the suspended solids (SS) removed, as a percentage of the initial suspended solids, and by the final floc sizes attained at 20 minutes flocculation. The initial SS were determined by running a parallel jar test with no polymer additions. At the end of the 20 minutes flocculation period, the settleable solids were allowed to settle for 20 minutes. The top 200 mL was then decanted and used for SS analysis. This supernatant of the untreated effluent was taken as the initial SS (100%). Supernatant from treated effluent was compared as a % removed to the untreated effluent.

Tables C1-C5 in the Appendices display both the raw and calculated data for %SS removal, as well as the final floc sizes attained (mm) for all the polymers tested at pH 4, 6, 7, 8, 10. Figures 4.3 and 4.4 represent Zetag 92 and Magnafloc 336 respectively, and graphically display the changes in floc size and changes in %SS removal with increasing polymer dose and increasing pH. Both figures only show results for trial 1 only.

A second trial was conducted, 7 days further to the first one. This trial was carried out to confirm the behaviour of the polymers and to ascertain if effluent obtained on a different day will display similar trends in both floc growth and SS removal. Table C6 shows the initial supernatant SS of both trials varied, with about 305 mg/L SS in trial 1 and about 210 mg/L SS in trial 2. Changing the pH also led to changes in the supernatant SS.

When 7.5 mg/L polyelectrolyte was used as the primary coagulant, it appeared there was slight resistance for the supernatant to filter through the filter paper. When it did filter through, there was a very thin, slightly slimy film coated on parts of the paper.

4.2.1 Zetag 92 (Figure 4.3)

Increasing the dosage from 2.5 mg/L to 7.5 mg/L at all pH levels led to an increase in the SS removed. At pH 4, Zetag 92 removed very little at 7.5 mg/L, with 12.5% SS removed in trial 1 and 12.8% SS removed in trial 2. Floc sizes increased slightly from 0.4 mm (the initial floc size) to only 0.52 mm at 7.5 mg/L for both trials.

At pH 6, there is a larger improvement with a total of 17.2% SS removed for trial 1 and 20.5% SS removed for trial 2 when dosed with 7.5 mg/L. At pH 6, doubling the dosage only led to slight increases of SS removals. About 9% SS was removed with 2.5 mg/L, 13.3% with 5.0 mg/L and 17.2% with 7.5 mg/L. Removals for trial 2 are very similar (see Table C2).

Changing the pH from 4 to 6 led to very large increases in removals for all doses. For 2.5 mg/L SS removals went from 5.5% to 9.4% (an increase in efficiency of about 71% for trial 1 and about 63% for trial 2). For 5.0 mg/L it was an efficiency increase of 90% for trial 1 and 43% for trial 2 and at 7.5 mg/L it was 38% increase for trial 1 and a 60% increase for trial 2. Floc sizes also

increased with an increase in pH from 4 to 6. This was more evident in trial 2, which consistently showed larger floc sizes. At pH 6, final floc sizes attained at 7.5 mg/L were 0.63 mm, for both trials, compared to 0.52 mm at pH 4 (an increase of about 20% in floc size).

Increasing the pH to 7 continued the improvement in efficency of SS removal and floc growth. For trial 1 the increase in SS removal efficiency was approximately 9%, 29%, and, 32% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. For trial 2 the increase in SS removal efficency was approximately 50%, 29%, and, 25% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. Increasing the levels of polymer by three times only led to a doubling in % SS removal, with 10.2% and 22.7% SS removed at 2.5 mg/L and 7.5 mg/L, respectively for trial 1. For trial 2 15.6% and 25.6% SS removals were recorded at 2.5 mg/L and 7.5 mg/L, respectively. Relative to pH 6 floc sizes did increase with 0.76 mm at 7.5 mg/L, trial 1 and 1.1 mm at 7.5 mg/L, trial 2.

Increasing the pH from 7 to 8 led to much SS removals and larger flocs for all dosage levels. At trial 1, 2.5 mg/L resulted in 19% SS removal, doubling the dosage to 5.0 mg/L led to about 25% SS removal and tripling the dosage to 7.5 mg/L led to 35% SS removals. Trial 2 showed higher removals: 2.5 mg/L resulted in 25% SS removal, doubling the dosage to 5.0 mg/L led to about 28.6% SS removal and tripling the dosage to 7.5 mg/L led to 39.3% SS removals. Compared to pH 7, the overall relative increases in efficiency corresponded to 85% for 2.5 mg/L, 48% for 5.0 mg/L and 55% for 7.5 mg/L, trial 1. At trial 2 the increases were 60% for 2.5 mg/L, 71% for 5.0 mg/L and 54% for 7.5 mg/L. Floc sizes also attained a larger final size at pH 8.

Figure 4.3 shows that at pH 10 there is a big increase in both floc sizes and SS removals. At 2.5 mg/L, 33.9% SS was removed for trial 1 and about 33% SS was removed in trial 2. Doubling the dose to 5.0 mg/L led to a big increase in SS removal to 57.3% and 56.1% SS removals. Further increasing the dose to 7.5 mg/L led to very slight increases in SS removals with 58.9% and 61% SS removed for trial 1 and trial, respectively. The relative efficiency increases from pH 8 to pH 10 were very large. For trial 1 they were, 79%, 126%, and, 67% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. For trial 2 they were, 32%, 96%, and, 55% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. Floc sizes also attained much larger, final sizes, especially for 7.5 mg/L, where flocs attained sizes of 1.9 mm and 2.3 mm in trial 1 and trial 2, respectively.

Figure 4.3 Coagulation of starch wastewaters using Zetag 92 as a primary coagulant. **(A)** Removal efficiency of suspended solids (%) versus pH for three different coagulant dosages; **(B)** Final floc size attained, mm, at 20 minutes flocculation versus pH for three different coagulant dosages. Results are for Trial 1, Day 1 only.

4.2.2. Zetag 87

Zetag 87 performed similarly to Zetag 92. SS removals and floc sizes were lowest at pH 4 and pH 6. At pH 4 only 13.3% was removed at 7.5 mg/L, trial 1, compared to 12.8% removal at 7.5 mg/L, trial 2. No floc growth was observed for trial 1 and only minimal floc growth occurred in trial 2 (0.4 mm to 0.52 mm at 7.5 mg/L). At pH 6 19.5% SS was removed using 7.5 mg/L, trial 1 and 17.9% was removed using 7.5 mg/L, trial 2. Floc sizes grew to 0.63 mm at 7.5 mg/L in trial 1 and in trial 2.

Compared to pH 6, only slight increases were observed in the removals of suspended solids at pH 7. About 12.5% SS was removed at 2.5 mg/L, compared to 24.2% SS removed at 7.5 mg/L, trial 1. Similarly, about 14.4% SS was removed at 2.5 mg/L, compared to 26.7% SS removed at 7.5 mg/L, trial 2. Trial 2 yielded the largest flocs with 0.76 mm, 0.88 mm, and, 1.1 mm, compared to 0.63 mm, 0.76 mm, and, 0.76 mm in trial 1, for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively.

A big improvement in SS removals and final floc sizes was observed at pH 8. Trial 1 yielded 17.2% SS removals at 2.5 mg/L, 23% SS removals at 5.0 mg/L and about 30.3% SS removals at 7.5 mg/L. Trial 2 yielded slightly higher removals with 20.2% SS removals at 2.5 mg/L, 27.4% SS removals at 5.0 mg/L and 38.1% SS removals at 7.5 mg/L. Relative to pH 7, the increase in efficiency with an increase in pH is approximately, for trial 1: 38% for 2.5 mg/L, 40% for 5.0 mg/L and 25% for 7.5 mg/L. For trial 2 the efficiency increase was about 40% for 2.5 mg/L, 64% for 5.0 mg/L and 43% for 7.5 mg/L. Floc sizes were reported at 0.76 mm, 0.88 mm and 0.88 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively, for trial 1. Floc sizes were reported at 0.88 mm, 0.1.1 mm and 1.1 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively, for trial 2. Relative to pH 7 floc sizes increased by 16% and 0% for trial 1 and trial 2, respectively, with 7.5 mg/L.

High removals were obtained at pH 10 for both trials. Zetag 87 was slightly less efficient than Zetag 92 for both SS removals and final floc sizes attained. About 33% SS removal and 32% SS removal at 2.5 mg/L for trial 1 and trial 2 respectively were obtained. Doubling the dosage to 5.0 mg/L led to 52.4% SS removal and 53.7% removal for trial 1 and trial 2 respectively. Further increasing the dosage to 7.5 mg/L only led to a very slight increases in removals with 54.8% SS removal and 58.5% removal for trial 1 and trial 2

respectively. Compared to pH 8, pH 10 was able to achieve higher levels of suspended solids.

The increase in efficiency corresponding with an increase in pH from 8-10 was 92%, 128% and 81% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively, in trial 1. In trial 2 the increase in efficiency corresponding with an increase in pH from 8-10 was 57%, 96% and 54% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively.

At pH 10 floc sizes were larger, with trial 2 producing the larger flocs. Trial 1 produced 1.1 mm, 1.3 mm and 1.9 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. Trial 2 produced 1.3 mm, 1.6 mm and 1.9 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. The relative increases in floc sizes compared to pH 8 were for trial 1 45%, 48% and 110% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. For trial 2 they were 48%, 45% and 73% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively.

4.2.3 Zetag 57

Zetag 57 performed similarly to Zetag 87 and Zetag 92, although the final floc sizes attained were slightly smaller. At pH 4 and pH 6, the smallest floc sizes were produced and removal of suspended solids was least efficent. At pH 4 floc growth does not occur for trial 1 and growth is minimal in trial 2. At pH 6, trial 1, floc sizes attained a size of 0.52 mm at 5.0 mg/L but diminished to 0.4 mm at 7.5 mg/L. At trial 2 floc growth was static at 0.52 mm at all doses.

Slight increases in SS removals and floc growth were observed at pH 7. At 2.5 mg/L 13.3% and 12.2% SS were removed for trial 1 and trial 2 respectively. At 5.0 mg/L 15.6% and 20% SS were removed for trial 1 and trial 2 respectively. At 7.5 mg/L 25% and 27.8% SS were removed for trial 1 and trial 2 respectively. Increasing the dosage from 2.5 mg/L to 5.0 mg/L only led to slight increases in SS removal but increasing the dose to 7.5 mg/L led to much higher removals.

Flocs at pH 7 were relatively larger than flocs formed at pH 6. At pH 7 for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L flocs attained a size of 0.52 mm, 0.63 mm and 0.63 mm respectively for trial 1, and 0.63 mm, 0.76 mm, and, 0.88 mm respectively for trial 2.

Increasing the pH to pH 8 led to bigger final floc sizes and larger levels of SS removals. Trial 1 yielded about 20%, 20% and 31% SS removals for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Trial 2 yielded higher removals at about 21%, 26% and 38% SS removals for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

Relative to pH 7 increasing the pH to pH 8 led to increases in efficiency corresponding to 48%, 26% and 24% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively, in trial 1. In trial 2 the increase in efficiency corresponding with an increase in pH from 7-8 was 61%, 68% and 52% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. Floc sizes were reported at 0.52 mm, 0.63 mm and 0.63 mm at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively for trial 1. At trial 2 floc sizes were reported at 0.88 mm, 1.1 mm and 1.1 mm at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively.

At pH 10 highest SS removals were attained as well as the highest floc sizes for both trials, with trial 2 yielding the highest results. At 2.5 mg/L 28.2% and 34.1% SS were removed for trial 1 and trial 2 respectively. Increasing the dosage to 5.0 mg/L led to 39.5% and 50% SS removals for trial 1 and trial 2 respectively. Further increasing the dose to 7.5 mg/L led to 54.8% and 58.5% SS removals for trial 1 and trial 2 respectively. Relative to pH 8, pH 10 had the following increases in efficiency for trial 1 and trial 2 respectively: at 2.5 mg/L, 43% and 59%; at 5.0 mg/L, 101% and 91%; at 7.5 mg/L, 76% and 54%. Floc sizes were also larger at pH 10 compared to pH 8. Trial 1 yielded 0.88 mm, 1.1 mm and 1.3 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. Trial 2 yielded 1.1 mm, 1.3 mm and 1.9 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively. Relative to pH 8 the increases in floc sizes for pH 10 were for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L, respectively, 16%, 25% and 48% for trial 1 and 45%, 18% and 73% for trial 2.

4.2.4 Zetag 53

Zetag 53 performed similarly to all the other Zetag polymers. At pH 4, increasing the dosage from 2.5 mg/L to 7.5 mg/L led an increase from 2.3% SS removed to 11.7% SS removed for trial 1 and 6.4% SS removed to 14.1% SS removed, respectively for trial 2. Flocs attained only 0.52 mm for both trials.

Increasing the pH from pH 4 to pH 6 resulted in higher SS removals across all doses. At 2.5 mg/L about 8% SS was removed compared to approximately

16% SS removal at 7.5 mg/L in trial 1. Trial 2 produced higher SS removals at 2.5 mg/L, with 15.4% SS removed at 2.5 mg/L and only 16.7% SS removed at 7.5 mg/L. Trial 2 floc sizes remained as at pH 4 levels ie: at 0.52 mm, despite an increase in the dose. In trial 1 though, floc growth was only marginal with max levels of 0.52 mm.

Larger removals and larger flocs were obtained when the pH was increased to pH 7. Dosing with 5.0 mg/L led to 18% SS removals and 7.5 mg/L led to 24.2% SS removals, for trial 1. In trial 2, dosing with 5.0 mg/L led to 18.9% SS removals and 7.5 mg/L led to 27.8% SS removals. Flocs were observed to grow from 0.52 mm at 2.5 mg/L to 0.63 mm at 5.0 mg/L and 7.5 mg/L in trial 1 and in trial 2, floc growth was 0.63 mm, 0.76 mm and 0.88 mm when doses were increased from 2.5 mg/L, 5.0 mg/L and 7.5 mg/L.

Increasing the pH from pH 7 to pH 8 led to slight increases in floc sizes and SS removals. In trial 1, 2.5 mg/L, 5.0 mg/L and 7.5 mg/L led to 21.3%, 19.7%, and 31.1% SS removals, respectively. This corresponded to a relative increase in efficency with an increase in pH, of 30%, 9% and 29% respectively. In trial 2, 2.5 mg/L, 5.0 mg/L and 7.5 mg/L led to 23.8%, 26.2%, and 36.9% SS removals, respectively. This corresponded to a relative increase in efficiency with an increase in pH, of 79%, 39% and 33% respectively.

Floc growth was recorded at 0.76 mm, 0.88 mm and 0.88 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively, in trial 1. Trial 2 yielded larger final floc sizes with 0.88 mm, 1.1 mm and 1.1 mm for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

Relatively high levels of SS removals and final floc sizes were produced at pH 10. About 26%, 40% and 51% SS were removed when dosed with 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Higher results were obtained in trial 2 with 32.9%, 46.3% and 53.7% SS were removed when dosed with 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

Relative to pH 8 the increase in efficency at 2.5 mg/L was 21% and 38% for trial 1 and trial 2 respectively; at 5.0 mg/L was 105% and 77% for trial 1 and trial 2 respectively; and at 7.5 mg/L was 63% and 46% for trial 1 and trial 2 respectively. Large flocs also developed with larger sizes in trial 2 (1.3 mm and 1.6 mm at 2.5 mg/L and 5.0 mg/L, respectively).

4.2.5 Magnafloc 155

The SS removals at pH 4 for both trials are low, with 3.1%, 8.6% and 14.1% SS removals at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively for trial 1. Trial 2 produced similar results with 9%, 10.3% and 14.1% SS removals at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Flocs produced were small at 0.52 mm for both 5.0 mg/L and 7.5 mg/L in both trials.

At pH 6 only 7.5 mg/L produced significant SS removals with 23.4% and 23% SS removals for trial 1 and trial 2 respectively. Changing the pH from pH 4 to pH 6 increased the efficency of 7.5 mg/L by 66% for trial 1 and 63% for trial 2. Floc sizes for trial 1 at pH 6 were similar to pH 4, at 0.52 mm for 7.5 mg/L Magnafloc 155. Trial 2 produced the larger flocs at 0.63 mm at all doses.

Doubling the dose from 2.5 mg/L to 5.0 mg/L had little effect on SS removals at pH 6. This was also observed at pH 7.

Increasing the pH from pH 6 to pH 7 led to slight increases in the efficency of Magnafloc 155 to remove SS for both trials, although in trial 1 at 7.5 mg/L there was no change, ie: Magnafloc 155 behaved similarly at pH 7 and pH 6. For trial 1 removals were 14.8%, 17.2% and 23.4% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Increasing the pH to pH 7 represented an increase in efficiency of 57%, 46% and 0% at the respective doses. Trial 2 produced higher removals with 20%, 21.1% and 30% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Increasing the pH to pH 7 represented an increase in efficiency of 72%, 64% and 30% at the respective doses.

Floc sizes increased at pH 7. In trial 1 they reached 0.76 mm at 7.5 mg/L and 1.1 mm at 7.5 mg/L in trial 2. This represented an overall increase in floc sizes of 46% in trial 1 and 75% in trial 2 for 7.5 mg/L, when the pH was increased from pH 6 to pH 7.

Further increasing the pH to pH 8 had little effect on the removal of SS and floc size only increased marginally. Removals were lower in trial 2 at pH 8 than at pH 7. At pH 8, the removals of SS were for trial 1, 16.4%, 16.4% and 25.4% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. For trial 2 they were 19%, 20.2% and 26.2% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. In terms of increase in efficiency of Magnafloc 155 with an increase in pH from pH 7 to pH 8, they are 11%, -5% and 9% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L

respectively for trial 1 and for trial 2 they are -5%, -4% and -13% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

Floc sizes remained at 0.76 mm at all doses for trial 1 and reached 0.88 mm in trial 2. Trial 2 produced smaller flocs at pH 8 (0.88 mm) than at pH 7 (1.1 mm) for 7.5 mg/L dosage.

Very little change occurred when the pH was increased from pH 8 to pH 10. Floc sizes in trial 2 dropped to 0.76 mm for all doses (from 0.88 mm at pH 8). The SS removal efficency of Magnafloc 155 dropped slightly at pH 10, removing less SS at 2.5 mg/L and 7.5 mg/L in both trials compared to pH 8.

4.2.6 Magnafloc 156

At pH 4 dosing the effluent with Magnafloc 156 at 2.5 mg/L removed 6.3% SS in trial 1 and 5.1% in trial 2. Increasing the dose to 5.0 mg/L led to A small increase of SS in trial 1, with 8.6% SS removed and 12.8% SS removed in trial 2. When the dose was increased further to 7.5 mg/L, 14.8% SS was removed in trial 1 and 17.9% in trial 2. Floc sizes increased from 0.4 mm to 0.52 mm in both trial 1 and trial 2.

Increasing the pH from pH 4 to pH 6 led to larger SS removals and slight increases in floc size. At pH 6 increasing the dose from 2.5 mg/L to 5.0 mg/L did not lead to a change in the efficiency of SS removal. When the dose was increased to 7.5 mg/L 22.6% and 28% SS were removed in trial 1 and trial 2 respectively. Increases in floc size were marginal , with 0.63 mm attained in both trials, when dosed with 7.5 mg/L Magnafloc 156.

Increasing the pH from pH 6 to pH 7 led to higher removals of SS, but only at doses of 2.5 mg/L and 5.0 mg/L. At 7.5 mg/L the SS removal results were similar to those of pH 6. In trial 1 14.1%, 17.2% and 25% were removed at doses of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Relative to pH 6, the efficiency increase of Magnafloc 156 with an increase in pH was 28%, 46% and 11% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. In trial 2 18.9%, 24.4% and 32.2% were removed at doses of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Relative to pH 6, the efficency increase of Magnafloc 156 with an increase in pH was 6%, 58% and 15% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

Floc sizes increased slightly compared to pH 6 for trial 1, but were significantly larger in trial 2, with an increase in floc size from 0.63 mm at 7.5 mg/L at pH 6 to 1.1 mm at 7.5 mg/L at pH 7. This represent an overall 75% increase in floc sizes by increasing the pH from pH 6 to pH 7.

Increasing the pH from pH 7 to pH 8 led to little change in the performance of Magnafloc 156, for all doses. For trial 1 SS removed from the lowest to the highest doses were 14.8%, 16.4% and 24.6%. This compared to pH 7, represents a 5% increase in efficiency for 2.5 mg/L a 5% decrease in efficiency for 5.0 mg/L and a 2% decrease in efficiency for 7.5 mg/L. For trial 2 SS removed from the lowest to the highest doses were 19%, 19% and 23.8%. This compared to pH 7, represents a 0% increase in efficiency for 2.5 mg/L a 22% decrease in efficiency for 5.0 mg/L and a 26% decrease in efficiency for 7.5 mg/L. Floc growth remained static for trial 1, at 0.76 mm at all doses and decreased at trial 2. At 7.5 mg/L, trial 2, 0.88 mm final floc sizes were attained compared with 1.1 mm in the equivalent conditions at pH 7.

When the pH was increased from pH 8 to pH 10, little changes in the efficiency of Magnafloc 156 to remove SS was observed. For trial 1, 16.1%, 20.2% and 21.8% SS were removed at doses of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Compared to trial 1 at pH 8 this represented an overall efficiency change of 8% increase in efficiency for 2.5 mg/L a 23% decrease in efficiency for 5.0 mg/L and 11% decrease in efficiency for 7.5 mg/L. The floc sizes in trial 1 of pH 10 attained a size of 0.76 mm, which was the same size attained in trial 1, pH 8.

The changes in SS for trial 2, pH 10 were not significant compared to trial 2, pH 8. At pH 10, for the lowest to highest dose, 19.5%, 23.2% and 25.6% SS were removed. Compared to trial 2 of pH 8, this represented an overall efficiency change of 3% increase in efficiency for 2.5 mg/L, a 22% increase in efficiency for 5.0 mg/L and 8% increase in efficiency for 7.5 mg/L.

4.2.7 Magnafloc 336 (Figure 4.4)

The efficiency of Magnafloc as a primary coagulant in the removal of SS was also investigated. At pH 4, dosing the effluent with 2,5 mg/L resulted in very little removal of SS (4.7% in trial 1 and 5.1% in trial 2). Increasing the dose to 5.0 mg/L led to slightly higher removals (9.4% in trial 1 and 12.8% in trial 2). Increasing the dose further to 7.5 mg/L resulted in 14.8% and 19.2% SS
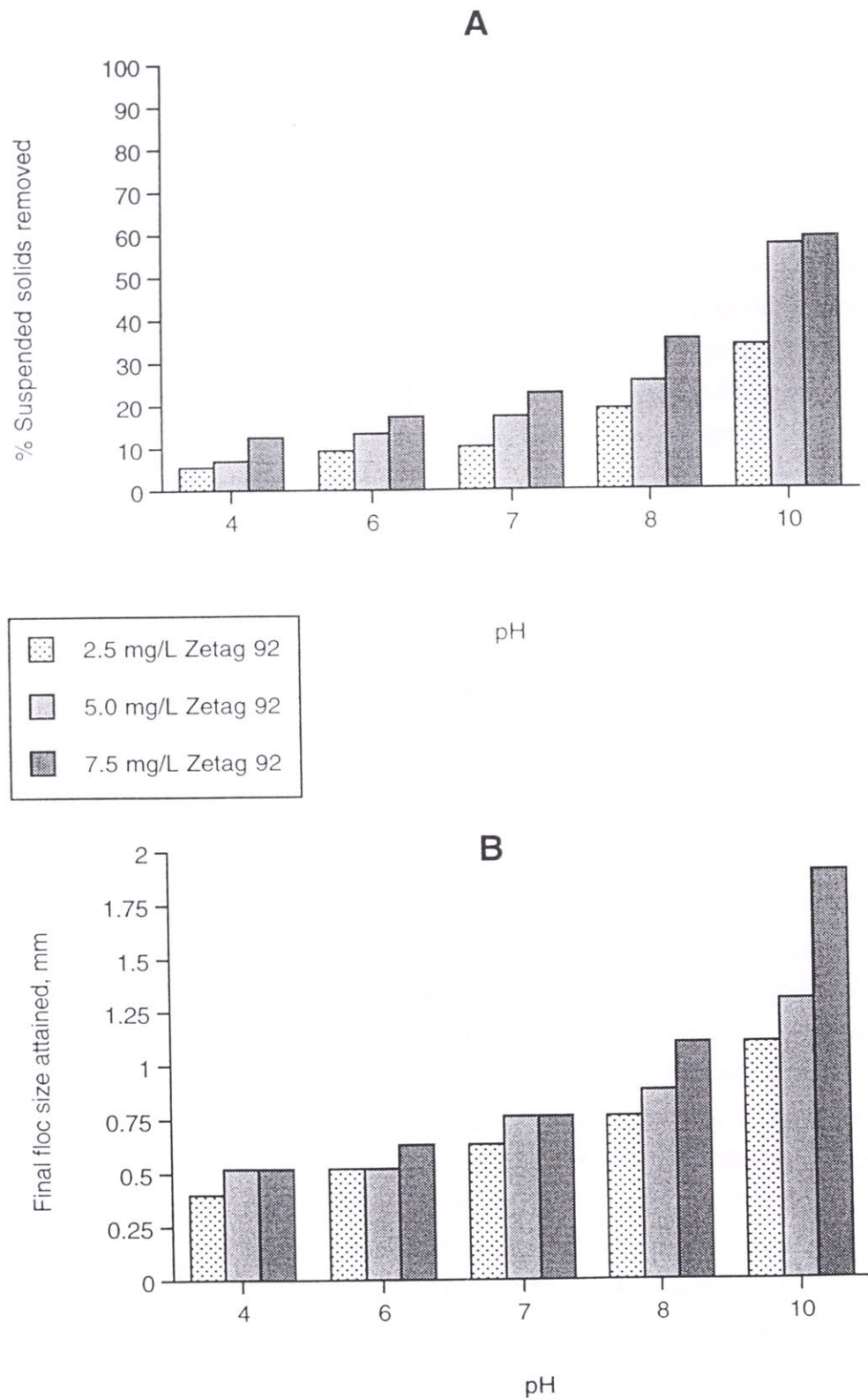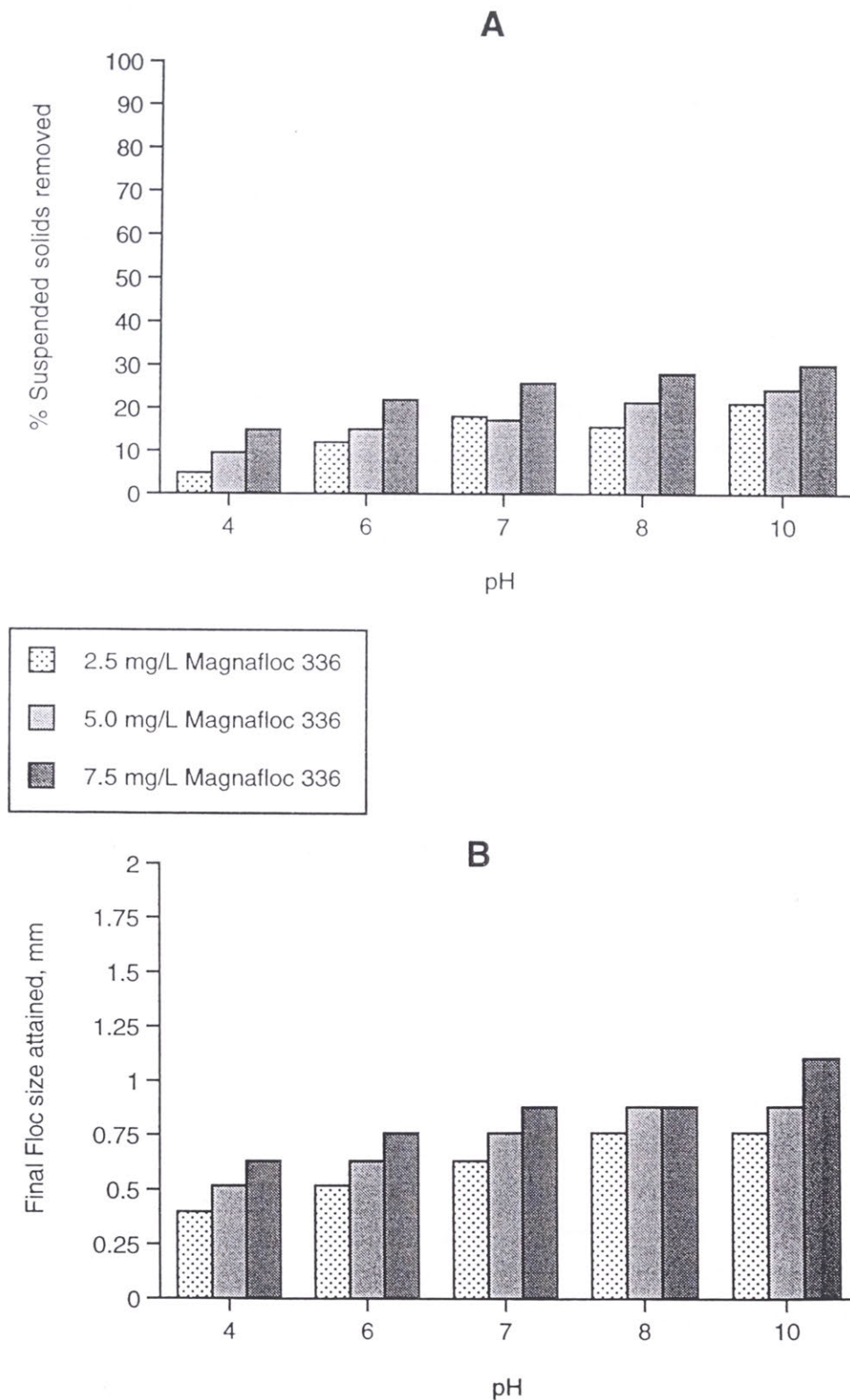
Figure 4.4 Coagulation of starch wastewaters using Magnafloc 336 as a primary coagulant. **(A)** Removal efficiency of suspended solids (%) versus pH for three different coagulant dosages; **(B)** Final floc size attained, mm, at 20 minutes flocculation versus pH for three different coagulant dosages. Results are for Trial 1, Day 1 only.

removals for trial 1 and trial 2 respectively. Floc growth followed the similar pattern of the other Magnafloc polymers: no growth was observed at 2.5 mg/L at both trials and increasing the doses to 7.5 mg/L led to maximum floc sizes of 0.52 mm (trial 1) and 0.63 mm (trial 2).

Increasing the pH from pH 4 to pH 6 led to higher levels of SS removal and larger final floc sizes. Increasing the doses from 2.5 mg/L to 7.5 mg/L led to increases from 11.8% SS removals to 21.8% removals in trial1. In trial 2, the increases were very slight, with 17.9% SS removed at 2.5 mg/L and 21.7% SS removed at 7.5 mg/L. The floc sizes increased at pH 6 relative to pH 4. At pH 6 the floc sizes attained a maximum size of 0.76 mm (trial 1 and trial 2) compared to 0.63 mm for both trials at pH 4.

At pH 7, higher levels of SS were removed and floc sizes reached their highest sizes at this pH. Trial 2 produced the largest flocs. In trial 1 flocs attained a final size of 0.63 mm, 0.76 mm, and, 0.88 mm at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. In trial 2 flocs attained a final size of 0.88 mm, 1.1 mm, and, 1.6 mm at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. The levels of SS removed did not change with an increase in dose from 2.5 mg/L to 5.0 mg/L, with SS removals at 18% and 17.2% respectively for trial 1 and 23.3% and 25.6% respectively for trial 2.

Increasing the dose to 7.5 mg/L led to a significant increase in the removal efficiency of Magnafloc 336, with 25.8% SS removed in trial 1 and 32.2% SS removed in trial 2. Comparing pH 7 to pH 6, the increase in efficiency of each dose with the increase in pH from pH 6 to pH 7 corresponded to 53%, 15%, and, 18% for trial 1 at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. For trial 2 the efficiency increase of each dose with the increase from pH 6 to pH 7 corresponded to 30%, 25%, and, 48% for trial 1 at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

When the pH was increased from pH 7 to pH 8, only slight changes in SS removal efficiencies were observed. Floc sizes at pH 8 remained static compared to pH 7 but for trial 2 the floc sizes actually decreased at pH 8: at 7.5 mg/L, pH 7, final floc sizes attained were 1.6 mm. At pH 8, equivalent conditions, 1.1 mm was the size attained. The SS removals at pH 8, 7.5 mg/L, were for trial 1, 27.9% (compared to 25.8% SS removal at pH 7, same dose). For trial 2 it was 29.8% (compared to 32.2% SS removal at pH 7, same dose).

At pH 10, only Magnafloc 336 was able to achieve the highest SS removals of all the anionic Magnafloc polymers. In trial 1, 21%, 24.2%, and, 29.8% SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. In trial 2, 22%, 28%, and, 32.9% SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. The floc sizes for trial 2 attained the same size as in pH 8 and in trial 1 the flocs increased to 1.1 mm at 7.5 mg/L (from 0.88 mm at pH 8).

4.2.8 Magnafloc 919

Magnafloc 919 was another polyelectrolyte trialled for its potential in removing SS from the effluent. At pH 4 dosing the effluent with 2.5 mg/L resulted in very little removal of SS (5.5% for trial 1 and 3.9% for trial 2). Increasing the dose to 5.0 mg/L led to large increases in SS removal efficiencies, with 10.2% removed in trial 1 and 14.1% removed in trial 2. When the dose was increased further to 7.5 mg/L, 17.2% SS were removed in trial 1 and 16.7% SS were removed in trial 2.

Floc growth was similar to the other Magnafloc polymers, with 0.52 mm final floc sizes attained at 5.0 mg/L and 7.5 mg L in trial 1 and 0.52 mm and 0.63 mm at 5.0 mg/L and 7.5 mg/L in trial 2, respectively. Increasing the pH from pH 4 to pH 6 caused higher removals of SS, but no increase in floc size. Increasing the doses from 2.5 mg/L to 7.5 mg/L led to increases from 11% SS removals to 18.7% removals respectively, for trial 1. Increasing the doses from 2.5 mg/L to 7.5 mg/L led to increases from 16.7% SS removals to 26.8% removals respectively, for trial 2.

At pH 7, higher levels of SS were removed and floc sizes reached a higher, final size compared to pH 6. For trial 1, floc sizes reached were 0.52 mm, 0.63 mm, and, 0.76 mm at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. In trial 2 floc sizes reached were 0.76 mm, 0.88 mm, and, 1.3 mm at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

The levels of SS removed at pH 7 did not vary greatly across the whole dosage range in trial 1, with 17.2%, 21.1% and 21.1% SS removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. In trial 2 18.9%, 21.1% and 32.2% SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

Comparing pH 7 to pH 6 the increase in the efficiency of each dose with the increase in pH from pH 6 to pH 7 corresponded to 56%, 107% and 13% for 2.5

mg/L, 5.0 mg/L and 7.5 mg/L respectively, trial 1. For trial 2 the increase in efficiency corresponded to 13%, 37% and 24% for 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

When the pH was increased from pH 7 to pH 8, there was very little change in SS removal efficiencies, with some doses displaying a decrease in efficiency. Floc sizes for pH 8, trial 1 reached 0.76 mm for all doses. For trial 2 all doses yielded flocs with a maximum size of 0.88 mm. The SS removed at pH 8 was for trial 1, 7.5 mg/L, 23% (compared to 21.1% at pH 7, 7.5 mg/L). The SS removed at pH 8 was for trial 2, 7.5 mg/L, 23.8% (compared to 32.2% at pH 7, 7.5 mg/L).

At pH 10, similar results were obtained as with pH 8. Floc growth was similar to that reached at pH 8. In trial 1, 14.5%, 20.2% and 24.2% of SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. For the respective doses in trial 2 18.3%, 26.8% and 23.2% SS were removed.

4.2.9 Magnafloc 1011

Magnafloc 1011 was the final anionic Magnafloc polyelectrolyte to be evaluated for its ability to remove suspended solids from the effluent. At pH 4 dosing the effluent from 2.5 mg/L to 7.5 mg/L led to significant increases in SS removal for both trials: trial 1 yielded 3.9% SS removals and 18% SS removals respectively; trial 2 yielded 2.6% SS removals and 14.1% SS removals respectively. Floc growth was similar to Magnafloc 919, with no growth at 2.5 mg/L and final floc sizes of 0.52 mm attained at 5.0 mg/L and 7.5 mg/L for both trials.

Increasing the pH from pH 4 to pH 6 led to higher SS removals. The SS removed for trial 1 at pH 6 was 11%, 9.4% and 17.2% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. The increase in the efficiency of Magnafloc 1011, from pH 4 to pH 6, for trial 1 was 182%, 34%, and, -4% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

The increase in the efficiency of Magnafloc 1011 was also observed in trial 2 with 19.2%, 17.9% and 23% SS removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively (compared to 2.6%, 7.7% and 14.1% in trial 2, at pH 4). Floc growth improved with an increase in pH reaching 0.63 mm at 7.5 mg/L in both trials.

At pH 7 higher levels of SS were removed and floc sizes reached a higher final size compared to pH 6. For trial 1 flocs reached 0.52 mm, 0.63 mm, and 0.76 mm for doses of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. For trial 2 flocs reached 0.76 mm, 1.1 mm, and 1.1 mm for doses of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. The levels of SS removed at pH 7 did not vary appreciably across the whole dosage range, with trial 1 showing 16.4%, 18% and 22.7% SS removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. In trial 2, 20%, 20% and 34.4% SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively.

When the pH was increased from pH 7 to pH 8, the efficiency of Magnafloc 1011 dropped slightly or remained static. Floc sizes for trial 1 actually increased but decreased for trial 2 from 1.1 mm at pH 7 to 0.88 mm at pH 8. The SS removals of trial 1 at pH 8 were 13.1%, 14.8% and 22.1% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Comparing these results with pH 7 (16.4%, 18% and 22.7%) very little difference was made by increasing the pH to pH 8.

The SS removals of trial 2 at pH 8 were 20.2%, 20.2% and 20.2% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L respectively. Comparing these results with pH 7 (20%, 20% and 34.4%) very little difference was made by increasing the pH to pH 8, except for a very big drop in SS removal efficiency with a dose of 7.5 mg/L.

Further increasing the pH from pH 8 to pH 10 resulted in little change for the capacity of Magnafloc 1011 to remove suspended solids. Floc sizes attained the same size as the flocs in pH 8.

4.2.10 Magnafloc 333

This non-ionic polymer was the poorest performer in its ability to form flocs and remove suspended solids from the effluent, relative to the other anionic and cationic polymers. Its highest levels of SS removal were at pH 8, at a dose of 7.5 mg/L, trial 2, where it only removed 19% SS. Under these conditions it also attained its largest floc growth at 0.63 mm.

4.3 THE EFFECTS OF MODIFIED EFFLUENT ON ALUMINIUM SULPHATE COAGULATION AND FERRIC SULPHATE COAGULATION.

A series of jar tests were conducted in order to evaluate the efficiency of aluminium sulphate and ferric sulphate as primary coagulants. The efficiency of these coagulants was evaluated only by the measure of suspended solids (SS) removed. These trials included modified effluent: effluent which had half the original levels of settleable solids removed, and, effluent which had a two-fold increase in the original settleable solids levels. These two modified effluents were compared to normal effluent.

The purpose of these modified effluents was to evaluate the performance of the coagulants on effluents which may undergo sudden changes in their organic/solids loadings, which may have a bearing on the overall coagulation efficiency.

Table D4.12 summarises the effluent characteristics of the normal and modified effluents. The unmodified effluent, in both trials, is similar in its supernatant suspended solids loadings. Table D4.11 displays data of the control/supernatant suspended solids. Changes in pH lead to slight changes in suspended solids concentrations.

Tables D4.2, D4.4, D4.6, D4.8 and D4.10, of the Appendices display results for coagulation and removal of suspended solids with aluminium sulphate at pH 4,6,7,8 and 10, respectively.

Tables D4.1, D4.3, D4.5, D4.7 and D4.9, of the Appendices display results for coagulation and removal of suspended solids with ferric sulphate at pH 4,6,7,8 and 10, respectively.

Figure 4.5 graphically displays the suspended solids removed, expressed as a percentage, for all three effluent types. Figure 4.5 also shows the changes of suspended solids removed at various pH levels and coagulant doses. All results shown in figure 5 pertain only to trial 1.

4.3.1 Aluminium Sulphate With Unmodified Effluent

At pH 4 4.8% SS was removed with 25 mg/L dosage. Doubling the dose to 50 mg/L effectively doubled the removal to 9.5%. When the dose was further

Figure 4.5. Effect of **aluminium sulphate** and **ferric sulphate**, over a broad dosage and pH range on the removal of suspended solids from unmodified starch effluent (1xTotal Settleable solids); starch effluent with a 50% reduction in Settleable solids (Total Settleable solids/2); and starch effluent with a 100% increase in Settleable solids (2xTotal Settleable solids)

▦	1 X Total Settleable Solids
▦	Total Settleable Solids/2
■	2 X Total Settleable Solids
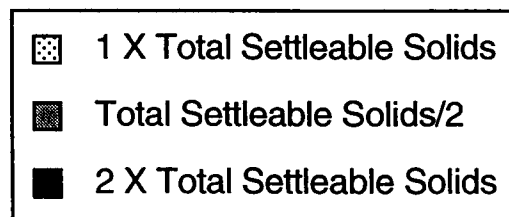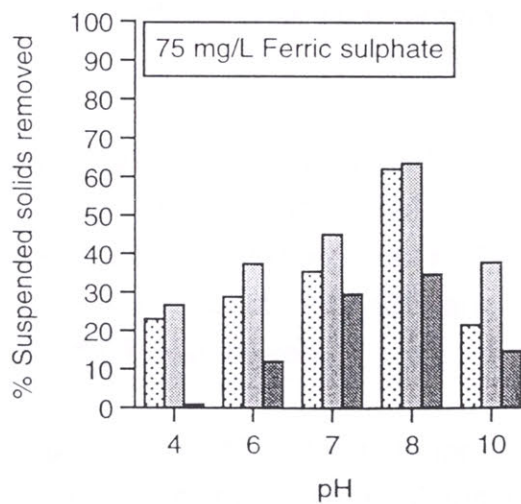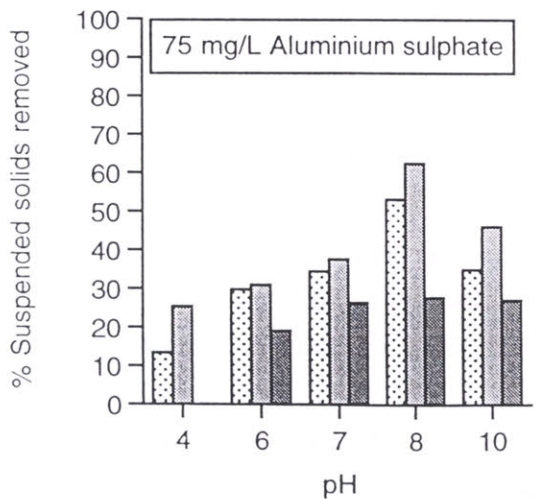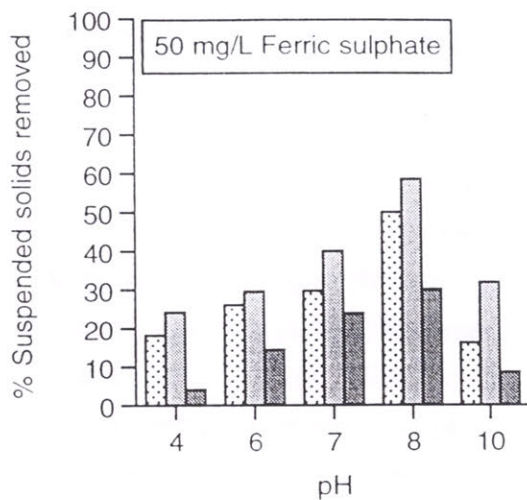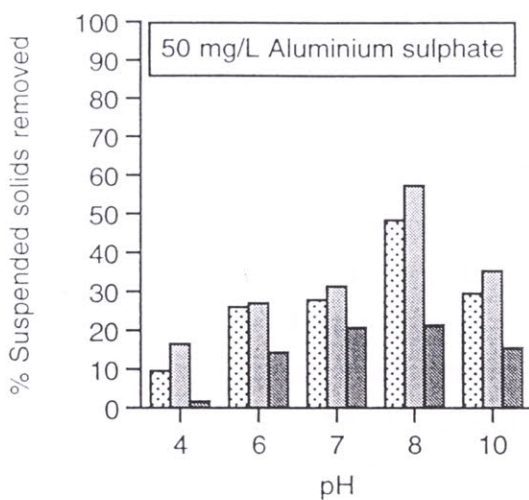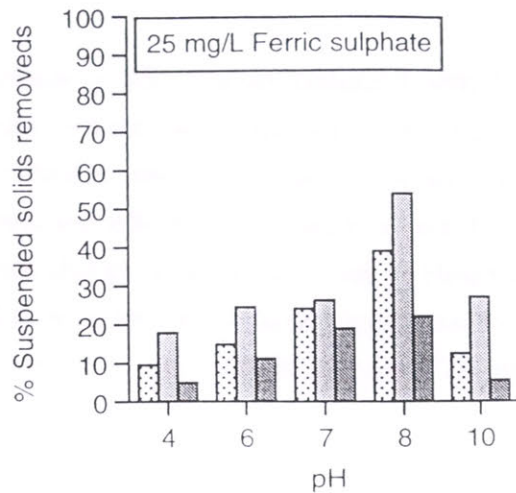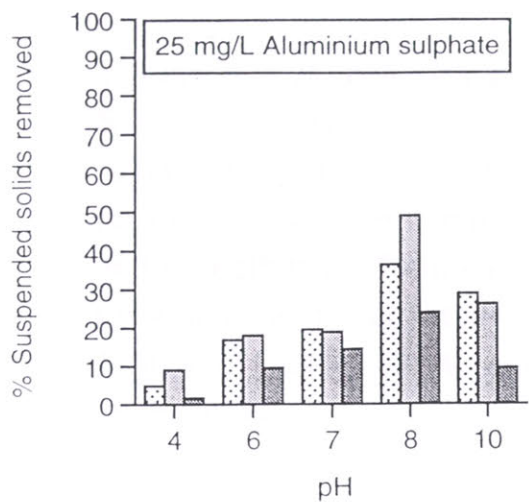
increased to 75 mg/L only a slight increase in SS removal resulted with 13.3% SS removed. The pattern for trial 2 was similar, although the increase in SS removal from 50 mg/L to 75 mg/L was smaller relative to trial 1. Increasing the pH from pH 4 to pH 6 improved the removal efficiency of alum at each dose, and for both trials. Figure 4.5 shows this sharp increase for trial 1. Relative to pH 4, the use of alum at pH 6 resulted in an overall efficiency increase for trial 1 of about 24%, 173%, and 123% for 25 mg/L, 50 mg/L and 75 mg/L, respectively.

Results of SS removal for pH 6 were, for trial 1: 16.7%, 25.9% and 29.6% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. For trial 2 the SS removals were: 15.1%, 24.8% and 27.4% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively.

When the pH was increased from pH 6 to pH 7, removals of SS were higher for both trials, although trial 2 had a relatively higher rate of SS removal. Trial 1 levels were 19.4%, 27.8% and 34.3% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. This increase, when compared to pH 6 represents an overall increase in efficiency of 16%, 7%, and, 16% for the lowest to highest dose of alum, when the pH of the effluent is increased from pH 6 to pH 7. Trial 2 had higher SS removals, with 23.2%, 29.3% and 39.3% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. This corresponds to an overall increase in the efficiency of alum with an increase in the pH. For the respective doses of 25 mg/L, 50 mg/L and 75 mg/L, the increase in efficiency of alum performance was 53%, 19% and 43%.

Increasing the pH from pH 7 to pH 8 led to very large SS removals. Figure 4.5 shows this very sharp, large increase in alum performance. Trial 1 yielded about 36%, 48% and 53% SS removals with 25 mg/L, 50 mg/L and 75 mg/L respectively. Trial 2 yielded very similar results with about 39%, 48% and 55% SS removals with 25 mg/L, 50 mg/L and 75 mg/L respectively.

When the pH was further increased from pH 8 to pH 10 a deterioration in the capacity of alum to remove SS was observed for both trials and at all doses. At 75 mg/L alum trial 1 yielded about 35% SS removal (compared to 53% at pH 8) and trial 2 yielded about 30% SS removal (compared to 55% at pH 8).

4.3.2 Aluminium Sulphate With Modified Effluent: 50% Reduction of Settleable Solids

Alum used with this type of effluent produced overall higher levels of SS removals compared to the unmodified effluent but the trends were very similar to eachother. Figure 5 shows that at all doses the modified effluent with reduced settleable solids produced higher SS removals. These higher results are prominent at pH 4 and pH 8.

Trial 1 displayed higher levels of SS removals to trial 2. Comparing the unmodified effluent to the reduced solids effluent, the latter effluent yielded almost double the SS removals for trial 1 (8.9%, 16.5% and 25.3% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively). Trial 2 produced high levels of SS removals with reduced settleable solids as well, although the magnitude was not as high as for trial 1.

When the pH was increased to pH 6 there was an improvement with SS removal for trial 1 , but very little change occurred in trial 2. In trial 1 the SS removals obtained were about 18%, 27% and 31% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. Compared to the unmodified effluent, pH 6, trial 1 there was very little change. This is graphically illustrated in Figure 4.5 where at pH 6 both unmodified and solids reduced effluent are very similar. Trial 2 also produced similar results to trial 1 at pH 6, with about 24%, 22% and 27% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively.

Figure 4.5 shows that at pH 7 there was only a slight increase in SS removals compared to pH 6, for all doses. This can also be seen in Tables D4.4 and D4.6 where trial 1 in both tables does not differ greatly between pH changes. Trial 2 showed a big increase in the levels of SS removed for 50 and 75 mg/L when the pH was increased from pH 6 to pH 7, with an approximate SS removal from 22% and 27% (pH 6) respectively to 38% and 42% (pH 7) respectively.

As with the unmodified effluent, at pH 8 very large removals of SS were obtained for both trials. Figure 4.5 shows that the reduced solids effluent yielded. higher SS removals than unmodified effluent, when dosed with the equivalent alum. At 75 mg/L 62% and 52% SS were removed in trial 1 and trial 2 respectively.

When the pH was further increased from pH 8 to pH 10, the capacity for alum to remove suspended solids was diminished at all doses (see Figure 4.5). At a dose of 75 mg/L trial 1 yielded 46% SS removal and trial 2 yielded about 38% SS removal. These results are significantly lower compared to the equivalent conditions at pH 8: 62% SS removal, trial 1 and 52% SS removal trial 2.

4.3.3 Aluminium Sulphate With Modified Effluent: 100% Increase In Settleable Solids

When alum was used with this modified effluent, similar trends were observed as with the unmodified effluent and the solids reduced effluent, but this solids increased effluent yielded relatively poor suspended solids removals. Figure 4.5 shows that at all pH levels and all dosages there was significant deterioration in the capacity for alum to remove suspended solids from effluent with an increased solids load.

At pH 6 there were large improvements in SS removals. At this pH, 9%, 14% and 19% SS were removed at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively, for trial 1. Trial 2 removed about 7%, 14% and 18% SS at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively.

Increasing the pH from pH 6 to pH 7 indicated a slight increase in the SS removal capacity of alum, at all doses. Trial 1 and trial 2 are similar in their SS removals although the latter trial yielded slightly higher results.

At pH 8 the highest SS removals were obtained but the removals were only marginally higher than pH 7. This is unlike the other two effluent types which yielded much larger results at pH 8. Figure 4.5 shows that at 25 mg/L, trial 1, 23.6% SS was removed, compared to 14.3%, trial 1, pH 7. At 50 mg/L and 75 mg/L only 21.3% and 27.6% SS were removed respectively compared to 20.6% and 26.2% respectively at pH 7. Increasing the pH from pH 7 to pH 8 did not increase the efficiency of alum.

At pH 10 no differences in SS removal rates were found with pH 8 for the dose of 75 mg/L, trial 1. This can be clearly seen in Figure 4.5. At 25 mg/L and 50 mg/L alum, the SS removal efficiency dropped, relative to pH 8, with about 9% and 15% SS removed at 25 mg/L and 50 mg/L.

4.3.4 Ferric sulphate With Unmodified Effluent

At pH 4 ferric sulphate removed about 10% of the SS at a dose of 25 mg/L, 18% at 50 mg/L and 23% at 75 mg/L. Trial 2 also produced very similar results with 8%, 14% and 22% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. Trial 1 with ferric sulphate, when compared to trial 1 with alum produced higher SS removals (see Figure 4.5). Trial 2 with ferric sulphate, on the other hand produced very similar removals to alum at all doses, at pH 4.

When the pH was increased to pH 6, SS removals also increased compared to pH 4. Both trials were similar. Comparing ferric sulphate with alum in Figure 4.5 the SS removals for ferric sulphate are very similar to the SS removals obtained with alum.

When the pH was increased from pH 6 to pH 7, removals of SS were higher for both trials. Trial 1 levels of SS removals reached about 24%, 30% and 35% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. This increase, when compared to pH 6, represents an overall increase in efficiency of about 63%, 14% and 23% at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively when the pH is increased from pH 6 to pH 7. Table D4.5 indicates that both trials at pH 7 removed similar levels of suspended solids. Table D4.6 indicates that alum and ferric sulphate removed similar levels of suspended solids.

Increasing the pH from pH 7 to pH 8 led to a very large increase in SS removals. Trial 1 yielded about 39%, 50% and 62% SS removals at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. Trial 2 yielded similar results with approximately 42%, 51% and 60% SS removals at doses of 25 mg/L, 50 mg/L and 75 mg/L, respectively. Figure 4.5 shows these large increases.

Comparing all ferric sulphate doses at pH 8 with alum doses at pH 8, the removal of the suspended solids were very similar at 25 mg/L and 50 mg/L. At 75 mg/L ferric sulphate attained higher SS removals.

When the pH was further increased from pH 8 to pH 10, a deterioration in the capacity of ferric sulphate to remove SS was observed for both trials and at all doses. This is shown in Figure 4.5. Compared to alum, ferric sulphate produced relatively poor SS removals at pH 10.

4.3.5 Ferric Sulphate With Modified effluent: 50% Reduction of Settleable Solids

Ferric sulphate used with this modified effluent produced the highest levels of SS removals over all the pH levels and all ferric sulphate doses. The trends were very similar with those of unmodified effluent and effluent with increased solids. At pH 4 ferric sulphate produced much higher removals of SS than did alum at 25 mg/L and 50 mg/L and very similar SS removals at 75 mg/L. More SS was removed in trial 2 than in trial 1.

When the pH was increased from pH 4 to pH 6, there was an improvement with SS removal for both trials and both trials yielded very similar removals. Comparing ferric sulphate with alum in Figure 4.5, the removals of SS in each ferric sulphate dose was similar to the equivalent doses for alum.

Figure 4.5 shows that by increasing the pH from pH 6 to pH 7 there was an increase in the SS removed compared to pH 6, for all doses. This can also be seen in Tables D4.3 and D4.5. Trial 2 displayed slightly lower levels of SS removal. In all doses, at pH 7, ferric sulphate produced higher SS removals than did alum at pH 7, with the equivalent doses.

As with the unmodified effluent, at pH 8 very large levels of SS were removed, in both trials. Both trials displayed very similar results, Table D4.7. Figure 4.5 shows that higher levels of SS were removed from the effluent compared to the unmodified effluent, when dosed with the equivalent level.

Comparing alum and ferric sulphate in Figure 4.5 indicates that ferric sulphate was able to remove higher levels of SS than alum at equivalent doses (25 mg/L and 50 mg/L). Similar SS removals were obtained at 75 mg/L for both alum and ferric sulphate.

When the pH was further increased to pH 10, the capacity for ferric sulphate to remove SS from the effluent was greatly diminished, as is shown in Figure 4.5, where at all doses the level of SS removed fell sharply. At pH 10, for trial 1 about 27%, 32% and 38% SS removals were obtained at 25 mg/L, 50 mg/L and 75 mg/L, respectively. This is much lower to results obtained at pH 8 (about 54%, 59% and 63% SS removals were obtained at 25 mg/L, 50 mg/L and 75 mg/L, respectively). Figure 4.5 shows that at pH 10, SS removals using ferric sulphate were similar to SS removals using alum at pH 10, for all doses.

4.3.6 Ferric Sulphate With Modified Effluent: 100% Increase In Settleable Solids

When ferric sulphate was used with this modified effluent, similar trends were noted with the other effluents, but SS removals were relatively poor with this solids increased effluent. Figure 4.5 shows relatively low levels of SS were removed throughout the whole ferric sulphate dosage range. Table D4.1 shows that at both trials very similar, low levels of SS were removed, except at trial 2 where about 14% SS was removed with 75 mg/L (compared to trial 1 where less than 1% SS was removed with 75 mg/L). Ferric sulphate was able to remove more SS at pH 4 than could alum (Figure 4.5).

Increasing the pH from pH 4 to pH 6 led to an increase in the SS removed. Table D4.3 shows that at both trials, similar results were obtained, except that at 75 mg/L trial 1 removed about 12% and trial 2 removed almost twice that amount with 24.5%. At pH 6, both ferric sulphate and alum produced similar removals at all doses.

Increasing the pH from pH 6 to pH 7 showed big improvements in the capacity of ferric sulphate to remove suspended solids. Figure 4.5 shows this increase at pH 7. For both trials, results were very similar. Comparing ferric sulphate at pH 7 to alum at pH 7, for all doses, shows both coagulants functioned similarly in their capacity for SS removals.

The highest SS removals were attained at pH 8. These removals were slightly higher than results obtained for pH 7. At pH 8, both trials yielded very similar results (Table D4.7). Results for trial 1 were 22%, 30% and 35% SS removals at 25 mg/L, 50 mg/L and 75 mg/L, respectively. At pH 7 the SS removals were 19%, 24% and 29% at 25 mg/L, 50 mg/L and 75 mg/L, respectively. Changing the pH from pH 7 to pH 8 only led to a slight increase in the removal efficiency of ferric sulphate. Comparing pH 8 of ferric sulphate to pH 8 of alum, in Figure 4.5, the SS removal capacities of both coagulants was similar.

At pH 10 (Table D9) the efficiency of SS removals dropped sharply relative to pH 7 and pH 8. For trial 1 the SS removals were about 5%, 9% and 15% at 25 mg/L, 50 mg/L and 75 mg/L, respectively. For trial 2 SS removals were about 10%, 15% and 19% at 25 mg/L, 50 mg/L and 75 mg/L, respectively. Ferric sulphate at pH 10 removed less SS than alum at pH 10 (Figure 4.5).

4.4 PHYSICAL PARAMETERS OF THE JAR TEST

The effects of various physical parameters such as flocculation intensity, flocculation duration, dosage rates, dosage sites and rapid mix durations were investigated in a series of jar tests which were conducted with alum and Zetag 92.

Alum was used in the jars at final concentrations of 25 mg/L, pH 8 and Zetag 92 was used at a dose of 10 mg/L, (final concentrations in the jars) pH 8. All the work evaluated these coagulants as separate primary coagulants. They were never used in a combination. Figures 4.6-4.9 display the results of most of the tests, for the first trial only.

Each experiment was duplicated with two trials, conducted 7 days apart. This was done to ascertain how changes to effluent, if any, effect the physical aspects of the jar tests. Table E4.10 summarises the parameters of the effluents studied, including the suspended solids concentrations of the supernatant. Generally, the suspended solids concentrations of the whole effluent fell into the typical ranges found in the effluent profile work.

Effluent containing relatively higher suspended solids concentrations in the whole effluent fraction tended to have higher supernatant suspended solids levels, for example in Table E4.10, the whole SS concentration is 2260 mg/L, the supernatant SS level is 345 mg/L. Those effluents with the lower levels of SS in the whole fraction tended to have lower supernatant suspended solids levels, for example in Table E4.10, the whole SS concentration of 1865 mg/L, results in supernatant SS levels of 250 mg/L.

4.4.1 Alum and Rapid Mixing

Table E4.1 and Figure 4.6 A (trial 1 only) display the results of jar tests conducted to investigate the effects of modifying the duration of rapid mixing. Alum was used as the primary (and only) coagulant, and was added directly into the vortex at a steady rate. Both trials yielded similar results and trends. Figure 4.6A indicates the optimum rapid mix time to be about 10 seconds (the same was found for trial 2). At 10 seconds, about 40% and 38% SS were removed for trial 1 and trial 2, respectively. The largest floc sizes were attained at 10 seconds as well, reaching 0.88 mm and 1.3 mm for trial 1 and trial 2, respectively.

At 20 seconds there is a drop in SS removal from about 40% SS removal at 10 seconds to 26% SS removal. Floc sizes were also reduced from 0.88 mm to 0.63 mm (both trials). Extending the rapid mixing time for a further 10 seconds to 30 seconds had no effect on floc growth as it attained 0.63 mm in both trials. In trial 1 the SS removals stayed static at approximately 28% (compared to 26% SS removals at 20 seconds) and in trial 2 the SS removals dropped slightly to approximately 22% (compared to 23% SS removals at 20 seconds).

When rapid mixing was extended beyond the 10 seconds, a deterioration in the SS removal capacity of the system was noted. Also, floc sizes were reduced. When rapid mixing was extended to 60 seconds and beyond, no floc growth was recorded and SS removals dropped to 22.8%, 10.5% and 8.8% for rapid mixing of 60 seconds, 120 seconds and 180 seconds, respectively (trial 1 results). These results resembled those where no rapid mixing was used: 17.5% SS was removed and no floc growth was recorded.

4.4.2 Alum and Flocculation Duration

These trials examined the effects of varying the flocculation time, using alum as the primary coagulant. Alum was injected at a steady rate directly into the vortex. Rapid mix continued for a full 10 seconds from the initial point of addition of the alum. Flocculation was then allowed to progress up to 60 minutes, with points of 0 minutes, 10, 20 and 30 minutes. At zero time, the stirrers were immediately switched off at 10 seconds after rapid mixing and the solids in the beaker allowed to settle for 20 minutes.

Table E4.2 and Figure 4.6B indicate that no flocculation (0 minutes) results in very poor SS removals: about 12% and 9% SS were removed at trial 1 and trial 2 respectively. No floc growth was observed.

The largest flocs attained were at 10 minutes flocculation, with 0.88 mm reached for both trials. At these flocculation times, though, SS removals were only 30% and 25% for trial 1 and trial 2 respectively. The highest levels of SS removals were obtained at 20 minutes flocculation. At 20 minutes flocculation, about 40% and 39% SS were removed at trial 1 and trial 2 respectively. Floc growth attained for both trials was 0.63 mm.
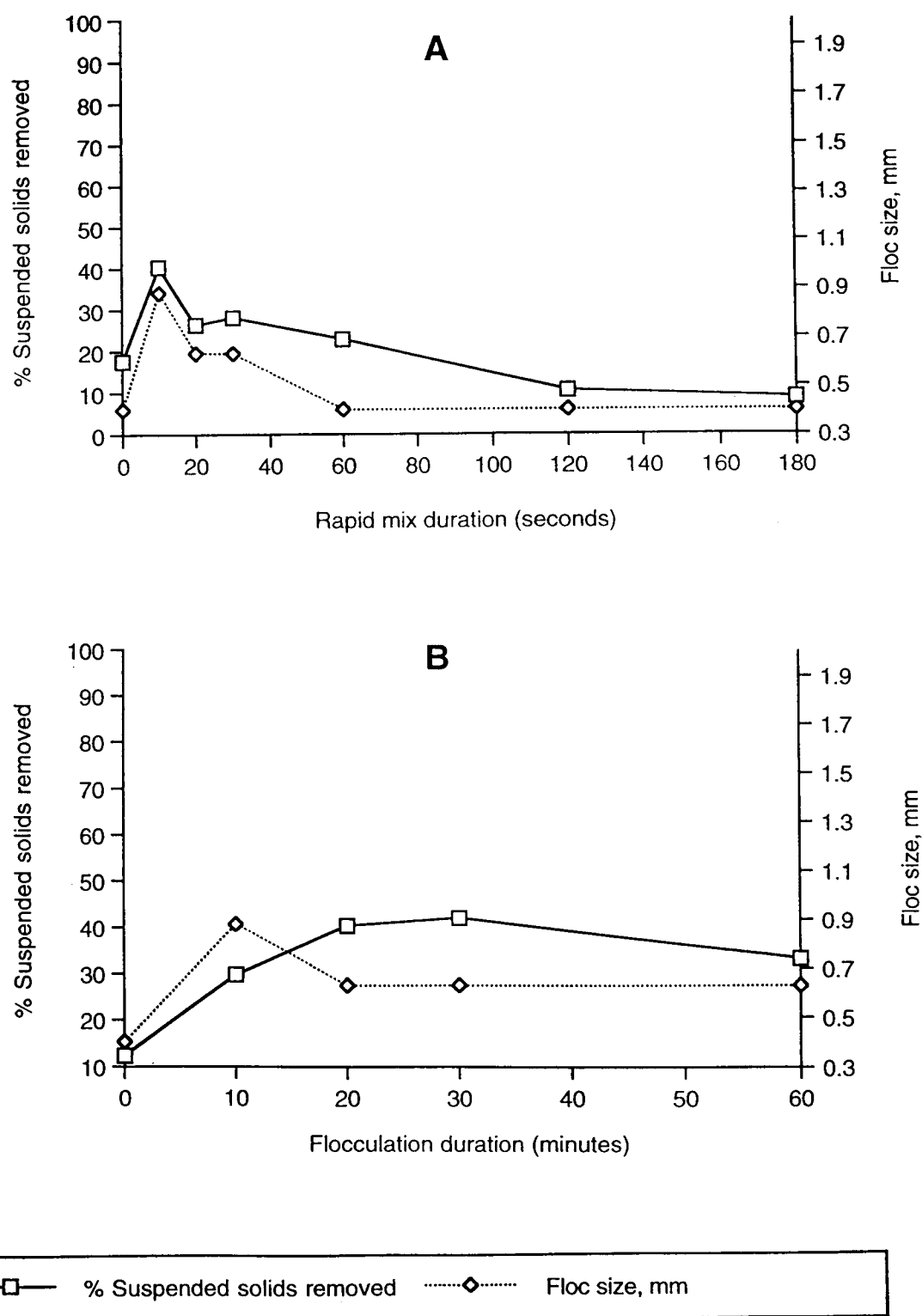
Figure 4.6. The effects of varying **rapid mix duration (A)** and varying **flocculation duration (B)** on suspended solids removals (%) and final floc floc sizes attained (mm) when using **aluminium sulphate** (50 mg/L, pH 8) as the primary coagulant.

Extending the flocculation duration beyond 20 minutes led to similar SS removals and floc sizes at 30 minutes for trial 1 (42% SS removed and 0.63 mm final floc size). Suspended solids removals deteriorated very slightly in trial 2 (about 35% SS removed compared to 39%) but floc sizes remained at 0.63 mm.

At 60 minutes the SS removal efficiency has dropped and resembles SS removals at 10 minutes. SS removals obtained for trial 1 and trial 2 respectively are about 33% and 26%. Floc sizes in trial 1 remain at 0.63 mm but decreased in trial 2 to 0.4 mm.

4.4.3 Alum and Flocculation Intensity

This work examined the effects of changing the flocculation intensity on SS removals and floc growth. Alum was used as the primary coagulant, with 10 seconds rapid mix and flocculation time of 20 minutes. The intensity of flocculation was measured in revolutions per minute (rpm) of the mixing blade. Results are indicated in Table E 4.3 and Figure 4.7A.

The results of both trials indicate the optimum flocculation intensity to be 30 rpm. At 30 rpm, about 41% SS are removed in trial 1 and about 38% are removed in trial 2. Both trials yielded floc sizes of 0.88 mm at 30 rpm.

At 20 rpm about 30% SS are removed in both trials and floc sizes attained 0.63 mm. When the intensity of the flocculation was increased beyond 30 rpm the efficiency of SS removals decreased slightly. At 40 rpm the SS removals dropped to 32% in both trials, from about 40% at 30 rpm for both trials. Floc sizes in trial 1 dropped to 0.63 mm from 0.88 mm, but remained at 0.88 mm in trial 2.

Increasing the flocculation intensity to 60 rpm and 80 rpm led to further reductions in the capacity of the system to remove SS. At these flocculation intensities floc growth was reduced to 0.4 mm for both trials. SS removals dropped to a low 5.3% and 6% at 80 rpm for trial 1 and trial 2 respectively.

4.4.4 Alum and Dosage Rates

The effects of changing the rates of coagulant addition were also examined for their impact on SS removals and floc growth. The actual dispensing of the
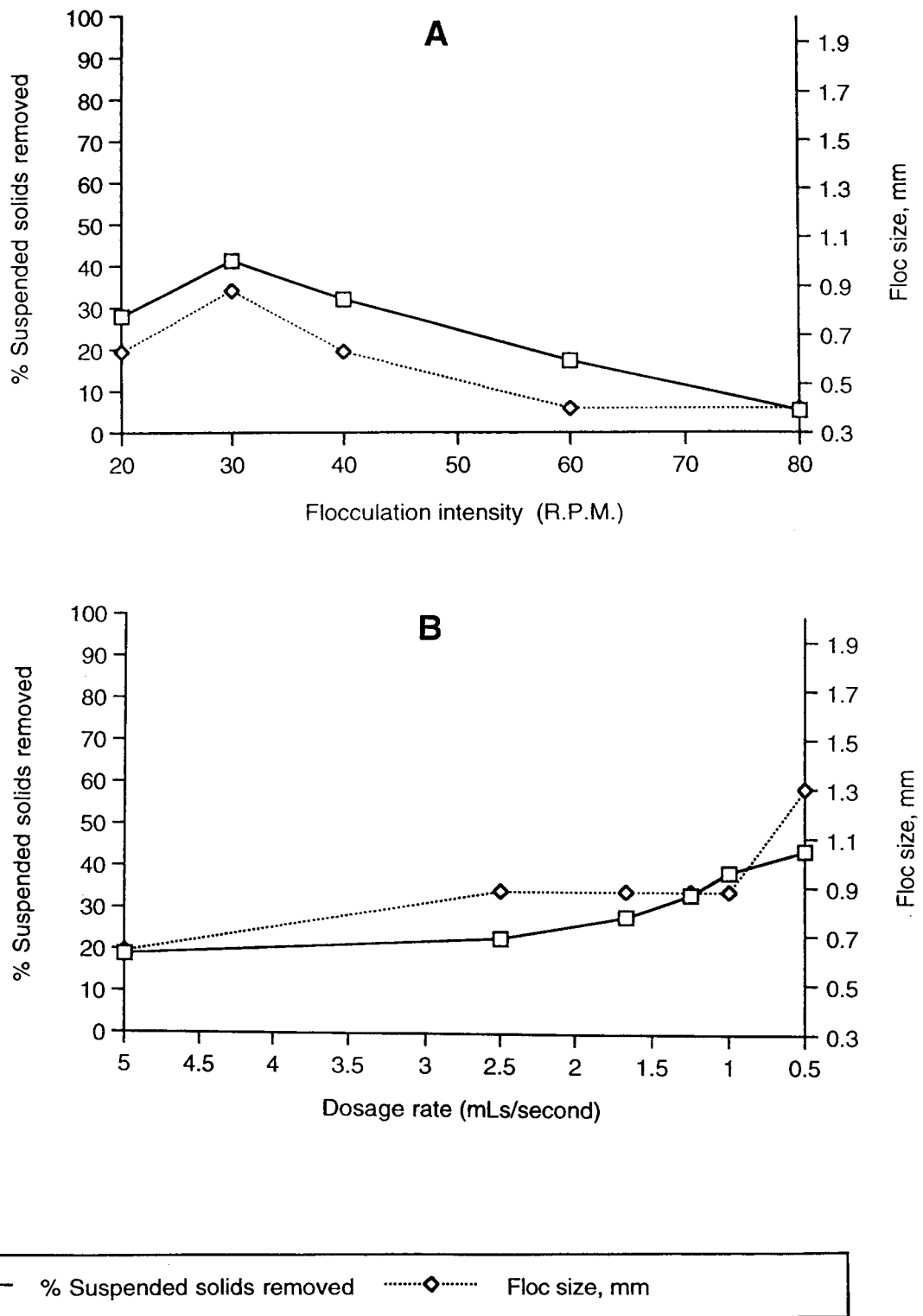
Figure 4.7. The effects of varying **flocculation intensity (A)** and varying **dosage rate (B)** on suspended solids removals (%) and final floc floc sizes attained (mm) when using **aluminium sulphate** (50 mg/L, pH 8) as the primary coagulant.

coagulant was done by hand with a syringe and timer. 5 mLs of alum at an initial concentration of 2500 mg/L was added into 500 mLs of effluent into the beaker, giving a final concentration of 25 mg/L. A dosage rate of 5 mLs/sec represents the whole 5 mLs of alum being injected into the vortex within one second. A dosage rate of 0.5 mLs/sec represents the whole 5 mLs of alum being injected into the vortex within the 10 second rapid mix period.

The results of both trials indicate that a dosage rate of 1 mL/sec or 0.5 mLs/sec is optimum with maximum SS removals attained and the largest floc sizes attained. For trial 1, SS removed was 39% and 44% for 1 mL/sec and 0.5 mLs/sec respectively. The respective floc sizes were 0.88 mm and 1.3 mm. For trial 2, SS removed was 38% and 44% for 1 mL/sec and 0.5 mLs/sec respectively. The respective floc sizes were 0.88 mm and 0.88 mm.

Higher dosage rates of alum (1.25 mLs/sec, 1.67 mLs/sec and 2.5 mLs/sec) all produced final floc sizes of 0.88 mm for both trials. Removals of SS were 38%, 30% and 26%, respectively for trial 1 and 33.3%, 28% and 22.7%, respectively for trial 2. At 5 mLs/sec the lowest floc sizes were attained at 0.63 mm for both trials. Only 20% SS and about 19% SS were removed in trial 2 and trial 1 respectively.

4.4.5 Alum and Dosage Sites

Certain dosage sites was also examined for their effects on SS removal and final floc size. Table E4.5 shows the results of SS removals and floc sizes attained when alum was dosed directly into the vortex, or dosed onto the surface of the effluent sample, in between the vortex and the beaker wall. The dosage rate used was 1 mL/sec (5 mLs in 10 seconds) and this covered the whole rapid mix time frame of 10 seconds.

Dosing directly into the vortex produced the highest SS removals with 38.7% and 40% SS removed in trial 1 and trial 2 respectively. Both trials yielded the highest floc sizes with 1.3 mm attained in both trials. Dosing onto the surface produced the lowest SS removals with 22.7% and 28% SS removed in trial 1 and trial 2 respectively. Both trials yielded the lowest floc sizes with 0.63 mm attained in both trials.

The trials conducted using alum were repeated with the cationic polyelectrolyte Zetag 92. Dosage site studies were not conducted.

4.4.6 Zetag 92 and Rapid mixing

Figure 4.8A and Table E 4.6 show the results of two trials conducted using Zetag 92 as a primary coagulant, dosed at about 1 mL/sec. The stock concentration of Zetag 92 was at 500 mg/L and 10 mLs were added into the beaker to give a final concentration of 10 mg/L.

Rapid mixing of 20-40 seconds produced highest levels of SS removals and the largest flocs. The results obtained at 20 seconds and 40 seconds were very similar, for both trials. For trial 1 and trial 2 at 20 seconds rapid mix about 71% and 73% SS were removed respectively. At 40 seconds rapid mixing, for trial 1 and trial 2, 72% and 67% SS were removed respectively. Final floc sizes at 20 second and at 40 seconds for both trials were 1.9 mm. Figure 4.8A shows very little change occurring between 20 seconds and 40 seconds for SS removals and floc growth.

At 60 seconds rapid mixing there is a decline in the efficiency of the system to remove SS. Trial 1 and trial 2 yield about 64% and 60% removals in SS. Floc sizes decrease in trial 2 from 1.9 mm to 1.3 mm but in trial 1 floc sizes maintain a size of 1.9 mm. This can also be seen in Figure 4.8A.

Extending the rapid mixing to 120 seconds led to larger decreases in SS removals with about 57% and 52% SS removed for trial 1 and trial 2, respectively. Floc sizes in trial 2 remained at 1.3 mm and in trial 1 floc sizes decreased to 1.3 mm (from 1.9 mm).

4.4.7 Zetag 92 and Flocculation Duration

The effect of changing flocculation duration was also observed for its effects on SS removals and floc growth. Table E4.7 and Figure 4.8B show these effects. At 20 40 and 60 minutes floccculation SS removals and final floc sizes attained were very similar for all flocculation times and both trials yielded similar results. In trial 1 about 67%, 69% and 67% SS were removed at 20, 40 and 60 minutes, respectively. The final floc sizes attained at all these times were 1.9 mm. In trial 2 about 71%, 71% and 69% SS were removed at 20, 40 and 60 minutes, respectively. The final floc sizes attained at all these times were 1.9 mm. clearly, increasing the time of flocculation from 20 minutes to 60 minutes does not change the levels of SS removed or the final floc size.
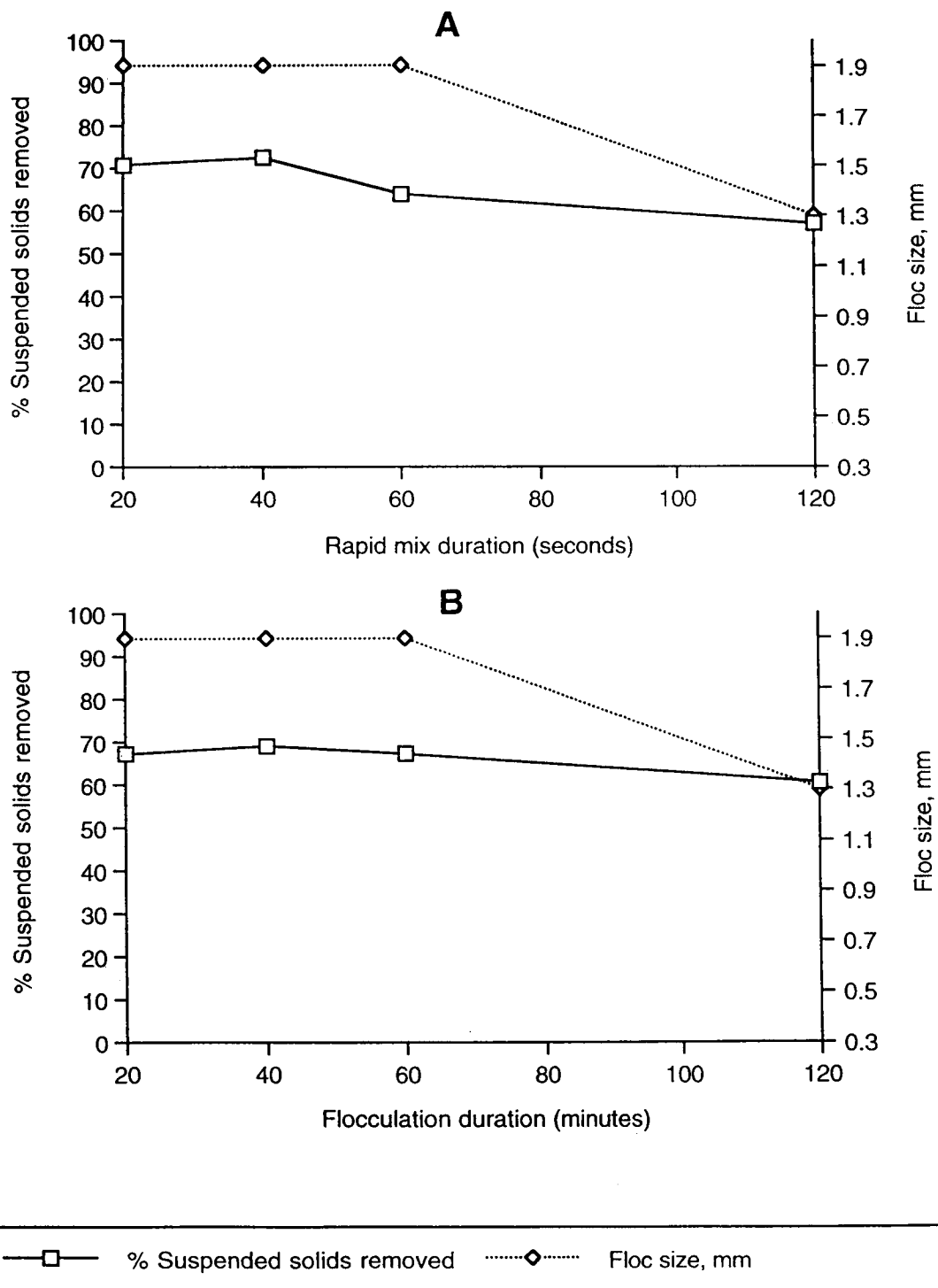
Figure 4.8. The effects of varying **rapid mix duration (A)** and varying **flocculation duration (B)** on suspended solids removals (%) and final floc floc sizes attained (mm) when using cationic **Zetag 92** (10 mg/L, pH 8) as the primary coagulant.

At 120 minutes the efficiency decreased slightly to 60% and 64% SS removed for trial 1 and trial 2, respectively. Floc sizes also decreased to 1.3 mm (from 1.9 mm) for both trials.

4.4.8 Zetag 92 and Flocculation Intensity

The effect of changing flocculation intensity was also observed for its effects on SS removals and floc growth. Table E4.8 and Figure 4.9B show these effects. The intensity of flocculation was measured in revolutions per minute (rpm) of the mixing blade.

For trial 1 flocculation intensity at both 30 rpm and 40 rpm produced very similar SS removals with about 71% and 72% SS removed respectively. Floc sizes at both levels attained a size of 1.9 mm. Results to trial 2 were very similar although trial 2 yielded slightly higher SS removals. For trial 2 flocculation intensity at both 30 rpm and 40 rpm produced SS removals with about 75% and 77% SS removed respectively. Floc sizes at both levels also attained a size of 1.9 mm.

Increasing the intensity to 60 rpm led to slight decreases in SS removed with 67% and 71% SS removed for trial 1 and trial 2 respectively. Floc sizes in trial 1 decreased to 1.3 mm but remained at 1.9 mm in trial 2.

Further increasing the intensity of flocculation to 80 rpm led to further decreases in SS removal capacities for both trials. About 59% and 64% SS were removed for trial 1 and trial 2 respectively. Floc sizes in trial 1 decreased further to 0.88 mm and floc sizes in trial 2 decreased to 1.3 mm.

4.4.9 Zetag 92 and Dosage Rates

The final trials conducted varied the dosage rates of Zetag 92. The stock polymer solution was 500 mg/L and the final concentration of Zetag 92 in the beaker was 10 mg/L. A total of 10 mLs was therefore dosed or injected into the vortex by a hand held syringe. The timing was monitored with a timer. The dosage rates were: 10 mLs/sec (the whole 10 mLs was injected within virtually one second); 5 mLs/sec, 2 mLs/sec, 1 mL/sec (10 mLs in 10 seconds) and 0.5 mLs/sec (10 mLs in 20 seconds). As soon as the polymer was dosed, 10 mLs of distilled water was rinsed through the syringe and flushed into the vortex to ensure all the polymer was added.
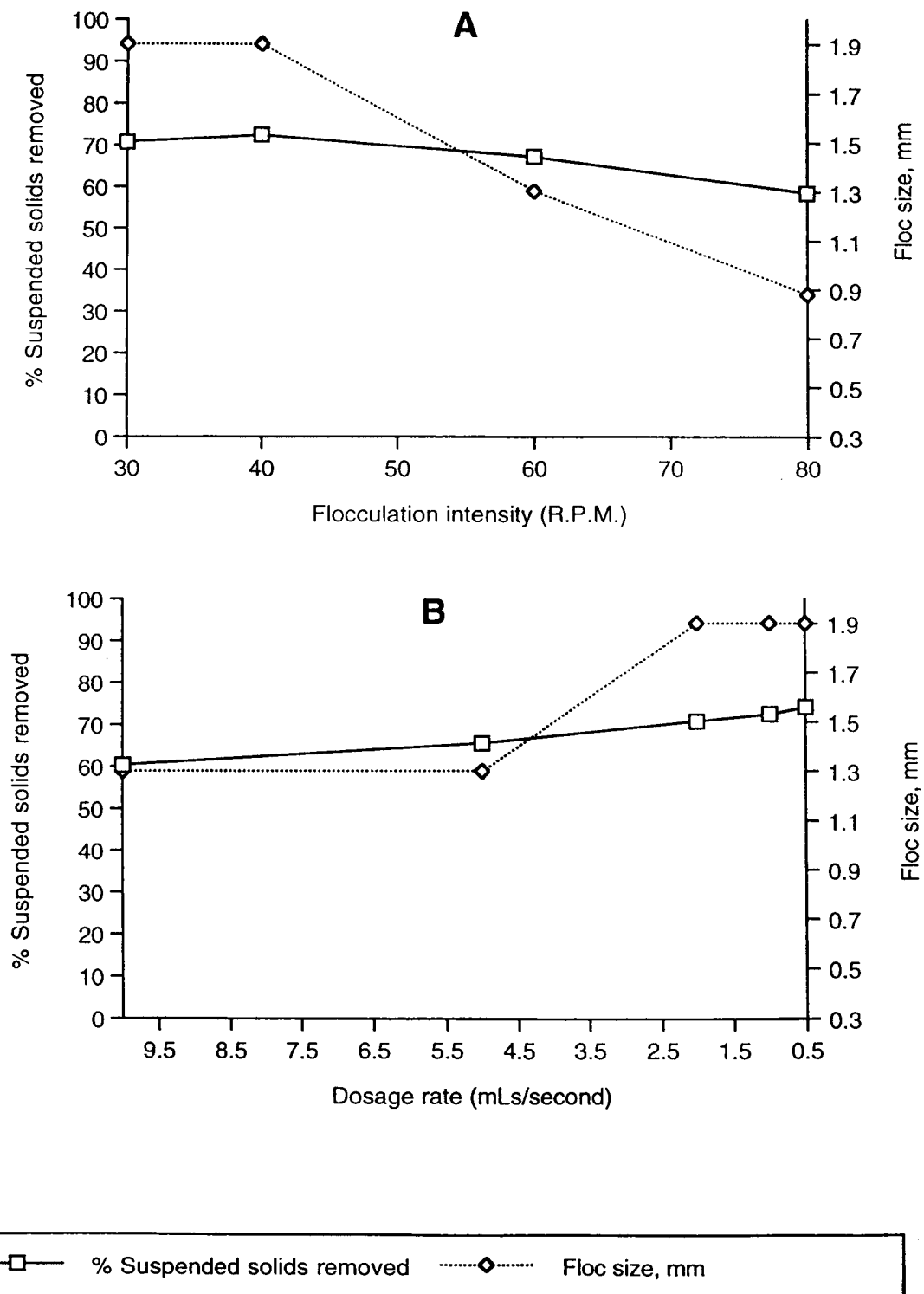
Figure 4.9. The effects of varying **flocculation intensity (A)** and varying **dosage rate (B)** on suspended solids removals (%) and final floc floc sizes attained (mm) when using cationic **Zetag 92** (10 mg/L, pH 8) as the primary coagulant.

The highest levels of SS removals were obtained for dosage rates of 2 mLs/sec, 1 mL/sec and 0.5 mLs/sec. At these three doses 71%, 72% and 74% SS were removed in trial 1, and 71%, 75% and 73% SS were removed in trial 2, respectively. The final floc sizes attained at all these dose rates were 1.9 mm for both trials (see Table E4.9). Figure 4.9B clearly shows that increasing the dose rate from 0.5 mLs/sec to 2 mLs/sec had no effect on SS removal and floc growth. Increasing the dosage rate to 5 mLs/sec and 10 mLs/sec led to a decrease in floc size from 1.9 mm to 1.3 mm. This was typical of both dose rates and both trials. The SS removal efficiency also dropped slightly, with about 66% and 60% for 5 mLs/sec and 10 mLs/sec, respectively in trial 1 and about 67% and 62% SS removal for 5 mLs/sec and 10 mLs/sec, respectively in trial 2.

4.5 ALUMINIUM SULPHATE COAGULATION AND FERRIC SULPHATE COAGULATION.

A series of jar tests were conducted in order to evaluate the efficiency of aluminium sulphate and ferric sulphate as primary coagulants. The efficiency of these coagulants was evaluated only by the measure of suspended solids (SS) removed and final floc sizes attained, mm. The final pH of each test was also determined. This work is similar to the work conducted for section 4.3 but this work profiled aluminium sulphate and ferric sulphate over a very wide dosage range.

4.5.1 Aluminium sulphate

Tables F5.1-F5.5 and Figures 4.10A, 4.11A and 4.12A detail the results obtained with aluminium sulphate coagulation of suspended solids. The results indicate that at all pH levels, highest removals were obtained with 75 mg/L. Figure 4.10A indicates that at pH 8 the highest levels of SS removals were obtained (with the optimum dose being between 75 mg/L and 100 mg/L), followed by pH 9. At pH 7 and pH 10, the levels of SS removed were very similar. The pH at which the least amount of SS was removed was at pH 6. Removals at pH 6, 7, 8, 9, and 10 were, for 75 mg/L, 26.7%, 32.6%, 54.2%, 46.8% and 31.2% respectively.

At pH 6, SS removal was quite low, reaching only 21.9% SS removal at 50 mg/L. When the dose was increased beyond 75 mg/L to 100 mg/L a decline in the efficiency of SS removal was observed, with levels reaching 15%, 10%

and about 6% SS removals with 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 1.1 mm, 0.63 mm, 0.52 mm and 0.4 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 5.15 was recorded.

At pH 7, SS removal improved, reaching 28.7% SS removal at 50 mg/L. When the dose was increased beyond 75 mg/L to 100 mg/L a decline in the efficiency of SS removal was observed, with levels reaching 32%, 27% and about 16% SS removals with 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 1.3 mm, 1.1 mm, 1.1 mm and 0.52 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 5.41 was recorded.

At pH 7 a big improvement was noted compared to pH 6. Higher levels of SS were removed at the equivalent doses, and larger flocs were formed.

At pH 8, SS removals attained their highest levels, reaching 46.8% SS removal at 50 mg/L, and 54.2% SS removal at 75 mg/L. When the dose was increased to 100 mg/L only a slight increase in the efficiency of SS removal was observed, with levels reaching 56%. Dosing beyond 100 mg/L, led to a decline in the levels of SS removal with 41% and 21.1% SS removals attained at 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 1.6 mm, 1.6 mm, 1.3 mm and 1.1 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 6.59 was recorded and at 100 mg/L a final pH of 6.27 was recorded.

At pH 8 a very big improvement was noted compared to pH 7. Higher levels of SS were removed at the equivalent doses, and larger flocs were formed.

At pH 9, SS removals attained high levels, reaching 36.1% SS removal at 50 mg/L, and 46.8% SS removal at 75 mg/L. Dosing beyond 75 mg/L, led to a decline in the levels of SS removal with 39.1%, 31.1% and 26.1% SS removals attained at 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 1.3 mm, 1.1 mm, 1.1 mm and 0.88 mm were the final floc sizes
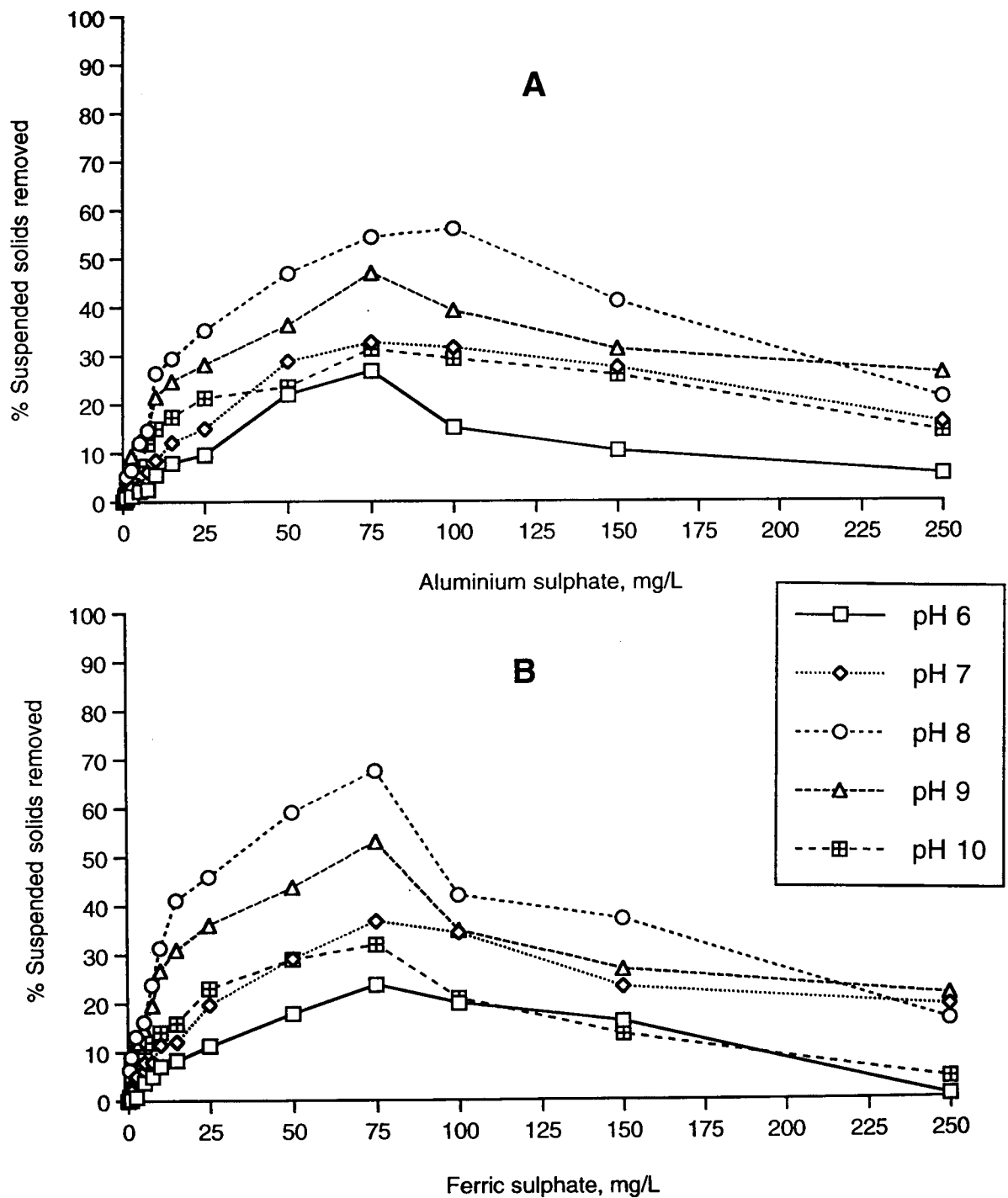
Figure 4.10. Effect of **aluminium sulphate (A)** and **ferric sulphate (B)** on Suspended solids removal from starch effluent over a broad dosage range and pH range.
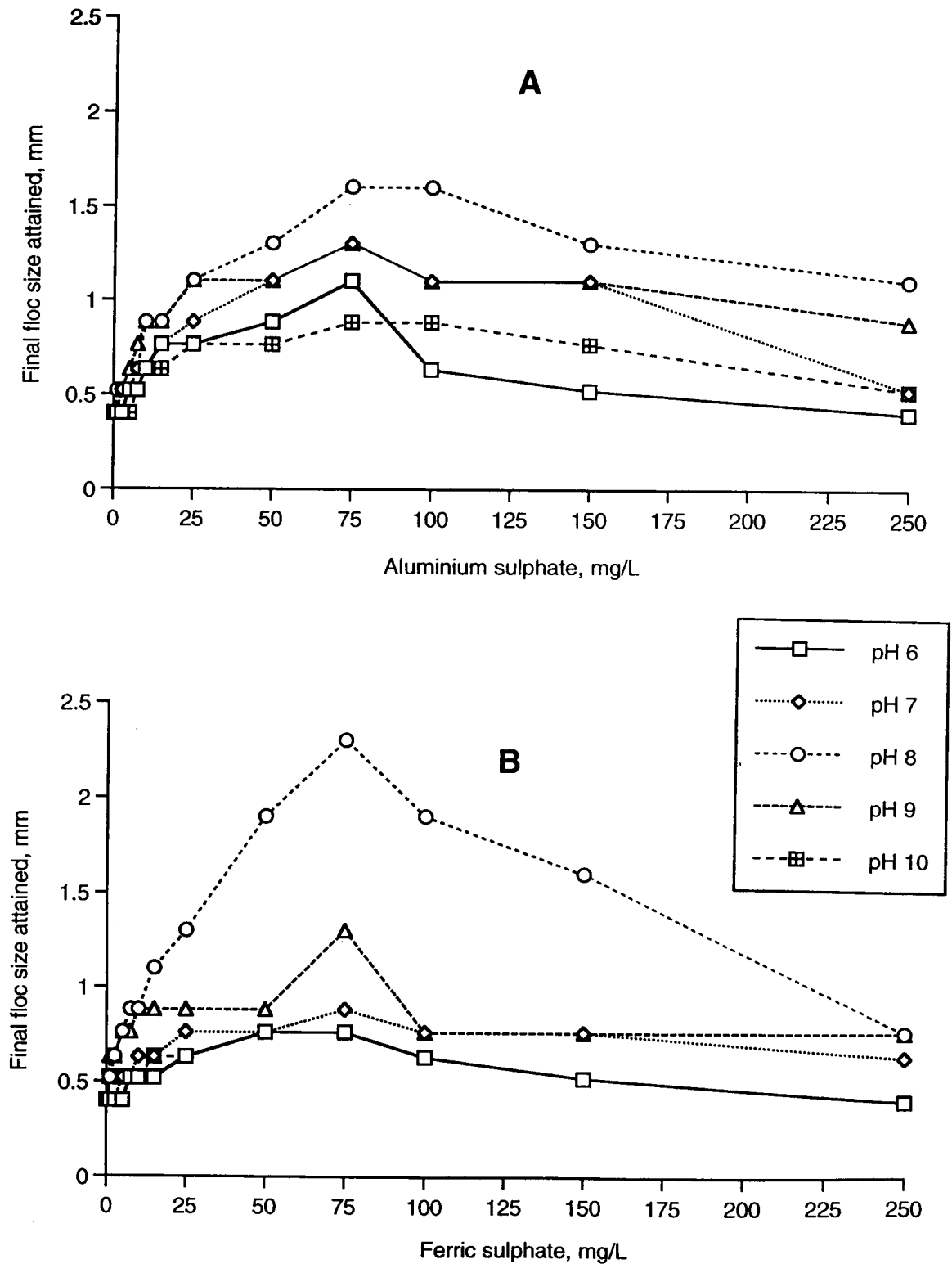
Figure 4.11. Effect of **aluminium sulphate (A)** and **ferric sulphate (B)** on final floc sizes attained (mm) in starch effluent over a broad dosage range and pH range.
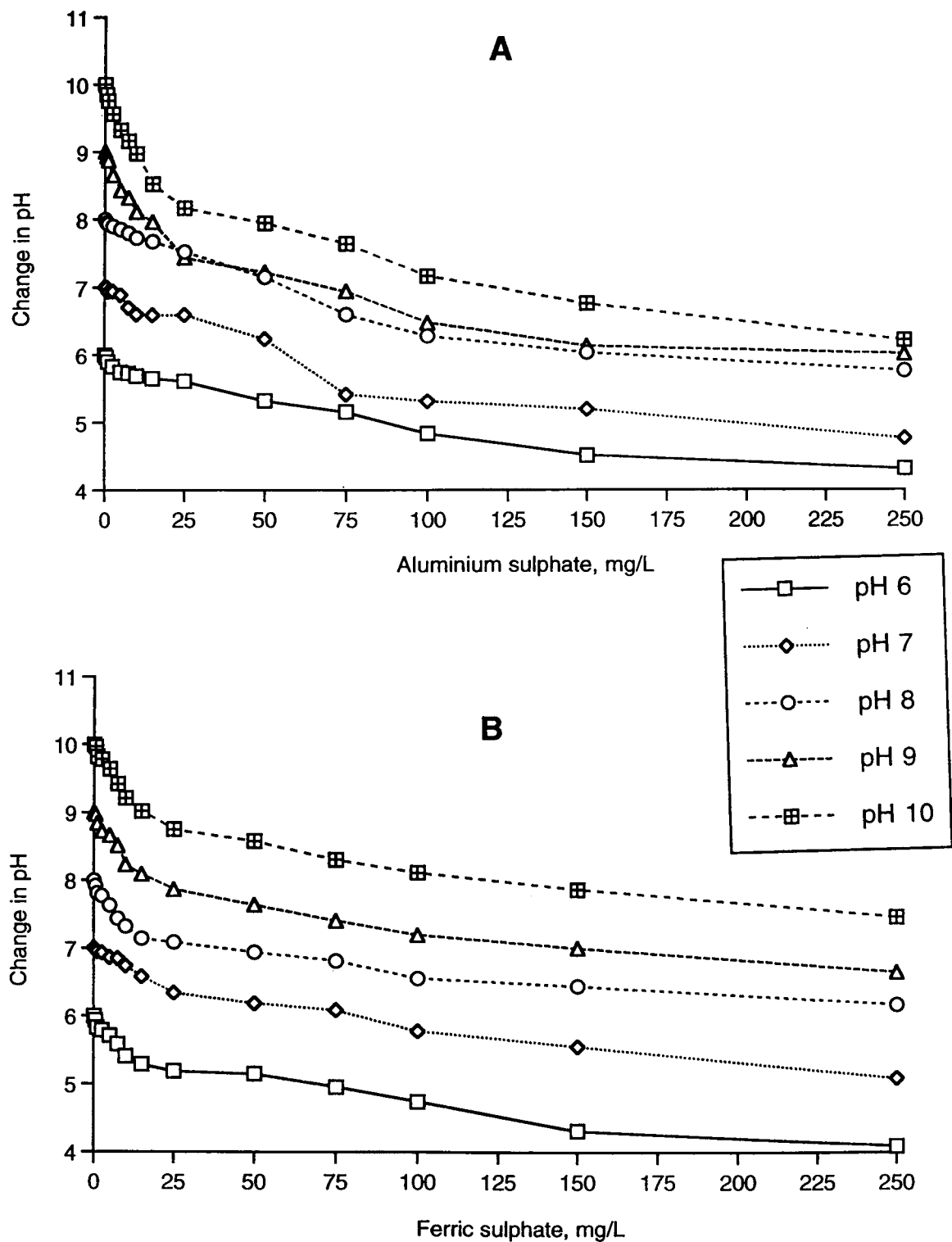
Figure 4.12. Effect of **aluminium sulphate (A)** and **ferric sulphate (B)** on final pH in starch effluent over a broad dosage range and pH range.

attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 6.93 was recorded.

At pH 9 a slight reduction in the efficiency of SS removal and final floc size was noted compared to pH 8. Slightly lower levels of SS were removed at the equivalent doses, and smaller flocs were formed.

At pH 10, SS removals attained relatively lower levels compared to the other pH levels (except pH 6), reaching 23.5% SS removal at 50 mg/L, and 31.2% SS removal at 75 mg/L. Dosing beyond 75 mg/L, led to a decline in the levels of SS removal with 29.2%, 25.8% and 14.2% SS removals attained at 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 0.88 mm, 0.88 mm, 0.76 mm and 0.52 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 7.63 was recorded.

At pH 10 a big reduction in the efficiency of SS removals and final floc size was noted compared to pH 9 and the other pH levels. Lower levels of SS were removed at the equivalent doses, and much smaller flocs were formed.

4.5.2 Ferric sulphate

Tables F5.6-F5.10 and Figures 5.10B, 5.11B and 5.12B detail the results obtained with aluminium sulphate coagulation of suspended solids. The results indicate that at all pH levels, highest removals were obtained with 75 mg/L. Figure 5.10B indicates that at pH 8 the highest levels of SS removals were obtained followed by pH 9. At pH 7 and pH 6, the levels of SS removed were very similar. The pH at which the least amount of SS was removed was at pH 10. Removals at pH 6, 7, 8, 9, and 10 were, for 75 mg/L, 23.8%, 36.8%, 67.5%, 53% and 32% respectively.

At pH 6, SS removal was low, reaching only 17.9% SS removal at 50 mg/L. When the dose was increased beyond 75 mg/L to 100 mg/L a decline in the efficiency of SS removal was observed, with levels reaching 20%, 16.1% and about 0.6% SS removals with 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 0.76 mm, 0.63 mm, 0.52 mm and 0.4 mm were the

final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 4.96 was recorded.

Comparing ferric sulphate coagulation and removal of SS with aluminium sulphate at pH 6 (Tables F5.1 and F5.6) the levels of SS removal are very similar at each dose. Aluminium sulphate coagulation also produced much larger flocs at pH 6 (1.1 mm at 75 mg/L), compared to ferric sulphate (0.76 mm at 75 mg/L). At all doses ferric sulphate resulted in slightly lower pH readings (5.15 at 75 mg/L) compared to aluminium sulphate (4.96 at 75 mg/L).

At pH 7, SS removal improved, reaching 29.1% SS removal at 50 mg/L. When the dose was increased beyond 75 mg/L to 100 mg/L a decline in the efficiency of SS removal was observed, with levels reaching about 34%, 23% and about 19% SS removals with 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 0.88 mm, 0.76 mm, 0.76 mm and 0.63 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 6.09 was recorded.

Ferric sulphate coagulation showed a big improvement at pH 7 compared to pH 6. Higher levels of SS were removed at the equivalent doses, and larger flocs were formed.

Comparing ferric sulphate coagulation and removal of SS with aluminium sulphate at pH 7 (Tables F5.2 and F5.7) the levels of SS removals are very similar with most doses. Aluminium sulphate coagulation again produced much larger flocs at pH 7 (1.3 mm at 75 mg/L), compared to ferric sulphate (0.88 mm at 75 mg/L). At all doses ferric sulphate resulted in slightly lower pH readings (5.41 at 75 mg/L) compared to aluminium sulphate (6.09 at 75 mg/L).

At pH 8, SS removals attained their highest levels, reaching 59.1% SS removal at 50 mg/L, and 67.5% SS removal at 75 mg/L. Dosing beyond 75 mg/L, led to a decline in the levels of SS removal with 42%, 37.1% and 16.2% SS removals attained at 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 2.3 mm, 1.9 mm, 1.6 mm and 0.76 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 6.82 was recorded.

Ferric sulphate coagulation showed a big improvements at pH 8 compared to pH 7. Higher levels of SS were removed at the equivalent doses, and much larger flocs were formed.

Comparing ferric sulphate coagulation and removal of SS with aluminium sulphate at pH 8 (Tables F5.3 and F5.8) the levels of SS removals are much larger than those attained by alum. Figure 4.11 shows aluminium sulphate coagulation produced smaller flocs at pH 8 (1.6 mm at 75 mg/L), compared to ferric sulphate (2.3 mm at 75 mg/L). At most doses ferric sulphate resulted in slightly lower pH readings. This was reversed when from doses of 75 mg/L and greater, ferric sulphate resulted in slightly higher pH readings (6.82 at 75 mg/L) compared to aluminium sulphate (6.59 at 75 mg/L).

At pH 9, SS removals attained high levels, reaching 43.7% SS removal at 50 mg/L, and 53% SS removal at 75 mg/L. Dosing beyond 75 mg/L, led to a decline in the levels of SS removal with 34.9%, 26.8% and 21.4% SS removals attained at 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 1.3 mm, 0.76 mm, 0.76 mm and 0.76 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 7.41 was recorded.

At pH 9 a slight reduction in the efficiency of SS removals and final floc size was noted compared to pH 8. Slightly lower levels of SS were removed at the equivalent doses, and smaller flocs were formed.

Comparing ferric sulphate coagulation and removal of SS with aluminium sulphate at pH 9 (Tables F5.4 and F5.9) the levels of SS removals are much larger than those attained by alum. Aluminium sulphate coagulation produced similar flocs at pH 9 (1.3 mm at 75 mg/L), compared to ferric sulphate (1.3 mm at 75 mg/L). At most doses ferric sulphate resulted in higher pH readings (7.41 at 75 mg/L) compared to aluminium sulphate (6.93 at 75 mg/L).

At pH 10, SS removals attained relatively lower levels compared to the other pH levels reaching 29% SS removal at 50 mg/L, and 32% SS removal at 75 mg/L. Dosing beyond 75 mg/L, led to a decline in the levels of SS removal with 21%, 13.5% and 4.2% SS removals attained at 100 mg/L, 150 mg/L and 250 mg/L, respectively. These declines in SS removal efficiency corresponded with a decline in the final floc sizes attained: 0.76 mm, 0.63 mm,

0.52 mm and 0.4 mm were the final floc sizes attained at 75 mg/L, 100 mg/L, 150 mg/L and 250 mg/L, respectively. At 75 mg/L a final pH of 8.31 was recorded.

At pH 10 a big reduction in the efficiency of SS removals and final floc size was noted compared to pH 9 and the other pH levels. Lower levels of SS were removed at the equivalent doses, and much smaller flocs were formed.

Comparing ferric sulphate coagulation and removal of SS with aluminium sulphate at pH 10 (Tables F5.5 and F5.10) the levels of SS removals were similar to those attained by alum. Aluminium sulphate coagulation produced similar flocs at pH 10 (0.88 mm at 75 mg/L), compared to ferric sulphate (0.76 mm at 75 mg/L). At all doses ferric sulphate resulted in higher pH readings (8.31 at 75 mg/L) compared to aluminium sulphate (7.63 at 75 mg/L).

4.6 THE USE OF INORGANIC COAGULANTS AS PRIMARY COAGULANTS AND ORGANIC POLYMERS AS COAGULANT AIDS.

A series of jar tests were conducted in order to evaluate the efficiency of aluminium sulphate and ferric sulphate as primary coagulants. These trials also evaluated the use of two polyelectrolytes, Zetag 92 (cationic) and Magnafloc 336 (anionic) as coagulant aids. The efficiency of these trials were evaluated by the measure of the removal of suspended solids (SS), total organic carbon (TOC), turbidity (NTU), total oil and grease (TOG), capillary suction time of the sludge (CST), and final floc size attained (mm).

4.6.1 Suspended Solids Removal

4.6.1.1 Zetag 92 and Aluminium sulphate (Figure 4.13).

When aluminium sulphate was used as the only coagulant, SS removals increased when the pH was increaed from pH 8 to pH 10. At pH 8 25% SS was removed compared to 45% SS removed at pH 10, when alum was used at 75 mg/L. The highest SS removals were obtained with the highest alum doses of 75 mg/L, at both pH levels, although pH 10 yielded higher SS reductions.

The addition of Zetag 92 as a coagulant aid improved the SS removals at both pH levels. At pH 8, Figure 4.13 shows an increase in SS removal with the introduction of Zetag 92. Increasing the concentration of Zetag 92 to 2.5 mg/L led to slight increases in SS removals. Further increasing the concentration of Zetag 92 led to higher removals with approximately 27%, 35% and 43% SS removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. These results were produced with 75 mg/L alum.

When the pH was increased from pH 8 to pH 10, very high increases in SS removals were obtained. Figure 4.13 shows this big increase in SS removal efficiency at pH 10. At 75 mg/L alum, approximately 43%, 56% and 59% SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. At pH 10, Zetag 92 performed similarly at conentrations of 5.0 mg/L and 7.5 mg/L.

4.6.1.2 Zetag 92 and Ferric sulphate (Figure 4.14).

When ferric sulphate was used as the only coagulant, SS removals increased when the pH was increaed from pH 8 to pH 10. At pH 8 31% SS was removed compared to 43% SS removed at pH 10, when ferric sulphate was used at 75 mg/L. The highest SS removals were obtained with the highest ferric sulphate doses of 75 mg/L, at both pH levels, although pH 10 yielded much higher SS reductions.

The addition of Zetag 92 as a coagulant aid improved the SS removals at both pH levels. At pH 8, Figure 4.14 shows an increase in SS removal with the introduction of Zetag 92. Increasing the concentration of Zetag 92 to 2.5 mg/L led to increases in SS removals. Further increasing the concentration of Zetag 92 led to higher removals with approximately 35%, 41% and 43% SS removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. These results were produced with 75 mg/L ferric sulphate.

When the pH was increased from pH 8 to pH 10, very high increases in SS removals were obtained. Figure 4.14 shows this big increase in SS removal efficiency at pH10. At 75 mg/L ferric sulphate, approximately 55%, 60% and 66% SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively.

Comparing aluminium sulphate to ferric sulphate, without Zetag 92, ferric sulphate was able to remove slightly higher levels of SS at pH 8. Increasing

Figure 4.13. Effect of the cationic polymer **Zetag 92** on suspended solids removal, alone or in conjunction with **aluminium sulphate**

Figure 4.14. Effect of the cationic polymer **Zetag 92** on suspended ‑solids removal, alone or in conjunction with **ferric sulphate**

	Zetag 92, 0 mg/L		Zetag 92, 2.5 mg/L
	Zetag 92, 5.0 mg/L		Zetag 92, 7.5 mg/L
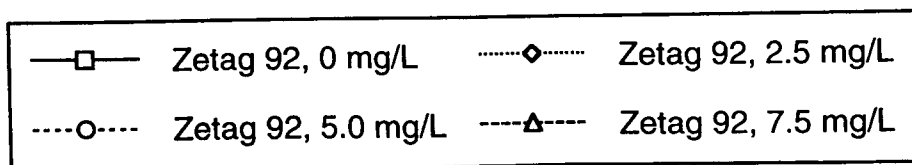
the pH from pH 8 to pH10 led to similar SS removals for both aluminium sulphate and ferric sulphate.

When Zetag 92 was used as a coagulant aid, ferric sulphate was able to remove higher levels of SS than aluminium sulphate. This was true at pH 8, for all concentrations of Zetag 92. At pH 10, the addition of Zetag 92 to aluminium sulphate and to ferric sulphate resulted in similar SS removals at all Zetag 92 concentrations.

4.6.1.3 Magnafloc 336 and Aluminium sulphate (Figure 4.15).

When aluminium sulphate was used as the only coagulant, SS removals increased when the pH was increaed from pH 8 to pH 10. At pH 8 25% SS was removed compared to 45% SS removed at pH 10, when alum was used at 75 mg/L. The highest SS removals were obtained with the highest alum doses of 75 mg/L, at both pH levels, although pH 10 yielded higher SS reductions.

The addition of Magnafloc 336 as a coagulant aid improved the SS removals at both pH levels. At pH 8, Figure 4.15 shows an increase in SS removal with the introduction of Magnafloc 336. Increasing the concentration of Magnafloc 336 to 2.5 mg/L led to significant increases in SS removals. Further increasing the concentration of Magnafloc 336 led to higher removals with approximately 40%, 55% and 52% SS removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. These results were produced with 75 mg/L alum.

When the pH was increased from pH 8 to pH 10, high increases in SS removals were obtained only for Magnafloc 336 concentrations of 2.5 mg/L and 5.0 mg/L. SS removals at 7.5 mg/L were only slightly higher at pH 10 compared to pH 8. Figure 4.15 shows this increase in SS removal efficiency at pH10. At 7.5 mg/L Magnafloc 336, SS removals were very similar to those of 5.0 mg/L.

4.6.1.4 Magnafloc 336 and Ferric sulphate (Figure 4.16).

When ferric sulphate was used as the only coagulant, SS removals increased when the pH was increaed from pH 8 to pH 10. At pH 8 31% SS was removed compared to 43% SS removed at pH 10, when ferric sulphate was used at 75

mg/L. The highest SS removals were obtained with the highest ferric sulphate doses of 75 mg/L, at both pH levels, although pH 10 yielded much higher SS reductions.

The addition of Magnafloc 336 as a coagulant aid improved the SS removals at both pH levels but only slightly. At pH 8, Figure 4.16 shows a slight increase in SS removal with the introduction of Magnafloc 336. Increasing the concentration of Magnafloc 336 to 2.5 mg/L led to increases in SS removals. Further increasing the concentration of Magnafloc 336 only led to slightly higher removals with approximately 39%, 45% and 50% SS removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. These results were produced with 75 mg/L ferric sulphate.

When the pH was increased from pH 8 to pH 10, high increases in SS removals were obtained. Figure 4.16 shows this increase in SS removal efficiency at pH10. At 75 mg/L ferric sulphate, approximately 52%, 55% and 58% SS were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively.

Comparing aluminium sulphate to ferric sulphate, without Magnafloc 336, ferric sulphate was able to remove slightly higher levels of SS at pH 8. Increasing the pH from pH 8 to pH10 led to similar SS removals for both aluminium sulphate and ferric sulphate.

When Magnafloc 336 was used as a coagulant aid, aluminium sulphate removed slightly higher levels of SS than did ferric sulphate at pH 8. At pH 10, the addition of Magnafloc 336 to aluminium sulphate and to ferric sulphate resulted in similar SS removals at all Magnafloc 336 concentrations.

When Magnafloc 336 and Zetag 92 were used as the primary coagulants (that is, no aluminium or ferric sulphate) they performed similarly in their capacity to reduce SS levels in the effluent. Increasing the pH to pH 10 led to greater improvements in the capacity of Zetag 92 to remove SS levels in the effluent although Magnafloc 336 only increased slightly in its capacity to remove SS from the effluent with the increase in pH.

With the addition of alum, Magnafloc 336 gave a signifcant increase in SS removal compared to Zetag 92, for pH 8. With the addition of ferric sulphate both Magnafloc 336 and Zetag 92 yielded similar SS removals at pH 8. At

Figure 4.15. Effect of the anionic polymer **Magnafloc 336** on suspended solids removal, alone or in conjunction with **aluminium sulphate**

Figure 4.16. Effect of the anionic polymer **Magnafloc 336** on suspended solids removal, alone or in conjunction with **ferric sulphate**

Legend	
—□— Magnafloc 336, 0 mg/L	·····◇····· Magnafloc 336, 2.5 mg/L
----O---- Magnafloc 336, 5.0 mg/L	----△---- Magnafloc 336, 7.5 mg/L
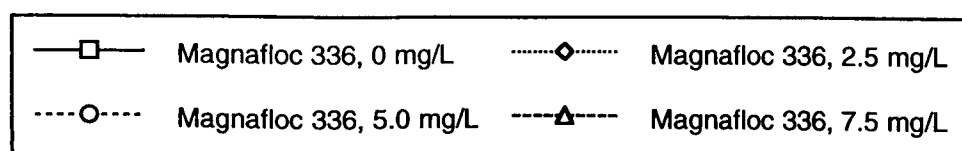
pH10, addition of alum greatly raised the SS removing capacity of Zetag 92 but only slightly increased the capacity of Magnafloc 336 to remove SS. This trend at pH 10 was also observed with ferric sulphate.

4.6.2 Total Organic Carbon Removal

4.6.2.1 Zetag 92 and Aluminium sulphate (Figure 4.17).

When aluminium sulphate was used as the only coagulant, TOC removals increased when the pH was increaed from pH 8 to pH 10. It was observed that at 50 mg/L the highest TOC removals were obtained at pH 8. Increasing the alum concentration to 75 mg/L actually caused a decrease in TOC removal at this pH. At pH 8, approximately 21%, 25% and 15% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L. At pH 10 it was observed that at 25 mg/L the highest TOC removals were obtained. Increasing the alum concentration to 50 mg/L and 75 mg/L actually caused a decrease in TOC removal at this pH. At pH 10, approximately 26%, 21% and 18% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L, respectively.

The addition of Zetag 92 as a coagulant aid improved the TOC removals at both pH levels. At pH 8, Figure 4.17 shows an increase in TOC removal with the introduction of Zetag 92. Increasing the concentration of Zetag 92 to 2.5 mg/L led to increases in TOC removals. Further increasing the concentration of Zetag 92 led to higher removals with approximately 32%, 40% and 43% TOC removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. These results were produced with 50 mg/L alum.

When the pH was increased from pH 8 to pH10, much higher increases in TOC removals were obtained. Figure 4.17 shows these big increases in TOC removal efficiency at pH10. At 25 mg/L alum, approximately 31%, 46% and 52% TOC were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively.

It was noted that while addition of Zetag 92 to alum increased TOC removal from the effluent the same trends were observed regarding a drop in efficiency, that is, at pH 8, alum concentrations exceeding 50 mg/L and at pH 10 alum concentrations exceeding 25 mg/L-50 mg/L led to a decrease in the capacity of alum to remove TOC.

4.6.2.2 Zetag 92 and Ferric sulphate (Figure 4.18).

When ferric sulphate was used as the only coagulant, TOC removals increased when the pH was increaed from pH 8 to pH 10. It was observed that at 50 mg/L the highest TOC removals were obtained at pH 8. Increasing the ferric sulphate concentration to 75 mg/L actually caused a decrease in TOC removal at this pH. At pH 8, approximately 21%, 20% and 12% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L. At pH 10 it was observed that at 25 mg/L the highest TOC removals were obtained. Increasing the ferric sulphate concentration to 50 mg/L and 75 mg/L actually caused a decrease in TOC removal at this pH. At pH 10, approximately 29%, 25% and 20% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L, respectively.

The addition of Zetag 92 as a coagulant aid improved the TOC removals at both pH levels. At pH 8, Figure 4.18 shows an increase in TOC removal with the introduction of Zetag 92. Increasing the concentration of Zetag 92 to 2.5 mg/L led to increases in TOC removals. Further increasing the concentration of Zetag 92 led to higher removals with approximately 33%, 38% and 49% TOC removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. These results were produced with 50 mg/L ferric sulphate.

When the pH was increased from pH 8 to pH 10, much higher increases in TOC removals were obtained. Figure 4.18 shows these big increases in TOC removal efficiency at pH 10. At 25 mg/L ferric sulphate, approximately 35%, 50% and 57% TOC were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively.

It was noted that while addition of Zetag 92 to ferric sulphate increased TOC removal from the effluent the same trends were observed regarding a drop in efficiency, that is, ferric sulphate concentrations exceeding 50 mg/L at pH 8 and ferric sulphate concentrations exceeding 25 mg/L at pH 10 led to a decrease in the capacity of ferric sulphate to remove TOC.

Comparing aluminium sulphate to ferric sulphate, without Zetag 92, both coagulants removed very similar levels of TOC at pH 8. Increasing the pH from pH 8 to pH10 led to similar TOC removals for both aluminium sulphate and ferric sulphate. When Zetag 92 was used as a coagulant aid, ferric sulphate and aluminium sulphate both removed very similar levels of TOC at both pH levels.
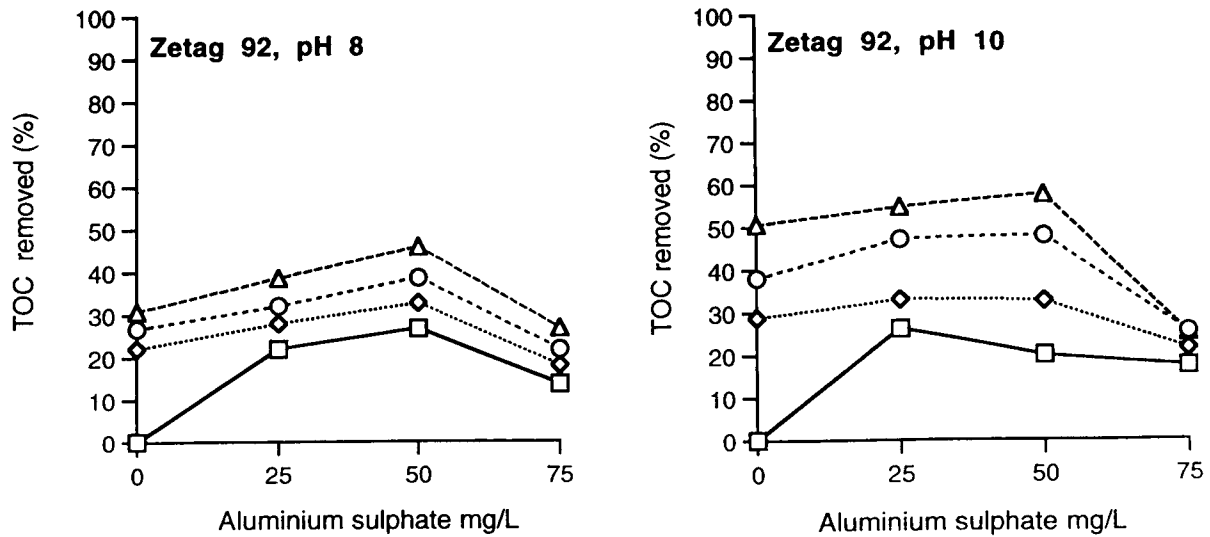
Figure 4.17. Effect of the cationic polymer **Zetag 92** on TOC removal, alone or in conjunction with **aluminium sulphate**
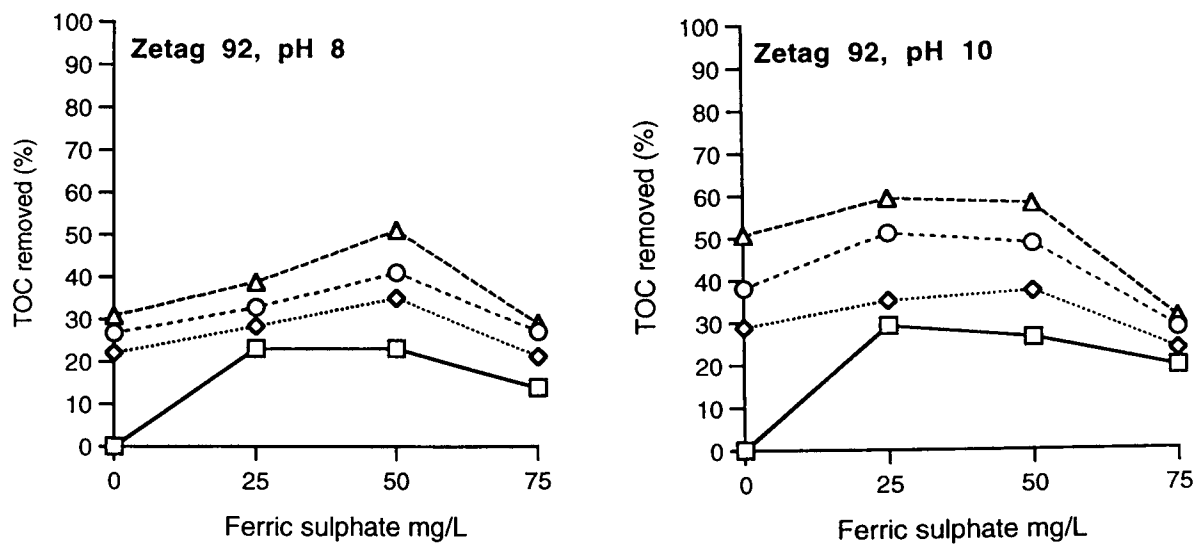
Figure 4.18. Effect of the cationic polymer **Zetag 92** on TOC removal, alone or in conjunction with **ferric sulphate**

—□— Zetag 92, 0 mg/L	·····◇····· Zetag 92, 2.5 mg/L	
----O---- Zetag 92, 5.0 mg/L	----△---- Zetag 92, 7.5 mg/L	
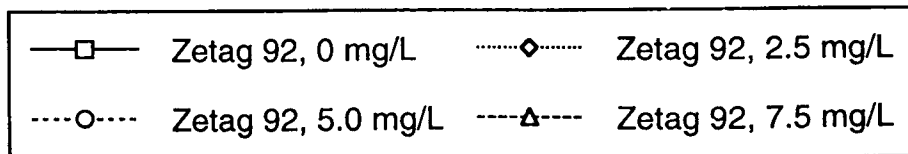
4.6.2.3 Magnafloc 336 and Aluminium sulphate (Figure 4.19).

When aluminium sulphate was used as the only coagulant, TOC removals increased slightly when the pH was increased from pH 8 to pH 10. It was observed that at 50 mg/L the highest TOC removals were obtained at pH 8. Increasing the alum concentration to 75 mg/L caused a decrease in TOC removal at this pH. At pH 8, approximately 21%, 25% and 15% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L. At pH 10 it was observed that at 25 mg/L the highest TOC removals were obtained. Increasing the alum concentration to 50 mg/L and 75 mg/L actually caused a decrease in TOC removal at this pH. At pH 10, approximately 26%, 21% and 18% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L, respectively.

At pH 8, Figure 4.19 shows an increase in TOC removal with the introduction of Magnafloc 336. Increasing the concentration of Magnafloc 336 to 2.5 mg/L led to increases in TOC removals. Further increasing the concentration of Magnafloc 336 led to higher removals with approximately 30%, 39% and 42% TOC removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. These results were produced with 50 mg/L alum.

When the pH was increased from pH 8 to pH10, only slight increases in TOC removals were obtained when Magnafloc 336 was the only coagulant. Figure 19 show these slight increases in TOC removal efficiency at pH10. As a coagulant aid Magnafloc 336 only had a slight increase in TOC removal efficiency compared to pH 8. At 25 mg/L alum, approximately 30%, 35% and 39% TOC were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively.

It was noted that while addition of Magnafloc 336 to alum increased TOC removal from the effluent the same trends were observed regarding a drop in efficiency, that is, at pH 8, alum concentrations exceeding 50 mg/L and at pH 10 alum concentrations exceeding 25 mg/L led to a decrease in the capacity of alum to remove TOC.

4.6.2.4 Magnafloc 336 and Ferric sulphate (Figure 4.20).

When ferric sulphate was used as the only coagulant, TOC removals increased when the pH was increased from pH 8 to pH 10. It was observed that at 50 mg/L the highest TOC removals were obtained at pH 8. Increasing

the ferric sulphate concentration to 75 mg/L actually caused a decrease in TOC removal at this pH. At pH 8, approximately 21%, 20% and 12% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L. At pH 10 it was observed that at 25 mg/L the highest TOC removals were obtained. Increasing the ferric sulphate concentration to 50 mg/L and 75 mg/L actually caused a decrease in TOC removal at this pH. At pH 10, approximately 29%, 25% and 20% TOC was removed at 25 mg/L, 50 mg/L and 75 mg/L, respectively.

The addition of Magnafloc 336 as a coagulant aid improved the TOC removals at pH 8 only. At pH 8, Figure 4.20 shows an increase in TOC removal with the introduction of Magnafloc 336. Increasing the concentration of Magnafloc 336 to 2.5 mg/L led to increases in TOC removals. Further increasing the concentration of Magnafloc 336 led to higher removals with approximately 28%, 38% and 41% TOC removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. These results were produced with 50 mg/L ferric sulphate.

When the pH was increased from pH 8 to pH 10, much higher increases in TOC removals were obtained only when Magnafloc 336 was used as the primary (and only) coagulant. When ferric sulphate was used as the primary coagulant, the addition of Magnafloc 336 actually caused a decrease in TOC removal. Figure 4.20 shows these decreases in TOC removal efficiency at pH 10. At 25 mg/L ferric sulphate, approximately 33%, 39% and 42% TOC were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. These TOC removal figures are slightly lower or similar to those generated by Magnafloc 336 only, (29%, 39% and 51% at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively).

It was noted that while addition of Magnafloc 336 to ferric sulphate at pH 8 increased TOC removal from the effluent the same trends were observed regarding a drop in efficiency, that is, ferric sulphate concentrations exceeding 50 mg/L at pH 8 led to a decrease in the capacity of ferric sulphate to remove TOC. At pH 10, TOC removal efficiency generally decreased with increasing concentrations of ferric sulphate.

Comparing aluminium sulphate to ferric sulphate, without Magnafloc 336 both coagulants removed very similar levels of TOC at pH 8. Increasing the pH from pH 8 to pH10 only led to slight increases in TOC removals for aluminium sulphate. When Magnafloc 336 was used as a coagulant aid, aluminium

Figure 4.19. Effect of the anionic polymer **Magnafloc 336** on TOC removal, alone or in conjunction with **aluminium sulphate**
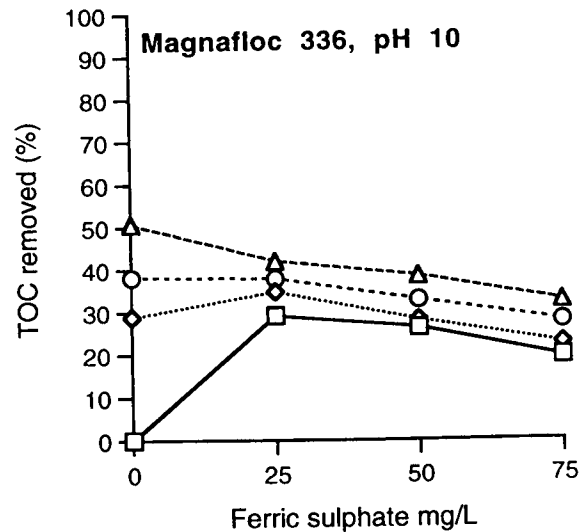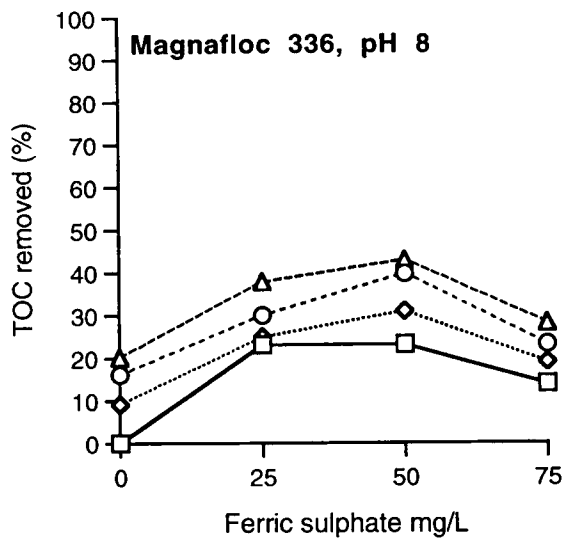
Figure 4.20. Effect of the anionic polymer **Magnafloc 336** on TOC removal, alone or in conjunction with **ferric sulphate**

—□— Magnafloc 336, 0 mg/L		⋯◇⋯ Magnafloc 336, 2.5 mg/L	
⋯O⋯ Magnafloc 336, 5.0 mg/L		⋯△⋯ Magnafloc 336, 7.5 mg/L	
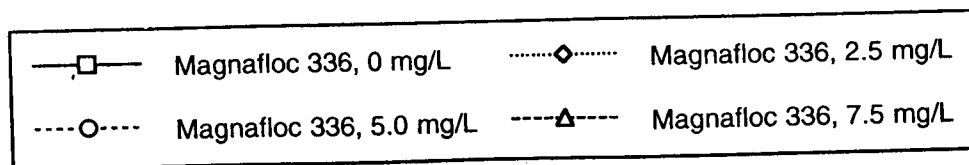
sulphate removed very similar levels of TOC at both pH levels. Ferric sulphate with Magnafloc 336 actually led to a decrease in TOC removal efficiency.

When Magnafloc 336 and Zetag 92 were used as the primary coagulants (that is, no aluminium or ferric sulphate) they performed similarly in their capacity to reduce TOC levels in the effluent, although Zetag 92 removed slightly higher levels of TOC than Magnafloc at pH 8. Increasing the pH to pH 10 led to greater improvements in the capacity of Zetag 92 to remove TOC levels in the effluent although Magnafloc 336 only increased very little in its capacity to remove TOC from the effluent with the increase in pH.

With the addition of alum, Magnafloc 336 gave slightly lower TOC removals compared to Zetag 92, for pH 8. With the addition of ferric sulphate both Magnafloc 336 and Zetag 92 yielded similar TOC removals at pH 8. At pH10, addition of alum greatly raised the TOC removing capacity of Zetag 92 but only slightly increased the capacity of Magnafloc 336 to remove TOC. This trend at pH 10 was also observed with ferric sulphate.

4.6.3 Turbidity Removal

4.6.3.1 Zetag 92 and Aluminium sulphate (Figure 4.21).

When aluminium sulphate was used as the only coagulant, Turbidity (NTU) removals increased when the pH was increaed from pH 8 to pH 10. At pH 8 59% NTU was removed compared to 72% NTU removed at pH 10, when alum was used at 75 mg/L. The highest NTU removals were obtained with the highest alum doses of 75 mg/L, at both pH levels, although pH 10 yielded higher NTU reductions.

The addition of Zetag 92 as a coagulant aid improved the NTU removals at both pH levels. At pH 8, Figure 4.21 shows an increase in NTU removal with the introduction of Zetag 92. Increasing the concentration of Zetag 92 to 2.5 mg/L led to big increases in NTU removals. Further increasing the concentration of Zetag 92 led to higher removals with approximately 69%, 73% and 75% NTU removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. These results were produced with 75 mg/L alum.

When the pH was increased from pH 8 to pH 10, high increases in NTU removals were obtained. Figure 4.21 shows these big increases in NTU

removal efficiency at pH 10. At 75 mg/L alum, approximately 82%, 83% and 87% NTU were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. At pH 10, Zetag 92 performed similarly at conentrations of 2.5 mg/L and 5.0 mg/L.

4.6.3.2 Zetag 92 and Ferric sulphate (Figure 4.22).

When ferric sulphate was used as the only coagulant, NTU removals increased when the pH was increased from pH 8 to pH 10. At pH 8 59% NTU was removed compared to 75% NTU removed at pH 10, when ferric sulphate was used at 75 mg/L. The highest NTU removals were obtained with the highest ferric sulphate doses of 75 mg/L, at both pH levels, although pH 10 yielded much higher NTU reductions.

The addition of Zetag 92 as a coagulant aid improved the NTU removals at both pH levels. At pH 8, Figure 4.22 shows an increase in NTU removal with the introduction of Zetag 92. Increasing the concentration of Zetag 92 led to higher removals with approximately 70%, 75% and 75% NTU removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively. These results were produced with 75 mg/L ferric sulphate. Increasing the ferric sulphate dose beyond 50 mg/L had little effect on turbidity removal.

When the pH was increased from pH 8 to pH 10, high increases in NTU removals were obtained. Figure 4.22 shows these big increases in NTU removal efficiency at pH10. At 75 mg/L ferric sulphate, approximately 82%, 89% and 90% NTU were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Zetag 92 respectively.

Comparing aluminium sulphate to ferric sulphate, without Zetag 92, ferric sulphate removed higher levels of NTU at pH 8. Increasing the alum dose beyond 50 mg/L at pH 8 led to further NTU removals but increasing the dose beyond 50 mg/L for ferric sulphate had little effect. Increasing the pH from pH 8 to pH10 led to similar NTU removals for both aluminium sulphate and ferric sulphate.

When Zetag 92 was used as a coagulant aid, ferric sulphate and aluminium sulphate performed similarly in their trends to remove turbidity, although at pH 8, for all concentrations of Zetag 92 ferric sulphate removed higher NTU
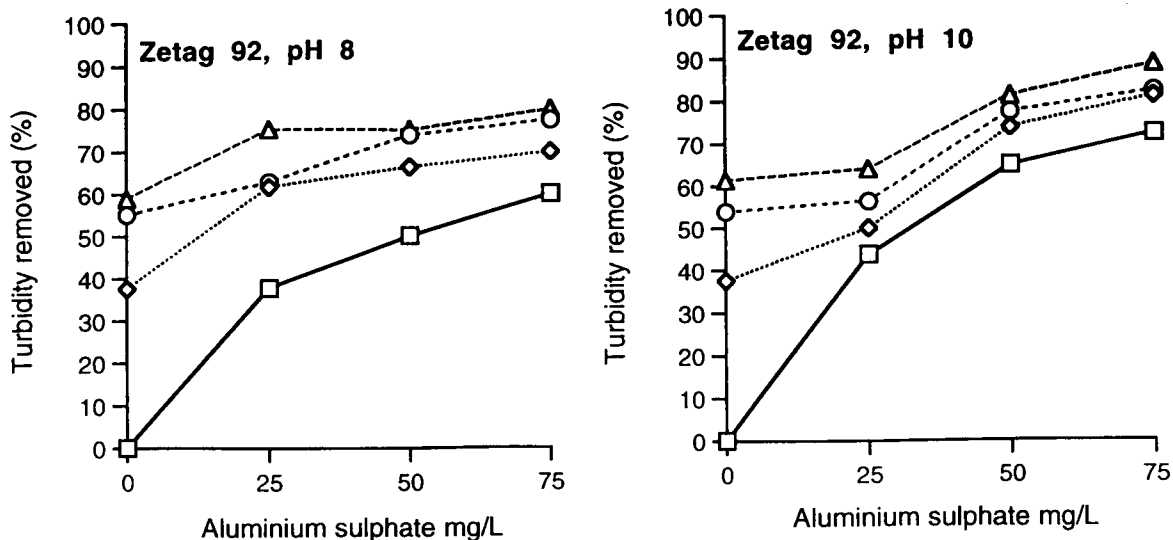
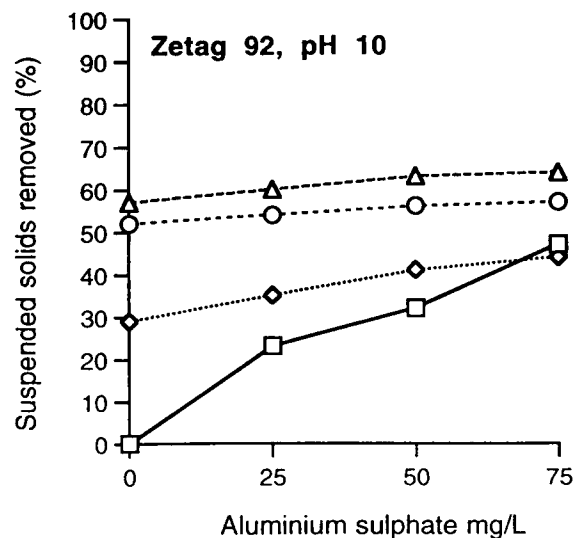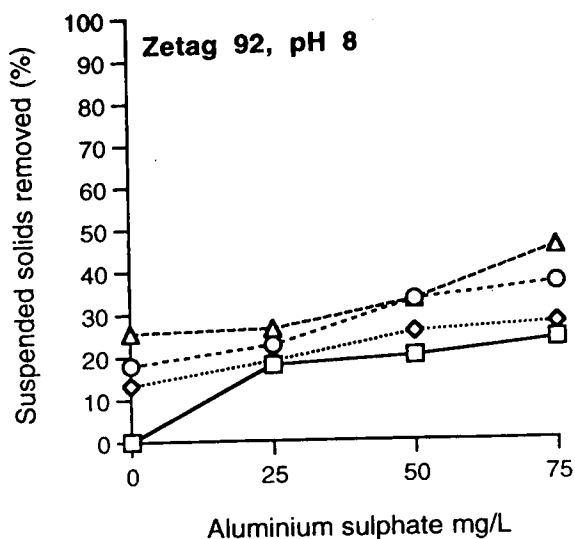**Figure 4.21.** Effect of the cationic polymer **Zetag 92** on Turbidity removal, alone or in conjunction with **aluminium sulphate**
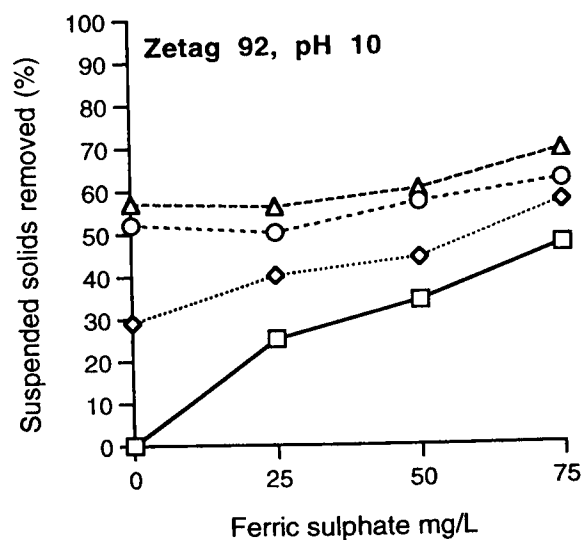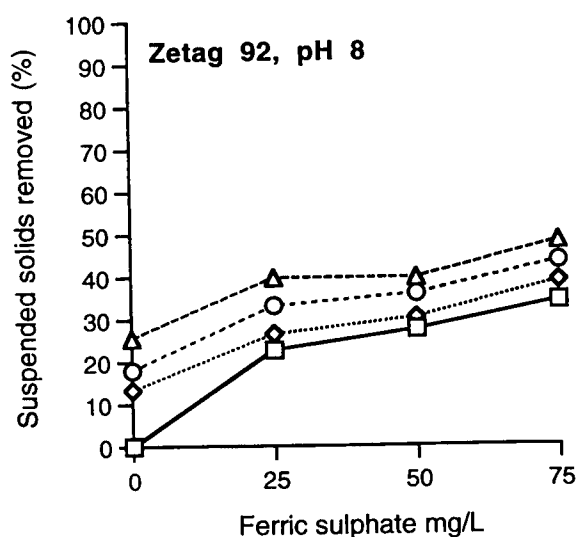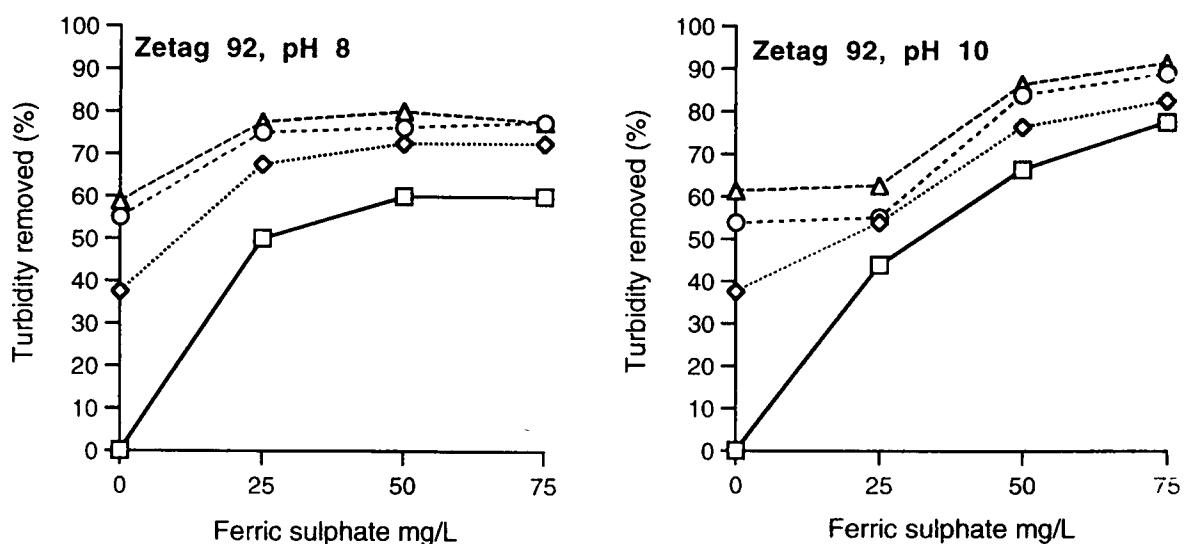

**Figure 4.22.** Effect of the cationic polymer **Zetag 92** on Turbidity removal, alone or in conjunction with **ferric sulphate**
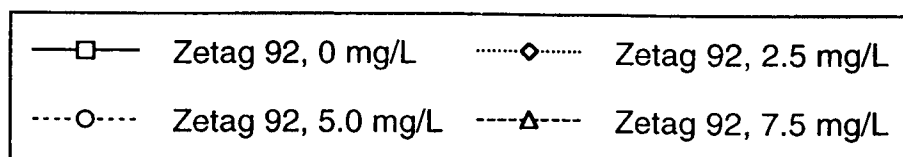
levels. At pH 10, the addition of Zetag 92 to aluminium sulphate and to ferric sulphate resulted in similar NTU removals at all Zetag 92 concentrations.

4.6.3.3 Magnafloc 336 and Aluminium sulphate (Figure 4.23).

When aluminium sulphate was used as the only coagulant, NTU removals increased when the pH was increased from pH 8 to pH 10. At pH 8, 59% NTU was removed compared to 72% NTU removed at pH 10, when alum was used at 75 mg/L. The highest NTU removals were obtained with the highest alum doses of 75 mg/L, at both pH levels, although pH 10 yielded higher NTU reductions.

The addition of Magnafloc 336 as a coagulant aid improved the NTU removals at both pH levels. At pH 8, Figure 4.23 shows large increases in NTU removal with the introduction of Magnafloc 336. Increasing the concentration of Magnafloc 336 led to high removals with approximately 63%, 71% and 74% NTU removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. These results were produced with 75 mg/L alum. Beyond 50 mg/L alum dose there was little effect on turbidity removal when Magnafloc 336 was used at pH 8.

When the pH was increased from pH 8 to pH 10, high increases in NTU removals were obtained. Magnafloc 336 concentrations of 2.5 mg/L and 5.0 mg/L yielded similar NTU removals. Turbidity removals at 7.5 mg/L were slightly higher. Figure 4.23 shows this increase in NTU removal efficiency at pH10. Dosing beyond 50 mg/L alum removed very little NTU for all Magnafloc 336 doses.

4.6.3.4 Magnafloc 336 and Ferric sulphate (Figure 4.24).

When ferric sulphate was used as the only coagulant, NTU removals increased when the pH was increaed from pH 8 to pH 10. At pH 8 59% NTU was removed compared to 75% NTU removed at pH 10, when ferric sulphate was used at 75 mg/L. The highest NTU removals were obtained with the highest ferric sulphate doses of 75 mg/L, at both pH levels, although pH 10 yielded much higher NTU reductions.

The addition of Magnafloc 336 as a coagulant aid improved the NTU removals at both pH levels significantly. At pH 8, Figure 4.24 shows a big increase in

NTU removal with the introduction of Magnafloc 336. Increasing the concentration of Magnafloc 336 to 2.5 mg/L led to slight increases in NTU removals. Further increasing the concentration of Magnafloc 336 only led to slightly higher removals with approximately 50%, 52% and 66% NTU removal at concentrations of 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. The increasing ferric sulphate concentration led to NTU removals which were only slightly higher. At 25 mg/L ferric sulphate there is only a slight increase in NTU removal. Further increasing the ferric sulphate dose did not lead to a significant improvement in turbidity removal.

When the pH was increased from pH 8 to pH 10, high increases in NTU removals were obtained. Figure 4.24 shows this increase in NTU removal efficiency at pH10. Increasing the dose of ferric sulphate beyond 50 mg/L did not increase the removal of NTU significantly. At 50 mg/L ferric sulphate, approximately 82%, 85% and 89% NTU were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively. At 75 mg/L ferric sulphate, approximately 88%, 90% and 91% NTU were removed at 2.5 mg/L, 5.0 mg/L and 7.5 mg/L Magnafloc 336 respectively.

Comparing aluminium sulphate to ferric sulphate, without Magnafloc 336, ferric sulphate was able to remove higher levels of NTU at pH 8. Increasing the pH from pH 8 to pH10 led to similar NTU removals for both aluminium sulphate and ferric sulphate.

When Magnafloc 336 was used as a coagulant aid, aluminium sulphate removed slightly higher levels of NTU than did ferric sulphate at pH 8. At pH 10, the addition of Magnafloc 336 to aluminium sulphate and to ferric sulphate resulted in higher NTU removals for both inorganic coagulants. Ferric sulphate yielded slightly higher NTU tremovals.

When Magnafloc 336 and Zetag 92 were used as the primary coagulants (that is, no aluminium or ferric sulphate) Magnafloc 336 was able to remove higher levels of turbidity at pH 8. Increasing the pH to pH 10 led to greater improvements in the capacity of Zetag 92 to remove NTU levels in the effluent although Magnafloc 336 decreased only slightly in its capacity to remove NTU from the effluent with the increase in pH.

With the addition of alum, Magnafloc 336 gave similar NTU removal compared to Zetag 92, for pH 8, although Zetag 92 gave slightly higher results. With the
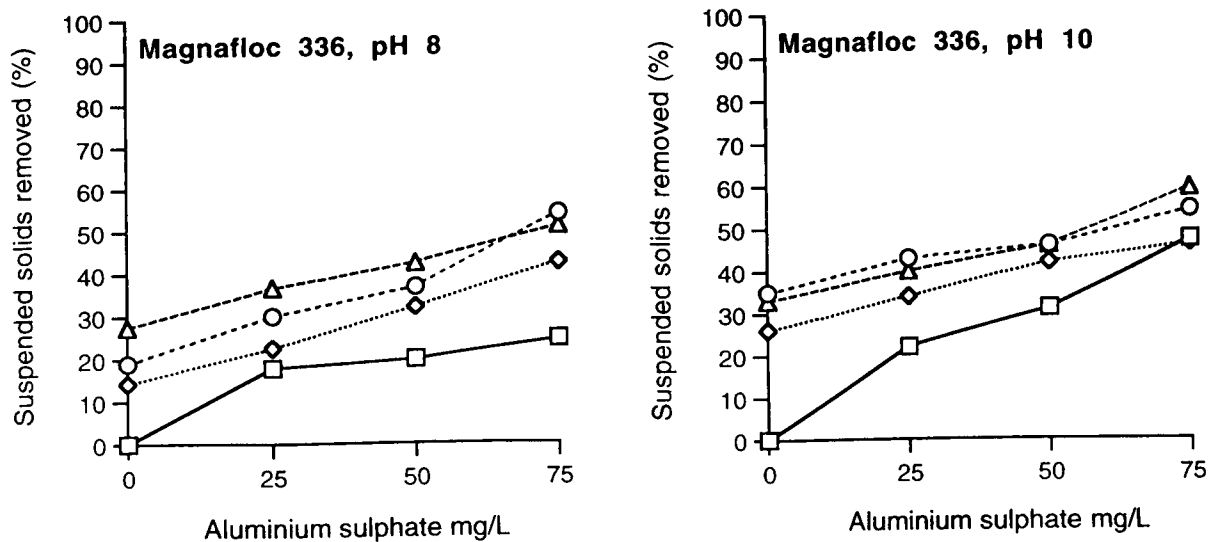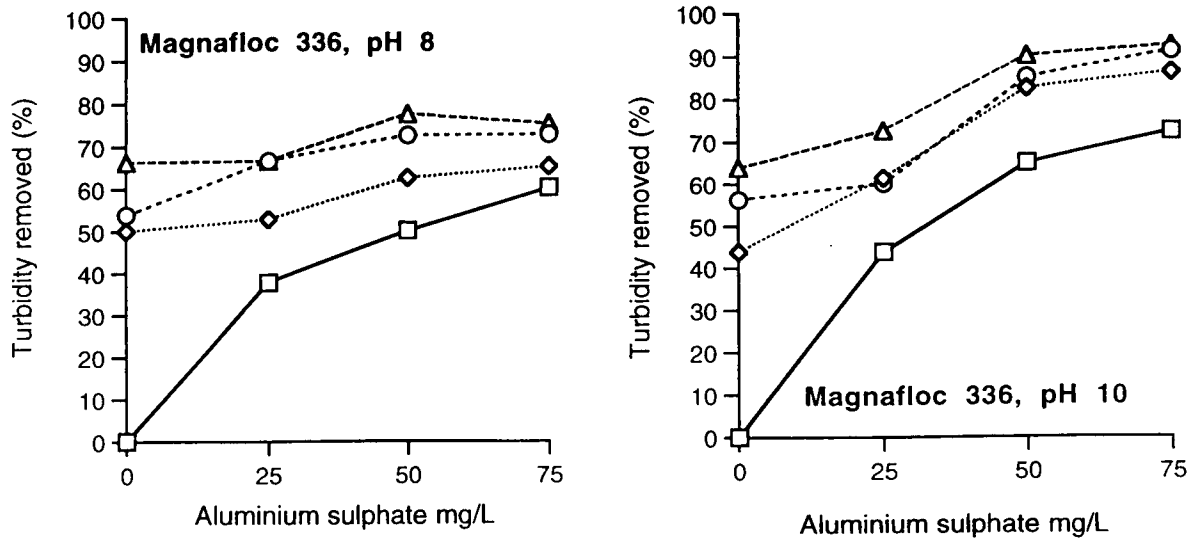
Figure 4.23. Effect of the anionic polymer **Magnafloc 336** on Turbidity removal, alone or in conjunction with **aluminium sulphate**
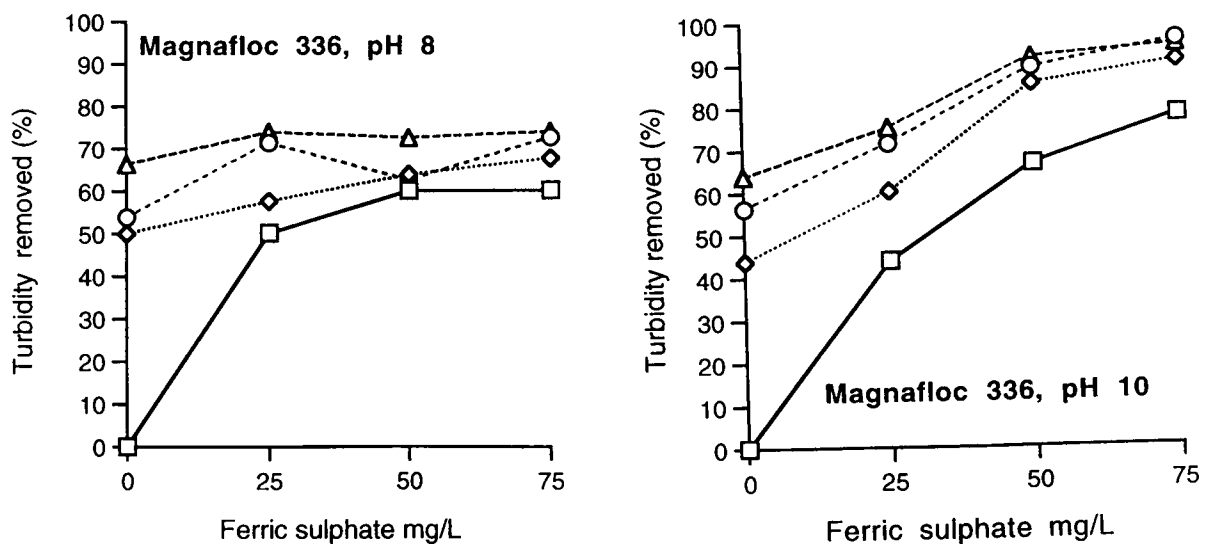
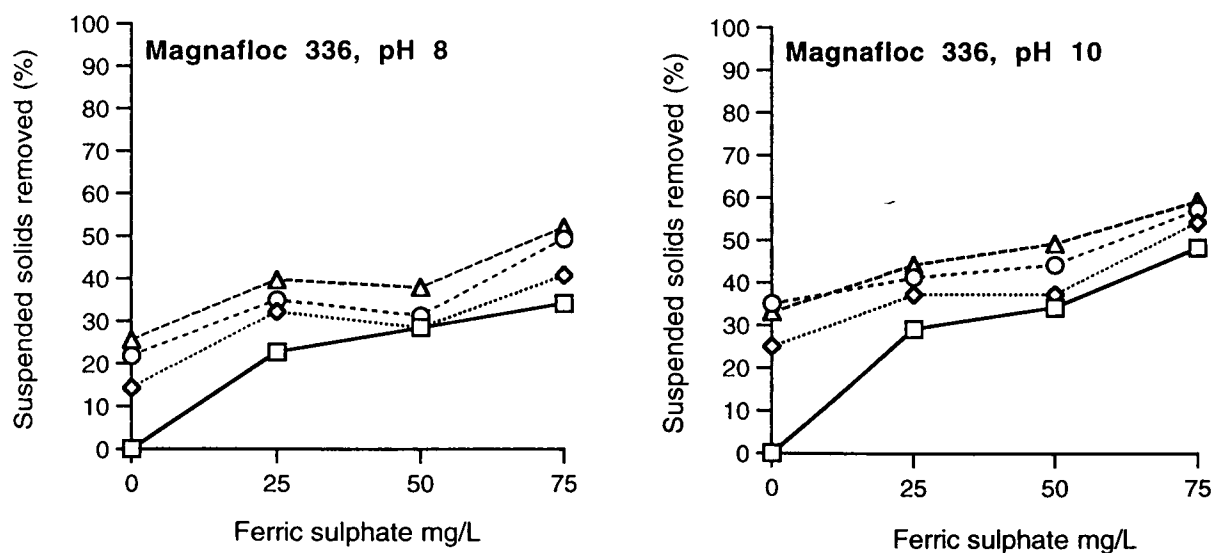**Figure 4.24.** Effect of the anionic polymer **Magnafloc 336** on Turbidity removal, alone or in conjunction with **ferric sulphate**

—□— Magnafloc 336, 0 mg/L	·······◇·······	Magnafloc 336, 2.5 mg/L
----O---- Magnafloc 336, 5.0 mg/L	----△----	Magnafloc 336, 7.5 mg/L
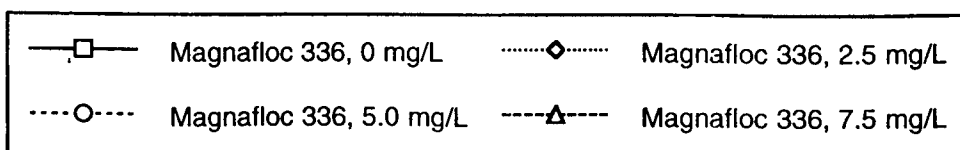
addition of ferric sulphate both Magnafloc 336 and Zetag 92 yielded similar NTU removals at pH 8, although again, Zetag 92 yielded higher NTU removals. At pH10, addition of alum greatly raised the NTU removing capacity of Magnafloc 336 but only slightly increased the capacity of Zetag 92 to remove NTU. This trend at pH 10 was also observed with ferric sulphate.

4.6.4 Total Oil and Grease (TOG) Removal

4.6.4.1 Zetag 92 and Aluminium sulphate (Figure 4.25).

Low levels of TOG were removed with alum as the primary coagulant at both pH 8 and pH 10. At 25 mg/L alum, approximately 16% TOG was removed at pH 8 and about 12% TOG was removed at pH 10. Further increasing the alum concentration had little effect on TOG removals at both pH levels.

The addition of Zetag 92 at pH 8 greatly improved the removal of TOG from the effluent. Figure 4.25 shows that Zetag 92 removed approximately 45% TOG. Increasing the concentration of Zetag 92 and increasing the concentration of alum resulted in very slight increases of TOG reduction, with the highest TOG removal recorded at about 54% (75 mg/L alum, 5.0 mg/L Zetag 92).

Increasing the pH from pH 8 to pH 10 actually decreased the efficiency of TOG removal. Zetag 92 at all doses removed approximately 33% TOG. Increasing alum concentrations had little effect on TOG reduction, with the highest TOG removal recorded at about 40% (75 mg/L alum, all Zetag 92 doses).

4.6.4.2 Zetag 92 and Ferric sulphate (Figure 4.26).

Low levels of TOG were removed with ferric sulphate as the primary coagulant at both pH 8 and pH 10. At 25 mg/L ferric sulphate, approximately 19% TOG was removed at pH 8 and about 15% TOG was removed at pH 10. Further increasing the ferric sulphate concentration had little effect on TOG removals at both pH levels.

The addition of Zetag 92 at pH 8 greatly improved the removal of TOG from the effluent. Figure 4.26 shows that Zetag 92 removed approximately 50% TOG. Increasing the concentration of Zetag 92 and increasing the concentration of ferric sulphate had some effect on TOG reduction, with the highest TOG removal recorded at about 64% (75 mg/L ferric sulphate, 5.0 mg/L Zetag 92).
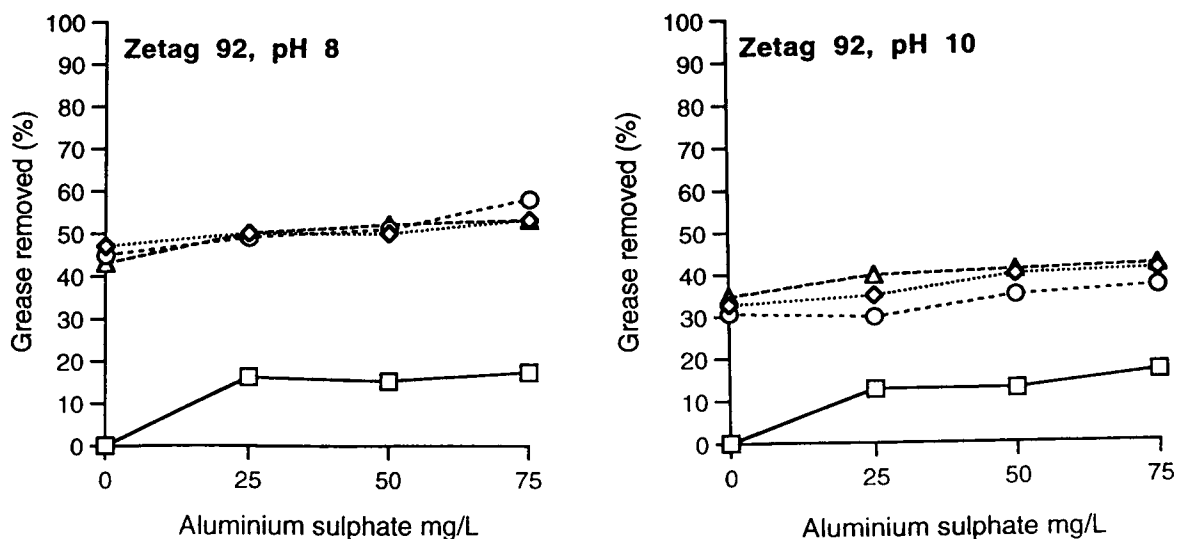
Figure 4.25. Effect of the cationic polymer **Zetag 92** on Total Oil and Grease removal, alone or in conjunction with **aluminium sulphate**
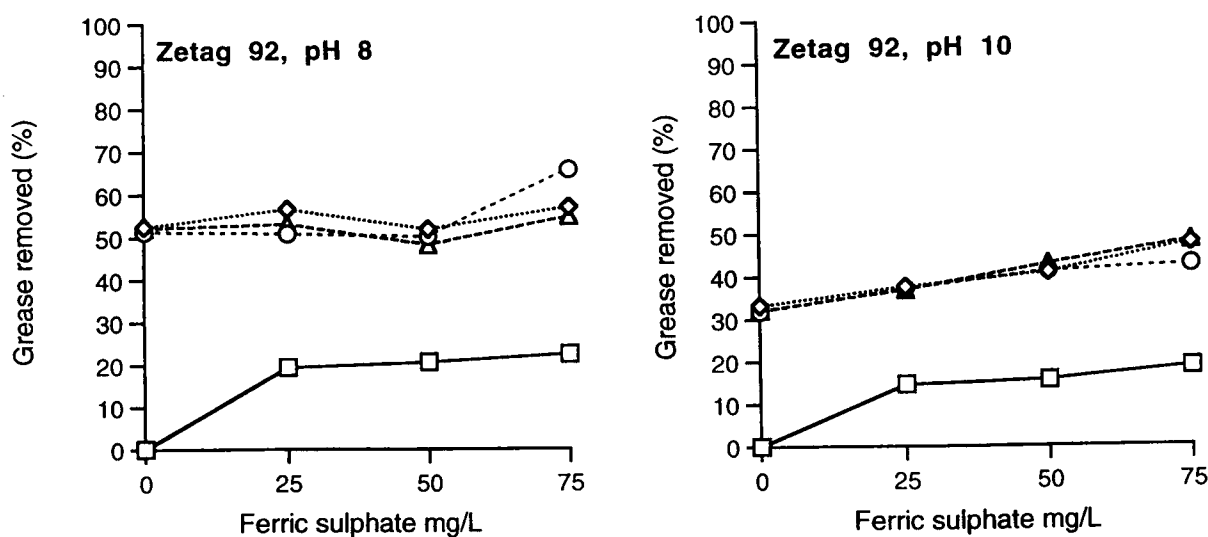
Figure 4.26. Effect of the cationic polymer **Zetag 92** on Total Oil and Grease removal, alone or in conjunction with **ferric sulphate**

——□—— Zetag 92, 0 mg/L	⋯⋯◇⋯⋯	Zetag 92, 2.5 mg/L
----O---- Zetag 92, 5.0 mg/L	----△----	Zetag 92, 7.5 mg/L
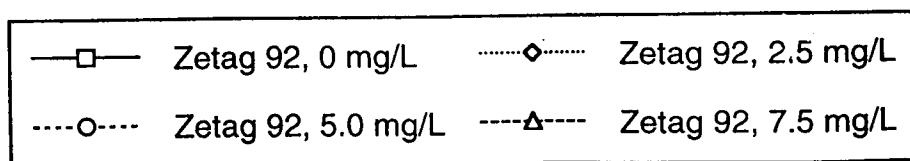
Increasing the pH from pH 8 to pH 10 actually decreased the efficiency of TOG removal. Zetag 92 at all doses removed approximately 32% TOG. Increasing ferric sulphate concentrations had a slight effect on TOG reduction, with the highest TOG removal recorded at about 40%-46% (75 mg/L ferric sulphate, all Zetag 92 doses).

Generally, ferric sulphate was able to remove slightly higher levels of TOG than alum, across both pH levels and at all Zetag 92 doses.

4.6.4.3 Magnafloc 336 and Aluminium sulphate (Figure 4.27).

Low levels of TOG were removed with alum as the primary coagulant at both pH 8 and pH 10. At 75 mg/L alum, approximately 30% TOG was removed at pH 8 and about 15% TOG was removed at pH 10. The efficiency of TOG removal actually decreased significantly with an increase in pH from pH 8 to pH 10.

The addition of Magnafloc 336 at pH 8 greatly improved the removal of TOG from the effluent. Figure 4.27 shows that Magnafloc 336 removed approximately 48% TOG. Increasing the concentration of Magnafloc 336 and increasing the concentration of alum resulted in virtually no effect on TOG reduction, with the highest TOG removal recorded at about 51% (75 mg/L alum, 5.0 mg/L Magnafloc 336).

Increasing the pH from pH 8 to pH 10 actually decreased the efficiency of TOG removal. Magnafloc 336 at all doses removed approximately 45% TOG. Increasing alum concentrations had little effect on TOG reduction, with the highest TOG removal recorded at about 43% (75 mg/L alum, 5.0 mg/L Magnafloc 336).

4.6.4.4 Magnafloc 336 and Ferric sulphate (Figure 4.28).

Low levels of TOG were removed with ferric sulphate as the primary coagulant at both pH 8 and pH 10. At 25 mg/L ferric sulphate, approximately 19% TOG was removed at pH 8 and about 15% TOG was removed at pH 10. Further increasing the ferric sulphate concentration had little effect on TOG removals at both pH levels.
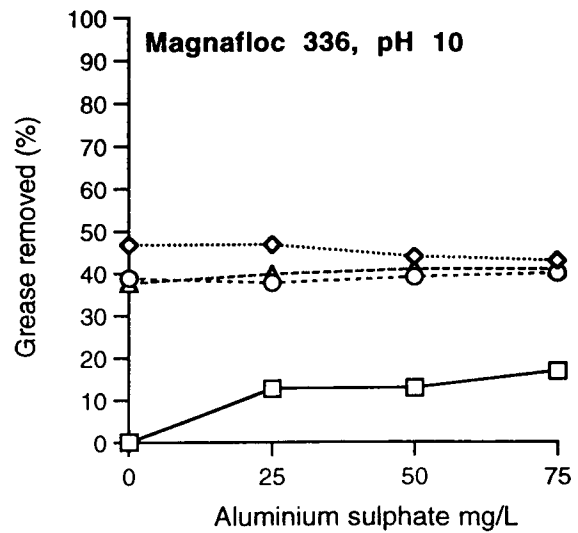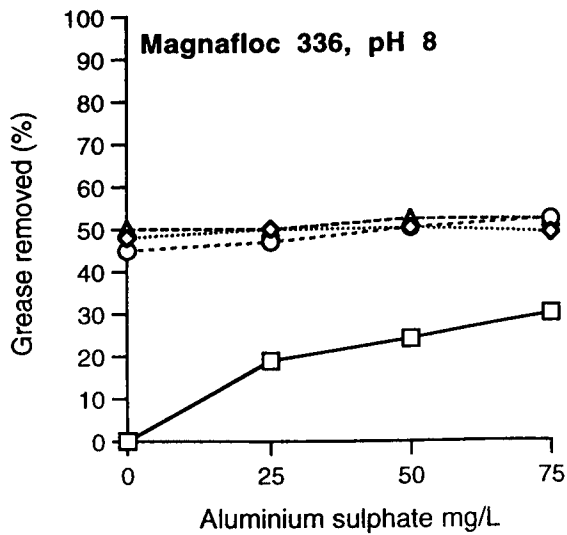
Figure 4.27. Effect of the anionic polymer **Magnafloc 336** on Total Oil and Grease removal, alone or in conjunction with **aluminium sulphate**
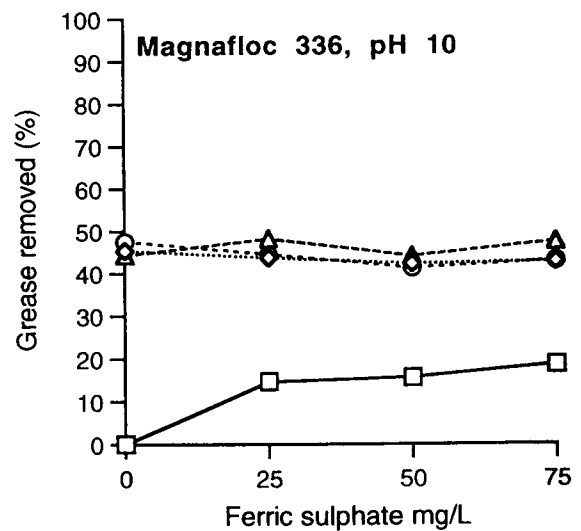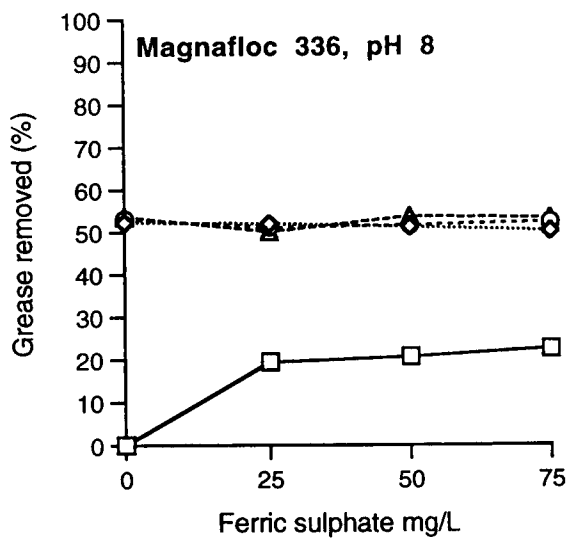
Figure 4.28. Effect of the anionic polymer **Magnafloc 336** on Total Oil and Grease removal, alone or in conjunction with **ferric sulphate**
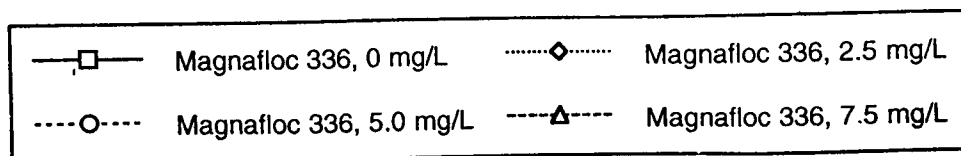
The addition of Magnafloc 336 at pH 8 greatly improved the removal of TOG from the effluent. Figure 4.28 shows that Magnafloc 336 removed approximately 52% TOG. Increasing the concentration of Magnafloc 336 and increasing the concentration of ferric sulphate had no effect on TOG reduction, with the highest TOG removal recorded at about 52% (75 mg/L ferric sulphate, 5.0 mg/L Magnafloc 336).

Increasing the pH from pH 8 to pH 10 actually decreased the efficiency of TOG removal for Magnafloc 336 slightly. Magnafloc 336 at all doses removed approximately 45% TOG. Increasing ferric sulphate concentrations had no effect on TOG reduction, with the highest TOG removal recorded at about 47% (75 mg/L ferric sulphate, all Magnafloc 336 doses).

Generally, ferric sulphate was able to remove slightly higher levels of TOG than alum, across both pH levels and at all Magnafloc 336 doses.

At pH 8 both Zetag 92 and Magnafloc 336 behaved almost identically, both as primary coagulants and as coagualnt aids. At pH 10 Magnafloc 336 removed slightly higher levels of TOG than Zetag 92 when both polymers were used as the primary coagulants. The addition of alum and ferric sulphate resulted in very similar removals for both polymers at pH 10.

4.6.5 Capillary Suction Time (CST)

4.6.5.1 Zetag 92 and Aluminium sulphate (Figure 4.29).

CST was reduced with increasing alum concentrations at pH 8, from about 21 CST at 0 mg/L alum to about 10 CST with 75 mg/L alum. At pH 10, similar results were obtained but at 75 mg/L CST increased slightly to 11 CST (compared to 9 CST at 50 mg/L).

The addition of Zetag 92 at pH 8 resulted in a significant decrease in CST. With no alum, the CST dropped to 13 CST (for all Zetag 92 doses) from 21 CST. Alum addition of 25 mg/L and 50 mg/L dropped the CST further to approximately 6.5 and 5 CST at 5 mg/L and 7.5 mg/L Zetag 92 respectively. Increasing the alum to 75 mg/L actually increased the CST slightly, from about 5 to 7 CST.

The addition of Zetag 92 at pH 10 also led to a drop in the CST to about 13 from 21 CST for all Zetag 92 doses. Increasing the alum dose led to further decreases in CST, although at 75 mg/L alum, Zetag 92 concentrations of 2.5 and 5.0 mg/L gave a higher CST reading, at 7 and 9 CST respectively (compared to 6 CST at 50 mg/L alum).

4.6.5.2 Zetag 92 and Ferric sulphate (Figure 4.30).

CST was reduced with increasing ferric sulphate concentrations at pH 8, from about 21 CST at 0 mg/L ferric sulphate to about 10 CST with 75 mg/L ferric sulphate. At pH 10, similar results were obtained but at 75 mg/L CST dropped further to 8 CST.

The addition of Zetag 92 at pH 8 resulted in a significant decrease in CST. With no ferric sulphate, the CST dropped to about 13 CST (for all Zetag 92 doses) from 21 CST. Ferric sulphate addition at all doses dropped the CST further to approximately 5, 5.5 and 7 CST at 7.5 mg/L, 5.0 mg/L and 2.5 mg/L Zetag 92 respectively.

The addition of Zetag 92 at pH 10 also led to a drop in the CST to about 13 from 21 CST for all Zetag 92 doses. Increasing the ferric sulphate dose led to further decreases in CST, although increasing the ferric sulphate concentrations beyond 50 mg/L, for all Zetag 92 concentrations gave a slightly higher CST reading.

Comparing ferric sulphate to alum, both coagulants were able to reduce the CST substantially. With Zetag 92, ferric sulphate was able to reduce the CST further at both pH levels. At pH 8 ferric sulphate could reduce the CST above 50 mg/L whereas alum yielded an increase in CST beyond 50 mg/L. At pH 10 both alum and ferric sulphate yielded higher CST results when dosed beyond 50 mg/L although alum resulted in higher CST levels.

4.6.5.3 Magnafloc 336 and Aluminium sulphate (Figure 4.31).

CST was reduced with increasing alum concentrations at pH 8, from about 21 CST at 0 mg/L alum to about 10 CST with 75 mg/L alum. At pH 10, similar results were obtained but at 75 mg/L CST increased slightly to 11 CST (compared to 9 CST at 50 mg/L).
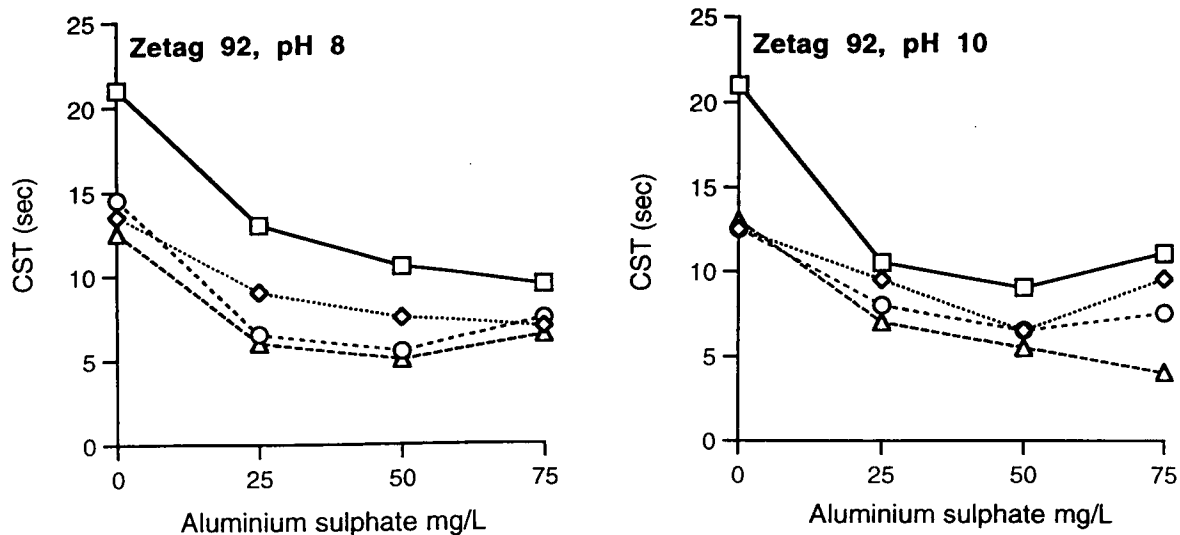
Figure 4.29. Effect of the cationic polymer **Zetag 92** on CST (seconds) of sludge, alone or in conjunction with **aluminium sulphate**
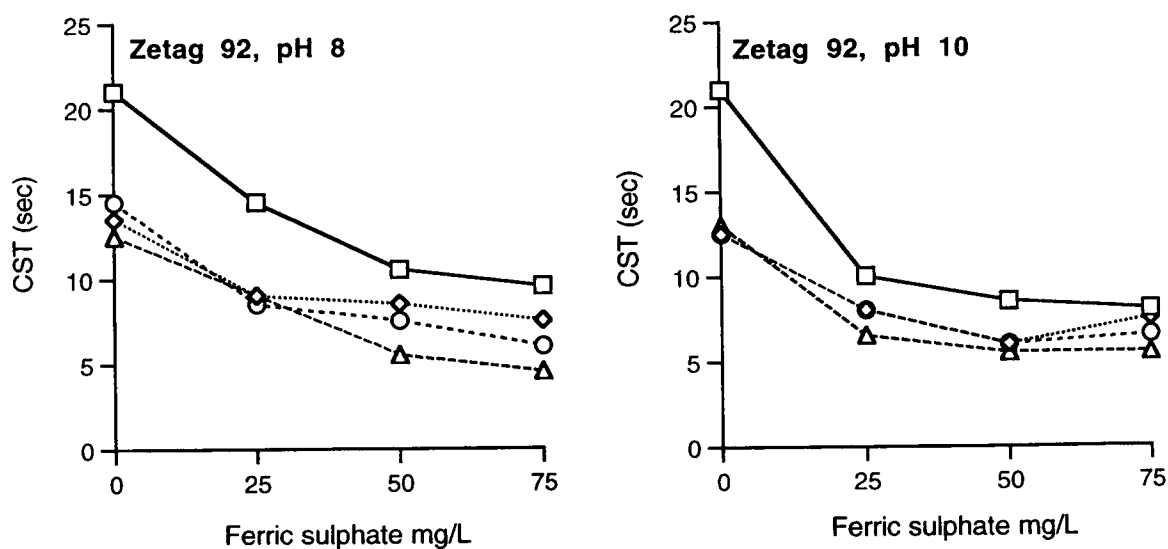
Figure 4.30. Effect of the cationic polymer **Zetag 92** on CST (seconds) of sludge, alone or in conjunction with **ferric sulphate**

—□— Zetag 92, 0 mg/L	········◇········ Zetag 92, 2.5 mg/L
----○---- Zetag 92, 5.0 mg/L	----△---- Zetag 92, 7.5 mg/L
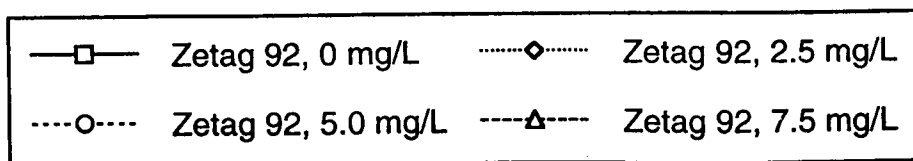
The addition of Magnafloc 336 at pH 8 resulted in a significant decrease in CST. With no alum, the CST dropped to approximately 13 CST (for all Magnafloc 336 doses) from 21 CST. Alum addition dropped the CST further to approximately 7 CST for all Magnafloc 336 doses.

The addition of Magnafloc 336 at pH 10 also led to a drop in the CST to about 13 from 21 CST for Magnafloc 336 doses of 5.0 mg/L and 7.5 mg/L. A CST of 10 was acheived with 2.5 mg/L. Increasing the alum dose led to further decreases in CST, although at 50 mg/L alum and higher, the CST increased slightly for all Magnafloc 336 concentrations.

4.6.5.4 Magnafloc 336 and Ferric sulphate (Figure 4.32).

CST was reduced with increasing ferric sulphate concentrations at pH 8, from about 21 CST at 0 mg/L ferric sulphate to about 10 CST with 75 mg/L ferric sulphate. At pH 10, similar results were obtained but at 75 mg/L CST dropped further to about 8 CST.

The addition of Magnafloc 336 at pH 8 resulted in a significant decrease in CST. With no ferric sulphate, the CST dropped to about 13 CST (for all Magnafloc 336 doses) from 21 CST. Ferric sulphate addition dropped the CST further at 25 mg/L but beyond 50 mg/L dosage of ferric sulphate the CST actually increased.

The addition of Magnafloc 336 at pH 10 also led to a drop in the CST to about 13-10 from 21 CST for all Magnafloc 336 doses. Increasing the ferric sulphate dose led to further decreases in CST, although increasing the ferric sulphate concentrations beyond 25 mg/L, for all Magnafloc 336 concentrations gave a slightly higher or static CST reading.

Comparing ferric sulphate to alum, both coagulants were able to reduce the CST substantially, although ferric sulphate was able to reduce the CST further than alum at both pH levels. With Magnafloc 336, ferric sulphate was able to reduce the CST further at both pH levels. At pH 10 both alum and ferric sulphate yielded higher CST results when dosed beyond 50 mg/L although alum resulted in higher CST levels.

Comparing the performance of the two polymers, Zetag 92 in combination with alum produced slightly lower CST levels than Magnafloc 336 with alum, for
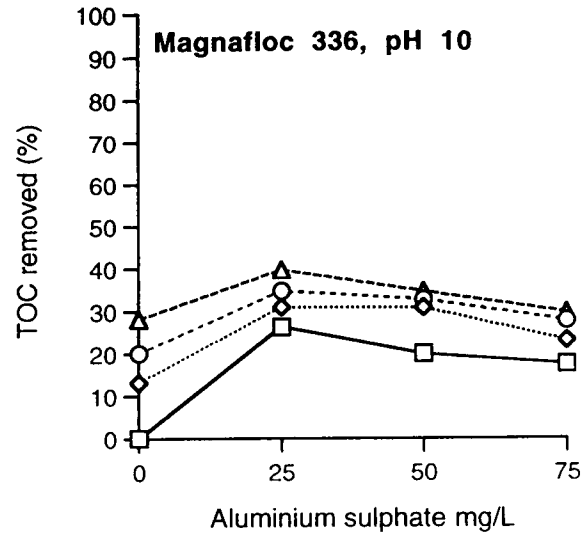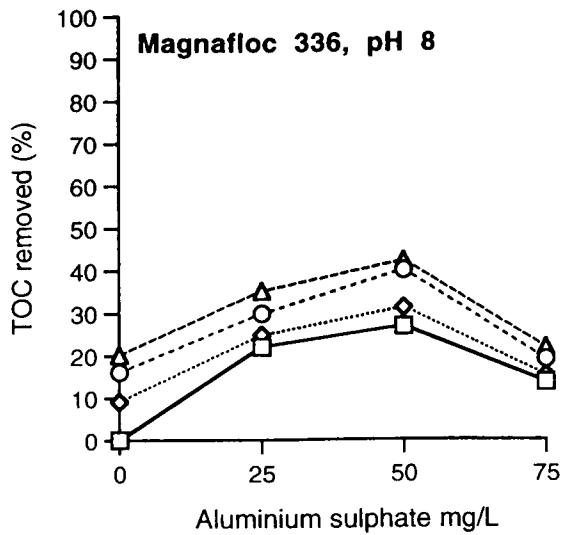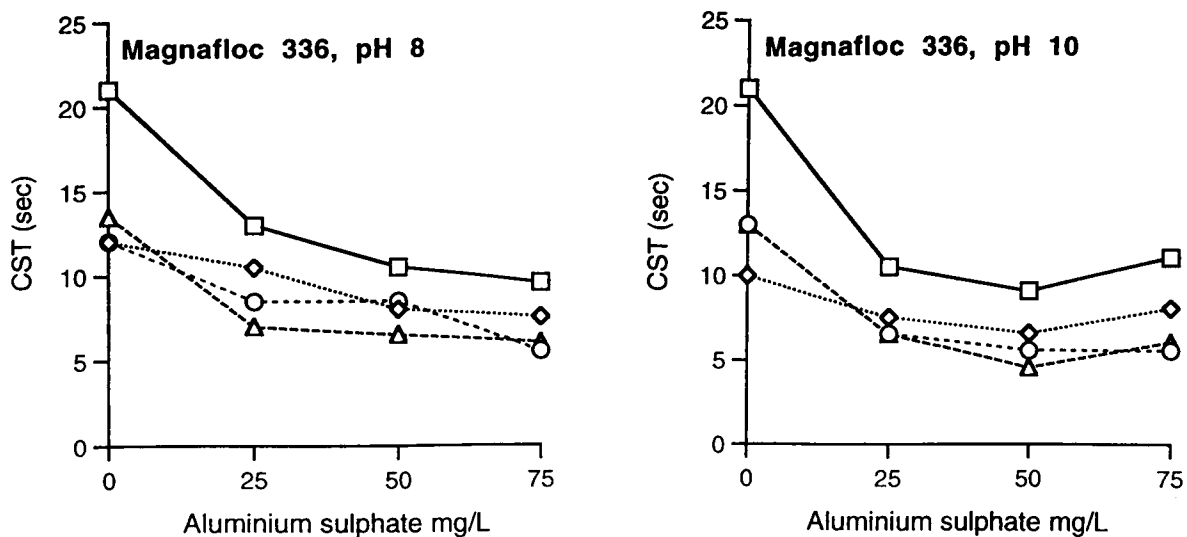
Figure 4.31. Effect of the anionic polymer **Magnafloc 336** on CST (seconds) of sludge, alone or in conjunction with **aluminium sulphate**
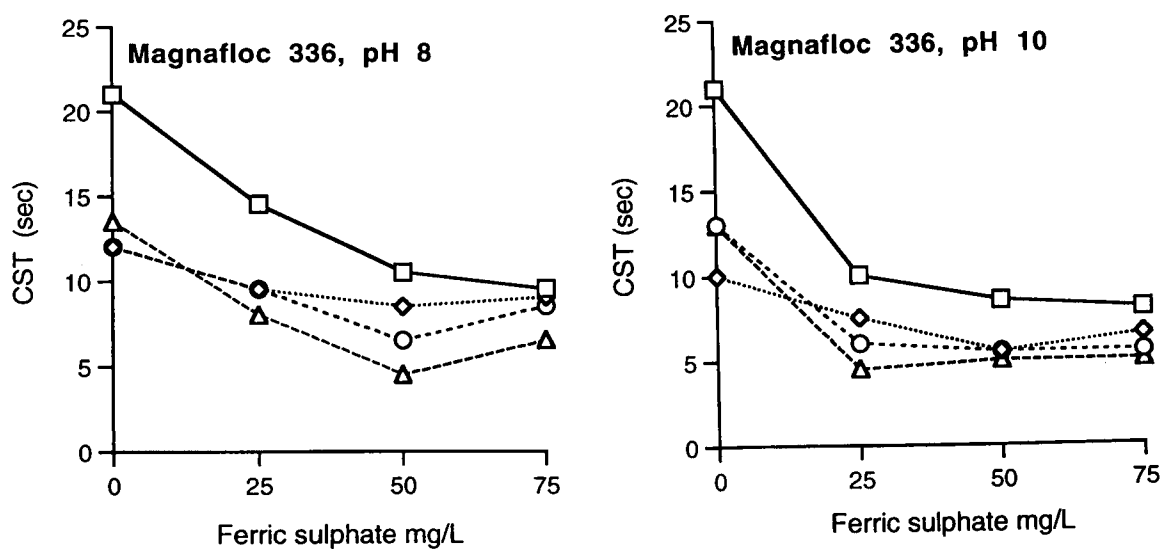
Figure 4.32. Effect of the anionic polymer **Magnafloc 336** on CST (seconds) of sludge, alone or in conjunction with **ferric sulphate**

─□─ Magnafloc 336, 0 mg/L	⋯◇⋯ Magnafloc 336, 2.5 mg/L
⋯O⋯ Magnafloc 336, 5.0 mg/L	─△─ Magnafloc 336, 7.5 mg/L
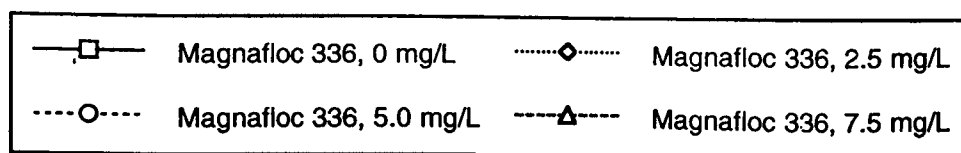
both pH 8 and pH 10. The use of ferric sulphate with Zetag 92, at both pH levles also produced slightly lower CST results.

4.6.6 Floc Formation/Growth (mm)

4.6.6.1 Zetag 92 and Aluminium sulphate (Figure 4.33).

Beyond 50 mg/L alum, floc growth ceased. Floc sizes grew from 0.4 mm to 0.63 mm at 25 mg/L alum and to 0.76 mm at 50 mg/L alum.
These figures and trends were identical for alum at both pH 8 and pH 10.

Zetag 92 addition led to much larger floc formation, with the largest flocs forming (1.6 mm) at Zetag concentrations of 7.5 mg/L and alum concentrations of 75 mg/L at pH 8 and 50-75 mg/L at pH 10.

4.6.6.2 Zetag 92 and Ferric sulphate (Figure 4.34).

Beyond 50 mg/L ferric sulphate, floc growth ceased. Floc sizes grew from 0.4 mm to 0.63 mm at 25 mg/L ferric sulphate and to 0.76 mm at 50 mg/L ferric sulphate. These figures and trends were identical for ferric sulphate at both pH 8 and pH 10.

Zetag 92 addition led to much larger floc formation, with the largest flocs forming (1.6 mm) at all Zetag concentrations and ferric sulphate concentrations of 50-75 mg/L at pH 8. At pH 10 the highest floc sizes attained were about 1.9 mm, and these were achieved with Zetag 92 at 7.5 mg/L and ferric sulphate at 75 mg/L.

Both alum and ferric sulphate produced identical floc sizes at both pH levels, when used as the sole coagulants. The addition of Zetag 92 produced, again, similar results for both alum and ferric sulphate. At pH 10, ferric sulphate produced larger flocs than alum.

4.6.6.3 Magnafloc 336 and Aluminium sulphate (Figure 4.35).

Beyond 50 mg/L alum, floc growth ceased. Floc sizes grew from 0.4 mm to 0.63 mm at 25 mg/L alum and to 0.76 mm at 50 mg/L alum.These figures and trends were identical for alum at both pH 8 and pH 10.
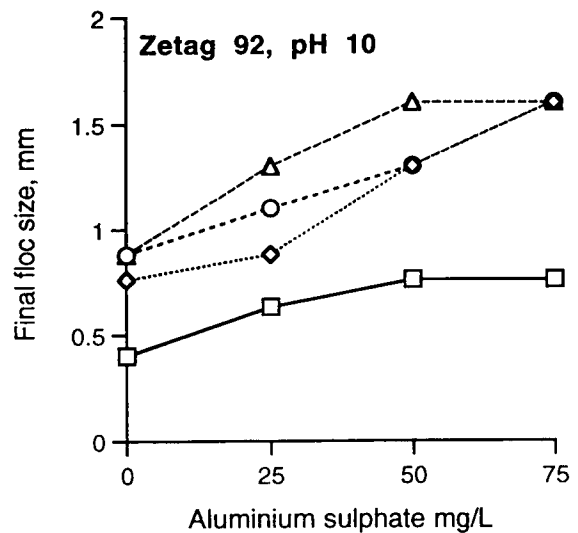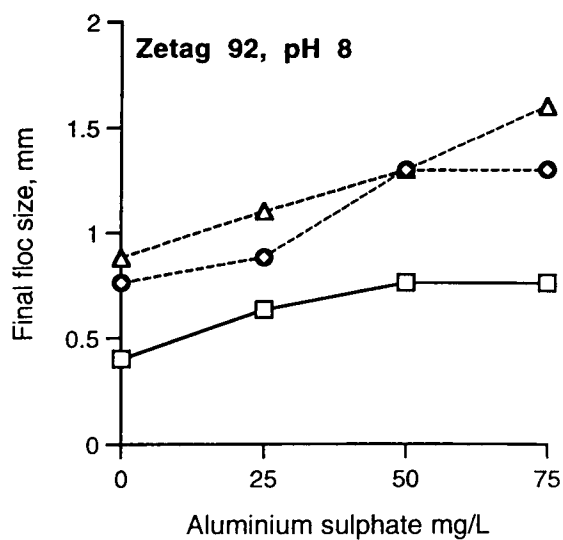
Figure 4.33. Effect of the cationic polymer **Zetag 92** on final floc sizes attained (mm), alone or in conjunction with **aluminium sulphate**
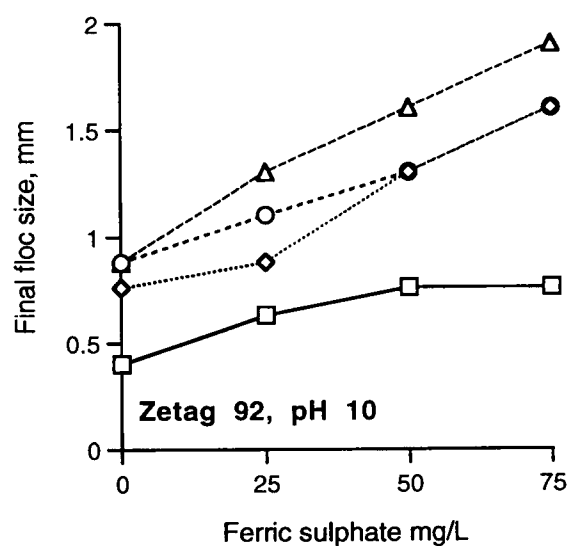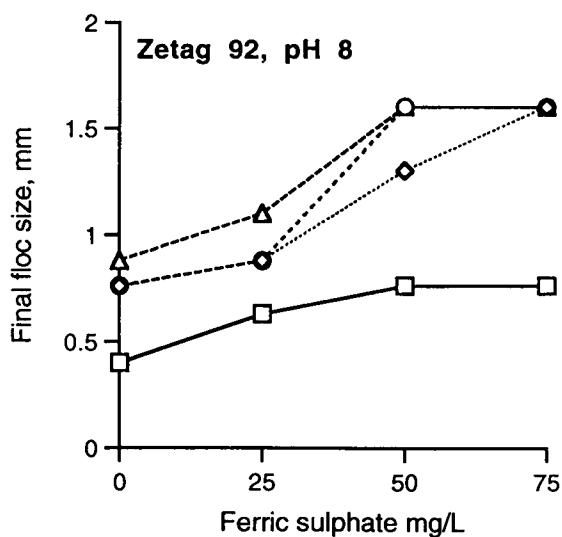
Figure 4.34. Effect of the cationic polymer **Zetag 92** on final floc sizes attained (mm), alone or in conjunction with **ferric sulphate**

| ─□─ Zetag 92, 0 mg/L | ┄◇┄ Zetag 92, 2.5 mg/L |
| ┄O┄ Zetag 92, 5.0 mg/L | ┄Δ┄ Zetag 92, 7.5 mg/L |
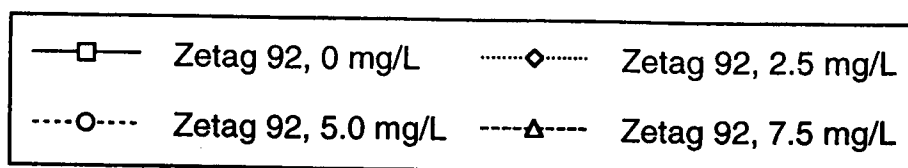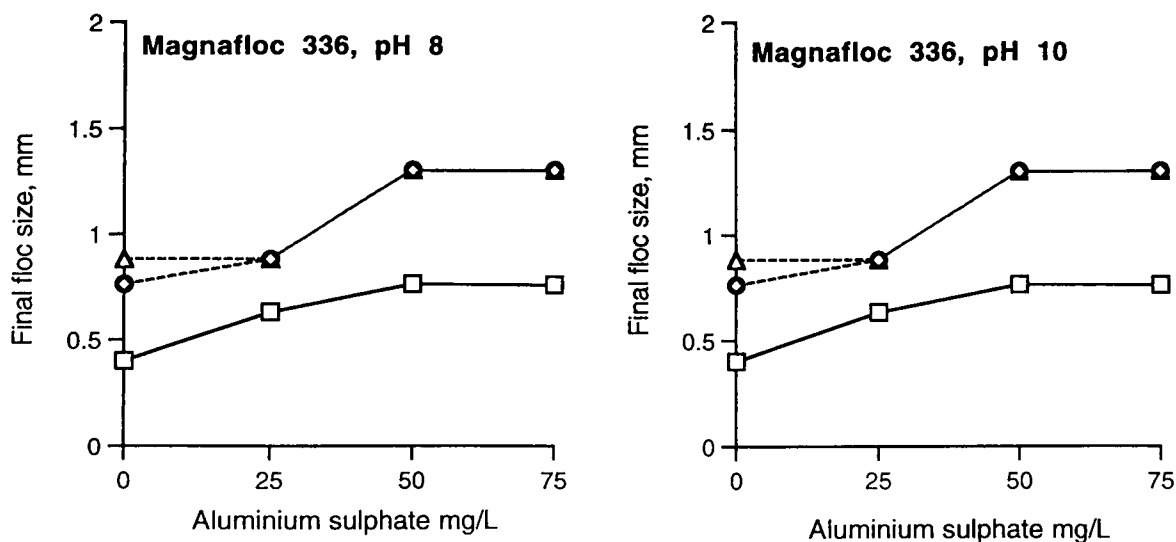
Figure 4.35. Effect of the anionic polymer **Magnafloc 336** on final floc sizes attained (mm), alone or in conjunction with **aluminium sulphate**
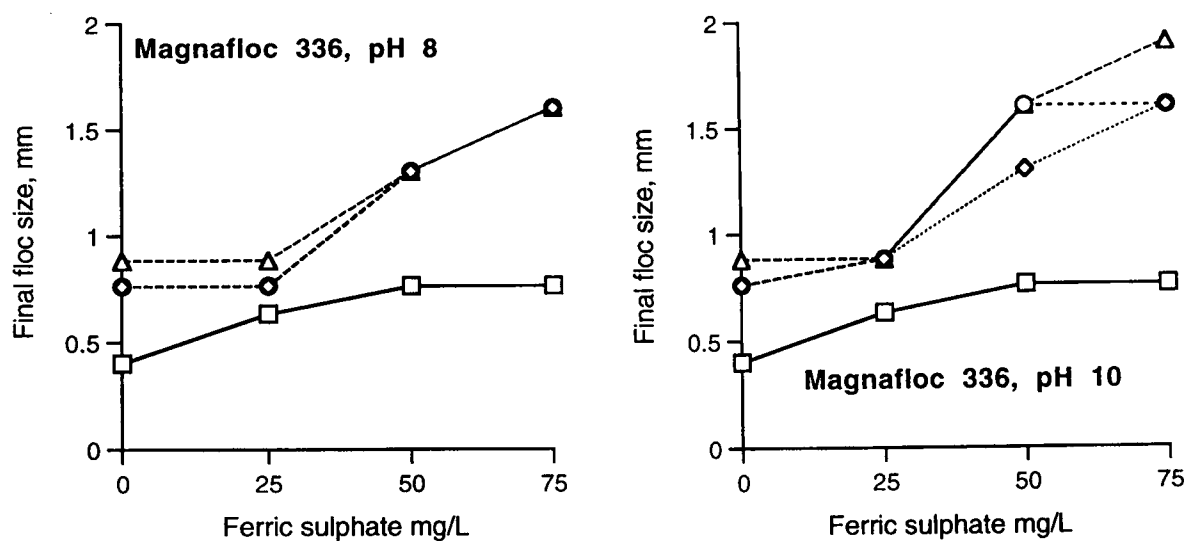
Figure 4.36. Effect of the anionic polymer **Magnafloc 336** on final floc sizes attained (mm), alone or in conjunction with **ferric sulphate**

──□── Magnafloc 336, 0 mg/L	┄┄◇┄┄ Magnafloc 336, 2.5 mg/L	
┄┄O┄┄ Magnafloc 336, 5.0 mg/L	┄┄△┄┄ Magnafloc 336, 7.5 mg/L	
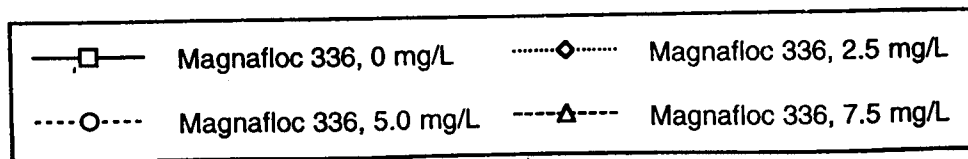
Magnafloc 336 addition led to much larger floc formation, with the largest flocs forming (1.3 mm) at Magnafloc 336 concentrations of 7.5 mg/L and alum concentrations of 50-75 mg/L at pH 8 and 50-75 mg/L at pH 10.

4.6.6.4 Magnafloc 336 and Ferric sulphate (Figure 4.36).

Beyond 50 mg/L ferric sulphate, floc growth ceased. Floc sizes grew from 0.4 mm to 0.63 mm at 25 mg/L ferric sulphate and to 0.76 mm at 50 mg/L ferric sulphate. These figures and trends were identical for ferric sulphate at both pH 8 and pH 10.

Magnafloc 336 addition led to much larger floc formation, with the largest flocs forming (1.6 mm) at all Magnafloc 336 concentrations and ferric sulphate concentrations of 75 mg/L at pH 8. At pH 10 the highest floc sizes attained were about 1.9 mm, and these were achieved with Magnafloc 336 at 7.5 mg/L and ferric sulphate at 75 mg/L.

Both alum and ferric sulphate produced identical floc sizes at both pH levels, when used as the sole coagulants. The addition of Magnafloc 336 produced, again, similar trends for both alum and ferric sulphate, although the latter produced much larger flocs at both pH levels.

Comparing the two polymers Zetag 92 and Magnafloc 336, the former produced larger flocs at pH 10 with either alum or ferric sulphate compared to identical conditions with Magnafloc 336. At pH 8 alum with Zetag 92 and with Magnafloc 336 produced similar floc sizes but ferric sulphate with Zetag 92 produced slightly larger flocs than ferric sulphate with Magnafloc 336.

4.7 CHITOSAN

Chitosan was trialled for its effectiveness as a primary coagulant, as well as a coagulant aid (aluminium sulphate was used as a primary coagulant in this instance). A total of four trials were conducted and the results of these trials can be found in Tables H7.1-H7.10 of the Appendices chapter. Figure 4.37 graphically illustrates the effects of pH changes and alum additions on chitosan. These results are from trial 2.

The initial trial (Table H7.1) evaluated chitosan over a wide dosage range (0-40 mg/L) and a broad pH range (7-11). The most effective removals were

4

Figure 37. Effect of the cationic polyelectrolyte **Chitosan** on Suspended solids removal and final floc sizes attained (mm), alone or in conjunction with **aluminium sulphate**

▦	5 mg/L Chitosan
▨	10 mg/L Chitosan
■	20 mg/L Chitosan
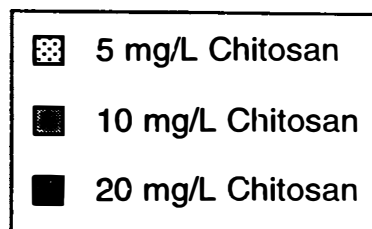
obtained with 10 mg/L chitosan at pH 9, which removed 75.6% of the suspended solids. A similar result (76.5%) was obtained at pH 10 with 40 mg/L. At pH 11, very poor removals and no floc growth were obtained.

Supernatant from samples treated with 40 mg/L chitosan, at all pH levels, did not filter with ease: it took relatively long to filter through the filter paper and upon completion, a thin slimy layer resulted on the surface of the paper.

At pH 7, pH 8 and pH 9 optimum removals were obtained with 10 mg/L chitosan (about 69%, 49% and 76%, respectively). At 20 mg/L, there actually was a slight decrease in efficiency of chitosan to remove SS. Increasing the chitosan concentration to 40 mg/L resulted in only a slight increase in SS removal.

Increasing the chitosan concentration from 5 mg/L to 20 mg/L at pH 7 and pH 11 did not result in any floc growth. Flocs grew slightly to 0.6 mm at 40 mg/L. At pH 8, all chitosan conentrations, flocs attained a final size of 1.3 mm.

At pH 9 floc sizes remained at 0.88 mm for both 5 mg/L and 10 mg/L eventhough at the latter dose about 76% SS was removed, compared to 61% at 5 mg/L. Floc sizes grew at least 4 times larger with an increase in the chitosan conentration to 20 mg/L and 40 mg/L. Eventhough the flocs attained such a large size at these chitosan concentrations, less SS was removed (about 69%, compared to 76% at 10 mg/L).

At pH 10, we begin to see a decrease in the floc growth for the lower chitosan concentrations. This also corresponds to a decrease in the efficiency of chitosan to remove SS at pH 10.

Trial 2 evaluated chitosan as a coagulant aid. It was trialled at a smaller concentration range (5 mg/L, 10 mg/L and 20 mg/L) and a smaller pH range (7-11). Alum was used as the primary coagulant. Tables H7.2-H7.4 and Figure 4.37 contain the data for trial 2.

Compared to trial 1 chitosan in trial 2 was not as effective at removing SS from the effluent. Chitosan at 10 mg/L, pH 9 appears to be the optimum dosage and pH for maximum SS removal when chitosan is the primary coagulant. At pH 7 and pH 9, when the concentration of chitosan was doubled, the removal efficiency actually decreased. At pH 8 a very small increase in efficiency was

observed and at pH 10 there was also an increase in the efficiency of SS removal. At pH 8 there was very little increase in SS removal from 5 mg/L to 20 mg/L.

At pH 7 there was no increase in floc sizes despite an increase in the removal efficiencies of chitosan when the concentration was increased from 5 mg/L to 10 mg/L. At pH 8, there was an increase in floc size from 1.3 mm at 5 mg/L to 1.9 mm at 10 mg/L and 20 mg/L. At pH 9 the floc sizes increased from 0.88 mm at 5 mg/L and 10 mg/L to 3.8 mm at 20 mg/L. At pH 10, floc growth begins to deteriorate relative to pH 8 and pH 9.

Table H7.3 contains the results of alum at 5 mg/L, trial 2. Excellent removals were obtained in all tests, compared to the use of chitosan as a primary coagulant, trial 2 (Table H7.2). Optimum results were obtained at pH 9 with about 70% SS removals, 76% SS removals and 81% SS removals at 5 mg/L, 10 mg/L and 20 mg/L chitosan. Removal efficiency increased with increases in chitosan concentration. All pH levels showed large final floc sizes except pH 7 which seems to have decreased in size with an increase in chitosan concentration from 10 mg/L to 20 mg/L (0.88 mm to 0.6mm respectively).

Table H7.4 contains the results of alum at 10 mg/L, trial 2. Excellent removals were obtained in all tests, compared to the use of chitosan as a primary coagulant, trial 2 (Table H7.2). Doubling the level of alum to 10 mg/L did not impact significantly on the SS removal efficiency of chitosan, when alum was used at 5 mg/L. SS removals only increased slightly with an increase in alum. Final floc sizes were larger in the system using alum at 10 mg/L.

Trial 3 also evaluated chitosan as a coagulant aid. It was trialled at a smaller concentration range (5 mg/L, 10 mg/L and 20 mg/L) and a smaller pH range (7-11). Alum was used as the primary coagulant. Tables H7.5-H7.7 contain the data for trial 3. Trial 3 was a duplicate of trial 2.

Compared to trial 1, chitosan in trial 3 was very effective at removing SS from the effluent, displaying higher SS removal efficiencies than trial 1 and trial 2. Chitosan at 10 mg/L, pH 9, appears to be the optimum dosage and pH for maximum SS removal when chitosan is the primary coagulant, removing about 79% SS. At pH 8 and pH 9, when the concentration of chitosan was doubled from 5 mg/L to 10 mg/L, the removal efficiency increased. Further increasing the chitosan to 20 mg/L actually led to slight decreases in SS

removal efficiencies. At pH 7 an increase in efficiency was observed when the concentration of chitosan was doubled from 5 mg/L to 10 mg/L. Further increasing the chitosan to 20 mg/L only led to slight increases in SS removal efficiencies.

At pH 7 there was very little increase in floc sizes despite an increase in the removal efficiencies of chitosan when the concentration was increased from 5 mg/L to 10 mg/L. At pH 8, there was an increase in floc size from 0.6 mm at 5 mg/L to 0.9 mm at 10 mg/L and 2.6 mm at 20 mg/L. At pH 9 the floc sizes increased from 1.3 mm at 5 mg/L to 2.6 mm at 20 mg/L. At pH 10, floc growth begins to deteriorate relative to pH 8.

Table H7.6 contains the results of alum at 5 mg/L, trial 3. Excellent removals were obtained in all tests, compared to the use of chitosan as a primary coagulant, trial 3 (Table H7.5). Optimum results were obtained at pH 9 with about 80% SS removals, and 81% SS removals at 10 mg/L and 20 mg/L chitosan. Removal efficiency generally increased with increases in chitosan concentration from 5 mg/L to 10 mg/L. Further increasing the chitosan to 20 mg/L did not lead to and significant change in SS removals. All pH levels showed large final floc sizes except pH 7 which seems to have increased slightly.

Trial 3 with 5 mg/L alum yielded higher SS removals than did trial 2 with 5 mg/L alum. Trial 3 also produced larger final floc sizes.

Table H7.7 contains the results of alum at 10 mg/L, trial 3. Excellent removals were obtained in all tests, compared to the use of chitosan as a primary coagulant, trial 3 (Table H7.5). Doubling the level of alum to 10 mg/L did not impact significantly on the SS removal efficiency of chitosan, when alum was used at 5 mg/L. SS removals only increased slightly with an increase in alum. Final floc sizes were slightly larger in the system using alum at 10 mg/L.

Trial 3 with 10 mg/L alum yielded very similar SS removals and very similar final floc sizes to trial 2 with 10 mg/L alum.

Trial 4 was the final trial which evaluated chitosan as a primary coagulant and in combination with alum (Tables H7.8-H7.10). Compared to trial 1, chitosan in trial 4 was also very effective at removing SS from the effluent, displaying similar SS removal efficiencies to trial 1, 2 and 3. Chitosan at 10 mg/L, pH 9,

appears to be the optimum dosage and pH for maximum SS removal when chitosan is the primary coagulant, removing about 74% SS. At pH 8 and pH 9, when the concentration of chitosan was doubled from 5 mg/L to 10 mg/L, the removal efficiency increased. Further increasing the chitosan to 20 mg/L actually led to slight decreases in SS removal efficiencies. At pH 7 an increase in efficiency was observed when the concentration of chitosan was doubled from 5 mg/L to 10 mg/L. Further increasing the chitosan to 20 mg/L only led to slight decreases in SS removal efficiencies.

At pH 7 there was very little increase in floc sizes despite an increase in the removal efficiencies of chitosan when the concentration was increased from 5 mg/L to 10 mg/L. At pH 8, there was an increase in floc size from 1.3 mm at 5 mg/L and 10 mg/L to 1.9 mm at 20 mg/L. At pH 9 the floc sizes increased from 1.3 mm at 5 mg/L to 3.8 mm at 20 mg/L. At pH 10, floc growth begins to deteriorate relative to pH 8 and pH 9.

Table H7.9 contains the results of alum at 5 mg/L, trial 4. Excellent removals were obtained in all tests, compared to the use of chitosan as a primary coagulant, trial 4 (Table H7.8). Optimum results were obtained at pH 9 with about 77% SS removals, and 79% SS removals at 10 mg/L and 20 mg/L chitosan. Removal efficiency generally increased with increases in chitosan concentration from 5 mg/L to 10 mg/L. Further increasing the chitosan to 20 mg/L did not lead to and significant change in SS removals for pH 7, 8 and 9. All pH levels showed large final floc sizes except pH 7 which seems to have increased slightly and decreased again at 20 mg/L.

Trial 4 with 5 mg/L alum yielded similar SS removals to trial 3 with 5 mg/L and trial 2 with 5 mg/L alum. Trial 4 also produced similar final floc sizes.

Table H7.10 contains the results of alum at 10 mg/L, trial 4. Excellent removals were obtained in all tests, compared to the use of chitosan as a primary coagulant, trial 4 (Table H7.8). Doubling the level of alum to 10 mg/L did not impact significantly on the SS removal efficiency of chitosan, when alum was used at 5 mg/L. SS removals only increased slightly with an increase in alum. Final floc sizes were slightly larger in the system using alum at 10 mg/L.

Trial 4 with 10 mg/L alum yielded very similar SS removals to the other equivalent trials and larger final floc sizes to the other equivalent trials.

CHAPTER 5. <u>DISCUSSION</u>

<u>5.1 EFFLUENT CHARACTERISATION</u>

According to the Office of Technology Transfer (1975), grab samples tend to be of little use, as the wastewater composition from food industries varies widely throughout the production day. The results obtained from the effluent profiles in this study indicate that generally, the wastewater composition from this factory did not vary significantly on a daily or even monthly basis.

Table 4.1 indicates that total solids and suspended solids did not fluctuate widely within each profile, and this is shown by the Relative Standard Deviations (%). With the exception of early December 1990 and late May 1991 (both of which had relatively high standard and relative standard deviations for all the parameters tested), the total averages of all the other profiles did not vary significantly from profile to profile. Other typical characteristics of the effluent, such as BOD, pH and soluble solids can be found in Sfinas and FitzGerald (1993).

Figures 4.1A and 4.1B graphically display two typical profiles. The profile for mid June 1991 is very typical of most of the profiles. The sudden and only sharp increase at 44 hours for the profile of late July 1990 was due to a breakdown in the operation of the process (personal communication, 1989). This breakdown led to the subsequent release of large levels of solids to the waste stream. Table B.8 of Appendix B also shows that at 16 hours for the profile of mid March 1991, there is a sudden sharp increase in the solids loading of the effluent. This was due to slight changes in operational procedures.

Besides mechanical breakdowns, shutdowns, cleaning, changes in operational procedures or even sloppy handling, there are a number of other reasons for the unpredictability of wastewater characteristics. According to Niranjan and Shilton (1994) these include: critical variations in wastewater pH; fluctuations in the hydraulic flow of the wastewater; large variations in the wastewater temperature; presence of unknown or unsuspected constituents; and a wide range of variations in wastewater suspended/total/dissolved solids. It is this last aspect which affects the quality of the effluent in this study.

This is due to the use of a variety of flour types in the starch and gluten manufacturing process.

According to Radley (1976b) flours can be classified as either hard or soft. The former type tends to yield strong elastic dough and therefore strong elastic gluten. The latter flour type produces dough which easily breaks down and crumbles during the dough washing process. This breakdown releases higher levels of suspended and dissolved solids into the waste streams as well as increasing the BOD. It is therefore important to know and control the flour type used, as its behaviour will impact on the process and may well add a significant amount of organic loading to the effluent. This is why it is important to have a flour supply (and ultimately a wheat grain supply) which is physically and chemically constant.

A constant flour supply can be difficult to obtain in Australia because wheat is grown on a wide range of soil types and this, together with varied climatic conditions contributes to a highly variable crop in any one season, and from one season to another. It is for this reason that wheat starch manufacturers use a variety of wheat blends to obtain, as ideally as is possible, their desired characteristics of gluten and starch.

5.1.2. Effluent Storage

Figures 4.2A and 4.2B indicate that storing the effluent at ambient temperatures for at least three days leads to significant changes in the suspended and soluble (and therefore total) solids, as well as the pH. Microbial spoilage of the particulate organic matter led to the increase of dissolved or soluble solids and the subsequent decrease in suspended solids. The production of organic acids is typical of the microbial spoilage of starch (Poock, 1985). The results obtained here agree with this and indicate that storage at ambient temperature led to a significant drop in the pH, within 24 hours (from pH 7.5 at day 0 to 3.4 at day 1). Cold storage of the effluent led to much lower rates of particulate solids breakdown (and soluble solids increase) and only slight drops in the pH.

5.2 PRELIMINARY EVALUATION OF POLYELECTROLYTES

Trials were conducted to asses a number of polyelectrolytes for their ability to coagulate and flocculate suspended solids from the effluent. The four cationic

polymers all proved to be more effective at producing larger flocs and removing higher levels of suspended solids than the five anionic polymers and the one non-ionic polymer.

The trends between the two trials were virtually identical, but the results of trial 2 consistently yielded higher suspended solids removals and larger flocs. The results were not always significantly larger. It should be stressed that the initial supernatant solids of trial 2 were lower (about 200 mg/L) than that of trial 1 (approximately 300 mg/L). These results contradict the views put forward by Mishra (1989) in that an increase in particle concentration increases the efficiency of the bridging mechanism. This is because the increased particle concentration increases the interparticle collision frequencies with the other particles, and when the loops are still in a relatively extended state, they are allowed to bridge between particles. It appears that our results in fact yielded more efficient coagulation with a lower concentration of particles. In other words, a system with less colloids yields slightly better results.

One hypothesis for these variations between the trials may be that less colloids create a more efficient destabilisation mechanism. With a slightly lower concentration of particles, there may be more availability of polymer functional groups to bind to the colloid surfaces. An increase in binding, and therefore bridging, leads to larger floc formation and therefore higher suspended solids removal. Having a higher concentration of colloids on the other hand, may compete for functional groups on the polymer and once the polymers are saturated, there may still be an excess of colloids. Therefore the results in trial 1 may be indicating that for the initial levels of about 300 mg/L supernatant suspended solids, slightly more polymer may be needed to achieve similar removals as in trial 2.

It should also be stressed that even though both trials had effluent characteristics which were quite similar (Appendix C, Table C2.11) there may have been factors such as variations in soluble solids or variations in solute concentration or type. These factors may affect polyelectrolyte performance or the charge of the colloid surface, and hence the interactions of the polymer and particle.

5.2.1 Cationic Polyelectrolytes

In all cases increasing the concentration of the cationic polymer led to increases in floc size and increases in efficiency of suspended solids removal. This was true for all pH levels tested. Increasing the pH from pH 4 to pH 7 led to slight increases in floc size and suspended solids removals. At pH 8 the suspended solids removal increased significantly and increasing the pH to pH 10 led to very high suspended solids removals and larger flocs. At pH 10, 7.5 mg/L yielded similar suspended solids removals with 5 mg/L, so it was felt the optimum concentration of cationic polymer was about 5 mg/L.

Further work would need to be conducted to fine tune the optimum concentration of the cationic polymers. For these trials comparisons were made only with 2.5 mg/L to 5 mg/L and it may well be possible the optimum dose lies closer to 2.5 mg/L. The behaviour of the polyelectrolytes also needs to be investigated at pH 9 as well, as the optimum pH may be closer at this level.

Of the four cationic polymers tested, Zetag 92 yielded the largest floc growth and highest suspended solids removals. Zetag 92 also had the highest molecular weight of the cationic polymers (approximately 2×10^7) and had the highest charge density with a viscosity reading of 4000 cP at 0.8% w/v (This is from the technical data provided by Allied Colloids see Appendix A). According to Eilbeck and Mattock (1987) the higher the charge density on the polymer, then the higher is its viscosity in solution. Since there was no technical information on the actual charge density given, it was felt that the viscosity of the polymer would give us an indication of the charge density, and so a relative comparison could be made between the cationic polymers. Below is a summary obtained from the technical data, Allied Colloids, Appendix A, regarding the molecular weight and the viscosities (at 0.8% w/v, cP):

Polymer	Molecular Weight (approx)	cP
Zetag 92	2×10^7	4000
Zetag 87	1×10^7	4000
Zetag 57	1×10^7	2300
Zetag 53	Ultra High	1300

The results obtained with Zetag 92 agree with those found by Stump and Novak (1979); Lee and Fuller (1985) who found that the higher molecular weight polymers yielded better clarification. This was attributed to the fact that

they have a higher number of functional units per segment and so adsorb faster and cover a larger surface area of colloids or particulates.

Because the charge density influences the configuration of the polymer in solution, the charge density will also affect coagulation efficiency, since the degree of density of the charge will influence the uncoiling and extending of the polymer in solution. Generally the higher the charge, the greater the tendency to unfold in solution and extend out. Therefore, with a high charge density, an extended and unfolded polymer will be able to bind a lot more effectively and over a wider area, on the colloidal particles.

The results come into conflict with Gadiel (1978). According to Gadiel, in acidic pH cationic polymers will extend due to charge repulsion along the chain. In alkaline pH the positively charged functional groups become neutralised and the polymer tends to become randomly coiled. Therefore coagulation would be expected to be more effective at lower pH and relatively worse at higher pH, using cationic polymers. Our results demonstrated that all the cationic polymers tested worked best at the higher pH.

One hypothesis is that while there may have been a neutralisation effect due to the alkaline conditions, the charge density was strong enough to allow the polymers to extend and unfold and attach via the bridging mechanism, at pH 10. At pH 8, there may have been a combined effect of charge neutralisation and bridging.

There is another hypothesis, which is effectively an extension of the hypothesis above, to explain the excellent suspended solids removals at pH 10. At pH 10 the starch granules (both broken and intact), as well as the pentosans and other fibrous matter tend to swell and become more viscous (Radley 1976a). These physical changes to the particulates may enhance the polymer-particle bridging mechanism.

Generally at the lower pH levels of pH 4 and pH 6, the predominant mechanism of coagulation was probably by the bridging mechanism, since both the polymers and the particles have like charge, although at pH 6 the charge neutralisation mechanism may also be taking place since the particles may have a partial or net negative charge. At pH 6, pH 7 and pH 8 the electrostatic patch model may be operative since the negatively charged colloids are interacting with the positively charged polyelectrolytes and at

pH 10 the bridging mechanism may predominate again.

5.2.2 Anionic Polyelectrolytes

Below is a summary obtained from the technical data, Allied Colloids, Appendix A, regarding the molecular weight and the viscosities (at 0.8% w/v, cP) of the five anionic flocculants tested:

Polymer	Molecular Weight (approx)	cP
Magnafloc 155	High	1600
Magnafloc 156	High	2200
Magnafloc 336	High	2000
Magnafloc 919	Ultra High	2100
Magnafloc1011	Very High	1700

All the five anionic polymers tested were able to remove suspended solids slightly more effectively at pH 4, pH 6 and pH 7 compared to the cationic range, same pH. When the pH was increased beyond pH 7 to pH 8 and pH 10 there were insignificant increases, if any, of suspended solids removals and floc growth. Only Magnafloc 336 was able to yield high suspended solids removal and larger flocs compared to the other anionics. In contrast, at pH 8 and pH 10 the cationic polymers displayed significant increases in their capacity to remove suspended solids and form larger flocs.

The results indicate that coagulation of suspended solids with anionic Magnafloc polymers was not molecular weight dependant, as the results of Magnafloc 1011, Magnafloc 919 and Magnafloc 155 were all very similar, despite a large difference in molecular weights. These results differ to those of the Zetag coagulation trials, which showed suspended solids removals increased with an increase in the molecular weight of the polymer.

It is not very clear by which mechanism coagulation and flocculation would have taken place with the anionic polymers. According to both Eckenfelder (1980) and Gregory (1993), the primary mechanism of coagulation is probably due to the negatively charged polymer functional groups which replace the anionic groups on the surfaces of the colloids. This permits hydrogen bonding between the polymer and the particle and then permits the bridging to occur. This is why according to Gregory (1993), anionic polymers are effective for the flocculation of negatively charged particles, despite the like sign of charge.

Increasing the Magnafloc dosage led to an increase in suspended solids removal, with the highest removals consistently obtained at 7.5 mg/L. This was true for all Magnafloc polymers. Also noted was the improved performance of all the anionics with increases in the pH to pH 7. As the pH increased, the charges on the colloidal surfaces probably approached a net negative charge. With an increasing shift toward a negatively charged surface (due to the increasing pH) the efficiency of the anionics to coagulate the particles also increased. It may be that beyond pH 7 (ie, pH 8 and pH 10) the surface of the colloids were saturated with negatively charged groups that polymer addition at these pH levels had little effect.

5.2.3 The Non-ionic Polyelectrolyte, Magnafloc 333.

The highest suspended solids removed, as well as the largest flocs were at a dose of 7.5 mg/L, pH 8, trial 2. Approximately 19% suspended solids were removed and the final floc sizes attained were about 0.63 mm. This polyelectrolyte was the poorest performer of all the polymers evaluated. Generally, increasing the pH and the polymer concentration led to an increase in the suspended solids removal efficiency, as well as floc size. The mechanism of destabilisation for the non-ionic polymers, according to Eckenfelder (1980), is they adsorb and flocculate by hydrogen bonding between the particles and polar functional groups on the polymer. Given the results, the mechanisms by which Magnafloc 333 worked is unclear, but at the lower pH range (4 and 6) the suspended solids removals and floc sizes obtained were very poor.

Unfortunately, the only information to be obtained on this polymer is that it has a "very high molecular weight" (Appendix A) but a relatively low charge density compared to the other polymers in this study, with only a viscosity reading of 750 cP at 0.8% solution. It may be that the low charge density did not allow the polymer to fully extend in solution and therefore effectively bridge the particles. The nature of the functional groups could be that at low pH (and therefore an excess of hydrogen ions) the polymer was ineffective for either extending and/or bridging.

5.3 PRELIMINARY EVALUATION OF ALUMINIUM SULPHATE (ALUM) AND FERRIC SULPHATE AS PRIMARY COAGULANTS ON NORMAL AND MODIFIED EFFLUENT.

Initial trials were conducted evaluating alum and ferric sulphate for their ability to remove suspended solids from the effluent. The effectiveness of these primary coagulants was monitored as a function of concentration of coagulant and pH. Two trials were conducted to evaluate the coagulant efficiency on effluent obtained on different days. Table D3.12 (Appendix D) summarises the characteristics of the two effluent types. The main differences between the two effluent types lie with the turbidity values of both the whole effluent and the supernatant. The whole and supernatant suspended solids were very similar for both trials.

Despite the initial supernatant suspended solids levels being similar for both effluent types, there were slight differences in suspended solids removals between the two trials. In almost all cases the trends between the trials were similar. The differences in suspended solids removals between the two effluent types may have been due to a number of reasons: even though suspended solids levels were similar, the soluble solids levels between the two may have varied significantly; solute concentrations may have varied; and slight changes in operator technique may have caused some difference. Also duplicate analysis was not carried out for the suspended solids and this could have contributed to errors.

Another aspect of these studies with alum and ferric sulphate was to investigate the effects of shock loads in the effluent. The effluent solids were artificially increased and decreased with the levels of settleable solids (see section 3.3.1, 3.3.2). This was done to assess the effects of very large and dramatic changes in the colloidal or total solids concentration on coagulation efficiency.

According to Benefield et al (1982) if the coagulating system has a low concentration of colloids (and therefore low colloidal surface area) there is limited opportunity for the hydrolysis products to come into contact with the colloidal surface and so there is limited destabilisation of the colloids. On the other hand, if the effluent has a high concentration of colloids then it allows for ample contact between hydrolysis products and the colloidal surfaces, and therefore more effective coagulation.

The results obtained with these trials indicate that in fact coagulation of effluent with a reduced level of solids yielded higher suspended solids removals compared to the normal effluent and dramatically higher suspended solids removals compared to effluent with increased solids. Figure 4.5 shows that both the normal effluent and the effluent with reduced solids had the highest suspended solids removed at all pH levels but especially at pH 8. In nearly all doses and pH levels the effluent with reduced solids had slightly higher levels of suspended solids removed compared to the normal effluent when alum was the primary coagulant, and significantly higher removals compared to the normal effluent when ferric sulphate was the primary coagulant.

In all cases the effluent with increased solids had the poorest removal of suspended solids with either coagulant. It appears there was a need for more coagulant. These results are in agreement with Narkis and Rebhun (1981) who coagulated clay from drinking water. They found that adding humic acids to the water required higher flocculant doses and concluded that additional solids loading to a water (or wastewater) system increased flocculant demand.

A hypothesis that could be put forward with these results is that by reducing the levels of colloids the colloids have more exposure to the hydrolysis products and the metal hydroxides formed because there is more coagulant available per colloid. By significantly increasing the level of solids in the system (2 x total settleable solids), the same concentrations of hydrolysis products and metal hydroxides cannot come into sufficient contact with the increased number of particulates as there is now less coagulant available per colloid/particle. So while coagulation is still taking place, there remain a higher level of uncoagulated colloids and so the level of suspended solids remains high.

The modified effluent used in this thesis, with reduced solids may have had a higher colloidal concentration relative to that of the low colloid concentration system proposed by Benefield et al (1982). This may explain why our reduced solids effluent produced very high suspended solids removals and the low colloidal system proposed by Benefield et al (1982) had relatively poor coagulation.

The two basic mechanisms by which coagulation occurs with metal salts are by charge neutralisation and enmeshment coagulation. These two mechanisms are believed to normally work simultaneously (Schwoyer, 1986).

The formation of the metal hydroxide precipitate generates a number of positively charged hydrolysis products. The main role of the precipitated metal hydroxide is to provide a higher particulate or colloidal surface area which enhances destabilisation kinetics. The precipitate forms a type of floc matrix with the colloids and this matrix entraps or enmeshes other colloids (hence the term enmeshment coagulation). During the precipitation of the metal hydroxide, the positively charged hydrolysis products work by the mechanism of charge neutralisation on the negatively charged colloids, thus destabilising the colloids by neutralisation. In this way, there is a combination of neutralisation and enmeshment of neutralised particles into a floc matrix.

According to the AWWA Coagulation Committee (1989), in order for the charge neutralisation mechanism to be effective, the positive hydrolysis products need to come into sufficient contact with the negatively colloids. This can only be effected by efficient particle collisions. Because these metal hydrolysis products are formed in a matter of microseconds, the coagulant needs to be dispersed in the rapid mix phase as quickly as possible so that most of the reactive products will be available to produce charge neutralisation.

The pH of the effluent is crucial to the efficiency of these mechanisms since the pH determines what species of the hydrolysis products will predominate and what charge the colloidal surface may take on (Qureshi and Malmberg, 1985). The results in Tables D3.1 to D3.10 (Appendix D) and in Figure 4.5 reveal for both trials that maximum suspended solids removals were always attained at pH 8, which was the approximate, initial pH. This was always the case for both alum and ferric sulphate, for all three concentrations tested and at all three different effluent types. The advantage of the optimum pH being around pH 8 is that prior to discharge the effluent is normally dosed with lime to attain a final pH of around pH 8, therefore the effluent would not need pH adjustment for coagulation with the metal salts.

At pH 4 relatively low levels of suspended solids removals were attained. Increasing the pH to pH 6 significantly improved the suspended solids removals, and further increasing the pH to pH 7 only led to a slight

improvement in suspended solids removals. It may well be that at the lower pH values, or at the pH values below the isoelectric points of the metal hydroxide precipitate, the hydrolysis products take on a positive charge. Possibly at around pH 4 the colloids take on a predominantly positive charge as well and this probably gives rise to repulsion of the colloid to the hydrolysis products. As a result very poor coagulation takes place because the charge neutralisation mechanism would not be very effective as there would not be many negative ions on the colloidal surface to neutralise the positively charged hydrolysis products. At pH 4, the metal hydroxide precipitate probably never formed as it is very soluble at low pH (aluminium is amphoteric). This results in very little effective coagulation. For pH 4, at 25 mg/L and 75 mg/L alum approximately 5% and 10% suspended solids were removed respectively, for the normal unmodified effluent. At pH 4, at 25 mg/L and 75 mg/L ferric sulphate about 12% and 23% suspended solids were removed respectively, for the normal unmodified effluent. At this low pH, it is unlikely that charge neutralisation or enmeshment would have occurred, yet when the concentration of the coagulants was tripled the suspended solids removal efficiency doubled. According to Montgomery (1985), coagulation may have taken place via double layer compression (see Section 2.6).

Increasing the pH had a marked effect with the removal of the suspended solids when using alum or ferric sulphate. As pH increased the particles acquire a net negative charge on their surface. Their surfaces interact with the positively charged metal hydrolysis products, and this allows for charge neutralisaton. The production of the metal hydroxide precipitate is probably the other mechanism that works, and it is this mechanism that probably initiates floc growth.

At higher pH levels such as at pH 10 (which produced relatively poor suspended solids removals compared to pH 8) the negatively charged metal hydrolysis products predominate and these are ineffective for destabilisation. According to Schowyer (1986) the solubility of $Al(OH)_3$ and $Fe(OH)_3$ increases significantly either above or below the pH of minimum solubility of the respective coagulant. Below pH 6, $Al(OH)_3$ solubilises rapidly. at about pH 6 the soluble aluminium ion concentration is about 3×10^{-3} mg/L. At pH 4 the soluble aluminium ion concentration is approximately 2.8×10^4 mg/L and at pH 10, the soluble aluminium ion concentration is approximately 30 mg/L. This may explain why at pH10, the capacity for aluminium to coagulate suspended solids diminished.

The solubility of $Fe(OH)_3$ is some orders of magnitude less soluble relative to $Al(OH)_3$, over a wider pH range. For instance, at pH 4 and pH 12, the concentration of the ferric ion in solution is about 6×10^{-3} mg/L and at pH 8, it is about 1×10^{-6} mg/L. Ferric sulphate yielded much higher suspended solids removals in nearly all cases, for all pH levels, at the reduced solids effluent and the normal effluent.

5.4 EVALUATION OF PHYSICAL PARAMETERS OF THE JAR TEST

Various parameters such as rapid mix duration (secs), flocculation duration (mins), flocculation intensity (rpm), dosage rates (mL/sec) and dosage sites were evaluated for their impact on the jar test. two trials were conducted, and both yielded very similar trends and very similar results.

5.4.1 Rapid Mix Duration (RM)

The function of the rapid mix phase is important as it aims to achieve complete and thorough mixing of the coagulant in the water to be treated (AWWA Coagulation Committee, 1989). The optimum RM time for alum was found to be at 10 seconds. No trials were done to evaluate the impact of RM between 1 second and 10 seconds, so the actual optimum time for RM with alum may in fact be shorter.

Figure 4.6A demonstrated that no RM yielded very poor suspended solids removal and no floc growth. RM beyond 10 seconds also led to poor suspended solids removals and poor floc growth. Increasing the RM led to a greater deterioration of suspended solids removal and floc growth. These results agree with Bhatia and Cheremisinoff (1979) and with Francois and Van Haute (1984), who found that RM should not be normally increased beyond about 5 seconds, and that beyond any optimum RM time, there is a detrimental effect on flocculation efficiency.

For the mechanisms of alum and ferric sulphate coagulation to be effective, RM need to be optimised. It is essential that the coagulant is uniformly distributed for maximum contact with the particles before the hydrolysing reactions are complete and the metal hydroxides form. This is because the hydrolysis and polymerisation reactions are very rapid.

According to Sylvester and Toure (1978) and Keys and Hogg (1979) increasing the RM time beyond the optimum time leads to floc break up as the mechanisms for primary floc formation are disrupted because the shear forces split and erode the flocs. Our results agree with Sylvester and Toure (1978) and Keys and Hogg (1979) and also support the observations by Janssens (1992) that extending the RM beyond the optimum time leads to floc erosion and therefore floc break up.

By contrast, the 20 seconds RM for Zetag 92 proved to be the optimum time length, Figure 4.8A. RM between 0 and 20 seconds was not evaluated, so potentially, the RM time may be shorter. The results of extending the RM beyond 20 seconds was not as dramatic to the results of alum. It is clear that prolonged shearing did result in some floc break up and therefore a decrease in suspended solids removal but it appears the polymer was able to withstand the shear forces compared to alum coagulation.

According to Janssens (1992) polymers can break apart by the high shear forces via chain breakage of bridges between the particles. When our RM was extended beyond 20 seconds to 120 seconds floc sizes decreased from 1.9 mm to 1.3 mm (which represents a 32% size reduction) and suspended solids removal dropped from about 70% to 61% (which represents a 13% efficiency reduction).

A hypothesis that may explain why prolonged shearing did not greatly diminish coagulation with Zetag 92 could be the polymer itself. Once the polymer is fully extended it may form a type of network or matrix which captures or binds the particulates. If the polymer is strong enough or has a strong polymer backbone, it may withstand the shearing. It should also be noted that our jar test did not have stators so the polymer did not encounter very turbulent mixing.

Kennedy et al (1994) extracted protein from whey using chitosan and found 2 hours was the optimum RM time. Beyond this 2 hour optimum time frame, floc disruption occurred. Unfortunately, no intensity of rapid mix was given to allow us to make relative comparisons to this work.

5.4.2 Flocculation Duration

According to Tchobanoglous and Burton (1991) flocculation is an essential step after destabilisation because it accelerates floc formation and influences the strength, size and density of the flocs. This ultimately determines the level of particles which have been destabilised. For a given velocity gradient, there will be a limiting flocculation time beyond which the flocs will not grow further. When the flocs reach a certain size and exceed the maximum stable diameter for a given velocity gradient, the flocs begin to break up (Ghosh et al, 1985).

Figure 4.6B shows that beyond 10 seconds for alum, floc sizes break down and remain at the same size for up to 60 seconds. In Figure 4.8B, flocs produced with Zetag 92 also remain the same size for up to 60 seconds but beyond this time floc sizes break down. In both cases the breakdown of floc sizes corresponded with a decrease in suspended solids removals.

5.4.3 Dosage Sites

The effects of dosing at different points in the beaker were evaluated with alum only. Dosing directly into the vortex resulted in 39% suspended solids removal and final floc sizes of 1.3 mm. When alum was dosed on the surface of the effluent, between the vortex and the beaker wall, there was approximately 25% suspended solids removal and final floc sizes of 1.3 mm attained.

Our results agree with Hudson (1981) who strongly recommends that the coagulant be added directly to the agitator blade and not the surface, as the hydrolysis products form extremely rapidly and need to be dispersed thoroughly and rapidly to achieve maximum coagulation. Inefficient dispersion leads to over treated and under treated regions and ultimately, coagulation efficiency decreases (Qureshi and Malmberg, 1985; Janssens, 1992). The results obtained indicate that dosing on to a surface did not maximise the distribution of alum and as a result, coagulation efficiency was relatively poor compared to dosing into the vortex.

5.4.4 Flocculation Intensity

The effect of changing the flocculation intensity was also investigated. Increasing the flocculation intensity beyond 30 rpm for alum led to significant

decreases in suspended solids removal and floc growth, Figure 4.7A. When less than 30 rpm was used to flocculate the coagulated solids the removals of suspended solids and floc growth were less than those attained at 30 rpm.

Figure 4.9A shows the effect of changing flocculation intensity with Zetag 92. At both 30 and 40 rpm suspended solids removals and floc sizes were very similar. Increasing the flocculation intensity beyond 40 rpm led to very large decreases in the final floc sizes (approximately 53% reduction in size) but suspended solids were reduced only slightly. Clearly, increasing the flocculation intensity or velocity gradient caused floc breakage. This happens because the local shear stresses exceed the binding forces of the aggregates, causing them to break up (Montgomery, 1985). Our results also indicate that Zetag 92 produced stronger flocs and was able to withstand the increasing shear forces compared to alum. These results agree with Muhle (1993), who pointed out that flocs produced by inorganic coagulants were weaker than those produced by organic polymers.

5.4.5 Dosage Flow Rates

The results for both alum and Zetag 92 coagulation (Figure 4.7B and Figure 4.9B) indicate that increasing dosage rates leads to smaller final floc sizes and a decreased capacity to remove suspended solids. For alum the optimum dosage rate was about 0.5 mL/sec or 1 mL/sec and for Zetag 92 the optimum dosage rate was about 0.5 mL/sec to 2 mL/sec. These results are in agreement with Keys and Hogg (1979) and Fisher and Glatz (1988a; 1988b), who found coagulation could be optimised by a slower, continuous rate of coagulant addition.

When the dosage rate was increased beyond these optimum rates, for both alum and Zetag 92, a decrease in the coagulating efficiency was noted. This may have been due to inadequate dispersion of the coagulating agents, resulting in regions of over treatment and under treatment.

5.5 FURTHER EVALUATIONS OF ALUM AND FERRIC SULPHATE AS PRIMARY COAGULANTS.

Further trials to those in Section 4.3/5.3 were conducted. These trials were only carried out with normal/unmodified effluent. Two effluent types were used, one for alum coagulation and the other for ferric sulphate coagulation. A very wide concentration range of coagulant was used, spanning from 0 mg/L to 250 mg/L, as it was decided to fine tune the optimum dosage levels. The pH levels were from pH 6 to pH 10 and included pH 9. Suspended solids removals were assessed, as was the final floc size attained at 20 minutes. The final pH of each experiment was also determined in order to correlate coagulation efficiency with the pH of the reaction.

Figures 4.10, 4.11 and 4.12 detail the information of suspended solids removal, floc growth and final pH, respectively. Appendix F details the raw and final data of these trials. The effluent used for both effluent had an initial pH reading of 7.95 (trial 1) and 8.21 (trial 2). The effluent used for alum coagulation had an initial supernatant suspended solids concentration of about 680 mg/L. When the pH was decreased to pH 7 and pH 6, the suspended solids concentration had an apparent increase to about 730 mg/L and 710 mg/L respectively. Increasing the pH to pH 9 and pH 10 led to an apparent decrease in the suspended solids concentration to about 650 mg/L for both pH levels.

Using the initial suspended solids readings at pH 8 as the starting/initial concentration, there is an apparent increase of 7% and 4% suspended solids at pH 6 and pH 7 respectively and an apparent decrease in suspended solids of 4% and 5% at pH 9 and pH 10 respectively. While these variations are relatively small, they may be due to the behaviour of the starch granules and proteins in the effluent. According to Simmonds (1989), at high pH the starch granules tend to swell, leak and rupture. Therefore the large particulate granules probably begin to fragment into smaller particles or begin to solubulise. Some proteinaceous material also begins to solubilise at high pH. This may give rise to the apparent decrease in suspended solids. When the pH is decreased, some soluble proteins and soluble starches precipitate (Curtis, 1983) and this may give the effect of an apparent increase in the suspended solids concentration. The effluent used for ferric sulphate coagulation showed insignificant changes with variations in pH.

The results obtained in these trials revealed very similar trends and results with the initial trials (section 4.3). At 25 mg/L, 50 mg/L and 75 mg/L alum the suspended solids removals were about 37%, 47% and 53% respectively, pH 8, in section 4.3 and were 35,%, 47% and 54% in this trial, pH 8. Similarly for ferric sulphate, 25 mg/L, 50 mg/L and 75 mg/L the suspended solids removals were about 39%, 50% and 61% respectively, pH 8, in section 4.3 and were 46%, 60% and 68% respectively in this trial, pH 8. These two trials were conducted about seven months apart and clearly show that using the same dosage results in similar suspended solids removals.

The results obtained for suspended solids removals for both alum and ferric sulphate indicate that about 75 mg/L of both these coagulants is the optimum concentration, at all pH levels tested. The optimum pH was found to be pH 8. At this pH, with 75 mg/L alum and 75 mg/L ferric sulphate, approximately 54% and 68% suspended solids were removed respectively. The final floc sizes attained were 1.6 mm and 2.3 mm respectively and these were the largest flocs produced. The final pH attained was pH 6.6 for alum and pH 6.8 for ferric sulphate, both at 75 mg/L and both at a starting pH of pH 8.

Generally, Figures 4.10 and 4.11 show that increasing the coagulant from 0 mg/L to 75 mg/L led to steady floc growth and suspended solids removals. At pH 8 and pH 9, ferric sulphate yielded the highest suspended solids removals, and at pH 8 yielded the largest flocs. The effects of pH on the mechanisms of charge neutralisation and enmeshment have already been covered in section 5.3. The superior performance of ferric sulphate over alum in these trials is supported by other workers as well. Work conducted by Karim and Sistrunk (1985) evaluated ferric sulphate and alum for their potential to treat effluent generated from potato processing factories. They found that ferric sulphate consistently yielded higher suspended solids removals and COD removals.

Work conducted by Rusten and Sandberg (1991) in the treatment of yeast wastewater found ferric chloride gave higher COD removal efficiencies than did alum. Even though they used ferric chloride they demonstrated that the ferric ion coagulant was superior to the aluminium ion coagulant.

Dosing beyond 75 mg/L of either coagulant led to a deterioration in both floc growth and suspended solids removals. Increasing the dosage to 250 mg/L

from 75 mg/L continued to decrease the floc sizes and decreased the suspended solids removal efficiency. These problems were encountered with both coagulants at all the pH levels tested. This effect is largely due to restabilisation, in which a system is overdosed with the adsorbable species.

When excessive levels of coagulant are added, excessive levels of the positively charged hydrolysis products are formed, which neutralise the negatively charged colloids. These colloids then take on a positively charged surface due to the excessive levels of the positively charged hydrolysis products. As a result, restabilisation takes place. As the coagulant doses were increased beyond the optimum dose the effects of restabilisation became pronounced. These restabilisation problems were also encountered by Alaerts et al (1982) who found that in treating sewage wastewater 100 mg/L alum yielded very low phosphorus levels but doubling the dose to 200 mg/L led to higher final phosphorus levels. This was due to restabilisation caused by the excess positive charge liberated by the excess alum.

The final pH was recorded as an indication at which pH the coagulation reactions took place, so a correlation of coagulation performance could be done. The alkalinity of the effluent is therefore important to know and control. Even though our work did not look at the alkalinity of the effluent, it was felt that the final pH would indicate the buffering capacity of the effluent.

According to Barnes et al (1981) when the metal salt is added to the water, the hydrolysis reaction produces the metal hydroxide precipitate as well as sulphuric acid. If the buffering capacity of the effluent is low the pH may drop significantly, depending on the concentration of the coagulant. If the drop in pH falls outside the range of minimum solubility of the metal hydroxide precipitate, the formation of the precipitate is poor and coagulation efficiency decreases.

If the pH drops, or if the starting pH is low, the positively charged hydrolysis products may predominantly coagulate by charge neutralisation, which helps to explain why at pH 4 and pH 6 (section 4.3) and pH 6 of this trial coagulation still took place.

The control of pH is also very important for the formation of the polymeric metal hydroxide species, as their formation leads to interparticle bridging which enhances floc formation (section 2.5.1). At the lower to middle end of the pH

scale, these polymeric species are positively charged and can bind to the negatively charged colloids and span across an d bind onto other colloids. This bridging action promotes the initial aggregation of particles.

Fisher and Glatz (1988a) looked at precipitating egg proteins and proposed that interparticle bridging via these polymeric species is one mechanism which dominates for primary particle aggregation in the precipitation of proteins. Given that the effluent contains high levels of soluble and particulate proteinaceous material, this mechanism may assist in inducing enmeshment and ultimately floc growth.

5.6 EVALUATIONS OF ALUM AND FERRIC SULPHATE AS PRIMARY COAGULANTS WITH SYNTHETIC POLYELECTROLYTES FOR COAGULANT AIDS

Trials were conducted using alum and ferric sulphate as primary coagulants and Zetag 92 and Magnafloc 336 as the coagulant aids. A number of effluent characteristics were investigated and these included suspended solids, turbidity, final floc sizes, capillary suction time (CST), total organic carbon (TOC) and total oil and grease (TOG). These trials were conducted to determine the effective and optimum combinations of the inorganic coagulants with the polymers.

5.6.1 Suspended Solids, Turbidity and Final Floc Sizes.

The results for suspended solids as well as turbidity removals and final floc sizes attained indicate that the optimum pH for their removal is at pH 10, when the polyelectrolytes are used as aids. At pH 8 (and pH 9 for suspended solids and floc growth only, see Appendix G) significant removals of suspended solids and turbidity were attained, and floc sizes reached relatively large sizes, but increasing the pH from pH 8 to pH 10 led to significant improvements in the coagulation efficiency.

In virtually all trials, the increase in suspended solids removals was accompanied by an increase in the turbidity removed and an increase in floc size. In all the trials ferric sulphate produced the highest turbidity and suspended solids removal than alum although both coagulants yielded similar final floc sizes.

The results obtained agree with work done by others. Bough (1974) coagulated vegetable cannery wastewaters and found that ferric sulphate was superior to alum. Trials conducted by Karim and Sistrunk (1985) to evaluate the effects of coagulation on peeled potato wastewaters also revealed that ferric sulphate and ferric chloride outperformed alum as a primary coagulant.

Work done by Jun et al (1994) in treating tofu wastewaters found that at pH 6 ferric sulphate was far superior to alum. Starting with a turbidity of 75 NTU dosing with 200 mg/L ferric sulphate reduced the turbidity to 23 NTU while 200 mg/L alum had little effect, with a final turbidity of 73 NTU. Doubling the dose of both these coagulants to 400 mg/L led to a further reduction of turbidity to 5 NTU and 65 NTU for ferric sulphate and alum respectively.

While our results confirm similar findings in that ferric sulphate was a better coagulant than alum our results indicate ferric sulphate was only marginally better in removing suspended solids, turbidity, TOC and TOG, and not significantly better as found by Bough (1974); Karim and Sistrunk (1985); and Jun et al (1994).

At pH 8 75 mg/L ferric sulphate was able to remove about 33% of the suspended solids compared to 23% removed with 75 mg/L alum. Increasing the pH to pH 10 and using the same doses, about 46% suspended solids were removed with both coagulants. The addition of Zetag 92 greatly improved the removal of suspended solids and turbidity at both pH 8 and pH 10. At 75 mg/L ferric sulphate and 7.5 mg/L Zetag 92 approximately 69% suspended solids and 92% turbidity was removed. With 75 mg/L alum and 7.5 mg/L Zetag 92 approximately 62% suspended solids and 88% turbidity was removed, see Figures 4.13, 4.14, 4.21 and 4.22.

Figures 4.33 and 4.34 indicate that an increase in floc size corresponded with an increase in suspended solids and turbidity removals. These figures also indicate that ferric sulphate in combination with Zetag 92 yielded larger flocs compared to alum.

Magnafloc 336 as a coagulant aid also improved the removal of suspended solids and turbidity when it was used with either primary coagulant and its performance was very similar to Zetag 92 at pH 10. Figures 4.15 and 4.16 show that at 75 mg/L ferric sulphate with 7.5 mg/L Magnafloc 336 approximately 60% suspended solids were removed (compared to about 69%

removal with 7.5 mg/L Zetag 92). At 75 mg/L alum, with 7.5 mg/L Magnafloc 336 approximately 60% suspended solids were removed (compared to about 62% removal with 7.5 mg/L Zetag 92).

Figures 4.23 and 4.24 show that at 75 mg/L for either primary coagulant, with 7.5 mg/L Magnafloc 336, about 93% turbidity is removed. These results are very similar to turbidity removals with Zetag 92. Floc sizes (Figures 4.35 and 4.36) also attained large sizes but were significantly smaller than the flocs attained with Zetag 92.

In all these trials, increasing the dose of the primary coagulant or coagulant aid (both singularly and in combination) did not result in resuspension, indicating that doses of the coagulants were not high enough to cause charge reversal.

5.6.2 Total Organic Carbon (TOC) Removals.

The results obtained for TOC removal potentially indicate that the effluent underwent some form of resuspension. TOC was conducted to yield information on the effects of coagulation on the total solids of the effluent, ie, the soluble and suspended fractions. Since soluble solids determinations were not conducted in any of the trials, it was felt that TOC determinations could be used to give an indication of changes to the composition of soluble or suspended solids.

The results of the TOC analysis are in Figures 4.17 to 4.20 and are quite different to the results of the suspended solids, Figures 4.13 to 4.16. At pH 8, the optimum concentration of alum or ferric sulphate appears to be about 25 mg/L to 50 mg/L. Increasing the pH to pH 10 shifted the optimum to 25 mg/L for both the metal salts. Both the metal salts at pH 8 and pH 10 performed similarly. Dosing beyond the optimum led to significant reductions in the TOC removal efficiency and this suggests there are higher levels of organic carbon in the supernatant.

The results of the suspended solids indicate that increasing the dosage of both the metal salt and the polymer led to an increase in the removal of suspended solids. If the suspended solids are being removed at an increasing rate with increasing dosage, and assuming the soluble solids levels are unaffected, then TOC removal would be expected to increase.

The addition of Zetag 92 or Magnafloc 336 significantly improved TOC reductions. Increasing the pH from pH 8 to pH 10 led to much larger reductions with Zetag 92 (compared to Magnafloc 336) and both the metal salts performed similarly with or without Zetag 92.

According to Randtke (1988) soluble compounds tend to get poorly removed in most coagulation processes because they do not get incorporated into the floc matrix. Semmens and Ayers (1985) also found that soluble solids are poorly removed, when they trialed coagulation to remove humic substances from river water. They found the low to very low molecular weight compounds were not effectively removed. Karim and Sistrunk (1985) coagulated wastewaters generated from steam-peeled potatoes. The suspended solids removals attained were high (about 65%) but COD removals were poor. This was attributed to the fact that soluble compounds are poorly removed in the coagulation process.

Given that soluble solids are poorly removed, the TOC would have been expected to get reduced because high levels of the suspended solids were removed. One hypothesis that could be put forward is that the addition of coagulant alters the effluent system chemically and creates higher levels of soluble (and possibly suspended) solids.

When controls are run, a jar test is simply set up and allowed to stir at the same time and intensity as with the normal jar tests and then allowed to settle for 20 minutes. The top 200 mL of the supernatant is then analysed. The sludge is not analysed (except for CST). Whatever concentration a parameter is in the supernatant of the control is taken as the initial level and all trials are relatively compared to the control and normally expressed as a percentage difference.

It may be possible that during the trials where alum or ferric sulphate were tested, there were interactions with particles normally in the settleable solids fraction, and this gave rise to higher soluble solids which contributed to the increase in TOC of the supernatant. For instance certain proteins or carbohydrates may undergo partial hydrolysis during the metal salts additions, thus contributing to the soluble solids fractions.

This increase in TOC was also noted with the addition of the coagulant aids. It may be possible that residual polymer may remain in the supernatant therefore contributing to the organic carbon.

While many workers have found soluble organics are poorly removed in the coagulation process Licsko (1993) found that soluble solids are removed in coagulation but at low rates. It was found that a proportion of the dissolved organics can form complexes with the metal salts and thus get incorporated into the floc matrix.

5.6.3 Total Oil and Grease (TOG) Removal.

Figures 4.25 to 4.28 show the results of TOG removal. Both ferric sulphate and alum performed similarly in their capacity to remove TOG. At both pH 8 and pH 10 the optimum dose for both the primary coagulants was 25 mg/L. Increasing the dose beyond the optimum had little effect.

The addition of Zetag 92 or Magnafloc 336 resulted in significantly higher removals compared to the primary coagulants only. The primary coagulants had little effect on the polymers. The TOG removal by the polymers deteriorated slightly at pH 10. Clearly maximum TOG removal was attained at pH 8 with about 2.5 mg/L of either polymer and no metal salt coagulant.

There are a number of possible explanations for these results. The coagulation of TOG by either of the metal salts may be by the sweep-floc mechanism, but because of the size of the grease molecules coagulation was not as effective. The coagulation of TOG with the polymers is probably due to the bridging mechanism. The physically large and complex structures of most oils and greases, which in this effluent would include lipoproteins and lipopolysaccharides along with the highly branched structures of the polymers would allow for complex binding/bridging between the two.

About 2.5 mg/L polymer was optimum for removal of about 50% of the TOG. Increasing the polymer dose three-fold had little effect on removing more TOG. This may be due to the other fraction of the TOG being bound up to the settleable solids and are not available to polymer binding.

The use of the metal coagulants as primary coagulants did not enhance coagulation with the polymers. This was unusual because a substantial

amount of TOG was removed with the primary coagulants (about 18%) and this effect was not observed when the coagulant aid was added after the addition of the metal coagulant. It would have been expected that at 25 mg/L of either primary coagulant, a significant increase in TOG removal should have been observed with the polymer additions. The results indicate there was very little if any effect. It appears that any floc formation with alum or ferric sulphate did not assist the bridging mechanism.

5.6.4 Capillary Suction Time (CST).

The only work carried out with the sludge was the CST test. This is a common laboratory procedure used to quantify sludge dewaterability (Kayode and Gregory, 1988). Dentel et al (1988) feels that the CST test evaluates the dewatering capacity of sludges and hence allows us to make estimations of the strength of the hydroxide or polymeric flocs formed and allows us to make comparative evaluations between the sludges. The CST test was conducted to obtain a comparison of the dewaterability of the various sludges obtained with these trials.

There were no significant differences between alum or ferric sulphate when they were used as primary coagulants although at pH 10 ferric sulphate sludges had lower CST values implying the ferric hydroxide flocs were stronger. The use of both Zetag 92 and Magnafloc 336 as coagulant aids significantly decreased the CST and both yielded similar CST values.

Correlations have been found between very strong flocs and low CST (Erickson and Alm, 1991) and it appears that increasing the primary coagulant dose as well as adding polymer strengthens the flocs because the CST dropped significantly. Increasing the polymer two and three-fold led to lower CST values, indicating stronger flocs.

Figures 4.29, pH 10 and 4.31, pH 10, indicate an increase in CST although very slight, at 75 mg/L alum. This implies that the flocs have been weakened or possibly excess alum or excess aluminium hydroxide floc impedes the flow of water from the floc matrix. This was also observed with Magnafloc 336 (Figure 4.32) at pH 8, where at 50 mg/L ferric sulphate there is an increase in CST for 5 mg/L and 7.5 mg/L polymer. This latter case may be due to excess polymer which when incorporated into the matrix, impedes the flow of water from the sludge.

5.7 CHITOSAN

Chitosan was trialed for its effectiveness as both a primary coagulant and a coagulant aid. Chitosan was used as an alternative to the synthetic polyelectrolytes trialed, because as a naturally occurring, biodegradable polymer it may have the added advantage of performing similarly or better than the synthetic polymers.

A number of trials were conducted and all trials yielded similar floc sizes and suspended solids removals. Figure 4.37 only shows the results for trial 2. The results demonstrate that as a primary coagulant and coagulant aid, chitosan works well in a pH range of pH 7 to pH 10, even though at pH 8 it had relatively low removals. The addition of 5 mg/L alum as the primary coagulant significantly increased suspended solids removals and floc growth.

With 10 mg/L chitosan and 5 mg/L alum approximately 77%, 67%, 76% and 68% suspended solids were removed at pH 7, pH 8, pH 9, and pH 10, respectively. Doubling the alum dose to 10 mg/L and keeping the chitosan at 10 mg/L, approximately 79%, 70%, 77% and 72% suspended solids were removed at pH 7, pH 8, pH 9, and pH 10, respectively. Clearly, doubling the dose of alum made very little difference in increasing suspended solids removals although floc sizes increased appreciably.

Similarly, there were not great differences noted between 10 mg/L and 20 mg/L chitosan when alum was used as the primary coagulant. Only slight increases in suspended solids removals were noted when chitosan was doubled although floc sizes did grow significantly.

From these results it appears that chitosan can be used over the broad pH range of pH 7 to pH 10 quite effectively. While the optimum dose of chitosan appears to be in the range of 10 mg/L to 20 mg/L, the optimum dose for alum may actually be slightly lower than the 5 mg/L used in this study.

The results of the trials with chitosan have demonstrated that it is a very effective coagulant aid. The results obtained in this study confirm and support findings by other workers who have used chitosan to coagulate wastewaters generated from food industries.

Bough et al (1975) used chitosan to treat effluent generated from poultry meat processing industries. Using dissolved air flotation, about 5 mg/L chitosan at pH 7 could remove about 93% of the turbidity and 94% of the suspended solids. Increasing the dose to 10 mg/L led to resuspension of the suspended solids. Our results revealed that no resuspension occurred even in trial 1 where 40 mg/L chitosan was used (Appendix H).

This work by Bough et al (1975) revealed that the COD fraction remained high and this was due to the inability of chitosan to remove the dissolved solids fraction. While this study did not investigate the effects of coagulation on the soluble solids fraction of the effluent, Figures 4.17 to 4.20 indicate that soluble solids are poorly removed.

Bough (1975a) also treated vegetable canning wastes with chitosan at pH 6. Chitosan was also compared to a range of synthetic polymers and was found to be the most effective coagulant.

In treating egg breaking wastewaters, Bough (1975b) found chitosan, at 10 mg/L, pH 7, to be very effective as a primary coagulant. It was able to remove 91% of the turbidity and about 83% of the suspended solids. Again, very poor removals of the soluble solids were recorded.

Moore et al (1987) used chitosan as a primary coagulant to successfully treat snap and dry bean effluent. At 20 mg/L and pH 8 about 90% of the turbidity and 85% of the suspended solids were removed.

The results obtained with the above mentioned workers, including the results of this study, reveal that chitosan is effective either as a primary coagulant, or as a coagulant aid which requires relatively low primary coagulant doses. It appears that using chitosan anywhere between 5 mg/L and 20 mg/L, within a pH range of pH 7 to pH10, produces optimum conditions for potentially high turbidity and suspended solids removals. The immediate advantages of this are realised in that the pH of the effluent would not need major adjustments if chitosan is to be used as the only coagulant. Also the uses of metal salts are made redundant or minimised, leading to fewer problems of sludge disposal.

Work carried out by other researchers using chitosan have used much higher doses. No doubt, this is reflective of the strength of the effluent. Work conducted by Kennedy et al (1994) trialed chitosan as a primary coagulant to

extract protein from whey. The optimum pH was at pH 6 and the optimum coagulant dose for maximum protein extraction was between 200 mg/L to 300 mg/L.

Protein solids were also successfully recovered from tofu wastewaters using 300 mg/L chitosan as the primary coagulating agent. Jun et al (1994) were able to reduce the turbidity of these wastewaters by 97% at pH 5.8 (optimum pH and optimum chitosan concentration). Chitosan was equally effective as a coagulant aid or primary coagulant, so the authors felt there was no need to use a metal salt, thus making the process potentially more cost effective. The protein extracted from the effluent had high potential for use as an ingredient in livestock feed formulations.

Chitosan is rapidly becoming more acceptable and widespread in its use in both food and drugs, as it is now realised chitosan is non toxic. According to Weiner (1992) the USA Food and Drug Administration now considers chitosan as a food additive in animal feed if it is only used as a precipitating agent of proteinaceous material from food processing plants, as long as it does not exceed 0.1% of animal feed. Chitosan is also finding increased use in water treatment, such as water filtration.

CHAPTER 6. <u>CONCLUSIONS AND</u> <u>RECOMMENDATIONS</u>

<u>6.1 CONCLUSIONS</u>

A number of studies were conducted for this thesis to assess the efficiency of certain coagulating agents in their ability to reduce the organic waste load of an effluent generated from a starch and gluten manufacturing factory. The main test used to assess these was the jar test.

Ferric sulphate and alum were the two metal salts used as the primary coagulants. A range of synthetic polyelectrolytes were also trialed for their ability to function as primary coagulants and as coagulant aids. They were the Zetag range of polymers (cationic); the Magnafloc range of polymers (anionic) and one non-ionic, Magnafloc 333. Chitosan a cationic derivative of the naturally occurring polymer chitin was also evaluated.

The suspended solids and floc sizes in the supernatant of the settled effluent were used as the indicators of coagulation efficiency, although in some trials TOC, TOG and turbidity were also used as indicators of coagulation efficiency. In almost all cases an increase in floc sizes corresponded to an increase in suspended solids removals.

The optimum results for all the trials can be summarised below as follows:

* Alum as a primary coagulant was able to remove about 54% suspended solids at pH 8. This result is actually an average of the results obtained in Section 4.3 and Section 4.5. This average represents the accurate removals since in both trials very similar results were obtained. This was the highest suspended solids removed using alum as the primary coagulant, 75 mg/L, pH 8.

* Using ferric sulphate at 75 mg/L and at pH 8, higher levels of suspended solids were removed, about, 63%. Again, this result is an average of the various trials which used ferric sulphate as the primary coagulant. Again this result of 63% suspended solids removed reflects very similar suspended

solids removed as both trials yielded similar results. At 50 mg/L and pH 8, ferric sulphate removed about 53% suspended solids.

* The cationic polyelectrolytes (Zetag) proved to be more effective at removing suspended solids than the anionic polyelectrolytes (Magnafloc). At pH 8, 7.5 mg/L Zetag 92 removed approximately 37% suspended solids. Increasing the pH to pH 10, 7.5 mg/L Zetag 92 removed approximately 60% suspended solids and at 5.0 mg/L Zetag 92 removed about 57% suspended solids. Using Magnafloc at 7.5 mg/L, pH 8, removed about 29% suspended solids.

* Using the polyelectrolytes as coagulant aids also revealed that the cationic polyelectrolytes were superior in their ability to remove suspended solids as well as TOC, TOG and turbidity, and to significantly reduce the CST of the sludges generated.

* The use of Zetag 92 (7.5 mg/L) with ferric sulphate (75 mg/L) as the primary coagulant yielded the highest removals of TOC, TOG and turbidity. At pH 8 and pH10, similar levels of TOC, TOG and turbidity were removed, although at pH10, 69% of the suspended solids were removed compared to 48% at pH 8. While alum and Zetag 92 also proved to be very effective in coagulating suspended solids, ferric sulphate with Zetag 92 consistently yielded higher removals.

* Chitosan was also trialed as a primary coagulant and a coagulant aid (where alum was the only primary coagulant trialed). According to Kawamura (1991), chitosan is a biodegradable and non-toxic cationic polyelectrolyte, which produces less sludge than sludges generated in alum coagulation. Even though chitosan is more expensive than many of its synthetic counterparts, an increase in demand will probably lower the price. As drinking water quality standards become more stringent with regards to residual synthetic polymers and residual aluminium, the demand for inorganic and synthetic organic coagulants will no doubt decrease, so the use of chitosan may well rise, thus potentially lowering its cost.

* The results obtained in this thesis with chitosan revealed that as a primary coagulant, it was able to remove very high suspended solids levels over a broad pH range, at only 10 mg/L. At this concentration it removed about 69%, 54%, 75%, and 58% suspended solids at pH 7, pH 8, pH 9, and pH 10,

respectively. With a concentration of 5 mg/L alum as a primary coagulant, and at

10 mg/L chitosan removed approximately 78%, 67%, 78%, and 70% at pH 7, pH 8, pH 9, and pH 10, respectively. These results are averages from the four trials using chitosan (Appendix H). Doubling the concentration of alum to 10 mg/L had little effect, as did doubling the concentration of chitosan to 20 mg/L.

* The results of coagulation with chitosan clearly demonstrate that chitosan at 5 mg/L was far superior than any of the synthetic polyelectrolytes at 5 mg/L or 7.5 mg/L, and chitosan was effective over a wider pH range. Excellent suspended solids removals could also be obtained with very little alum, again showing the cost effective potential of chitosan.

* The excellent results obtained with chitosan and its use with alum may be potentially better. The trials in this thesis did not attempt to fine-tune this work, so it is possible the optimum chitosan concentration may lie between 5 mg/L and 10 mg/L and alum's optimum concentration may lie between 1 mg/L and 5 mg/L. Since ferric sulphate displayed better removal efficiency, even at lower doses, than alum in the other trials, the use of ferric sulphate may prove to be better that alum.

* It is clear that ferric sulphate was more efficient in its ability to remove suspended solids than was alum. Chitosan, alone or in combination with alum, proved to be a very efficient coagulant and was superior to the synthetic polymers trialed.

* This thesis study also investigated various physical parameters of the jar test, in order to determine the optimum conditions which would produce maximum and efficient coagulant dispersion and floc growth, and therefore to maximise coagulation efficiency. Alum and Zetag 92 were used as the test coagulants. The effluent was adjusted to pH 8 before the addition of the coagulants. They were not used in combination for these trials.

* In almost all trials maximum suspended solids removal was accompanied by the formation of the largest flocs. A dosage flow rate of 1 mL/second was found to be optimum for coagulation for both alum and Zetag 92. Approximately 44% and 38% suspended solids were removed when alum was dosed at 0.5 mL/second and 1 mL/second, respectively. With Zetag 92, dosing at either

1 mL/second or 0.5 mL/second resulted in about 74% suspended solids removal. Dosing with either coagulant was done directly into the vortex, as this achieved the highest coagulation. The optimum rapid mix times were found to be 10 seconds for alum and 20 seconds for Zetag 92 yielding removals of 39% and 72% respectively. The optimum flocculation intensity was found to be 30 rpm for both coagulants, yielding 40% and 73% suspended solids removals with alum and Zetag 92 respectively.

The trials conducted in this thesis study showed that the jar test was an effective tool for rapid, reproducible comparisons between various coagulating agents. The results showed that chemical treatment can remove a significant level of the colloidal matter, or suspended solids, and, coagulation-flocculation with sedimentation may be a suitable pretreatment process in the effluent generated from a starch and gluten treatment process.

Chitosan proved to be a very effective cationic polymer and ferric sulphate also proved to be a very effective primary coagulant. Both these coagulants were effective in the pH range of pH 7 to pH 10 and this is the pH range (specifically around pH 8) in which the effluent is discharged. Therefore minimal pH adjustments would be necessary.

Given that a large proportion of the colloidal solids would have been removed in the coagulation-flocculation step the remaining colloidal particles and soluble organics could be treated in a biological process to further reduce the organic load and possibly render the effluent suitable for discharge or to be used as recycle water.

The sludge could have potential use as a value added product. Certainly, the use of chitosan would increase the potential for the sludge safe enough to be added into animal and possibly human feed. This would make the recovery of the sludge financially viable.

6.2 RECOMMENDATIONS AND FUTURE WORK

Various issues and questions were brought up as a result of the trials conducted for this thesis.

The jar test used in this study needs to be further refined in order to achieve coagulation that would reflect a large scale process. To this end, proper

velocity gradient measurement, as well as better dosing or dispersion techniques would need to be adopted.

Dissolved air flotation could be trialed as an alternative to sedimentation. This could also prove to be effective and may give higher levels of supernatant clarification and higher sludge solids (and therefore a relatively higher value added product).

Trials should be conducted on how to further treat the effluent after coagulation. Activated carbon treatment or biological treatment could further reduce the organic fraction. Future work should therefore include analysis of soluble or dissolved solids in order to determine their concentrations and rates of removal.

The work with chitosan should be expanded. Further experiments should be conducted to determine the real optimum values and work should be conducted using ferric sulphate as the primary coagulant.

Analysis of the sludge should be carried out for its protein content/amino acid profile, chitosan content, ash, etc. This information will assist in determining what degree of quality the sludge will have for use as a value added product.

APPENDIX A

APPENDIX A

Ajax Chemicals

A. C. N. 000 002 031

Subject: ALUMINIUM SULFATE (SOLID) CAT #881

Description: Granulated solid aluminium sulfate
$Al_2(SO_4)_3$, $18H_2O$.

Appearance: White granular solid.

Specification:

pH (1% solution)	3.0–3.5
Aluminium Sulfate as Al_2O_3	16.0± 0.5%
Unreacted Al_2O_3	1.5% max.
Water Insolubles (other than Al_2O_3)	0.2% max.
Fe content	100 ppm max.

Application: Soluble aluminium salt for chemical manufacturing, flocculant in water treatment. Dry acid salt in fire extinguishers.

Storage/Handling: See Safety Data Bulletin

Pack: 33.3kg nett multiwall paper bags. (30 to 1 tonne).

Samples and Service: Samples, prices, technical service and further information available on request.

A division of Clyde Industries Limited
(Incorporated in New South Wales)

Tioxide

IRON CHEMICALS DIVISION

PRODUCT NAME: FerriClear

CHEMICAL DESCRIPTION: Ferric sulphate solution

CHEMICAL CHARACTERISATION: $Fe_2(SO_4)_3$

APPEARANCE: Red-Brown viscous liquid

PHYSICAL DATA

Melting/Softening Point	10°C
Boiling Point	112°C
pH	<1
Solubility in water	Miscible
Density	1.605 g/ml at 20°C
	1.612 g/ml at 10°C
Viscosity	55 centipoise at 20°C
	130 centipoise at 10°C
Odour	No
Physical form	20°C - liquid
	0°C - liquid
Solution stability	Stable

STORAGE AND HANDLING

Non-corrosive to:
1. Stainless Steel
2. Rubber
3. P.V.C., Polyethylene, Polypropylene
4. Glass Fibre Reinforced Polyester
5. Glazed Earthenware
6. Lead

Corrosive to:
1. Mild Steel
2. Copper
3. Brass
4. Bronze
5. Zinc
6. Silver
7. Aluminium
8. Natural Fibres and Nylon

Incompatible Substances: Nil

Hazardous Decomposition Products: Under very high temperatures, SO_3 could be evolved.

FerriClear

SPECIFICATIONS AND HANDLING

FerriClear is an aqueous liquor containing ferric sulphate, chemical formula $Fe_2(SO_4)_3$ and is produced in Burnie, Tasmania by the Iron Chemicals Division of Tioxide Australia Pty. Ltd.

Specification

	% W/W
Ferric Sulphate, $Fe_2(SO_4)_3$	44.5-45.5
Ferric Iron, Fe^{3+}	12.5-12.7
Ferrous Iron, Fe^{2+}	0.2 max
Sulphate: Iron, $\dfrac{SO_4^{-}}{Fe^{3+}}$	2.60-2.65
Specific Gravity (average)	1.60g ml⁻¹ (15°C)
Freezing Point	-18°C
Boiling Point	112°C

FerriClear Uses

FerriClear is suitable for most water, effluent and sewage applications:

(1) As a primary coagulant in the treatment of potable and industrial waters.

(2) As a coagulant in the treatment of various industrial effluents.

(3) Phosphate removal from sewage, raw waters and industrial effluents, particularly dairy effluents.

(4) As a primary coagulant in physico-chemical treatment of sewage.

(5) As a coagulant at the primary sedimentation stage of conventional biological sewage treatment as an alternative solution to overloaded works or to improve the throughput of a sewage works.

(6) As a coagulant on conventional sewage treatment after the secondary biological treatment to obviate the need for tertiary treatment plant.

(7) As a coagulant to give a final polish to sewage and other effluents, particularly when environmental limits are difficult to achieve.

FerriClear has numerous other applications.

More detailed descriptions of applications are available on other data sheets.

FerriClear Handling

When handling FerriClear, protective cloting, gloves and goggles should be worn and any spillage on the skin should be washed off immediately with water without soap. Any splash into the eyes should be promptly and thoroughly irrigated and medical attention obtained.

Spillages should be contained and recovered or washed away with large volumes of water.

FerriClear _ Delivery

FerriClear is supplied in bulk tankers of the following capacities:

10 m³ — 16 tonne nett
12.5 m³ — 20 tonne nett

Small bulk deliveries are available in those states represented by a Tioxide agent.

Tanker connections are 50mm male KAMLOK fitting and discharge is usually by pump via a flexible hose. Storage tanks require a flexible hose connection with a 50mm female KAMLOK fitting. A 3-phase power outlet is usually required.

FerriClear
Storage and dosing —
materials of construction

Bulk storage tanks should always be sized to allow sufficient time for delivery after ordering. Consideration should be given to tanker access. Diaphragm type metering pumps are the most suitable for dosing FerriClear.

FerriClear is non-corrosive to:
1. Stainless Steels and Lead
2. Most rubbers
3. P.V.C., Polyethylene, Polypropylene
4. Glass fibre reinforced Polyester
5. Glazed Earthenware

and corrosive to:
1 Mild Steel, Galvanised Steel
2. Copper, Brass, Bronze, Zinc
3. Silver
4. Aluminium
5. Natural Fibres and Nylon

The reaction product produced when FerriClear is used to treat water or wastewater, ferric hydroxide, is non-corrosive.

● Safety Data Sheets are available.

● It is advisable to discuss all bulk storage and delivery proposals with Tioxide personnel.

Tioxide

FerriClear

FerriClear coagulation and flocculation in water treatment

When added to raw water, FerriClear forms a ferric hydroxide floc which is stable over the entire pH range normally encountered in water treatment. Effective range is approximately pH 4.5 to 12.

The exact quantity and conditions required to treat a particular water is determined by undertaking a jar test, described later.

The ferric hydroxide floc when first formed presents a large surface area at which many other more complex reactions occur, and phenomena such as co-precipitation, absorption and adsorption, ensure that organic matter, suspended solids and other contaminants are trapped. Plant conditions should then ensure that the flocs will continue to grow to a size and density that is consistent with requirements for the relevant treatment process — sedimentation, dissolved air floatation, direct filtration, etc.

FerriClear for Phosphate Removal

Phosphate removal from sewage, raw waters and industrial effluents, particularly dairy effluents.

FerriClear for potable water treatment and industrial water pretreatment

The tolerance of FerriClear to sudden changes in raw water quality makes it a logical choice for Potable and industrial Water Treatment. A wide range between the upper, optimum, and lower threshold doses ensures that sudden and fluctuating changes do not result in loss of treated water quality.

It should be stressed, however, that these properties of FerriClear should be considered as a safety factor and in no way should it be used as a substitute for good plant practice of regular jar testing to maintain optimum operating conditions.

FerriClear is particularly effective in removing organic colour such as found in peaty waters. Generally the optimum coagulation pH is slightly lower than other coagulants, and also its ability to coagulate by design at relatively low pH's are factors which can result in more efficient colour removal. This property of FerriClear is beneficial in minimising organic fouling of demineralisation plant resins.

FerriClear is one of the most efficient coagulants available for lime-softening processes because of the stability of its floc at high pH levels. The low doses generally encountered have little effect on pH and residual hardness.

The high pH tolerance of FerriClear is also an important property required of a coagulant in iron and manganese removal processes.

These and other properties provide benefits for the particular treatment process, whether it be horizontal or upflow sedimentation, dissolved air floatation, direct filtration or similar processes.

FerriClear for sewage treatment

FerriClear is effective in coagulating and settling colloidal and semi-colloidal solids from sewage in the primary sedimentation stage resulting in a decreased load on subsequent treatment stages. In addition sludge from chemically treated sewerage is generally found to be much easier to dewater than sludge from conventional processes.

The use of FerriClear at the secondary sedimentation stage can provide a polishing process and obviate the need for further tertiary treatment plant. Furthermore, some activated sludge bulking problems can be overcome with FerriClear.

FerriClear is particularly effective in removing phosphorus from sewage effluent. A result of 90% phosphorus removal is common, and the density of precipitate formed usually makes subsequent flocculation unnecessary. Addition can be at the primary or secondary sedimentation stages.

An alternative solution for heavily loaded plants or for improving other plants is FerriClear treatment to improve performance of the treatment stages.

FerriClear for industrial wastewater treatment

FerriClear has been found to be cost effective in treating a wide range of industrial effluents for removal of soluble, colloidal and suspended matter, both organic and inorganic, reduction of B.O.D. and C.O.D. levels, and for cracking and settling emulsions such as cutting oil and latex emulsions.

A generalisation of treatment procedures cannot be made, but effluents that can be successfully treated with FerriClear include:

Animal processing wastes
Dairy effluents — particularly phosphate removal
Clay effluents
Dye and chemical works wastes
Food processing wastes
Foundry and rolling mill wastes
Gas scrubbing liquors   •
Mining wastes
Oil and grease wastes
Paint effluents — particularly from eletrophoretic
    processes
Paper mill effluents
Rubber latex and associated compounds
Tannery wastes
Water soluble oils

Technical advice and assistance is available on how FerriClear can be best applied to suit your effluent problem.

Laboratory evaluations with FerriClear

A test solution of FerriClear is prepared by diluting 6.25 ml or 10.0 g of FerriClear to 1000 mL with distilled water. (Make up fresh daily).

By adding 1 mL of test solution to 1 litre of sample under test we have the equivalent of 10 mg/litre dose of FerriClear.

The following factors represent what may be significant when undertaking jar tests with FerriClear.

1. FerriClear dose
2. Size of floc
3. Turbidity of clarified water
4. pH of filtered water
5. Colour of filtered water
6. Lime or other alkali dose
7. Polymer dose if evaluated
8. B.O.D. content of clarified wastewater
9. Suspended solids of clarified wastewater

The jar test is ideally a test which reproduces as closely as practicable the processes encountered in a water or wastewater treatment plant.

The jar test is performed using a flocculator or gang stirrer and a normal procedure would be:

1. Addition of sample with fast stirring
2. Fast stir for 2 minutes (100 rpm)
3. Slow stir for 10 minutes (25 rpm)
4. Settle for 15 minutes

Carry out this procedure on a number of samples using a range of dose rates to establish the optimum dose.

Using the optimum FerriClear dose, carry out further tests over the pH range 4 to 10 to establish the optimum pH.

pH adjustment is usually by addition of lime slurry or some other alkali.

If polyelectrolytes are to be evaluated, then varying dose rates of polyelectrolyte should be added to samples using the optimum FerriClear dose and optimum pH level. A variety of polyelectrolytes should be evaluated to establish the most suitable. Dosing of polyelectrolytes to the sample should take place within the two minutes fast stir period following FerriClear and alkali addition.

Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9070

ZETAG AND MAGNAFLOC
High efficiency polyelectrolytes

The synthetic polyelectrolytes of the ZETAG-MAGNAFLOC series have been specifically designed to give high efficiency results wherever solids-liquids separation processes are involved.

Intensive development work and many years field experience on a world-wide basis has indicated the need for a range of products so that selection can be made of the optimum product for a particular substrate and for a particular dewatering process.

A product from the ZETAG-MAGNAFLOC series will produce major economic and processing benefits for the following unit process.

Primary Sedimentation
Secondary Sedimentation
Solids Consolidation
Solids Dewatering on Rotary Vacuum Filters
                    Drying Beds
                    Filter Press
                    Centrifuge
                    Band Filters
                    etc

The HIGH PERFORMANCE ZETAG and MAGNAFLOC are giving lower Running Costs-Faster Thro'Puts-Lower Capital Costs-Extra Base Load Capacity-Extra Peak Load Capacity-More Reliable Operation-More Predictable Operation, in some cases operations otherwise impossible to achieve.

Products from the ZETAG-MAGNAFLOC series are easy to handle on simple mixing equipment and easy to apply on simple metering equipment.

Their high efficiency provides for Lower Volume Storage and Easier Handling Facilities.

Application of ZETAG and MAGNAFLOC

Solution Preparation

Laboratory

Solutions of the solid grade products can be conveniently prepared in the laboratory as follows:

To a dry 150ml bottle add 0.5g powder followed by 3ml meths, methanol or ethanol, in order to wet the powder. Pour on rapidly 97 ml water (temperature below 40°C) and immediately close the bottle and shake vigorously for 10-15 seconds. Occasional hand shaking during the following 30-60 minutes will complete the solution preparation.

Solutions of the liquid grades are prepared in the laboratory as follows:

Add 10g liquid grade product to a bottle add 90 ml of water to add shake vigorously. The product can be used at this concentration.

Plant

The particle size of the solid grade ZETAG and MAGNAFLOC is controlled to give rapid dissolving characteristics, but as with all fine powders, lumps which may be difficult to disperse may form on adding to water. The efficient dispersion of the products is best achieved by the use of an ALLIED COLLOIDS EDUCTOR which is a simple water-powered vacuum pump providing homogeneous solutions with the minimum of time and effort. It is recommended that the products are prepared at a concentration of 0.5% or below for use.

Solutions of the liquid grade products can be conveniently prepared by simply adding the required weight of product, by pump or manually, to a make-up tank which contains the required quantity of water. The tank should be provided with good agitation to ensure complete and homogeneous mixing of the product. It is recommended that stock solutions are diluted 10 fold or 20 fold with water for application.

Application

The recommended solution concentration for addition to the system may vary depending on the type of application and the product in use. Applied concentrations may vary from as high as for example 0.5% for a centrifugation application to as low as 0.05% for a sedimentation application.

The general range of recommended concentrations is 0.05% - 0.5% for the solid grade products and 5%-10% for the liquid grade products.

The general principles to be observed in the application procedure are:

1. Addition of the product at a point of local turbulence
2. Addition of the product as near as possible to the point where flocculation is required
3. Addition of the product throughout the total volume of the substrate to give homogeneous mixing
4. Avoid excessive turbulence after floc formation

Storage

The solid grade products are supplied as free flowing white powders. They should be stored in a cool, dry place and except when reagent is being withdrawn, packages should be carefully closed to prevent entry of moisture.

The liquid grade products are supplied as readily handleable liquids capable of being readily pumped and diluted. Positive displacement pumps, e.g. mono pumps or gear pumps are recommended for transfer of concentrated product. The products should be stored in a cool place where extremes of temperature are avoided.

Technical Service

Advice and assistance in selection of the correct grade for a particular application is given by experienced technologists and complete Technical Service is implicit in the sale of these products.

This includes initial Product Selection Tests-Planning and Execution of Trials-Supply of Equipment For Trials-Routine Efficiency Checks.

Trade Names

ZETAG and MAGNAFLOC are registered trade names of Allied Colloids.

Health and Safety

Detailed information on handling and any precautions to be observed in the use of the product(s) described in this leaflet can be found in our relevant Health and Safety Information Sheet.

Warranty

This information is given in good faith ʒut no liability is assumed, nor is freedom from any patent owned by Allied Colloids or others implied.

Date: 2/90. Issue No. 1.
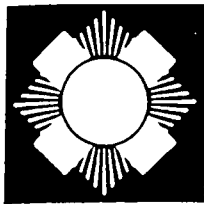
Allied Colloids (AUSTRALIA PTY. LIMITED)

TDP 9104

LABORATORY TESTING OF MAGNAFLOC FLOCCULANTS

Introduction

The MAGNAFLOC range consists of high molecular weight flocculants based on polyacrylamide. A wide range of cationic, non-ionic and anionic grades are available. In order to determine the best product for a given application, it is necessary in the first instance to perform laboratory tests.

Preparation of laboratory solutions

To a dry 100ml bottle add 0.25g MAGNAFLOC powder followed by 3ml methylated spirit, methanol or ethanol in order to wet the powder. Rapidly pour on 97ml water and quickly close the top of the bottle and shake vigorously for 10-15 seconds. Occasional shaking of the bottle during the following 1-2 hours will complete the solution preparation. To ensure rapid and uniform incorporation of the flocculant into the system, MAGNAFLOC solution should be diluted to 0.025% concentration, or further, before using each day. Stock solutions of MAGNAFLOC should be stored in the dark and discarded after 2-3 days.

Choice of test method

In general, laboratory tests should not be designed to reproduce plant conditions. Instead, the reagents should be tested under optimum conditions in the laboratory, and then these conditions be reproduced, as far as is possible, on the plant.

The optimum conditions for flocculation vary according to the pulp under consideration. There are, therefore, no universally applicable standard test procedures. A method should be designed, and followed, to suit the particular problem under consideration.

Before commencing laboratory tests, the problem should be defined in terms of the desired end result and the time and equipment available on the plant to achieve this. The test can then be designed to study the particular features concerned, that is, settling rate, filtration rate, sedimentation compaction and liquor clarity.

It is not necessarily true, for example, to assume that the reagent that will give the highest settling rate will also give the best clarity to supernatant liquor or filtration rate. To avoid ageing effects on the test slurry, the sample should be taken as late as possible prior to running the tests. Care must be taken to obtain a representative sample.

Settling tests

These tests are designed to simulate the action of MAGNAFLOC in a thickener. When possible, tests should be carried out on samples of at least 500ml in measuring cylinders of 30cm or more, in height. To avoid wall effects, the internal diameter of the cylinder should not be less than 5cm.

Using a syringe the 0.025% MAGNAFLOC solution should be introduced onto the surface of the pulp. The sample should then immediately be agitated in a standardised manner to ensure thorough mixing and homogeneous flocculation.

An efficient method of agitation is to close the cylinder with a hand or stopper and to invert gently but completely. With hot or corrosive pulps this is not possible and in these cases MAGNAFLOC can be mixed by raising and lowering a loosely fitting plunger inside the cylinder. The plunger can be solid or perforated. Initial tests should be carried out to establish the optimum agitation routine, which should be adhered to for all subsequent tests. For these initial tests, a dosage of reagent should be used which gives a well-defined floc structure, but modest settling rate.

In some cases, the pulp will flocculate completely with one or two inversions and deteriorate with further agitation. In other cases, a large number of inversions may give improved results. In a majority of cases, three gentle inversions will give optimum results, but this should not be assumed.

Selection of dosage is made by adding MAGNAFLOC as a 0.025% solution in successive cumulative amounts corresponding to 25 to 50g of reagent per tonne of solids. Repeat tests at or about the indicated dosage can then be carried out in which the reagent is added in one, two or three stages.

4ml 0.025% MAGNAFLOC solution per 1000ml slurry equals 1.0 part reagent per million of slurry (ppm) i.e. 1.0mg per litre.

A dosage of 1ppm per 1% solids corresponds to 100g reagent per metric tonne dry solids.

Initial tests indicated that the pulp is so dense as to settle extremely slowly it should, with advantage, be diluted by clean thickener overflow to give a pulp with free settling characteristics. This approximates the condition found in a thickener where the feed pulp is diluted by the supernatant liquor as it passes through the feed zone.

It is important to reproduce the procedure exactly for each test.

According to the test requirements, results may be recorded as a plot of interface height against time, the actual settling rate between two selected points, the pulp volume or residual suspended solids in the supernatant after the specified time has elapsed.

Observation of the susceptibility of the flocs to agitation provides a useful guide when considering the method and point of addition of the flocculant on the plant.

Filtration tests

a) Rotary drum/disc vacuum filtration

The required dose of 0.025% flocculant solution is added to a 500ml slurry sample. The flocculated slurry is well mixed, then gently poured into a beaker.

LABORATORY VACUUM FILTRATION APPARATUS
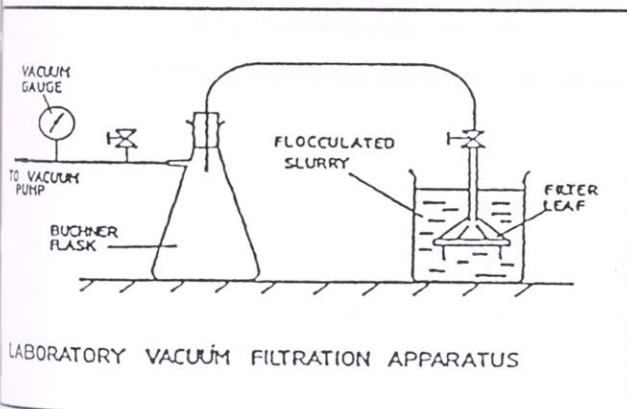
The filter leaf which is connected to a vacuum pump, Buchner flask and vacuum gauge is introduced into the flocculated slurry. The filter leaf should be kept gently moving up and down to maintain a homogeneous suspension, however vigorous agitation should be avoided. After the required pick-up time, lift the leaf vertically out of the sample and maintain vacuum for a set drying time with the leaf inverted. (This is to represent the drum/disc revolving).

On completion of the filtration cycle, release the vacuum, remove the cake, weigh, dry and reweigh to determine moisture content and dry cake yield.

b) Horizontal vacuum belt filtration

A volume of slurry that gives a cake thickness of 6-10mm is used for this testwork. The appropriate dose of flocculant, made up to a constant volume with dilution water, is added and mechanically mixed for a set period.

The flocculated slurry is then transferred to the filter chamber (Buchner funnel). If free drainage exists on plant, allowances should be made in the test method. The vacuum, set to match plant equipment, should then be applied and filtrate volume against time recorded. The time taken for water to disappear from the cake surface is noted and a further drying time suitable to plant conditions is allowed. The vacuum is released and the cake removed, weighed, dried and reweighed to determine wet cake moisture.

Health and safety information

MAGNAFLOC has a low order of oral toxicity and does not present abnormal handling problems.

Detailed information on handling and any precautions to be observed in the use of the product(s) described can be found in our relevant health and safety information sheets.

Warranty

The information contained in this leaflet is given in good faith but no liability is assumed, nor is freedom from any patent owned by Allied Colloids or others implied.

Date: 12/89. Issue No. 1.

Allied Colloids (AUSTRALIA PTY. LIMITED)

PD 9069

ZETAG 53

Cationic polyelectrolyte

Description

ZETAG 53 is a synthetic, ultra-high molecular weight cationic polyacrylamide supplied as a free-flowing white powder. It is completely soluble in water producing solutions of high viscosity.

ZETAG 53 is of high cationic charge.

Principal uses

ZETAG 53 is recommended for use in the treatment of sewage and industrial waste sludges in general. It is particularly recommended where thickening or dewatering of mixed sludges having a high secondary component, by centrifugation is required.

This product has also been found useful in the thickening of activated sludge by flotation.

ZETAG 53 is active over a wide range of pH values.

Viscosity-concentration graph
(Brookfield viscosity, cps. at 6 r.p.m.)

Typical properties

Appearance:	white granular powder
Bulk density:	approx 0.7 g/cm
Particle size:	98% less than 1400um
pH of 1% solution:	3.1 - 4.1
Viscosity at 25°C:	see table and graph

Application and storage

Recommended solution concentration:
Stock solution   0.25 - 0.5% max
Feed solution   0.025 - 0.05% max

Recommended storage periods:
Solid   up to 2 years
Stock solution   2 - 5 days
Feed solution   1 - 3 days

Storage of dry product and solutions for longer than the recommended periods may be acceptable under the correct conditions but could result in some loss of product efficiency. Storage of the solid should be in a cool, dry place and conditions of temperature and high humidity should be avoided. Under such condition the hydroscopic nature of the product may result in excessive moisture up-take and resultant caking. Packages should be kept sealed when not in use. Full details of solution preparation procedures for laboratory and plant are contained within the ZETAG and MAGNAFLOC leaflet. Further advice on solution preparation using an Allied Colloids AUTO JET-WET make-up system is available and details may be obtained on request.

Apparent Viscosity-Concentration data
(Fann Viscometer-Shear rate 5.51 sec^{-1})

Concentration, %	Viscosity, cP
0.2	200
0.4	500
0.6	900
0.8	1250
1.0	1550

Corrosive properties

Corrosion towards most standard materials of construction is very low. Stainless steel, glass fibre, polyethylene, polypropylene and rubberised surfaces are recommended as ideal. In some cases aluminium and galvanised surfaces can be adversely affected.

Packaging

ZETAG 53 is supplied in 25 kg nett weight, plastic bags in palletised shrink-wrapped units of total nett weight 750 kg.

ZETAG 53 is also available in semi-bulk palletised 'big bags' of 600 kg nett weight.
Technical service

Complete technical service is implicit in the sale of ZETAG 53. This includes advice and full assistance in all aspects of product selection, laboratory tests, plant trials etc.

Trade names

ZETAG, MAGNAFLOC and AUTO-JETWET are registered trade names of Allied Colloids.

Health and safety information

ZETAG 53 exhibits a very low order of oral toxicity and does not present any abnormal problems in its handling or general use.

Full details on health and safety aspects are available on request.

Warranty

The information contained in this leaflet is given in good faith but no liability is assumed nor is freedom from any patent owned by Allied Colloids or others implied.

Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9115

ZETAG 57 & 63
Cationic polyelectrolytes

Description

ZETAG 57 and 63 are synthetic, high molecular weight cationic polyelectrolytes supplied in MICRO-BEAD form which renders them essentially non-dusting and free flowing. They are completely soluble in water and, due to their micro-bead form, exhibit fast dissolving rates. ZETAG 63 is of medium cationic charge and ZETAG 57 has high cationic charge.

Principal uses

ZETAG 57 and 63 have been designed specifically to aid the dewatering of sewage and industrial sludges prior to dewatering of sewage and industrial sludges prior to dewatering on centrifuges, vacuum-filters and belt-presses. Both products are particularly suited to the flocculation of mixed and secondary sludges. Other applications include flotation of activated sludge and filter-pressing. Optimum pH activity ranges for ZETAG 57 and 63 is 0 - 9.0. Above pH 9.0 in some instances their activity may be adversely affected.

Viscosity — Concentration graph
(Brookfield viscosity, cps at 6 rpm)
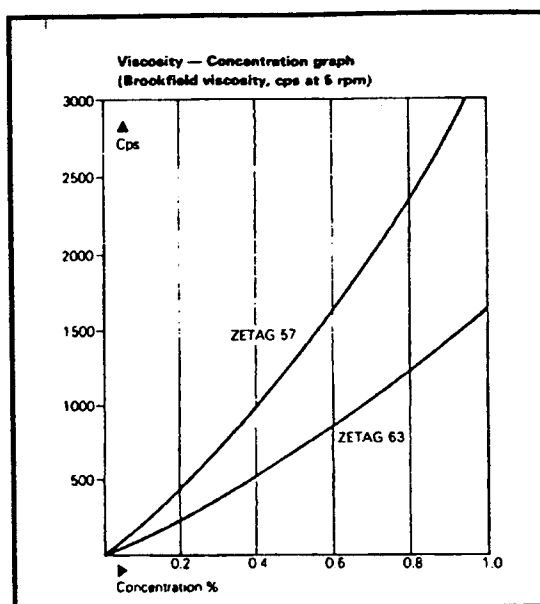
Typical properties

Appearance:	off-white, free flowing micro-beads
Molecular weight:	10 - 12 x 10^6
Bulk density:	approx. 0.78 g/cm^3
Particle size:	100% less than 1.0 mm
pH of 1% solution:	4.2 - 5.0
Viscosity at 25°C:	see graph

Application and storage

<u>Recommended solution concentrations:</u>
Stock solution	0.25 - 1.0% max
Feed solution	0.05 - 0.2% max

<u>Recommended storage periods:</u>
Solid	up to 1 year
Stock solution	up to 1 week
Feed solution	1 - 3 days

Storage of the dry solid should be in a cool, dry place.

Storage conditions of high temperature and high humidity should be avoided. Under such conditions the hygroscopic nature of the material may result in excessive moisture up-take and resultant caking.

Packages should be kept sealed when not in use.

Full details of solution preparation for laboratory and plant are given in the 'ZETAG and MAGNAFLOC' leaflet. Further advice on automatic solution preparation using an Allied Colloids AUTO JET-WET make-up system is available and details may be obtained on request.

Corrosive properties

Corrosion towards most standard materials of construction is low but aluminium and galvanised surfaces should be avoided.

Packaging

ZETAG 57 and 63 are supplied in 25 kg nett weight, plastic bags in shrink-wrapped units of total nett weight 750 kg.

Alternative forms of packaging include 'Big Bags' of 600 kg nett weight.

Technical service
Complete technical service is implicit in the sale of ZETAG products. This includes advice and practical assistance in carrying out laboratory selection tests and on-plant trials.

Trade names
ZETAG, MAGNAFLOC and AUTO JET-WET are registered names of Allied Colloids Limited.

Health and safety information
ZETAG 57 and 63 have a low order of oral toxicity and do not present abnormal handling problems.

Full details on health and safety aspects are available on request.

Warranty
The information contained in this leaflet is given in good faith but no liability is assumed, nor is freedom from any patent owned by Allied Colloids Limited or others implied.

Allied Colloids (AUSTRALIA PTY. LIMITED)

PD 9118

ZETAG 87
Cationic Polyelectrolyte

Description
ZETAG 87 is a synthetic cationic polyelectrolyte supplied in MICRO-BEAD form which renders it essentially non-dusting and free-flowing. It is completely soluble in water and, due to its micro-bead form, exhibits a fast dissolving rate. ZETAG 87 is of high cationic charge and molecular weight.

Principal Uses
ZETAG 87 has been designed specifically to aid dewatering of sludges on filter presses, belt presses and centrifuges. It is also useful in other sludge thickening and dewatering processes and is particularly well suited to the flocculation of sludges which have a high percentage of biological component. ZETAG 87 is effective over a wide range of pH.

Typical Properties
Appearance:	Off-white, free flowing micro bead
Molecular Weight:	$10 - 12 \times 10^6$
Bulk Density:	approx. 0.78 g/cm^3
Particle Size:	100% less than 1.0 mm
pH of 1% Solution:	3.5 - 4.5
Viscosity at 25°C:	See graph

VISCOSITY-CONCENTRATION GRAPH
(BROOKFIELD VISCOSITY, cps. at 5 r.p.m.)

Application and Storage
Recommended solution concentrations:-
Stock solution	0.25 - 1.0% max
Feed solution	0.05 - 0.2% max

Recommended storage periods:-
Solid	up to 1 year
Stock solution	up to 1 week
Feed solution	1 - 3 days

Storage of the dry solid should be in a cool, dry place. Storage conditions of high temperature and high humidity should be avoided. Under such conditions the hygroscopic nature of the material may result in excessive moisture up-take and resultant caking.

Packages should be kept sealed when not in use.

Full details of solution preparation procedure for laboratory and plant are given in the 'ZETAG and MAGNAFLOC' leaflet. Further advice on automatic solution preparation using and Allied Colloids fully automatic system is available and details may be obtained on request.

Corrosive Properties
Corrosion towards most standard materials of construction is low but aluminium and galvanised surfaces should be avoided.

Packaging
ZETAG 87 is supplied in 25 kg nett weight, multi-wall paper bags contained within palletised crates or shrink-wrapped packs of total nett weight 750 kg.

Technical Service
Complete technical Service is implicit in the sale of ZETAG products. This includes advice and practical assistance in carrying out laboratory selection tests and on-plant trials.
Trade Names

ZETAG and MAGNAFLOC are registered trade names of Allied Colloids Limited.

Health and Safety Information
ZETAG 87 has a low order of oral toxicity and does not present abnormal handling problems.

Full details on Health and Safety aspects are available on request.

Warranty
The information contained in this leaflet is given in good faith but no liability is assumed, nor is freedom from any patent owned by Allied Colloids Limited or others implied.
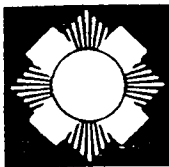
Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9068

ZETAG 92
Cationic polyelectrolyte

Description

ZETAG 92 is a synthetic, ultra-high molecular weight cationic polyelectrolyte supplied as a free-flowing white powder. It is completely soluble in water producing solutions of high viscosity. ZETAG 92 is of medium-high cationic charge.

Principal uses

ZETAG 92 has been designed specifically as a dewatering aid in centrifugation applications involving sewage and industrial effluent sludges, and is particularly suitable for flocculation of primary and digested sludges.

Other applications include vacuum filtration, sludge thickening and flotation and as a sedimentation aid in conjunction with a primary coagulant where phosphate removal is being practised. Optimum pH activity range for ZETAG 92 is 0 - 9.0. Above pH 9.0 in some instances its activity may be adversely affected.

Typical properties

Appearance:	White, free-flowing powder
Molecular weight:	$18 - 20 \times 10^6$
Bulk density:	Approx. 0.66 g/cm^3
Particle size:	95% less than 1.0 mm
pH of 1% solution:	4.2 - 5.0
Viscosity at 25°C:	see table

Application ant storage

Recommended solution concentration:
Stock solution	0.25 - 0.5% max
Feed solution	0.05 - 0.02% max

Recommended storage periods:
Solid	up to 2 years
Stock solution	up to 1 week
Feed solution	1 to 3 days

Storage of the dry solid should be in a cool dry place.

Storage conditions of high temperature and high humidity should be avoided. Under such conditions the hygroscopic nature of the material may result in excessive moisture up-take and resultant caking.

Packages should be kept sealed when not in use.

Full details of solution preparation procedure for laboratory and plant are given in the 'ZETAG and MAGNAFLOC' leaflet. Further advice on solution preparation using an Allied Colloids fully automatic system is available and details may be obtained on request.

Viscosity-concentration data
(Brookfield viscosity, cP at 5 rpm)

Concentration, %	Viscosity, cP
0.2	500
0.4	1200
0.6	2200
0.8	4000

Corrosive properties

Corrosion towards most standard materials of construction is low but aluminium and galvanised surfaces should be avoided.

Packaging

ZETAG 92 is supplied in 25 kg nett weight, plastic bags in palletised shrink-wrapped units of total nett weight 750 kgs.

ZETAG 92 is also available in semi-bulk palletised 'big bags' of 750 to 800 kgs nett weight.

Technical service
Complete technical service is implicit in the sale of ZETAG products. This includes advice and practical assistance in carrying out laboratory selection tests and on-plant trials.

Trade names
ZETAG and MAGNAFLOC are registered trade names of Allied Colloids.

Health and safety information
ZETAG 92 has a low order of oral toxicity and does not present abnormal handling problems.

Full details on health and safety aspects are available on request.

Warranty
The information contained in this leaflet is given in good faith but no liability is assumed, nor is freedom from any patent owned by Allied Colloids or other implied.
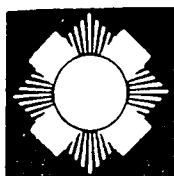
Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9103

MAGNAFLOC 155
ANIONIC FLOCCULANT

Description

MAGNAFLOC 155 is a high molecular weight anionic polyacrylamide flocculant supplied as a free flowing granular powder.

Principal uses

MAGNAFLOC 155 has found application in a wide variety of mineral processing operations including the following:

1. Base metal sulphide and oxide concentrates thickening and filtration.
2. Clarification of 'neutral' stage electrolytic zinc pulps.
3. Sedimentation of coal tailings.
4. Sedimentation of coal fines.
5. Filtration of coal fines.
6. Sedimentation and filtration of cobalt hydroxide.
7. Sedimentation of iron oxides in copper/cobalt leach systems and clarification of pregnant liquor.
8. Sedimentation of fine sands and clays.
9. Tailings dewatering.

Dosage depends on application but normally lies in the range 2g to 200g per tonne of dry substrate flocculated.

Apparent viscosity — Concentration graph
(Fann viscometer — Shear rate 4.95 sec⁻¹)

Typical properties

Physical form:	White granular powder
Particle size:	98% < 750um
Bulk density:	0.85
pH of 1% solution at 25°C:	5.5 - 6.5
Viscostiy at 25°C:	see graph and table

Application and storage

Recommended solution concentrations:	
Stock solution	0.25 - 0.5% max.
Feed solution	0.025 - 0.05% max.
Recommended storage periods:	
Solid	up to two years
Stock solution	1 - 2 days

Storage of polymer should be in cool, dry place.

Details on preparation and application can be obtained from an Allied Colloids representative

MAGNAFLOC 155 Solution viscosity data
(Fann Viscometer - 25°C - solvent - deionised water)

MAGNAFLOC 155 concentration (%)	Shear rate (sec^{-1})					
	4.95	9.90	165	330	495	989
	Viscosity (Cps)					
1.0	2047	1450	195	132	111	80
0.5	999	700	93	67	55	38
0.25	599	300	45	33	30	21
0.10	200	150	24	16	15	10

Shipping and handling
MAGNAFLOC 155 is supplied in 25kg polyethylene-lined, multi-walled bags. MAGNAFLOC 155 has a low order of toxicity and no special precautions are necessary in handling.

Corrosivity towards most standard material of construction is low, but aluminium and galvanised equipment should be avoided.

Technical service
Advice and assistance in the running of laboratory and plant tests to select the correct flocculant and determine the best application is given by representatives of Allied Colloids, who are experienced in mineral processing applications.

MAGNAFLOC is the registered trade name of Allied Colloids.

Health and safety information
MAGNAFLOC 155 exhibits a very low order of oral toxicity and does not present any abnormal problems in its handling or general use.

Detailed information on handling and any precautions to be observed in the use of the product(s) described in this leaflet can be found in our relevant Health and Safety Information sheet.

Warranty
The information contained in this leaflet is given in good faith, but no liability is assumed nor is freedom from any patent owned by Allied Colloids or others implied.

Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9106

MAGNAFLOC 156
Anionic flocculant

Description

MAGNAFLOC 156 is a high molecular weight anionic polyacrylamide flocculant supplied as a free flowing dust free micro-bead.

Principal uses

MAGNAFLOC 156 has application in a wide variety of mineral processing operations including the following:

1. Base metal sulphide and oxide concentrates thickening and filtration.
2. Sedimentation of coal tailings.
3. Sedimentation of coal fines.
4. Filtration of coal fines.
5. Sedimentation and filtration of metal hydroxides.
6. Sedimentation of fine sands and clays.
7. Tailings dewatering.
8. Brine clarification.
9. Phosphate slimes thickening.

Dosage depends on application but normally lies in the range 2g to 200g per tonne of dry substrate flocculated.

Apparent viscosity — Concentration graph
(Fann viscometer — Shear rate 4.35 sec⁻¹)
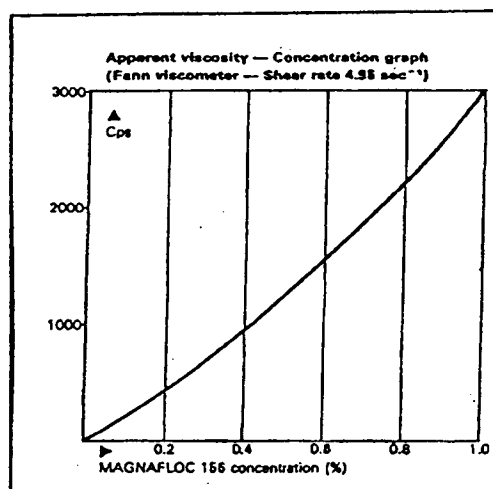
Typical properties

Physical form:	dust free micro-bead
Particle size:	85% < 425um
Bulk density:	0.8 - 0.9
pH of 1% solution at 25°C:	5.5 - 6.5
Viscosity at 25°C:	see graph and table overleaf

Application and storage

Recommended solution concentrations:
Stock solution   0.25 - 0.5% max.
Feed solution   0.025 - 0.05% max.

Recommended storage periods:
Solid   up to two years
Stock solution   1 - 2 days

Storage of polymer should be in a cool, dry place.

Details on preparation and application can be obtained from an Allied Colloids representative.

Solution viscosity data
(Fann viscometer - 25°C - solvent - deionised water)

MAGNAFLOC 156 concentration (%)	Shear rate (sec^{-1})					
	4.95	9.90	165	330	495	989
	Viscosity (Cps)					
1.0	2797	1425	243	150	119	67
0.5	1298	800	118	79	62	41
0.25	599	375	60	40	33	22
0.10	200	150	27	19	15	10

Shipping and handling
MAGNAFLOC 156 is supplied in 25kg polyethylene lined, multi-walled bages. MAGNAFLOC 156 has a low order of toxicity and no special precautions are necessary in handling.

Corrosivity towards most standard materials of construction is low, but aluminium and galvanized equipment should be avoided.

Technical service
Advice and assistance in the running of laboratory and plant tests to select the correct flocculant and determine the best application is given by representatives of Allied Colloids, who are experienced in mineral processing application.

MAGNAFLOC is the registered trade name of Allied Colloids.

Health and safety information
MAGNAFLOC 156 exhibits a very low order of oral toxicity and does not present any abnormal problems in its handling or general use.

Full details on health and safety aspects are available on request.

Warranty
The information contained in this leaflet is given in good faith, but no liability is assumed nor is freedom from any patent owned by Allied Colloids or other implied.

Date: 2/90. Issue No. 1.
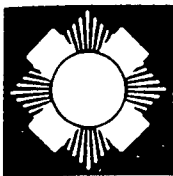
Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9161

MAGNAFLOC 336
Anionic flocculant

Introduction

The MAGNAFLOC 300 series of polyacrylamide based
synthetic flocculants has been developed especially for
those applications where an ultra high molecular weight
flocculant with good pulp mixing characteristics is required.
Although of very high molecular weight, the solution viscosity
is similar to that of conventional flocculants and does not
present special problems when preparing solutions or
adding the flocculant to mineral pulps.

The MAGNAFLOC 300 series is of particular value when
treating very slimey pulps at relatively high pulp densities
when larger than average flocculant dose levels are normally
required.

Description

MAGNAFLOC 336 is a high molecular weight anionic
polyacrylamide flocculant supplied as a free flowing granular
powder.
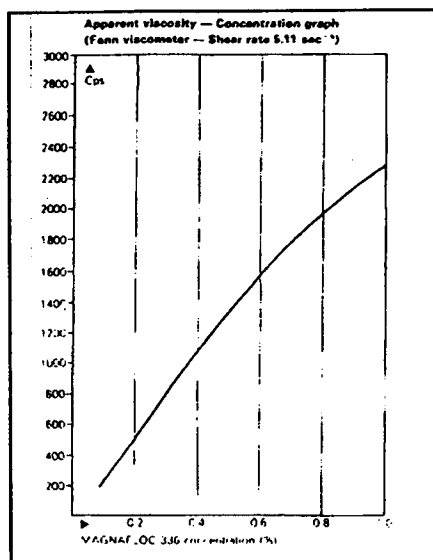
Principal uses

MAGNAFLOC 336 has found application in a wide variety of mineral processing operations including the following:

1. Base metal sulphide and oxide concentrates thickening and filtration.
2. Sedimentation of coal tailings.
3. Sedimentation of coal fines.
4. Filtration of coal fines.
5. Sedimentation and filtration of metal hydroxides.
6. Sedimentation of fine sands and clays.
7. Tailings dewatering.
8. Brine clarification.
9. Phosphate slimes thickening.

Dosage deqends on application but normally lies in the range 50 gm to 200 gm per tonne of dry substrate
flocculated.

Typical properties

Physical form:	White granular powder
Particle size:	98% <750u
Bulk density:	0.7 - 0.8
pH of 1% solution at 25°C:	6.5 - 7.5
Viscosity at 25°C:	See graph and table

Solution viscosity data

(Fann viscometer - 25°C - solvent - deionised water)

MAGNAFLOC 336 concentration(%)	Shear rate (sec-1)					
	5.11	10.22	170	340	511	1022
	Viscosity(Cps)					
1.0	2300	1300	180	120	60	38
0.5	950	550	78	62	49	41
0.25	450	255	44	33	26	22
0.10	180	125	22	17	14	11

Application and storage

Recommended solution concentrations:
Stock solution	0.25 - 0.5% max.
Feed solution	0.025 - 0.05% max.

Recommended storage periods:
Solid	up to two years
Stock solution	1 to 2 days

Storage of polymer should be in a cool, dry place.

Details on preparation and feeding can be obtained from an Allied Colloids representative.

Shipping and handling

MAGNAFLOC 336 is supplied in 25 kg polyethylene-lined, multi-walled bags. MAGNAFLOC 336 has a low order of toxicity and no special precautions are necessary in handling.

Corrosivity towards most standard material of construction is low, but aluminum and galvanized equipment should be avoided.

Technical service

Advice and assistance in the running of laboratory and plant tests to select the correct flocculant and determine the best application is given by representatives of Allied Colloids, who are experienced in mineral processing applications.

MAGNAFLOC is the registered trade name of Allied Colloids.

Health and safety information

MAGNAFLOC 336 exhibits a very low order of oral toxicity and does not present any abnormal problems in its handling or general use.

Full details on Health and Safety aspects are available on request.

Warranty

The information contained in this leaflet is given in good faith, but no liability is assumed nor is freedom from any patent owned by Allied Colloids or others implied.

Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9109

MAGNAFLOC 919
Anionic flocculant

Description

MAGNAFLOC 919 is an ultra high molecular weight anionic polyacrylamide flocculant supplied as a free flowing granular powder.

Principal uses

MAGNAFLOC 919 has found application in a wide variety of mineral processing operations including the following:

1. Coal tailings clarification and refuse dewatering.
2. Copper tailings clarification.
3. Iron ore tailings clarification.
4. Gold cyanidation, CCD washing.
5. Centrifugation of many mineral slurries including clays etc.

Dosage depends on application but normally lies in the range 2g to 200g per tonne of dry substrate flocculated.

Apparent viscosity — Concentration graph
(Fann viscometer — Shear rate 4.35 sec⁻¹)
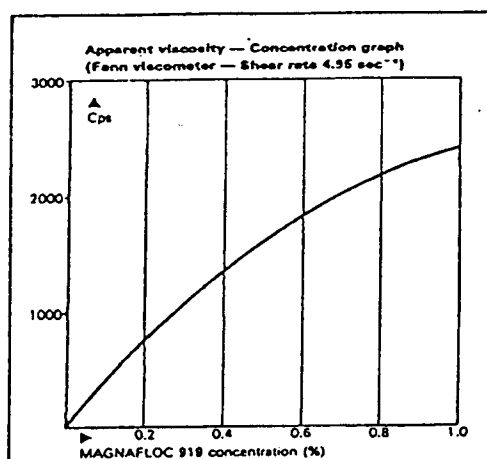
Typical properties

Physical form:	white granular powder
Particle size:	98% < 750um
Bulk density:	0.5 - 0.6
pH of 1% solution at 25°C:	6.0 - 7.0
Viscosity at 25°C:	see graph and table

Application and storage

Recommended solution concentrations:

Stock solution	0.25 - 0.5% max.
Feed solution	0.025 - 0.05% max.

Recommended storage periods:

Solid	up to two years
Stock solution	1 - 2 days

Storage of polymer should be in a cool, dry place.

Details on preparation and application can be obtained from an Allied Colloids representative.

Magnafloc 919 solution viscosity data
(Fann viscometer - 25°C - solvent - deionised water)

MAGNAFLOC 919 concentration (%)	Shear rate (sec^{-1})					
	4.95	9.90	165	330	495	.989
	Viscosity (Cps)					
1.0	2550	1590	260	160	128	75
0.5	1500	900	140	93	73	55
0.25	850	500	74	45	39	22
0.10	300	200	30	21	18	12

Shipping and handling

MAGNAFLOC 919 is supplied in 25kg polyethylene lined, multi-walled bages. MAGNAFLOC 919 has a low order of toxicity and no special precautions are necessary in handling.

Corrosivity towards most standard materials of construction is low, but aluminium and galvanized equipment should be avoided.

Technical service

Advice and assistance in the running of laboratory and plant tests to select the correct flocculant and determine the best application is given by representatives of Allied Colloids, who are experienced in mineral processing applications.

MAGNAFLOC is the registered trade name of Allied Colloids.

Health and safety information

MAGNAFLOC 919 exhibits a very low order of oral toxicity and does not present any abnormal problems in its handling or general use.

Full details on health and safety aspects are available on request.

Warranty

The information contained in this leaflet is given in good faith, but no liability is assumed nor is freedom from any patent owned by Allied Colloids or others implied.

Date: 2/90. Issue No. 1.
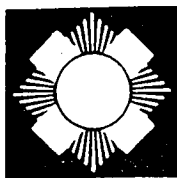
Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9102

MAGNAFLOC 1011
ANIONIC FLOCCULANT

Description

MAGNAFLOC 1011 is a very high molecular weight anionic polyacrylamide flocculant supplied as a free flowing granular powder.

Principal uses

MAGNAFLOC 1011 has found application in a wide variety of mineral processing operations including the following:

1. Base metal sulphide and oxide concentrates thickening and filtration.
2. Sedimentation of coal tailings.
3. Sedimentation of coal fines.
4. Filtration of coal fines.
5. Deep cone thickening of coal tailings.
7. Tailings dewatering.
8. Iron ore tailings.
9. Clarification of acid leach pulp (copper).
10. Sulphur extraction.

Dosage depends on application but normally lies in the range 2g to 200g per tonne of dry substrate flocculated.

Apparent viscosity — Concentration graph
(Fann viscometer — Shear rate 4.95 sec⁻¹)

Typical properties

Physical form:	White granular powder
Particle size:	98% 750um
Bulk density:	0.85
pH of 1% solution at 25°C:	5.5 - 6.5
Viscostiy at 25°C:	see graph and table

Application and storage

Recommended solution concentrations:	
Stock solution	0.25 - 0.5% max.
Feed solution	0.025 - 0.05% max.
Recommended storage periods:	
Solid	up to two years
Stock solution	1 - 2 days

Storage of polymer should be in cool, dry place.

Details on preparation and application can be obtained from an Allied Colloids representative

MAGNAFLOC 1011 Solution viscosity data
(Fann Viscometer - 25°C - solvent - deionised water)

MAGNAFLOC 1011 concentration (%)	Shear rate (sec^{-1})					
	4.95	9.90	165	330	495	989
	Viscosity (Cps)					
1.0	1799	1050	114	76	70	-
0.5	1099	650	87	67	60	45
0.25	549	350	48	39	34	26
0.10	200	150	24	18	17	11

Shipping and handling
MAGNAFLOC 1011 is supplied in 25kg polyethylene-lined, multi-walled bags. MAGNAFLOC 1011 has a low order of toxicity and no special precautions are necessary in handling.

Corrosivity towards most standard material of construction is low, but aluminium and galvanised equipment should be avoided.

Technical service
Advice and assistance in the running of laboratory and plant tests to select the correct flocculant and determine the best application is given by representatives of Allied Colloids. who are experienced in mineral processing applications.

MAGNAFLOC is the registered trade name of Allied Colloids.

Health and safety information
MANGAFLOC 1011 exhibits a very low order of oral toxicity and does not present any abnormal problems in its handling or general use.

Detailed information on handling and any precautions to be observed in the use of the product(s) described in this leaflet can be found in our relevant Health and Safety Information sheet.

Warranty
The information contained in this leaflet is given in good faith, but no liability is assumed nor 's freedom from any patent owned by Allied Colloids or others implied.

Allied Colloids (AUSTRALIA PTY. LIMITED)

TPD 9145

MAGNAFLOC 333
NON-IONIC FLOCCULANT

Introduction

The MAGNAFLOC 300 series of polyacrylamide based synthetic flocculants has been developed especially for those applications where an ultra high molecular weight flocculant with good pulp mixing characteristics is required. Although of very high molecular weight, the solution viscosity is similar to that of conventional flocculants. The MAGNAFLOC 300 series is of particular value when treating very slimey pulps at relatively high pulp densities when larger than average flocculant dose levels are normally required.

Description

MAGNAFLOC 333 is a very high molecular weight non-ionic polyacrylamide flocculant supplied as a free flowing granular powder.

Principal uses

MAGNAFLOC 333 has found application in a wide variety of mineral processing operations including the following:

Apparent viscosity — Concentration graph
(Fann viscometer — Shear rate 5.11 sec⁻¹)
MAGNAFLOC 333 concentration (%)

1. Acid leach CCD (uranium)
2. Acid leach CCD (copper)
3. Sedimentation of jarosite predipitate in electrolytic zinc processes
4. Clarification of zinc electrolyte
5. Flocculation of copper tailings
6. China clay flocculation
7. Base metal sulphide and oxide concentrates thickening and filtration
8. Carbonate leach CCD (uranium)
9. Potash slimes clarification and dewatering
10. Phosphoric acid clarification
11. Iron ore tailings clarification
12. Pregnant liquor clarification (gold)
13. Tailings dewatering

Solution viscosity data
(Fann viscometer - 25°C - solvent - deionised water)

MAGNAFLOC 333 concentration (%)	Shear rate (sec⁻¹)					
	5.11	10.22	170	340	511	1022
	Viscosity (Cps)					
1.0	1200	750	126	87	73	61
0.5	300	200	38	32	28	25
0.25	100	50	15	12	11	10
0.10	75	50	6	6	5	5
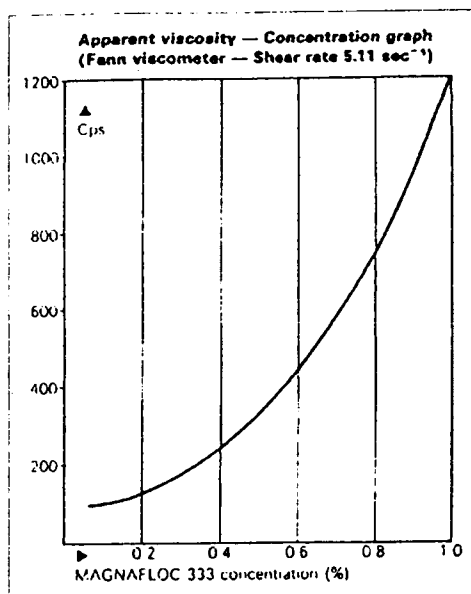
Typical properties

Physical form	White granular powder
Particle size	98% < 750 u
Bulk density	0.7 - 0.8
pH of 1% solution @ 25°C	3.0 - 4.0
Viscosity at 25°C	See graph and table

Application and storage

Recommended solution concentrations:

Stock solution	0.25 to 0.5% max.
Feed solution	0.025 to 0.05% max.

Recommended storage periods:

Solid	up to two years
Stock solution	1 to 2 days

Storage of polymer should be in a cool, dry place.

Details on preparation and feeding can be obtained from an Allied Colloids representative.

Shipping and handling

MAGNAFLOC 333 is supplied in 25kg polyethylene lined, bags. MAGNAFLOC 333 has a low order of toxicity and no special precautions are necessary in handling.

Corrosivity towards most standard material of construction is low, but aluminum and galvanised equipment should be avoided.

Technical service

Advice and assistance in the running of laboratory and plant tests to select the correct flocculant and determine the best application is given by representatives of Allied Colloids, who are experienced in mineral processing applications.

MAGNAFLOC is the registered trade name of Allied Colloids.

Health and safety information

MAGNAFLOC 333 exhibits a very low order of oral toxicity and does not present any abnormal problems in its handling or general use.

Full details on Health and safety aspects are available on request.

Warranty

The information contained in this leaflet is given in good faith, but no liability is assumed nor is freedom from any patent owned by Allied Colloids or others implied.

Date: 12/89. Issue No. 1. UK: 1056.

APPENDIX B

TABLE B.1 24 hour effluent profile for late June, 1990

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2295, 2265	2280	3540, 3560	3550	1200, 1200	1200
3	12.00 PM	2280, 2270	2275	3510, 3490	3500	1200, 1200	1200
4	4.00 PM	2285, 2280	2283	3365, 3365	3365	1200, 1200	1200
5	8.00 PM	2155, 2175	2165	3380, 3360	3370	1100, 1100	1100
6	12.00 AM	2125, 2130	2128	3240, 3300	3270	1100, 1100	1100
7	4.00 AM	2050, 2050	2050	3230, 3190	3210	900, 900	900
8	8.00 AM	2040, 2020	2030	3380, 3110	3245	950, 950	950

TABLE B.2 24 hour effluent profile for early July, 1990

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2170, 2160	2165	3030, 3000	3015	1300, 1300	1300
3	12.00 PM	2185, 2185	2185	3030, 2950	2990	1400, 1400	1400
4	4.00 PM	2220, 2210	2215	2920, 2830	2875	1600, 1600	1600
5	8.00 PM	2240, 2240	2240	2825, 2900	2863	1500, 1500	1500
6	12.00 AM	2190, 2190	2190	2820, 2895	2858	1200, 1200	1200
7	4.00 AM	2140, 2150	2145	2860, 3020	2940	1000, 1000	1000
8	8.00 AM	2100, 2120	2110	3095, 3045	3070	900, 900	900

TABLE B.3 48 hour effluent profile for late July, 1990

	A	B	C	D	E	F	G
		B					
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2395, 2480	2438	3640, 3790	3715	1200, 1200	1200
3	12.00 PM	2340, 2295	2318	3660, 3640	3650	1400, 1400	1400
4	4.00 PM	2285, 2275	2280	3580, 3470	3525	1300, 1300	1300
5	8.00 PM	2240, 2255	2248	3480, 3460	3470	1300, 1300	1300
6	12.00 AM	2240, 2240	2240	3600, 3560	3580	1200, 1200	1200
7	4.00 AM	2270, 2270	2270	3545, 3530	3538	1200, 1200	1200
8	8.00 AM	2285, 2280	2283	3520, 3460	3490	1200, 1200	1200
9	12.00 PM	2275, 2265	2270	3520, 3570	3545	1200, 1200	1200
1 0	4.00 PM	2230, 2240	2235	3560, 3590	3575	1100, 1100	1100
1 1	8.00 PM	2260, 2260	2260	3550, 3535	3543	1100, 1100	1100
1 2	12.00 AM	2245, 2260	2256	3470, 3450	3460	1100, 1100	1100
1 3	4.00 AM	3315, 3345	3330	3995, 4025	4010	1900, 1900	1900
1 4	8.00 AM	2295, 2235	2265	3500, 3460	3480	1100, 1100	1100

TABLE B.4 72 hour effluent profile for early November, 1990

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2180, 2220	2200	3570, 3320	3445	1100, 1100	1100
3	12.00 PM	2195, 2175	2185	3200, 3260	3230	1000, 1000	1000
4	4.00 PM	2165, 2185	2175	3250, 3150	3200	1000, 1000	1000
5	8.00 PM	2170, 2185	2178	3120, 3240	3180	950, 950	950
6	12.00 AM	2220, 2230	2225	3075, 3285	3180	900, 900	900
7	4.00 AM	2160, 2190	2175	3230, 3290	3260	1000, 1000	1000
8	8.00 AM	2200, 2185	2193	3265, 3220	3243	1000, 1000	1000
9	12.00 PM	2170, 2160	2165	3375, 3465	3420	900, 900	900
10	4.00 PM	2195, 2195	2195	3360, 3495	3428	800, 800	800
11	8.00 PM	2220, 2200	2210	3225, 3285	3255	800, 800	800
12	12.00 AM	2235, 2245	2240	3320, 3420	3370	900, 900	900
13	4.00 AM	2215, 2195	2205	3275, 3265	3270	900, 900	900
14	8.00 AM	2300, 2275	2288	3395, 3460	3428	900, 900	900
15	12.00 PM	2315, 2325	2320	3545, 3325	3435	1000, 1000	1000
16	4.00 PM	2335, 2300	2318	3340, 3470	3405	1200, 1200	1200
17	8.00 PM	2325, 2295	2310	3420, 3485	3453	1300, 1300	1300
18	12.00 AM	2255, 2150	2203	4280, 4190	4235	2000, 2000	2000
19	4.00 AM	2280, 2270	2275	3230, 3240	3235	1200, 1200	1200
20	8.00 AM	2230, 2230	2230	3065, 3075	3070	1100, 1100	1100

TABLE B.5 24 hour effluent profile for early December, 1990

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	725, 715	720	2125, 1985	2055	250, 250	250
3	12.00 PM	690, 690	690	2045, 2095	2070	200, 200	200
4	4.00 PM	520, 525	523	1865, 1840	1853	200, 200	200
5	8.00 PM	425, 425	425	1755, 1755	1755	200, 200	200
6	12.00 AM	395, 405	400	1525, 1500	1513	150, 150	150
7	4.00 AM	585, 615	600	1985, 1925	1955	200, 200	200
8	8.00 AM	630, 635	633	1980, 2080	2030	200, 200	200

TABLE B.6 24 hour effluent profile for late January, 1991

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2005, 2015	2010	3155, 3165	3160	950, 950	950
3	12.00 PM	1985, 1985	1985	3150, 3155	3153	850, 850	850
4	4.00 PM	1960, 1955	1958	3070, 3085	3078	800, 800	800
5	8.00 PM	1940, 1930	1935	3045, 3015	3030	750, 750	750
6	12.00 AM	1920, 1910	1915	2980, 2960	2970	550, 550	550
7	4.00 AM	1860, 1865	1863	2890, 2905	2898	500, 500	500
8	8.00 AM	1850, 1850	1850	2800, 2805	2803	500, 500	500

TABLE B.7 48 hour effluent profile for early Februaury, 1991

	A	B	C	D	E	F	G
	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
1							
2	8.00 AM	2330, 2290	2310	3700, 3715	3708	1400, 1400	1400
3	12.00 PM	2210, 2190	2200	3690, 3660	3675	1500, 1500	1500
4	4.00 PM	2140, 2145	2143	3660, 3650	3655	1200, 1200	1200
5	8.00 PM	2100, 2100	2100	3575, 3570	3573	1300, 1300	1300
6	12.00 AM	2055, 2025	2040	3465, 3545	3505	1100, 1100	1100
7	4.00 AM	2015, 2025	2020	3370, 3320	3345	1100, 1100	1100
8	8.00 AM	2005, 1975	1990	3315, 3295	3305	1000, 1000	1000
9	12.00 PM	1960, 1965	1963	3595, 3615	3605	850, 850	850
10	4.00 PM	1955, 1920	1938	3700, 3705	3703	750, 750	750
11	8.00 PM	2115, 2100	2108	3715, 3755	3735	700, 700	700
12	12.00 AM	2120, 2130	2125	3755, 3750	3753	1100, 1100	1100
13	4.00 AM	2200, 2270	2235	3800, 3805	3803	1200, 1200	1200
14	8.00 AM	2250, 2250	2250	3795, 3765	3780	1200, 1200	1200

TABLE B.8 48 hour effluent profile for mid - March, 1991

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2245, 2245	2245	3590, 3570	3580	1100, 1100	1100
3	12.00 PM	2225, 2220	2223	3540, 3490	3515	1000, 1000	1000
4	4.00 PM	2190, 2175	2183	3465, 3445	3455	950, 950	950
5	8.00 PM	2115, 2100	2108	3390, 3365	3378	850, 850	850
6	12.00 AM	2875, 2905	2890	4275, 4265	4270	2000, 2000	2000
7	4.00 AM	2455, 2495	2475	3845, 3790	3818	1300, 1300	1300
8	8.00 AM	2140, 2090	2115	3200, 3185	3193	750, 750	750
9	12.00 PM	2000, 2005	2003	3105, 2990	3048	700, 700	700
10	4.00 PM	1975, 1965	1970	2850, 2845	2848	650, 650	650
11	8.00 PM	1935, 1905	1920	2765, 2695	2730	500, 500	500
12	12.00 AM	1900, 1900	1900	2545, 2525	2535	500, 500	500
13	4.00 AM	2200, 2215	2208	2675, 2690	2683	750, 750	750
14	8.00 AM	2015, 2025	2020	2605, 2595	2600	700, 700	700

TABLE B.9 24 hour effluent profile for late April, 1991

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2235, 2205	2220	3595, 35 05	3550	1200, 1200	1200
3	12.00 PM	2270, 2270	2270	3540, 3560	3550	1200, 1200	1200
4	4.00 PM	2215, 2205	2210	3265, 3195	3230	950, 950	950
5	8.00 PM	2175, 2165	2170	3145, 3100	3123	750, 750	750
6	12.00 AM	2145, 2135	2140	2965, 2995	2980	600, 600	600
7	4.00 AM	2075, 2095	2085	2915, 2905	2910	450, 450	450
8	8.00 AM	2030, 2045	2038	2915, 2890	2903	300, 300	300

TABLE B.10 24 hour effluent profile for late May, 1991

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	1250, 1255	1253	2095, 2070	2083	550, 550	550
3	12.00 PM	1155, 1005	1080	2040, 2015	2028	400, 400	400
4	4.00 PM	980, 985	983	1755, 1705	1730	200, 200	200
5	8.00 PM	750, 750	750	1600, 1650	1625	125, 125	125
6	12.00 AM	700, 705	703	1375, 1370	1373	95, 95	95
7	4.00 AM	1455, 1465	1460	2275, 2275	2275	100, 100	100
8	8.00 AM	2005, 2010	2008	3395, 3285	3340	650, 650	650

TABLE B.11 24 hour effluent profile for mid - June, 1991

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2145, 2155	2150	3455, 3490	3473	950, 950	950
3	12.00 PM	2150, 2145	2148	3450, 3460	3455	900, 900	900
4	4.00 PM	2140, 2135	2138	3495, 3475	3485	900, 900	900
5	8.00 PM	2135, 2140	2138	3415, 3425	3420	950, 950	950
6	12.00 AM	2140, 2140	2140	3420, 3435	3428	950, 950	950
7	4.00 AM	2155, 2160	2158	3465, 3470	3468	900, 900	900
8	8.00 AM	2170, 2175	2173	3415, 3405	3410	900, 900	900

TABLE B.12 24 hour effluent profile for early July, 1991

	A	B	C	D	E	F	G
1	TIME	SUSPENDED SOLIDS (mg/L)	Average	TOTAL SOLIDS (mg/L)	Average	TURBIDITY (NTU)	Average
2	8.00 AM	2235, 2240	2238	3260, 3265	3263	950, 950	950
3	12.00 PM	2275, 2260	2268	3170, 3195	3183	700, 700	700
4	4.00 PM	2175, 2160	2168	3305, 3325	3315	700, 700	700
5	8.00 PM	2155, 2145	2150	3260, 3270	3265	750, 750	750
6	12.00 AM	2130, 2120	2125	3005, 2995	3000	650, 650	650
7	4.00 AM	2100, 2095	2098	2990, 2895	2942	500, 500	500
8	8.00 AM	2075, 2090	2083	2845, 2865	2855	250, 250	250

APPENDIX C

TABLE C2.1. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 7) OF THE USE OF
POLYELECTROLYTES AS PRIMARY COAGULANTS IN THE REMOVAL OF SUSPENDED SOLIDS AT **pH 4**
( ) = Final floc size attained, mm, at 20 minutes.

	A	B	C	D	E	F	G
1		Trial 1, Day 1 Initial/control Suspended solids = 315 mg/L and 325 mg/L. Therefore average = 320 mg/L					
2		TRIAL 1, 2.5 mg/L polymer	% removed	TRIAL 1, 5 mg/L polymer	% removed	TRIAL 1, 7.5 mg/L polymer	% removed
3	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
4	Zetag 92	300, 305 (0.4)	5.5	295, 300 (0.52)	7	280, 280 (0.52)	12.5
5	Zetag 87	305, 310 (0.4)	3.9	300, 300 (0.4)	6.3	275, 280 (0.4)	13.3
6	Zetag 57	310, 310 (0.4)	3.1	295, 295 (0.4)	7.8	270, 270 (0.4)	15.6
7	Zetag 53	315, 310 (0.4)	2.3	295, 300 (0.4)	7	280, 285 (0.52)	11.7
8	Magnafloc 155	310, 310 (0.4)	3.1	290, 295 (0.52)	8.6	275, 275 (0.52)	14.1
9	Magnafloc 156	300, 300 (0.4)	6.3	290, 295 (0.52)	8.6	275, 270 (0.52)	14.8
10	Magnafloc 336	305, 305 (0.4)	4.7	290, 290 (0.52)	9.4	270, 275 (0.63)	14.8
11	Magnafloc 919	305, 300 (0.4)	5.5	290, 285 (0.52)	10.2	265, 265 (0.52)	17.2
12	Magnafloc 1011	305, 310 (0.4)	3.9	295, 300 (0.52)	7	265, 260 (0.52)	18
13	Magnafloc 333	315, 315 (0.4)	1.6	310, 310 (0.4)	3.1	295, 290 (0.4)	8.6
14							
15		Trial 2, Day 7 Initial/control Suspended solids = 180 mg/L and 210 mg/L. Therefore average = 195 mg/L					
16		TRIAL 2, 2.5 mg/L polymer	% removed	TRIAL 2, 5 mg/L polymer	% removed	TRIAL 2, 7.5 mg/L polymer	% removed
17	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
18	Zetag 92	185, 180 (0.4)	6.4	175, 180 (0.52)	9	170, 170 (0.52)	12.8
19	Zetag 87	180, 180 (0.4)	7.7	180, 180 (0.52)	7.7	160, 180 (0.52)	12.8
20	Zetag 57	180, 180 (0.4)	7.7	180, 180 (0.52)	7.7	165, 160 (0.52)	16.7
21	Zetag 53	180, 185 (0.4)	6.4	180, 190 (0.52)	5.1	165, 170 (0.52)	14.1
22	Magnafloc 155	175, 180 (0.4)	9	170, 180 (0.52)	10.3	165, 170 (0.52)	14.1
23	Magnafloc 156	185, 185 (0.4)	5.1	170, 170 (0.52)	12.8	160, 160 (0.63)	17.9
24	Magnafloc 336	185, 185 (0.4)	5.1	170, 170 (0.63)	12.8	160, 155 (0.63)	19.2
25	Magnafloc 919	190, 185 (0.4)	3.9	170, 165 (0.52)	14.1	160, 165 (0.63)	16.7
26	Magnafloc 1011	190, 190 (0.4)	2.6	180, 180 (0.52)	7.7	165, 170 (0.52)	14.1
27	Magnafloc 333	195, 195 (0.4)	0	185, 190 (0.4)	3.8	180, 185 (0.4)	6.4

TABLE C2.2. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 7) OF THE USE OF
POLYELECTROLYTES AS PRIMARY COAGULANTS IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 6
( ) = Final floc size attained, mm, at 20 minutes.

	A	B	C	D	E	F	G
1		Trial 1, Day 1 Initial/control Suspended solids = 315 mg/L and 330 mg/L. Therefore average = 323 mg/L					
2		TRIAL 1, 2.5 mg/L polymer	% removed	TRIAL 1, 5 mg/L polymer	% removed	TRIAL 1, 7.5 mg/L polymer	% removed
3	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
4	Zetag 92	295, 290 (.52)	9.4	280, 280 (.52)	13.3	270, 265 (.63)	17.2
5	Zetag 87	300, 300 (.52)	7.1	275, 270 (.52)	15.6	260, 260 (.63)	19.5
6	Zetag 57	295, 290 (.4)	9.4	290, 285 (.52)	11	260, 260 (.4)	19.5
7	Zetag 53	300, 295 (.4)	7.9	290, 290 (.52)	10.2	270, 275 (.52)	15.6
8	Magnafloc 155	285, 300 (.52)	9.4	290, 280 (.52)	11.8	245, 250 (.52)	23.4
9	Magnafloc 156	285, 290 (.4)	11	285, 285 (.52)	11.8	250, 250 (.63)	22.6
10	Magnafloc 336	290, 280 (.52)	11.8	275, 275 (.63)	14.9	250, 250 (.76)	21.8
11	Magnafloc 919	290, 285 (.4)	11	290, 290 (.52)	10.2	265, 260 (.52)	18.7
12	Magnafloc 1011	285, 290 (.4)	11	290, 295 (.52)	9.4	265, 270 (.63)	17.2
13	Magnafloc 333	315, 315 (.4)	2.5	310, 310 (.4)	4	300, 300 (.4)	7.1
14							
15		Trial 2, Day 7 Initial/control Suspended solids = 190 mg/L and 205 mg/L. Therefore average = 198 mg/L					
16		TRIAL 2, 2.5 mg/L polymer	% removed	TRIAL 2, 5 mg/L polymer	% removed	TRIAL 2, 7.5 mg/L polymer	% removed
17	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
18	Zetag 92	175, 180 (.63)	10.4	175, 170 (.63)	12.9	160, 155 (.63)	20.5
19	Zetag 87	175, 175 (.52)	11.6	170, 170 (.63)	14.1	160, 165 (.63)	17.9
20	Zetag 57	175, 170 (.52)	12.9	165, 170 (.52)	15.4	165, 165 (.52)	16.7
21	Zetag 53	165, 170 (.52)	15.4	170, 170 (.52)	14.1	165, 165 (.52)	16.7
22	Magnafloc 155	170, 175 (.63)	11.6	170, 175 (.63)	12.9	150, 155 (.63)	23
23	Magnafloc 156	165, 160 (.52)	17.9	165, 170 (.52)	15.4	140, 145 (.63)	28
24	Magnafloc 336	165, 160 (.63)	17.9	155, 160 (.76)	20.5	155, 155 (.76)	21.7
25	Magnafloc 919	165, 165 (.52)	16.7	165, 170 (.52)	15.4	145, 145 (.52)	26.8
26	Magnafloc 1011	160, 160 (.52)	19.2	165, 160 (.63)	17.9	155, 150 (.63)	23
27	Magnafloc 333	190, 190 (.4)	4	180, 190 (.4)	6.6	180, 185 (.4)	7.8

TABLE C2.3. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 7) OF THE USE OF
POLYELECTROLYTES AS PRIMARY COAGULANTS IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 7
( ) = Final floc size attained, mm, at 20 minutes.

	A	B	C	D	E	F	G
1		Trial 1, Day 1 Initial/control Suspended solids = 320 mg/L and 320 mg/L. Therefore average = 320 mg/L = 100%						
2		TRIAL 1, 2.5 mg/L polymer	% removed	TRIAL 1, 5 mg/L polymer	% removed	TRIAL 1, 7.5 mg/L polymer	% removed
3	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
4	Zetag 92	290, 285 (.63)	10.2	265, 265 (.76)	17.2	245, 250 (.76)	22.7
5	Zetag 87	280, 280 (.63)	12.5	265, 270 (.63)	16.4	245, 240 (.76)	24.2
6	Zetag 57	275, 280 (.52)	13.3	275, 265 (.63)	15.6	240, 240 (.63)	25
7	Zetag 53	270, 265 (.52)	16.4	260, 265 (.63)	18	245, 240 (.63)	24.2
8	Magnafloc 155	270, 275 (.52)	14.8	260, 270 (.63)	17.2	240, 250 (.76)	23.4
9	Magnafloc 156	275, 275 (.52)	14.1	260, 270 (.63)	17.2	240, 240 (.76)	25
10	Magnafloc 336	265, 260 (.63)	18	260, 270 (.76)	17.2	240, 235 (.88)	25.8
11	Magnafloc 919	265, 265 (.52)	17.2	255, 250 (.63)	21.1	255, 250 (.76)	21.1
12	Magnafloc 1011	265, 270 (.52)	16.4	260, 265 (.63)	18	245, 250 (.76)	22.7
13	Magnafloc 333	305, 310 (.4)	3.9	290, 290 (.4)	9.4	275, 280 (.4)	13.3
14							
15		Trial 2, Day 7 Initial/control Suspended solids = 220 mg/L and 230 mg/L. Therefore average = 225 mg/L = 100%						
16		TRIAL 2, 2.5 mg/L polymer	% removed	TRIAL 2, 5 mg/L polymer	% removed	TRIAL 2, 7.5 mg/L polymer	% removed
17	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
18	Zetag 92	190, 190 (.76)	15.6	185, 190 (.88)	16.7	165, 170 (1.1)	25.6
19	Zetag 87	190, 195 (.76)	14.4	185, 190 (.76)	16.7	165, 165 (1.1)	26.7
20	Zetag 57	200, 195 (.63)	12.2	180, 180 (.76)	20	160, 165 (.88)	27.8
21	Zetag 53	195, 195 (.63)	13.3	180, 185 (.76)	18.9	160, 165 (.88)	27.8
22	Magnafloc 155	180, 180 (.76)	20	175, 180 (.88)	21.1	155, 160 (1.1)	30
23	Magnafloc 156	185, 180 (.76)	18.9	170, 170 (.88)	24.4	155, 150 (1.1)	32.2
24	Magnafloc 336	175, 170 (.88)	23.3	165, 170 (1.1)	25.6	155, 150 (1.6)	32.2
25	Magnafloc 919	185, 180 (.76)	18.9	175, 180 (.88)	21.1	155, 150 (1.3)	32.2
26	Magnafloc 1011	185, 180 (.76)	20	180, 180 (1.1)	20	145, 150 (1.1)	34.4
27	Magnafloc 333	205, 210 (.4)	7.8	200, 195 (.4)	12.2	190, 185 (.52)	16.7

TABLE C2.4. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 7) OF THE USE OF POLYELECTROLYTES AS PRIMARY COAGULANTS IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 8
( ) = Final floc size, mm, attained at 20 minutes.

	A	B	C	D	E	F	G
1		Trial 1, Day 1 Initial/control Suspended solids = 300 mg/L and 310 mg/L. Therefore average = 305 mg/L = 100%						
2		TRIAL 1, 2.5 mg/L polymer	% removed	TRIAL 1, 5 mg/L polymer	% removed	TRIAL 1, 7.5 mg/L polymer	% removed
3	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
4	Zetag 92	250, 245 (.76)	18.9	225, 230 (.88)	25.4	200, 195 (1.1)	35.2
5	Zetag 87	250, 255 (.76)	17.2	230, 240 (.88)	23	210, 215 (.88)	30.3
6	Zetag 57	245, 245 (.76)	19.7	245, 245 (.88)	19.7	210, 210 (.88)	31.1
7	Zetag 53	240, 240 (.76)	21.3	240, 250 (.88)	19.7	210, 210 (.88)	31.1
8	Magnafloc 155	255, 255 (.76)	16.4	250, 260 (.76)	16.4	230, 225 (.76)	25.4
9	Magnafloc 156	260, 260 (.76)	14.8	250, 260 (.76)	16.4	230, 230 (.76)	24.6
10	Magnafloc 336	255, 260 (.76)	15.6	240, 240 (.88)	21.3	220, 220 (.88)	27.9
11	Magnafloc 919	255, 260 (.76)	15.6	255, 255 (.76)	16.4	230, 240 (.76)	23
12	Magnafloc 1011	265, 265 (.76)	13.1	260, 260 (.76)	14.8	235, 240 (.76)	22.1
13	Magnafloc 333	285, 290 (.4)	5.7	270, 280 (.4)	9.8	260, 260 (.52)	14.8
14							
15		Trial 2, Day 7 Initial/control Suspended solids = 210 mg/L and 210 mg/L. Therefore average = 210 mg/L = 100%						
16		TRIAL 2, 2.5 mg/L polymer	% removed	TRIAL 2, 5 mg/L polymer	% removed	TRIAL 2, 7.5 mg/L polymer	% removed
17	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
18	Zetag 92	155, 160 (.88)	25	150, 150 (1.1)	28.6	130, 125 (1.3)	39.3
19	Zetag 87	165, 170 (.88)	20.2	150, 155 (1.1)	27.4	125, 135 (1.1)	38.1
20	Zetag 57	165, 165 (.88)	21.4	155, 155 (1.1)	26.2	130, 130 (1.1)	38.1
21	Zetag 53	160, 160 (.88)	23.8	150, 160 (.88)	26.2	130, 135 (1.1)	36.9
22	Magnafloc 155	170, 170 (.76)	19	165, 170 (.88)	20.2	150, 160 (.88)	26.2
23	Magnafloc 156	170, 170 (.76)	19	170, 170 (.76)	19	160, 160 (.88)	23.8
24	Magnafloc 336	170, 175 (.88)	17.9	160, 160 (1.1)	23.8	150, 145 (1.1)	29.8
25	Magnafloc 919	165, 170 (.88)	20.2	170, 170 (.88)	19	160, 160 (.88)	23.8
26	Magnafloc 1011	165, 170 (.88)	20.2	170, 165 (.88)	20.2	165, 170 (.88)	20.2
27	Magnafloc 333	180, 190 (.4)	11.9	175, 180 (.52)	15.5	170, 170 (.63)	19

TABLE C2.5. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 7) OF THE USE OF
POLYELECTROLYTES AS PRIMARY COAGULANTS IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 10
( ) = Final floc size attained, mm, at 20 minutes.

	A	B	C	D	E	F	G
1		Trial 1, Day 1 Initial/control Suspended solids = 305 mg/L and 315 mg/L. Therefore average = 310 mg/L = 100%					
2		TRIAL 1, 2.5 mg/L polymer	% removed	TRIAL 1, 5 mg/L polymer	% removed	TRIAL 1, 7.5 mg/L polymer	% removed
3	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
4	Zetag 92	210, 200 (1.1)	33.9	135, 130 (1.3)	57.3	125, 130 (1.9)	58.9
5	Zetag 87	205, 210 (1.1)	33.1	145, 150 (1.3)	52.4	140, 140 (1.9)	54.8
6	Zetag 57	225, 220 (.88)	28.2	180, 190 (1.1)	39.5	140, 140 (1.3)	54.8
7	Zetag 53	230, 230 (.88)	25.8	190, 180 (1.1)	40.3	155, 150 (1.3)	50.8
8	Magnafloc 155	265, 260 (.63)	15.3	250, 250 (.76)	19.4	250, 240 (.76)	21
9	Magnafloc 156	260, 260 (.63)	16.1	250, 245 (.76)	20.2	245, 240 (.76)	21.8
10	Magnafloc 336	240, 250 (.76)	21	235, 235 (.88)	24.2	215, 220 (1.1)	29.8
11	Magnafloc 919	270, 260 (.76)	14.5	245, 250 (.76)	20.2	230, 240 (.76)	24.2
12	Magnafloc 1011	260, 270 (.76)	14.5	245, 245 (.76)	21	235, 240 (.76)	23.4
13	Magnafloc 333	295, 290 (.4)	5.6	290, 285 (.4)	7.3	280, 275 (.4)	10.5
14							
15		Trial 2, Day 2 Initial/control Suspended solids = 210 mg/L and 200 mg/L. Therefore average = 205 mg/L = 100%					
16		TRIAL 2, 2.5 mg/L polymer	% removed	TRIAL 2, 5 mg/L polymer	% removed	TRIAL 2, 7.5 mg/L' polymer	% removed
17	Polymer	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
18	Zetag 92	135, 140 (1.3)	32.9	100, 80 (1.6)	56.1	80, 80 (2.3)	61
19	Zetag 87	140, 140 (1.3)	31.7	90, 100 (1.6)	53.7	90, 80 (1.9)	58.5
20	Zetag 57	130, 140 (1.1)	34.1	100, 105 (1.3)	50	80, 90 (1.9)	58.5
21	Zetag 53	135, 140 (1.1)	32.9	110, 110 (1.3)	46.3	90, 100 (1.6)	53.7
22	Magnafloc 155	165, 170 (.76)	18.3	160, 150 (.76)	24.4	155, 160 (.76)	23.2
23	Magnafloc 156	160, 170 (.76)	19.5	155, 160 (.76)	23.2	150, 155 (.76)	25.6
24	Magnafloc 336	160, 160 (.88)	22	145, 150 (1.1)	28	135, 140 (1.1)	32.9
25	Magnafloc 919	170, 165 (.88)	18.3	150, 150 (.88)	26.8	155, 160 (.88)	23.2
26	Magnafloc 1011	165, 170 (.76)	18.3	150, 160 (.88)	24.4	160, 155 (.88)	23.2
27	Magnafloc 333	190, 185 (.4)	8.5	180, 175 (.4)	13.4	170, 170 (.52)	17.1

APPENDIX D

TABLE D 3.1. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF FERRIC SULPHATE AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 4
AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/Lcoagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	475	9.5	430	18.1	405	22.9
4	HALF	325	17.7	300	24.1	290	26.6
5	DOUBLE	600	4.8	605	4	625	0.8
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/Lcoagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	510	8.1	475	14.4	435	21.6
9	HALF	375	8.5	320	22	320	22
10	DOUBLE	665	5	645	7.9	605	13.6

TABLE D3.2. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF ALUM AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 4
AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/Lcoagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	500	4.8	475	9.5	455	13.3
4	HALF	360	8.9	330	16.5	295	25.3
5	DOUBLE	620	1.6	620	1.6	630	0
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/Lcoagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	505	9	460	17.1	445	19.8
9	HALF	375	8.5	325	20.7	300	26.8
10	DOUBLE	680	2.9	660	5.7	635	9.3

TABLE D3.3. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF FERRIC SULPHATE AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 6 AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/L coagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	460	14.8	400	25.9	385	28.7
4	HALF	295	24.4	275	29.5	245	37.2
5	DOUBLE	560	11.1	540	14.3	555	11.9
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/L coagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	465	17.7	415	26.6	385	33.6
9	HALF	310	25.3	290	30.1	275	33.7
10	DOUBLE	625	10.1	590	15.1	525	24.5

TABLE D3.4. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF ALUM AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 6 AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/L coagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	450	16.7	400	25.9	380	29.6
4	HALF	320	17.9	285	26.9	270	30.8
5	DOUBLE	590	9.4	540	14.3	510	19
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/L coagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	480	15.1	425	24.8	410	27.4
9	HALF	315	24.1	325	21.7	305	26.5
10	DOUBLE	650	6.5	600	13.7	570	18

TABLE D3.5. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF FERRIC SULPHATE AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 7 AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/Lcoagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	410	24.1	380	29.6	350	35.2
4	HALF	295	26.3	240	40	220	45
5	DOUBLE	510	19	480	23.8	445	29.4
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/Lcoagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	435	22.3	380	32.1	325	42
9	HALF	320	23.8	265	36.9	265	36.9
10	DOUBLE	550	22	525	25.5	485	31.2

TABLE D3.6. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF ALUM AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 7 AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/Lcoagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	435	19.4	390	27.8	355	34.3
4	HALF	325	18.8	275	31.3	250	37.5
5	DOUBLE	540	14.3	500	20.6	465	26.2
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/Lcoagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	430	23.2	395	29.5	340	39.3
9	HALF	305	27.4	260	38.1	245	41.7
10	DOUBLE	565	19.9	535	24.1	495	29.8

TABLE D3.7. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF FERRIC SULPHATE AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 8 AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/L coagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	330	38.9	270	50	205	62
4	HALF	190	53.7	170	58.5	150	63.4
5	DOUBLE	495	22	445	29.9	415	34.6
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/L coagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	330	41.6	275	51.3	225	60.2
9	HALF	255	39.3	230	45.2	155	63.1
10	DOUBLE	555	22.4	495	30.8	475	33.6

TABLE D3.8. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF ALUM AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT pH 8 AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/L coagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	345	36.1	280	48.2	255	52.9
4	HALF	210	48.8	175	57.3	155	62.2
5	DOUBLE	485	23.6	500	21.3	460	27.6
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/L coagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	345	38.9	295	47.8	255	54.9
9	HALF	245	41.7	195	53.6	200	52.4
10	DOUBLE	595	16.8	585	18.2	555	22.4

TABLE D3.9: TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF FERRIC SULPHATE AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT **pH 10** AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/L coagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	490	12.5	470	16.1	440	21.4
4	HALF	310	27.1	290	31.8	265	37.6
5	DOUBLE	615	5.4	595	8.5	555	14.6
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/L coagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	455	20.2	455	20.2	395	30.7
9	HALF	340	20.9	330	23.3	215	50
10	DOUBLE	660	10.2	625	15	595	19

TABLE D3.10. TRIAL 1 (DAY 1) AND TRIAL 2 (DAY 8) OF THE USE OF ALUM AS A PRIMARY COAGULANT IN THE REMOVAL OF SUSPENDED SOLIDS AT **pH 10** AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1		TRIAL 1, 25 mg/L coagulant	% removed	TRIAL 1, 50 mg/L coagulant	% removed	TRIAL 1, 75 mg/Lcoagulant	% removed
2	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
3	NORMAL	400	28.6	395	29.5	365	34.8
4	HALF	315	25.9	275	35.3	230	45.9
5	DOUBLE	590	9.2	550	15.4	475	26.9
6		TRIAL 2, 25 mg/L coagulant	% removed	TRIAL 2, 50 mg/L coagulant	% removed	TRIAL 2, 75 mg/Lcoagulant	% removed
7	SETTLEABLE SOLIDS	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average	Residual suspended solids (mg/L)	average
8	NORMAL	480	15.8	450	21.1	400	29.8
9	HALF	360	16.3	325	24.4	265	38.4
10	DOUBLE	675	8.2	630	14.3	525	28.6

TABLE D3.11 CONTROL DATA FOR TABLES **3.1 - 3.10**. DATA IS FOR BOTH TRIAL 1 (DAY1) AND TRIAL 2 (DAY8), AND SHOWS THE **SUSPENDED SOLIDS** LEVELS (mg/L) IN THE SUPERNATANT AFTER TWENTY MINUTES SETTLING, FOR EFFLUENT AT THE THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS. THE TABLE ALSO SHOWS THE CHANGES IN THE SUSPENDED SOLIDS WITH CHANGES IN pH.

	A	B	C	D	E	F	G
1	pH	TRIAL 1, NORMAL	TRIAL 1, HALF	TRIAL 1, DOUBLE	TRIAL 2, NORMAL	TRIAL 2, HALF	TRIAL 2, DOUBLE
2	4	525	395	630	555	410	700
3	6	540	390	630	565	415	695
4	7	540	400	630	560	420	705
5	8	540	410	635	565	420	715
6	1 0	560	425	650	570	430	735

TABLE D3.12. EFFLUENT PARAMETERS FOR TRIAL 1 (DAY 1) AND
TRIAL 2 (DAY 8) AT THREE DIFFERENT SETTLEABLE SOLIDS LOADINGS

	A	B	C	D	E	F	G
1	Parameters	TRIAL 1, NORMAL	AVERAGE	TRIAL 1, HALF	AVERAGE	TRIAL 1, DOUBLE	AVERAGE
2	Total Suspended Solids (mg/L)	2145, 2205, 2100	2150	1120, 1200, 1175	1165	4050, 4100, 4095	4082
3	Supernatant Suspended Solids (mg/l	560, 595, 610	588	410, 410, 420	413	630, 620, 620	623
4	Total Turbidity (NTU)	850, 850	850	600, 600	600	1400, 1400	1400
5	Supernatant Turbidity (NTU)	100, 100	100	100, 100	100	125, 125	125
6	pH	8.31		8.31		8.31	
7	Parameters	TRIAL 2, NORMAL	AVERAGE	TRIAL 2, HALF	AVERAGE	TRIAL 2, DOUBLE	AVERAGE
8	Total Suspended Solids (mg/L)	1980, 1970, 1970	1980	1005, 1010, 995	1003	3875, 3870, 3950	3898
9	Supernatant Suspended Solids (mg/l	570, 570, 590	583	420, 420, 420	420	705, 715, 705	708
1 0	Total Turbidity (NTU)	1200, 1200	1200	750, 750	750	2000, 2000	2000
1 1	Supernatant Turbidity (NTU)	200, 200	200	200, 200	200	200, 200	200
1 2	pH	8.08		8.08		8.08	

APPENDIX E

TABLE E4.1 RAPID MIX DURATION.
20 MINUTES SETTLING, 30 MINUTES FLOCCULATION AT 30 R.P.M
ALUM, 25mg/L, pH 8

	A	B	C	D
1	Control = 300 mg/L and 270 mg/L Suspended Solids. Average = 285 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Rapid Mix duration (secs)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	0	235	17.5	0.4
5	10	170	40.4	0.88
6	20	210	26.3	0.63
7	30	205	28.1	0.63
8	60	220	22.8	0.4
9	120	255	10.5	0.4
10	180	260	8.8	0.4
11	Control = 350 mg/L and 340 mg/L Suspended Solids. Average = 345 mg/L			
12		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
13	Rapid Mix duration (secs)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
14	0	285	17.4	0.4
15	10	215	37.7	1.3
16	20	265	23.2	0.63
17	30	270	21.7	0.63
18	60	275	20.3	0.4
19	120	295	14.5	0.4
20	180	320	7.2	0.4

TABLE E4.2 FLOCCULATION DURATION.
20 MINUTES SETTLING, 10 SECONDS RAPID MIX, FLOCCULATION AT 30 R.P.M.
ALUM, 25mg/L, pH 8

	A	B	C	D
1	Control = 300 mg/L and 270 mg/L Suspended Solids. Average = 285 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Flocculation duration (min)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	0	250	12.3	0.4
5	10	200	29.8	0.88
6	20	170	40.4	0.63
7	30	165	42.1	0.63
8	60	190	33.3	0.63
9	Control = 350 mg/L and 340 mg/L Suspended Solids. Average = 345 mg/L			
10		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
11	Flocculation duration (min)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
12	0	315	8.7	0.4
13	10	260	24.6	0.88
14	20	210	39.1	0.63
15	30	225	34.8	0.63
16	60	255	26.1	0.4

TABLE E4.3 FLOCCULATION INTENSITY
20 MINUTES SETTLING, 10 SECONDS RAPID MIX, FLOCCULATION TIME, 20 MINUTES.
ALUM, 25mg/L, pH 8

	A	B	C	D
1	Control = 350 mg/L and 400 mg/L Suspended Solids. Average = 375 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Flocculation intensity (RPM)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	20	270	28	0.63
5	30	220	41.3	0.88
6	40	255	32	0.63
7	60	310	17.3	0.4
8	80	355	5.3	0.4
9	Control = 240 mg/L and 260 mg/L Suspended Solids. Average = 250 mg/L			
10		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
11	Flocculation Intensity (RPM)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
12	20	175	30	0.63
13	30	155	38	0.88
14	40	170	32	0.88
15	60	225	10	0.4
16	80	235	6	0.4

TABLE E4.4 COAGULANT DOSAGE RATES
20 MINUTES SETTLING, 10 SECONDS RAPID MIX, FLOCCULATION: 20 MINUTES AT 30 R.P.M.
5 mLs ALUM AT 25mg/L, pH 8

	A	B	C	D
1	Control = 350 mg/L and 400 mg/L Suspended Solids. Average = 375 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Dosage rate (mLs/sec)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	5	305	18.7	0.63
5	2.5	290	22.7	0.88
6	1.67	270	28	0.88
7	1.25	250	33.3	0.88
8	1	230	38.7	0.88
9	0.5	210	44	1.3
10	Control = 240 mg/L and 260 mg/L Suspended Solids. Average = 250 mg/L			
11		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
12	Dosage rate (mLs/sec)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
13	5	200	20	0.63
14	2.5	185	26	0.88
15	1.67	175	30	0.88
16	1.25	155	38	0.88
17	1	155	38	0.88
18	0.5	140	44	0.88

TABLE E4.5 COAGULANT DOSAGE SITES
20 MINUTES SETTLING, 10 SECONDS RAPID MIX, FLOCCULATION: 20 MINUTES AT 30 R.P.M.
5 mLs ALUM AT 25mg/L AT A DOSAGE RATE OF 1 mL/sec

	A	B	C	D
1	Control = 350 mg/L and 400 mg/L Suspended Solids. Average = 375 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Dosage site	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	Vortex, direct	230	38.7	1.3
5	Surface of effluent	290	22.7	0.63
6	Control = 240 mg/L and 260 mg/L Suspended Solids. Average = 250 mg/L			
7		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
8	Dosage site	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
9	Vortex, direct	150	40	1.3
10	Surface of effluent	180	28	0.63

TABLE E4.6 RAPID MIX DURATION
20 MINUTES SETTLING, FLOCCULATION: 20 MINUTES AT 30 R.P.M.
10 mLs ZETAG 92 (10 mg/L) AT A DOSAGE RATE OF 1 mL/sec.
EFFLUENT pH = 8

	A	B	C	D
1	Control = 280 mg/L and 300 mg/L Suspended Solids. Average = 290 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Rapid Mix duration (secs)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	20	85	70.7	1.9
5	40	80	72.4	1.9
6	60	105	63.8	1.9
7	120	125	56.9	1.3
8	Control = 250 mg/L and 270 mg/L Suspended Solids. Average = 260 mg/L			
9		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
10	Rapid Mix duration (secs)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
11	20	70	73.1	1.9
12	40	85	67.3	1.9
13	60	105	59.6	1.3
14	120	125	51.9	1.3

TABLE E4.7 FLOCCULATION DURATION

20 MINUTES SETTLING, FLOCCULATION:30 R.P.M., 20 SECONDS RAPID MIX
10 mLs **ZETAG 92** (10 mg/L) AT A DOSAGE RATE OF 1 mL/sec.
EFFLUENT pH = 8

	A	B	C	D
1	Control = 280 mg/L and 300 mg/L Suspended Solids. Average = 290 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Flocculation duration (min)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	20	95	67.2	1.9
5	40	90	69	1.9
6	60	95	67.2	1.9
7	120	115	60.3	1.3
8	Control = 250 mg/L and 270 mg/L Suspended Solids. Average = 260 mg/L			
9		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
10	Flocculation duration (min)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
11	20	75	71.2	1.9
12	40	75	71.2	1.9
13	60	80	69.2	1.9
14	120	95	63.5	1.3

TABLE E4.8 FLOCCULATION INTENSITY

20 MINUTES SETTLING, FLOCCULATION: 20 MINUTES, 20 SECONDS RAPID MIX
10 mLs **ZETAG 92** (10 mg/L) AT A DOSAGE RATE OF 1 mL/sec.
EFFLUENT pH = 8

	A	B	C	D
1	Control = 280 mg/L and 300 mg/L Suspended Solids. Average = 290 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Flocculation intensity (RPM)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	30	85	70.7	1.9
5	40	80	72.4	1.9
6	60	95	67.2	1.3
7	80	120	58.6	0.88
8	Control = 250 mg/L and 270 mg/L Suspended Solids. Average = 260 mg/L			
9		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
10	Flocculation intensity (RPM)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
11	30	65	75	1.9
12	40	60	76.9	1.9
13	60	75	71.2	1.9
14	80	95	63.5	1.3

TABLE E4.9 POLYMER DOSAGE RATES

20 MINUTES SETTLING, FLOCCULATION: 20 MINUTES/30 R.P.M 20 SECONDS RAPID MIX
10 mLs **ZETAG 92** (10 mg/L)
EFFLUENT pH = 8

	A	B	C	D
1	Control = 280 mg/L and 300 mg/L Suspended Solids. Average = 290 mg/L			
2		TRIAL 1, DAY 1	TRIAL 1, DAY 1	TRIAL 1, DAY 1
3	Dosage rate (mLs/sec)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
4	10	115	60.3	1.3
5	5	100	65.5	1.3
6	2	85	70.7	1.9
7	1	80	72.4	1.9
8	0.5	75	74.1	1.9
9	Control = 250 mg/L and 270 mg/L Suspended Solids. Average = 260 mg/L			
10		TRIAL 2, DAY 8	TRIAL 2, DAY 8	TRIAL 2, DAY 8
11	Dosage rate (mLs/sec)	Residual Suspended solids (mg/L)	% removed, average	Floc size (mm)
12	10	100	61.5	1.3
13	5	85	67.3	1.3
14	2	75	71.2	1.9
15	1	65	75	1.9
16	0.5	70	73.1	1.9

TABLE E4.10 RAW DATA AND CONTROL DATA

	A	B	C
1	**Raw effluent and Control effluent data for Tables 4.1 & 4.2**		
2		TRIAL 1	TRIAL 2
3	Total Suspended Solids (mg/L)	1950, 1920	2250, 2270
4	Average	1935	2260
5	Control Suspended Solids (mg/L)	300, 270	350, 340
6	Average	285	345
7	pH	8.09	7.86
8	**Raw effluent and Control effluent data for Tables 4.3 - 4.5**		
9		TRIAL 1	TRIAL 2
10	Total Suspended Solids (mg/L)	2050, 2150	1860, 1870
11	Average	2100	1865
12	Control Suspended Solids (mg/L)	350, 400	240, 260
13	Average	375	250
14	pH	7.66	8.21
15	**Raw effluent and Control effluent data for Tables 4.6 - 4.9**		
16		TRIAL 1	TRIAL 2
17	Total Suspended Solids (mg/L)	2350, 2390	2100, 2100
18	Average	2370	2100
19	Control Suspended Solids (mg/L)	280, 300	250, 270
20	Average	290	260
21	pH	8.34	7.39

APPENDIX F

Table F5.1 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Aluminium sulphate** as a primary coagulant at pH 6.

	A	B	C	D	E
1	Aluminium	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	720, 740, 740	0	0.4	6
4	0.5	730, 730	0.4	0.4	5.97
5	1	730, 725	0.8	0.4	5.9
6	2.5	725, 725	1.1	0.4	5.83
7	5	720, 715	2.1	0.52	5.74
8	7.5	715, 715	2.5	0.52	5.73
9	10	695, 690	5.5	0.63	5.69
1 0	15	670, 680	7.9	0.76	5.65
1 1	25	665, 660	9.6	0.76	5.61
1 2	50	570, 575	21.9	0.88	5.32
1 3	75	540, 535	26.7	1.1	5.15
1 4	100	620, 625	15.1	0.63	4.83
1 5	150	655, 660	10.3	0.52	4.51
1 6	250	695, 690	5.5	0.4	4.31

Table F5.2 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Aluminium sulphate** as a primary coagulant at pH 7.

	A	B	C	D	E
1	Aluminium	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	710, 715, 700	0	0.4	7
4	0.5	695, 700	1.5	0.4	6.99
5	1	695, 690	2.2	0.4	6.93
6	2.5	690, 685	2.9	0.52	6.94
7	5	685, 685	3.2	0.52	6.89
8	7.5	665, 665	6.1	0.63	6.7
9	10	650, 645	8.5	0.63	6.6
1 0	15	625, 620	12.1	0.76	6.59
1 1	25	600, 605	14.9	0.88	6.59
1 2	50	510, 500	28.7	1.1	6.23
1 3	75	480, 475	32.6	1.3	5.41
1 4	100	490, 480	31.5	1.1	5.31
1 5	150	510, 520	27.3	1.1	5.19
1 6	250	595, 595	16	0.52	4.76

Table F5.3 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Aluminium sulphate** as a primary coagulant at pH 8.

	A	B	C	D	E
1	Aluminium	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	670, 685, 690	0	0.4	8
4	0.5	665, 670	2.1	0.4	7.95
5	1	645, 650	5.1	0.52	7.93
6	2.5	635, 640	6.5	0.52	7.9
7	5	600, 600	12	0.52	7.85
8	7.5	585, 580	14.6	0.63	7.8
9	10	505, 500	26.3	0.88	7.73
1 0	15	485, 480	29.3	0.88	7.67
1 1	25	445, 440	35.1	1.1	7.52
1 2	50	360, 365	46.8	1.3	7.14
1 3	75	315, 310	54.2	1.6	6.59
1 4	100	300, 300	56	1.6	6.27
1 5	150	400, 405	41	1.3	6.03
1 6	250	540, 535	21.1	1.1	5.76

Table F5.4 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Aluminium sulphate** as a primary coagulant at pH 9.

	A	B	C	D	E
1	Aluminium	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	650, 650, 660	0	0.4	9
4	0.5	640, 640	2	0.4	8.93
5	1	620, 620	5.1	0.4	8.87
6	2.5	595, 590	9.3	0.52	8.65
7	5	575, 570	12.3	0.63	8.43
8	7.5	560, 560	14.2	0.76	8.32
9	10	515, 510	21.5	0.88	8.11
1 0	15	495, 490	24.6	0.88	7.96
1 1	25	470, 470	28	1.1	7.43
1 2	50	415, 420	36.1	1.1	7.21
1 3	75	345, 350	46.8	1.3	6.93
1 4	100	400, 395	39.1	1.1	6.47
1 5	150	450, 450	31.1	1.1	6.13
1 6	250	480, 485	26.1	0.88	6.01

Table F5.5 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Aluminium sulphate** as a primary coagulant at pH 10.

	A	B	C	D	E
1	Aluminium	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	660, 640, 650	0	0.4	1 0
4	0.5	650, 650	0	0.4	9.85
5	1	645, 650	0.4	0.4	9.76
6	2.5	625, 630	3.5	0.4	9.56
7	5	600, 605	7.3	0.4	9.32
8	7.5	575, 570	11.9	0.52	9.16
9	10	555, 550	15	0.63	8.97
1 0	15	540, 535	17.3	0.63	8.52
1 1	25	515, 510	21.2	0.76	8.16
1 2	50	500, 495	23.5	0.76	7.94
1 3	75	450, 445	31.2	0.88	7.63
1 4	100	460, 460	29.2	0.88	7.16
1 5	150	480, 485	25.8	0.76	6.76
1 6	250	560, 555	14.2	0.52	6.21

Table F5.6 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Ferric sulphate** as a primary coagulant at pH 6.

	A	B	C	D	E
1	Ferric	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	835, 835, 840	0	0.4	6
4	0.5	835, 835	0.2	0.4	5.94
5	1	835, 835	0.2	0.4	5.83
6	2.5	830, 830	0.8	0.4	5.79
7	5	800, 810	3.8	0.4	5.71
8	7.5	795, 795	5	0.52	5.59
9	10	775, 780	7.1	0.52	5.41
1 0	15	765, 770	8.3	0.52	5.29
1 1	25	740, 745	11.3	0.63	5.19
1 2	50	685, 690	17.9	0.76	5.15
1 3	75	640, 635	23.8	0.76	4.96
1 4	100	670, 670	20	0.63	4.74
1 5	150	705, 700	16.1	0.52	4.31
1 6	250	835, 830	0.6	0.4	4.09

Table F5.7 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Ferric sulphate** as a primary coagulant at pH 7.

	A	B	C	D	E
1	Ferric	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	850, 850, 850	0	0.4	7
4	0.5	840, 835	1.5	0.4	6.98
5	1	830, 835	2.1	0.4	6.95
6	2.5	810, 805	5	0.4	6.93
7	5	780, 785	7.9	0.52	6.86
8	7.5	780, 785	7.9	0.52	6.85
9	10	750, 755	11.5	0.63	6.74
1 0	15	750, 745	12.1	0.63	6.58
1 1	25	680, 685	19.7	0.76	6.34
1 2	50	605, 600	29.1	0.76	6.19
1 3	75	535, 540	36.8	0.88	6.09
1 4	100	560, 555	34.4	0.76	5.78
1 5	150	655, 650	23.2	0.76	5.55
1 6	250	690, 685	19.1	0.63	5.09

Table F5.8 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Ferric sulphate** as a primary coagulant at pH 8.

	A	B	C	D	E
1	Ferric	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	840, 845, 830	0	0.4	8
4	0.5	780, 790	6.3	0.4	7.92
5	1	760, 765	9	0.52	7.81
6	2.5	730, 725	13.2	0.63	7.77
7	5	705, 700	16.2	0.76	7.63
8	7.5	640, 635	23.9	0.88	7.44
9	10	570, 580	31.4	0.88	7.32
1 0	15	495, 490	41.2	1.1	7.14
1 1	25	450, 455	46	1.3	7.09
1 2	50	345, 340	59.1	1.9	6.94
1 3	75	275, 280	67.5	2.3	6.82
1 4	100	480, 490	42.1	1.9	6.56
1 5	150	525, 530	37.1	1.6	6.44
1 6	250	705, 700	16.2	0.76	6.18

Table F5.9 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Ferric sulphate** as a primary coagulant at pH 9.

	A	B	C	D	E
1	Ferric	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	825, 830, 835	0	0.4	9
4	0.5	805, 810	2.7	0.4	8.97
5	1	755, 750	9.3	0.63	8.84
6	2.5	730, 730	12	0.63	8.72
7	5	705, 710	14.8	0.76	8.66
8	7.5	665, 670	19.6	0.76	8.51
9	10	605, 610	26.8	0.88	8.23
1 0	15	575, 570	31	0.88	8.09
1 1	25	530, 530	36.1	0.88	7.87
1 2	50	465, 470	43.7	0.88	7.64
1 3	75	390, 390	53	1.3	7.41
1 4	100	540, 540	34.9	0.76	7.2
1 5	150	605, 610	26.8	0.76	7
1 6	250	655, 650	21.4	0.76	6.66

Table F5.10 Suspended solids (mg/L and %) removed, final floc size attained (mm) at 20 minutes flocculation and final pH of coagulation of starch wastewaters with **Ferric sulphate** as a primary coagulant at pH 10.

	A	B	C	D	E
1	Ferric	Residual Suspended	Suspended solids	Final floc size, mm	Final
2	sulphate, mg/L	solids, mg/L	removed (%)	attained at 20 min	pH
3	0	830, 835, 840	0	0.4	10
4	0.5	825, 830	0.9	0.4	9.97
5	1	800, 805	3.9	0.52	9.82
6	2.5	770, 765	8.1	0.52	9.78
7	5	750, 755	9.9	0.52	9.64
8	7.5	735, 735	12	0.52	9.42
9	10	720, 715	14.1	0.52	9.21
1 0	15	700, 705	15.9	0.63	9.02
1 1	25	640, 645	23.1	0.63	8.75
1 2	50	595, 590	29	0.76	8.58
1 3	75	565, 570	32	0.76	8.31
1 4	100	660, 660	21	0.63	8.12
1 5	150	725, 720	13.5	0.52	7.87
1 6	250	800, 800	4.2	0.4	7.48

TABLE 5.11 Raw Results for Section 5

	A	B	C	D	E
1	PARAMETERS	TRIAL 1		TRIAL 2	
2		Aluminium sulphate	average	Ferric sulphate	average
3	Total Suspended Solids (mg/L)	2125, 2135, 2130	2130	2195, 2190, 2205	2197
4	Supernatant/Control Suspended Solids (mg/L)	685, 670, 690	682	840, 830, 845	838
5	pH	7.95		8.21	

TABLE 5.12 Changes in
Supernatant/Control Suspended Solids (mg/L) corresponding to changes in pH

	A	B	C	D	E
1	pH	TRIAL 1		TRIAL 2	
2		Aluminium sulphate	average	Ferric sulphate	average
3	6	720, 740, 740	733	835, 840, 835	837
4	7	710, 700, 715	708	850, 850, 850	850
5	8	685, 670, 690	682	840, 845, 830	838
6	9	660, 650, 650	653	825, 830, 835	830
7	10	640, 660, 650	650	840, 830, 835	835

APPENDIX G

TABLE G6.1 Zetag 92 at **0 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removal

	A	B	C	D	E
1	pH	Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2		SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	265, 265	225, 210	200, 225	205, 200
4	9	240, 260	195, 190	175, 190	165, 170
5	10	250, 250	195, 190	180, 160	120, 145
6	pH	Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7		% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	0	17.9	19.8	23.6
9	9	0	23	27	33
10	10	0	23	32	47

TABLE G6.2 Zetag 92, **0 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	3180	2480	2330	2750
4	10	3200	2360	2560	2640
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	0	22	26.7	13.5
8	10	0	26.3	20	17.5

TABLE G6.3 Zetag 92, **0 mg/L** and Aluminium sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	200, 200	125, 125	100, 100	80, 80
4	10	200, 200	100, 125	65, 75	50, 60
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	0	37.5	50	60
8	10	0	43.8	65	72.5

TABLE G6.4 Zetag 92, **0 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	21, 21	14, 12	10, 11	11, 8
4	10	21, 21	11, 10	9, 9	10, 12
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	21	13	10.5	9.5
7	10	21	10.5	9	11

TABLE G6.5 Zetag, **0 mg/L** and Aluminium sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	61.3	51.4	52	50.7
4	10	62.7	54.8	54.8	52.3
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	0	16.2	15.2	17.3
7	10	0	12.6	12.6	16.6

TABLE G6.6. The effects of **Zetag 92, 0 mg/L** and Aluminium sulphate,as primary coagulant, on floc growth (mm). Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.4	0.63	0.76	0.76
4	9	0.4	0.63	0.76	0.76
5	10	0.4	0.63	0.76	0.76

TABLE G6.7 Zetag 92 at **2.5 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	230, 230	210, 220	195, 200	190, 195
4	9	190, 195	195, 200	175, 180	110, 120
5	10	175, 180	160, 165	145, 150	120, 160
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	13.2	18.9	25.5	27.4
9	9	23	21	29	54
10	10	29	35	41	44

TABLE G6.8 Zetag 92, **2.5 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2480	2290	2140	2610
4	10	2280	2140	2150	2510
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	22	28	32.7	17.9
8	10	28.8	33.1	32.8	21.6

TABLE G6.9 Zetag 92, **2.5 mg/L** and Aluminium sulphate. Turbidity (NTU) removed

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	100, 150	75, 80	65, 70	60, 60
4	10	125, 125	100, 100	50, 55	35, 40
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	37.5	61.3	66.3	70
8	10	37.5	50	73.8	81.3

TABLE G6.10 Zetag 92, **2.5 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	13, 14	10, 8	8, 7	6, 8
4	10	13, 12	9, 10	7, 6	10, 9
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	13.5	9	7.5	7
7	10	12.5	9.5	6.5	9.5

TABLE G6.11 Zetag, **2.5 mg/L** and Aluminium sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	33.4	31.5	31.5	29.6
4	10	42.2	40.9	37.8	37.2
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	47	50	50	53
7	10	32.7	34.8	39.7	40.7

TABLE G6.12. The effects of **Zetag 92, 2.5 mg/L** and Aluminium sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.88	1.3	1.3
4	9	0.76	0.88	1.3	1.3
5	10	0.76	0.88	1.3	1.6

TABLE G6.13 Zetag 92 at **5.0 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	220, 215	210, 200	170, 185	160, 175
4	9	165, 185	170, 180	150, 160	85, 110
5	10	115, 125	100, 130	110, 110	110, 105
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	17.9	22.6	33	36.8
9	9	30	30	38	61
10	10	52	54	56	57

TABLE G6.14 Zetag 92, **5.0 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2330	2160	1950	2490
4	10	1980	1690	1660	2380
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	26.7	32.1	38.7	21.7
8	10	38.1	47.2	48.1	25.6

TABLE G6.15 Zetag 92, **5.0 mg/L** and Aluminium sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	80, 100	60, 90	50, 55	50, 40
4	10	85, 100	80, 95	40, 50	40, 30
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	55	62.5	73.8	77.5
8	10	53.8	56.3	77.5	82.5

TABLE G6.16 Zetag 92, **5.0 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	14, 15	6, 7	6, 5	8, 7
4	10	12, 13	7, 9	7, 6	7, 8
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	14.5	6.5	5.5	7.5
7	10	12.5	8	6.5	7.5

TABLE G6.17 Zetag, **5.0 mg/L** and Aluminium sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	34.7	32.1	30.9	26.5
4	10	43.5	44.1	40.9	39.7
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	44.9	49	51	57.9
7	10	30.6	29.7	34.7	36.7

TABLE G6.18. The effects of **Zetag 92, 5.0 mg/L** and Aluminium sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.88	1.3	1.3
4	9	0.88	0.88	1.3	1.3
5	10	0.88	1.09	1.3	1.6

TABLE G6.19 Zetag 92 at **7.5 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	200, 195	195, 195	170, 185	140, 150
4	9	155, 150	170, 150	140, 135	75, 80
5	1 0	105, 110	90, 110	80, 105	90, 90
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	25.5	26.4	33	45.3
9	9	39	36	45	69
1 0	1 0	57	60	63	64

TABLE G6.20 Zetag 92, **7.5 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2200	1950	1720	2330
4	1 0	1580	1450	1350	2390
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	30.8	38.7	45.9	26.7
8	1 0	50.6	54.7	57.8	25.3

TABLE G6.21 Zetag 92, **7.5 mg/L** and Aluminium sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	75, 90	45, 55	40, 60	45, 35
4	1 0	75, 80	65, 80	35, 40	20, 25
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	58.8	75	75	80
8	1 0	61.3	63.8	81.3	88.8

TABLE G6.22 Zetag 92, **7.5 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	12, 13	7, 5	5, 5	7, 6
4	1 0	14, 12	6, 8	6, 5	4, 4
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	12.5	6	5	6.5
7	1 0	13	7	5.5	4

TABLE G6.23 Zetag, **7,5 mg/L** and Aluminium sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	35.9	31.5	30.2	29.6
4	1 0	40.9	37.8	37.2	36.5
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	43	50	52.1	53
7	1 0	34.7	39.7	40.7	41.8

TABLE G6.24. The effects of **Zetag 92, 7.5 mg/L** and Aluminium sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.88	1.09	1.3	1.6
4	9	0.88	1.09	1.3	1.3
5	1 0	0.88	1.3	1.6	1.6

TABLE G6.25 Zetag 92 at **0 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	265, 265	190, 220	190, 195	175, 175
4	9	240, 260	175, 180	165, 175	140, 150
5	10	250, 250	185, 190	165, 165	130, 135
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	0	22.6	27.4	34
9	9	0	29	32	42
10	10	0	25	34	47

TABLE G6.26 Zetag 92, **0 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	3180	2450	2450	2740
4	10	3200	2270	2360	2580
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	0	23	23	13.8
8	10	0	29.1	26.3	19.4

TABLE G6.27 Zetag 92, **0 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	200, 200	100, 100	80, 80	80, 80
4	10	200, 200	100, 125	65, 70	45, 45
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	0	50	60	60
8	10	0	43.8	66.3	77.5

TABLE G6.28 Zetag 92, **0 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	21, 21	13, 16	11, 10	10, 9
4	10	21, 21	10, 10	8, 9	8, 8
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	21	14.5	10.5	9.5
7	10	21	10	8.5	8

TABLE G6.29 Zetag, **0 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	61.3	49.4	48.7	47.6
4	10	62.7	53.6	53	51.1
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	0	19.4	20.6	22.3
7	10	0	14.5	15.5	18.5

TABLE G6.30. The effects of **Zetag 92, 0 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.4	0.63	0.76	0.76
4	9	0.4	0.63	0.76	0.76
5	10	0.4	0.63	0.76	0.76

TABLE G6.31 Zetag 92 at **2.5 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	230, 230	190, 200	180, 190	160, 165
4	9	190, 195	170, 150	1140, 160	130, 140
5	10	175, 180	150, 150	140, 140	110, 105
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	13.2	26.4	30.2	38.7
9	9	23	36	40	46
10	10	29	40	44	57

TABLE G6.32 Zetag 92, **2.5 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2480	2280	2070	2510
4	10	2280	2080	2010	2450
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	22	28.3	34.9	21.1
8	10	28.8	35	37.2	23.4

TABLE G6.33 Zetag 92, **2.5 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	100, 150	60, 70	55, 55	55, 55
4	10	125, 125	90, 95	45, 50	30, 40
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	37.5	67.5	72.5	72.5
8	10	37.5	53.8	76.3	82.5

TABLE G6.34 Zetag 92, **2.5 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	13, 14	9, 9	9, 8	8, 7
4	10	12, 13	8, 8	6, 6	8, 7
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	13.5	9	8.5	7.5
7	10	12.5	8	6	7.5

TABLE G6.35 Zetag, **2.5 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	30	27.4	30.4	27.3
4	10	41.9	39.2	37.2	32.9
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	52.4	56.5	51.7	56.7
7	10	33.2	37.5	40.7	47.5

TABLE G6.36. The effects of **Zetag 92, 2.5 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.88	1.3	1.6
4	9	0.76	0.88	1.3	1.6
5	10	0.76	0.88	1.3	1.6

TABLE G6.37 Zetag 92 at **5.0 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	215, 220	175, 180	170, 170	150, 150
4	9	175, 175	140, 145	135, 135	120, 120
5	10	120, 120	125, 125	105, 110	95, 95
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	17.9	33	35.8	43.4
9	9	30	43	46	52
10	10	52	50	57	62

TABLE G6.38 Zetag 92, **5.0 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2330	2140	1880	2320
4	10	1980	1570	1650	2290
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	26.7	32.7	40.9	27
8	10	38.1	50.9	48.4	28.4

TABLE G6.39 Zetag 92, **5.0 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	80, 100	45, 60	45, 50	45, 45
4	10	85, 100	80, 100	30, 35	20, 25
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	55	75	76.3	77.5
8	10	53.8	55	83.8	88.8

TABLE G6.40 Zetag 92, **5.0 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	14, 15	8, 9	7, 8	6, 6
4	10	12, 13	8, 8	6, 6	7, 6
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	14.5	8.5	7.5	6
7	10	12.5	8	6	6.5

TABLE G6.41 Zetag, **5.0 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	30.7	31	31.5	27.4
4	10	42.7	39.5	37	36.1
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	51.3	50.8	50	65.5
7	10	31.9	37	41	42.4

TABLE G6.42. The effects of **Zetag 92, 5.0 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.88	1.6	1.6
4	9	0.76	0.88	1.3	1.6
5	10	0.88	1.09	1.3	1.6

TABLE G6.43 Zetag 92 at **7.5 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	195, 200	160, 160	160, 160	135, 140
4	9	155, 150	130, 130	120, 115	110, 100
5	10	105, 110	110, 110	100, 100	80, 75
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	25.5	39.6	39.6	48.1
9	9	37	48	53	58
10	10	57	56	60	69

TABLE G6.44 Zetag 92, **7.5 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2200	1950	1560	2260
4	10	1580	1310	1350	2210
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	30.8	38.7	50.9	28.9
8	10	50.6	59.1	57.8	30.9

TABLE G6.45 Zetag 92, **7.5 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	75, 90	40, 50	40, 40	40, 50
4	10	75, 80	75, 75	30, 25	20, 15
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	58.8	77.5	80	77.5
8	10	61.3	62.5	86.3	91.3

TABLE G6.46 Zetag 92, **7.5 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	13, 12	10, 8	6, 5	5, 4
4	10	14, 12	7, 6	6, 5	6, 5
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	12.5	9	5.5	4.5
7	10	13	6.5	5.5	5.5

TABLE G6.47 Zetag, **7.5 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	30.2	29.6	32.7	28.5
4	10	42.6	39.7	35.9	32.6
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	52.1	53	48.1	54.5
7	10	32.1	36.7	42.7	48

TABLE G6.48. The effects of **Zetag 92, 7.5 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.88	1.09	1.6	1.6
4	9	0.88	1.09	1.6	1.6
5	10	0.88	1.3	1.6	1.9

TABLE G6.49 Magnafloc 336 at **0 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	265, 265	220, 215	210, 215	200, 200
4	9	240, 260	190, 195	185, 180	160, 175
5	10	250, 250	190, 200	170, 175	135, 130
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	0	17.9	19.8	24.5
9	9	0	23	27	33
10	10	0	22	31	47

TABLE G6.50 Magnafloc 336, **0 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	3180	2480	2330	2750
4	10	3200	2360	2560	2640
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	0	22	26.7	13.5
8	10	0	26.3	20	17.5

TABLE G6.51 Magnafloc 336, **0 mg/L** and Aluminium sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	200, 200	125, 125	100, 100	80, 80
4	10	200, 200	100, 125	65, 75	50, 60
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	0	37.5	50	60
8	10	0	43.8	65	72.5

TABLE G6.52 Magnafloc 336, **0 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	21, 21	14, 12	10, 11	11, 8
4	10	21, 21	11, 10	9, 9	10, 12
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	21	13	10.5	9.5
7	10	21	10.5	9	11

TABLE G6.53 Magnafloc 336, **0 mg/L** and Aluminium sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	63	51	47.9	44.1
4	10	62.7	54.8	54.8	52.3
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	0	19	24	30
7	10	0	12.6	12.6	16.6

TABLE G6.54. The effects of **Magnafloc 336, 0 mg/L** and Aluminium sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.4	0.63	0.76	0.76
4	9	0.4	0.63	0.76	0.76
5	10	0.4	0.63	0.76	0.76

TABLE G6.55 Magnafloc 336 at **2.5 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	220, 235	200, 210	180, 180	150, 155
4	9	220, 215	170, 170	150, 155	140, 140
5	10	180, 190	165, 165	140, 150	130, 140
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	14.2	22.6	32.1	42.5
9	9	13	32	39	44
10	10	26	34	42	46

TABLE G6.56 Magnafloc 336, **2.5 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2890	2390	2190	2700
4	10	2780	2210	2210	2460
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	9.1	24.8	31.1	15.1
8	10	13.1	30.9	30.9	23.1

TABLE G6.57 Magnafloc 336, **2.5 mg/L** and Aluminium sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	100, 100	90, 100	75, 75	70, 70
4	10	100, 125	75, 80	30, 40	25, 30
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	50	52.5	62.5	65
8	10	43.8	61.3	82.5	86.3

TABLE G6.58 Magnafloc 336, **2.5 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	12, 12	11, 10	8, 8	8, 7
4	10	10, 10	8, 7	7, 6	8, 8
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	12	10.5	8	7.5
7	10	10	7.5	6.5	8

TABLE G6.59 Magnafloc 336, **2.5 mg/L** and Aluminium sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	32.8	31.5	31.5	32.1
4	10	33.4	33.4	35.3	35.9
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	47.9	50	50	49
7	10	46.7	46.7	43.7	42.7

TABLE G6.60. The effects of **Magnafloc 336, 2.5 mg/L** and Aluminium sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.88	1.3	1.3
4	9	0.76	0.88	1.3	1.3
5	10	0.76	0.88	1.3	1.3

TABLE G6.61 Magnafloc 336 at **5.0 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	215, 215	185, 185	170, 165	120, 125
4	9	195, 200	150, 155	145, 145	115, 125
5	1 0	160, 165	145, 140	135, 135	115, 115
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	18.9	30.2	36.8	53.8
9	9	21	39	42	52
1 0	1 0	35	43	46	54

TABLE G6.62 Magnafloc 336, **5.0 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2670	2230	1910	2580
4	1 0	2560	2090	2150	2310
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	16	29.9	39.9	18.9
8	1 0	20	34.7	32.8	27.8

TABLE G6.63 Magnafloc 336, **5.0 mg/L** and Aluminium sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	90, 95	65, 70	60, 50	60, 50
4	1 0	75, 100	70, 90	20, 40	15, 20
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	53.8	66.3	72.5	72.5
8	1 0	56.3	60	85	91.3

TABLE G6.64 Magnafloc 336, **5.0 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	13, 11	9, 8	9, 8	5, 6
4	1 0	12, 14	7, 6	6, 5	5, 6
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	12	8.5	8.5	5.5
7	1 0	13	6.5	5.5	5.5

TABLE G6.65 Magnafloc 336, **5.0 mg/L** and Aluminium sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	34.7	33.4	31.5	30.2
4	1 0	38.4	39.1	38.4	37.8
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	44.9	47	50	52.1
7	1 0	38.8	37.6	38.8	39.7

TABLE G6.66. The effects of **Magnafloc 336, 5.0 mg/L** and Aluminium sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.88	1.3	1.3
4	9	0.76	0.88	1.3	1.3
5	10	0.76	0.88	1.3	1.3

TABLE G6.67 Magnafloc 336 at **7.5 mg/L** and Aluminium sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	190, 195	165, 170	150, 155	130, 130
4	9	175, 175	155, 160	140, 145	110, 115
5	1 0	165, 170	145, 155	135, 135	100, 105
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	27.4	36.8	42.5	50.9
9	9	30	37	43	55
1 0	1 0	33	40	46	59

TABLE G6.68 Magnafloc 336, **7.5 mg/L** and Aluminium sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2540	2060	1840	2490
4	1 0	2300	1930	2090	2250
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	20.1	35.2	42.1	21.7
8	1 0	28.1	39.7	34.7	29.7

TABLE G6.69 Magnafloc 336, **7.5 mg/L** and Aluminium sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	65, 70	60, 75	40, 50	40, 60
4	1 0	65, 80	50, 60	20, 20	15, 15
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	66.3	66.3	77.5	75
8	1 0	63.8	72.5	90	92.5

TABLE G6.70 Magnafloc 336, **7.5 mg/L** and Aluminium sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	15, 12	7, 7	6, 7	6, 6
4	1 0	13, 13	6, 7	4, 5	6, 6
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	13.5	7	6.5	6
7	1 0	13	6.5	4.5	6

TABLE G6.71 Magnafloc 336, **7.5 mg/L** and Aluminium sulphate as primary coagulant.Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	31.5	31.5	30.2	30.2
4	1 0	39.1	37.8	37.2	37.2
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	50	50	52.1	52.1
7	1 0	37.6	39.7	40.7	40.7

TABLE G6.72. The effects of **Magnafloc 336, 7.5 mg/L** and Aluminium sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.88	0.88	1.3	1.3
4	9	0.88	0.88	1.3	1.3
5	1 0	0.88	0.88	1.3	1.3

TABLE G6.73 Magnafloc 336 at **0 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	265, 265	205, 205	190, 190	175, 175
4	9	240, 260	185, 190	165, 170	145, 145
5	10	250, 250	175, 180	165, 165	130, 130
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	0	22.6	28.3	34
9	9	0	25	33	42
10	10	0	29	34	48

TABLE G6.74 Magnafloc 336, **0 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	3180	2450	2450	2740
4	10	3200	2270	2360	2580
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	0	23	23	13.8
8	10	0	29.1	26.3	19.4

TABLE G6.75 Magnafloc 336, **0 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	200, 200	100, 100	80, 80	80, 80
4	10	200, 200	100, 125	65, 70	45, 45
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	0	50	60	60
8	10	0	43.8	66.3	77.5

TABLE G6.76 Magnafloc 336, **0 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	21, 21	13, 16	11, 10	10, 9
4	10	21, 21	10, 10	8, 9	8, 8
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	21	14.5	10.5	9.5
7	10	21	10	8.5	8

TABLE G6.77 Magnafloc 336, **0 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	61.3	49.4	48.7	47.6
4	10	62.7	53.6	53	51.1
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	0	19.4	20.6	22.3
7	10	0	14.5	15.5	18.5

TABLE G6.78. The effects of **Magnafloc 336, 0 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.4	0.63	0.76	0.76
4	9	0.4	0.63	0.76	0.76
5	10	0.4	0.63	0.76	0.76

TABLE G6.79 Magnafloc 336 at **2.5 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	225, 230	180, 180	190, 190	155, 160
4	9	205, 205	165, 170	160, 160	125, 130
5	10	185, 190	155, 160	155, 160	115, 115
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	14.2	32.1	28.3	40.6
9	9	18	33	36	49
10	10	25	37	37	54

TABLE G6.80 Magnafloc 336, **2.5 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2890	2390	2200	2580
4	10	2280	2090	2300	2480
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	9.1	24.8	30.8	18.9
8	10	28.8	34.7	28.1	22.5

TABLE G6.81 Magnafloc 336, **2.5 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	100, 100	80, 90	70, 75	60, 70
4	10	100, 125	75, 85	30, 30	20, 20
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	50	57.5	63.8	67.5
8	10	43.8	60	85	90

TABLE G6.82 Magnafloc 336, **2.5 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	12, 12	9, 10	8, 9	9, 9
4	10	10, 10	7, 8	6, 5	6, 7
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	12	9.5	8.5	9
7	10	10	7.5	5.5	6.5

TABLE G6.83 Magnafloc 336, **2.5 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	30.1	30.3	30.9	31.5
4	10	34.3	35.4	36.3	35.9
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	52.2	51.9	51	50
7	10	45.3	43.5	42.1	42.7

TABLE G6.84. The effects of **Magnafloc 336, 2.5 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.76	1.3	1.6
4	9	0.76	0.76	1.3	1.3
5	10	0.76	0.88	1.3	1.6

TABLE G6.85 Magnafloc 336 at **5.0 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	215, 200	175, 170	180, 185	135, 135
4	9	185, 190	155, 150	150, 150	120, 115
5	10	160, 165	145, 150	140, 140	110, 105
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	21.7	34.9	31.1	49.1
9	9	25	39	40	53
10	10	35	41	44	57

TABLE G6.86 Magnafloc 336, **5.0 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2670	2230	1920	2450
4	10	1980	1990	2150	2310
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	16	29.9	39.6	23
8	10	38.1	37.8	32.8	27.8

TABLE G6.87 Magnafloc 336, **5.0 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	90, 95	50, 65	70, 80	50, 60
4	10	75, 100	55, 60	20, 25	10, 10
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	53.8	71.3	62.5	72.5
8	10	56.3	71.3	88.8	95

TABLE G6.88 Magnafloc 336, **5.0 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	13, 11	10, 9	7, 6	10, 7
4	10	12, 14	5, 7	5, 6	5, 6
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	12	9.5	6.5	8.5
7	10	13	6	5.5	5.5

TABLE G6.89 Magnafloc 336, **5.0 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	29.6	30.9	30.6	30.2
4	10	32.9	34.9	36.9	35.8
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	53	51	51.4	52.1
7	10	47.5	44.3	41.1	42.9

TABLE G6.90. The effects of **Magnafloc 336, 5.0 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.76	0.76	1.3	1.6
4	9	0.76	0.76	1.3	1.6
5	10	0.76	0.88	1.6	1.6

TABLE G6.91 Magnafloc 336 at **7.5 mg/L** and Ferric sulphate as primary coagulant. Suspended solids removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
3	8	195, 200	160, 160	165, 165	125, 130
4	9	175, 170	145, 140	140, 145	110, 110
5	10	165, 170	140, 140	125, 130	100, 105
6		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
7	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
8	8	25.5	39.6	37.7	51.9
9	9	31	43	43	56
10	10	33	44	49	59

TABLE G6.92 Magnafloc 336, **7.5 mg/L** and Ferric sulphate. TOC removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L	TOC remaining, mg/L
3	8	2540	1980	1820	2290
4	10	1580	1860	1970	2160
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, TOC	% removed, TOC	% removed, TOC	% removed, TOC
7	8	20.1	37.7	42.8	28
8	10	50.6	41.9	38.4	32.5

TABLE G6.93 Magnafloc 336, **7.5 mg/L** and Ferric sulphate. Turbidity (NTU) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	remaining NTU	remaining NTU	remaining NTU	remaining NTU
3	8	65, 70	50, 55	60, 50	40, 65
4	10	65, 80	40, 60	15, 20	10, 15
5		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
6	pH	% removed, NTU	% removed, NTU	% removed, NTU	% removed, NTU
7	8	66.3	73.8	72.5	73.8
8	10	63.8	75	91.3	93.8

TABLE G6.94 Magnafloc 336, **7.5 mg/L** and Ferric sulphate. Capillary Suction Time (sec) of sludge.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	CST (sec)	CST (sec)	CST (sec)	CST (sec)
3	8	15, 12	8, 8	4, 5	7, 6
4	10	13, 13	4, 5	5, 5	5, 5
5	pH	average CST (sec)	average CST (sec)	average CST (sec)	average CST (sec)
6	8	13.5	8	4.5	6.5
7	10	13	4.5	5	5

TABLE G6.95 Magnafloc 336, **7.5 mg/L** and Ferric sulphate as primary coagulant. Total oil and Grease (mg/L) removed.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)	Grease (mg/L)
3	8	29.3	31.5	29.3	29.6
4	10	35	32.6	35.2	33
5	pH	grease removed (%)	grease removed (%)	grease removed (%)	grease removed (%)
6	8	53.5	50	53.5	53
7	10	44.2	48	43.9	47.4

TABLE G6.96. The effects of **Magnafloc 336, 7.5 mg/L** and Ferric sulphate, as primary coagulant, on **floc growth (mm)**. Size is final floc size attained at 20 min flocculation.

	A	B	C	D	E
1		Coagulant, 0 mg/L	Coagulant, 25 mg/L	Coagulant, 50 mg/L	Coagulant, 75 mg/L
2	pH	Floc size, mm	Floc size, mm	Floc size, mm	Floc size, mm
3	8	0.88	0.88	1.3	1.6
4	9	0.88	0.88	1.3	1.6
5	10	0.88	0.88	1.6	1.9

TABLE G6.97 Characteristics of effluent used in Section 6. Initial pH = 8.04
Note, Total Suspended solids (mg/L) were determined only once at the initial pH.

	A	B	C	D
1	PARAMETER	pH 8	pH 9	pH 10
2	Total Suspended solids, mg/L	2300, 2275, 2280	2300, 2275, 2280	2300, 2275, 2280
3	Supernatant Suspended solids, mg/L	265, 265	240, 260	250, 250
4	Total Organic Carbon, mg/L	3180	Not determined	3200
5	Turbidity, NTU	200, 200	Not determined	200, 200
6	Capillary Suction Time, seconds	21, 21	Not determined	21, 21
7	Total Greases, mg/L	61.3	Not determined	62.7

APPENDIX H

TABLE H 7.1 Suspended solids removal and floc growth of starch wastewater using Chitosan as a primary coagulant at four different dosage levels and a pH range of 7-11.**TRIAL 1**

	A	B	C	D	E
1	Control =	780 mg/L Suspended Solids (SS) =	100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L	Chitosan, 40 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	430	243	275	295
5	8	413	398	400	373
6	9	308	190	253	235
7	10	323	318	230	183
8	11	670	660	625	603
9		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L	Chitosan, 40 mg/L
1 0	pH	% removed, SS	% removed, SS	% removed, SS	% removed, SS
1 1	7	44.9	68.8	64.7	62.2
1 2	8	47.1	49.0	48.7	52.2
1 3	9	60.5	75.6	67.6	69.9
1 4	10	58.6	59.2	70.5	76.5
1 5	11	14.1	15.4	19.9	22.7
1 6	Final floc size attained, mm, at 20 minutes flocculation				
1 7	7	0.4	0.4	0.4	0.6
1 8	8	1.3	1.3	1.3	1.3
1 9	9	0.9	0.9	3.8	3.8
2 0	10	0.4	0.6	0.9	3.8
2 1	11	0.4	0.4	0.4	0.6

TABLE H7.2 Suspended solids removed and floc growth of starch wastewater using Chitosan as a primary coagulant at three different dosage levels and a pH range of 7-10.**TRIAL 2**

	A	B	C	D
1	Control =	= 1017 mg/L Suspended Solids (SS)	= 100%	
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	588	363	420
5	8	610	613	603
6	9	433	283	410
7	10	535	398	320
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
1 0	7	42.2	64.3	58.7
1 1	8	40.0	39.8	40.7
1 2	9	57.5	72.2	59.7
1 3	10	47.4	60.9	68.5
1 4	Final floc size attained, mm, at 20 minutes flocculation			
1 5	7	0.4	0.4	0.4
1 6	8	1.3	1.9	1.9
1 7	9	0.9	0.9	3.8
1 8	10	0.6	0.6	1.3

TABLE H7.3 Suspended solids removed and floc growth of starch wastewaters using 5 mg/L Alum as a primary coagulant and Chitosan as a coagulant aid, at three different dosages and a pH range of 7-10. **TRIAL 2**

	A	B	C	D
1	Control =	= 1017 mg/L Suspended Solids (SS) =	100%	
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	463	243	263
5	8	425	340	388
6	9	310	240	190
7	10	293	315	245
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
1 0	7	54.5	76.1	74.2
1 1	8	58.2	66.6	61.9
1 2	9	69.5	76.4	81.3
1 3	10	71.2	69.0	75.9
1 4	Final floc size attained, mm, at 20 minutes flocculation			
1 5	7	0.6	0.9	0.6
1 6	8	1.3	1.9	1.9
1 7	9	1.9	1.9	2.6
1 8	10	1.9	2.6	2.6

TABLE H7.4 Suspended solids removed anf floc growth of starch wastewaters using 10 mg/L Alum as a primary coagulant and Chitosan as a coagulant aid at three different dosage levels and a pH range of 7-10 **TRIAL 2**

	A	B	C	D
1	Control =	= 1017 mg/L Suspended Solids (SS) = 100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	423	215	213
5	8	383	308	350
6	9	283	228	173
7	10	240	273	210
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
10	7	58.4	78.9	79.1
11	8	62.4	69.8	65.6
12	9	72.2	77.6	83.0
13	10	76.4	73.2	79.3
14	Final floc size attained, mm, at 20 minutes flocculation			
15	7	0.6	0.9	2.6
16	8	1.9	2.6	2.6
17	9	1.9	2.6	3.8
18	10	2.6	2.6	3.8

TABLE H7.5 Suspended solids removed and floc growth of starch wastewaters using Chitosan as a primary coagulant at three different dosage levels and a pH range of 7-10. **TRIAL 3**

	A	B	C	D
1	Control =	= 523 mg/L Suspended Solids (SS) = 100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	273	140	135
5	8	230	120	128
6	9	215	110	158
7	10	233	243	270
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
10	7	47.9	73.2	74.2
11	8	56.0	77.1	75.6
12	9	58.9	79.0	69.9
13	10	55.1	53.6	48.4
14	Final floc size attained, mm, at 20 minutes flocculation			
15	7	0.4	0.6	0.6
16	8	0.6	0.9	2.6
17	9	1.3	1.9	2.6
18	10	1.3	0.6	1.9

TABLE H7.6 Suspended solids removed and floc growth of starch wastewaters using 5 mg/l Alum as aprimary coagulant and Chitosan as a coagulant aid, at three different dosage levels and a pH range of 7-10. **TRIAL 3**

	A	B	C	D
1	Control =	= 523 mg/L Suspended Solids (SS) = 100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	218	108	118
5	8	200	163	173
6	9	138	105	98
7	10	133	140	143
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
10	7	58.4	79.4	77.5
11	8	61.8	68.9	67.0
12	9	73.7	79.9	81.4
13	10	74.7	73.2	72.8
14	Final floc size attained, mm, at 20 minutes flocculation			
15	7	0.6	0.9	0.9
16	8	1.9	1.9	1.9
17	9	1.9	1.9	2.6
18	10	2.6	2.6	2.6

TABLE H7.7 Suspended solids removed and floc growth of starch wastewaters using 10 mg/L Alum as a primary coagulant and Chitosan as a coagulant aid, at three different dosage levels and a pH range of 7-10. **TRIAL 3**

	A	B	C	D
1	Control =	= 523 mg/L Suspended Solids (SS) = 100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	205	173	178
5	8	188	120	135
6	9	130	105	90
7	10	120	140	110
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
10	7	60.8	67.0	66.1
11	8	64.2	77.1	74.2
12	9	75.1	79.9	82.8
13	10	77.1	73.2	79.0
14	Final floc size attained, mm, at 20 minutes flocculation			
15	7	0.9	0.9	2.6
16	8	1.9	2.6	2.6
17	9	1.9	2.6	3.8
18	10	2.6	3.8	3.8

TABLE H7.8 Suspended solids removed and floc growth of starch wastewaters using Chitosan as a primary coagulant at three different dosage levels and a pH range of 7-10. **TRIAL 4**

	A	B	C	D
1	Control =	= 717 mg/L Suspended Solids (SS) = 100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	413	233	270
5	8	400	373	380
6	9	325	188	258
7	10	328	315	238
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
10	7	42.5	67.6	62.3
11	8	44.2	48.0	47.0
12	9	54.7	73.8	64.1
13	10	54.3	56.1	66.9
14	Final floc size attained, mm, at 20 minutes flocculation			
15	7	0.4	0.4	0.6
16	8	1.3	1.3	1.9
17	9	1.3	1.3	3.8
18	10	0.9	1.3	2.6

TABLE H7.9 Suspended solids removed and floc growth of starch wastewaters using 5 mg/L Alum as a primary coagulant and Chitosan as a coagulant aid, at three different dosage levels and a pH range of 7-10. **TRIAL 4**

	A	B	C	D
1	Control =	= 717 mg/L Suspended Solids (SS) = 100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	335	163	183
5	8	310	255	268
6	9	230	165	148
7	10	220	228	170
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
10	7	53.3	77.3	74.5
11	8	56.8	64.4	62.7
12	9	67.9	77.0	79.4
13	10	69.3	68.3	76.3
14	Final floc size attained, mm, at 20 minutes flocculation			
15	7	0.6	0.9	0.6
16	8	2.6	1.9	2.6
17	9	1.9	1.9	2.6
18	10	2.6	1.9	2.6

TABLE H7.10 Suspended solids removed and floc growth of starch wastewaters using 10 mg/L Alum as a primary coagulant and Chitosan as a coagulant aid, at three different dosage levels and a pH range of 7-10. **TRIAL 4**

	A	B	C	D
1	Control =	= 717 mg/L Suspended Solids (SS) = 100%		
2		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
3	pH	SS remaining, mg/L	SS remaining, mg/L	SS remaining, mg/L
4	7	313	145	158
5	8	288	238	250
6	9	205	158	133
7	10	195	210	158
8		Chitosan, 5 mg/L	Chitosan, 10 mg/L	Chitosan, 20 mg/L
9	pH	% removed, SS	% removed, SS	% removed, SS
10	7	56.4	79.8	78.0
11	8	59.9	66.9	65.1
12	9	71.4	78.0	81.5
13	10	72.8	70.7	78.0
14	Final floc size attained, mm, at 20 minutes flocculation			
15	7	0.6	0.9	2.6
16	8	1.9	2.6	2.6
17	9	2.6	2.6	3.8
18	10	3.8	3.8	3.8

TABLE H.11. Raw and supernatant data for all four trials which evaluated Chitosan as a primary coagulant and as a coagulant aid.

	A	B	C	D	E
1	PARAMETER	TRIAL 1	TRIAL 2	TRIAL 3	TRIAL 4
2	Total Suspended Solids (mg/L)	2250, 2240, 2270	2350, 2295, 2305	2150, 2050, 2010	1750, 1760, 1740
3	Average	2253	2317	2070	1750
4	Control Suspended Solids (mg/L)	780, 780, 780	1000, 1000, 1050	555, 495, 520	710, 700, 740
5	Average	780	1017	523	717
6	pH	8.09	8.34	7.67	8.46

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