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Effect of Chemical Coagulation on the Removal of Fecal Coliforms from a Seasonal Discharge Sewage Lagoon

by

© Gordon R. Finch

A THESIS

SUBMITTED TO THE FACULTY OF GRADUATE STUDIES AND RESEARCH

IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE DEGREE

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ABSTRACT

Lagoons have found widespread application for treating wastewater generated by small communities in all climates. They are economical to construct and operate and usually perform to secondary effluent standards during the warm summer months. Winter conditions of ice and snow cover coupled with near 0° C water temperatures greatly reduce biological activity and natural disinfection. Controlled discharge lagoons store wastewater for periods of 120 to 365 days to prevent overloading the assimilation capacity of receiving waters during low flow periods. Fall discharges have a high degree of treatment, whereas spring releases meet essentially primary treatment standards. Spring discharges contain large numbers of indicator bacteria and consequently, are likely to contain pathogenic microorganisms. The high degree of persistence of many pathogens in cold water raises concerns about public health for subsequent water users.

Chlorine disinfection has been used in the past but it has many drawbacks. An alternative to disinfection is coagulation and flocculation of wastewater to remove microorganisms in addition to biochemical oxygen demand, suspended solids, and phosphorus.

This study examined several different factors influencing coagulation and flocculation of fecal coliform bacteria. The experimental program consisted of a series of

jar tests based on factorial designs and statistical analysis. These were later scaled-up to a larger vessel to ascertain scale effects. The research culminated in an application of alum to an operating seasonal discharge facility at Gibbons, Alberta. The results indicated that alum dose and final pH were the most significant parameters to achieve high levels of fecal coliform removals. In addition, good reductions in phosphorus, suspended solids, and biochemical oxygen demand were obtained. No scale-up factors were found and the results of the full scale application corresponded to the jar test results.

The treatment operating costs were approximately $ 0.07 per cubic metre of lagoon.

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1. INTRODUCTION

1.1 PROBLEM DEFINITION

Properly designed and operated waste stabilization ponds, also known as sewage lagoons, can produce effluents of sufficient organic, biological and chemical quality to meet secondary treatment standards. The climate in which a lagoon facility is operated will greatly affect the performance of the system; tropical and temperate regions have ideal conditions for lagoon employment. Cold climates cause lagoon effluent quality to deteriorate to the level of primary effluent during the winter months. This unsatisfactory situation has led to the use of controlled-discharge wastewater treatment facilities as an upgrading measure to protect the receiving environment.

This operating procedure has traditionally used two seasons, spring and autumn, for discharges. The former provides snowmelt water for additional dilution while the latter allows high levels of stabilization and natural disinfection to occur over the warm, sunny, summer months.

There are some drawbacks to an intermittent discharge system. The fall discharge has a high quality effluent provided algae are absent. However, since nutrients, sunlight, and the aerobic oxidation of organic matter create a large mass of algae, effluents often contain unacceptable amounts of suspended solids. The subsequent decomposition of the algae biomass in the receiving environment exerts a heavy oxygen demand and contributes to eutrophication of receiving waters. A variety of techniques have been used to remove algae including mechanical harvesting, chemical coagulation, various filter methods, flotation, and chemical oxidation.

The spring discharge has more complex problems. The algae population is usually low immediately following spring break-up but blooms of *Chlamydomonas* and *Euglena* occur quickly. These spring blooms pose similar problems to a fall discharge system. In addition, indicator microorganisms are highly persistent in cold water environments leading to effluent discharges containing large populations of these organisms. It is implied that large numbers of pathogens may also be released creating potential public health problems downstream.

Other public health ramifications involve the presence of transferable R factors (plasmids which confer hereditary resistance to common antibiotics, ultraviolet radiation, heavy metals, etc.) in bacterial populations. The common "harmless" flora of the intestinal tract are a reservoir of R factors and have been shown to transfer these to enteropathogenic organisms. Lagoons are an ideal environment for R^+ organisms to transfer drug resistance. Therefore it is desirable to limit the number of bacteria released into the receiving environment.

Disinfection is an obvious unit process to employ in upgrading biological water quality, yet it is controversial.

Objections originate from the use of chlorine which is effective in killing indicator organisms, however its efficacy in viral inactivation and protozoan cyst destruction is questionable. In addition, chlorine residuals as low as 0.10 mg/L are toxic to natural aquatic fauna, particularly fish. Recent concerns about carcinogenic chlorinated organic compounds such as chloroform and other trihalomethanes also weigh against the use of chlorine.

Ozone has been used for years in Europe as an alternative to chlorine but still accounts for a minor fraction of wastewater disinfection in North America because of its significantly higher cost compared to chlorine.

The water treatment industry has used coagulation, flocculation, and sedimentation to remove colloids (organic and inorganic), biocolloids (viruses, bacteria, protozoa), colour, and taste.

The principal reasons for upgrading lagoons are the inability of summer effluents to meet standards for biochemical oxygen demand and suspended solids because of algae, more stringent discharge requirements, and poor system performance as a result of overloading. Often the planning, design, and construction of new facilities takes months, in some cases years, to be implemented. Therefore interim upgrading is necessary. In the case of controlled discharge lagoons, coagulation and flocculation have proved to be cost effective in obtaining high reductions in biochemical oxygen demand, suspended solids, and phosphorus.

Microorganism removal has not been studied in detail. The purpose of this study is to explore the effectiveness of chemical coagulation and flocculation for removing indicator organisms prior to the discharge of a cold climate, seasonal retention lagoon.

1.2 OBJECTIVES

This study had four objectives:

1. to determine the optimum physical and chemical conditions for biocolloid removal through the use of factorial experimental designs;
2. to identify process scale-up and scale-down relationships between a full size facility and laboratory test vessels;
3. to complete a full scale test using simple, readily available equipment; and
4. to estimate the cost effectiveness of this technique for lagoon upgrading relative to other methods.

1.3 SCOPE

The main thrust of the study was to complete a full scale test of coagulation in a seasonal discharge lagoon. To achieve that goal, a literature survey of relevant material was undertaken to review the theory, design, and performance of lagoons; to identify the mechanisms of biocolloid stability and destabilization; to examine past expe[illegible]e in coagulating lagoon wastewater; to determine approp[illegible] experimental conditions; and to review past experience in scale-up of wastewater coagulation and flocculation facilities.

The research consisted of three segments:

1. bench scale experiments were performed using factorial designs to determine the optimum chemical and physical conditions for removal of fecal coliform bacteria by coagulation and flocculation;
2. scaled-up tests were conducted to identify factors which may affect the response of the full scale facility; and
3. coagulant was applied to an operating, full scale facility.

2. LITERATURE REVIEW

2.1 LAGOONS

Sewage lagoons are used extensively in Canada, accounting for nearly half of all wastewater treatment facilities (Table 1). There are several reasons for this (Gloyna 1971; Middlebrooks et al. 1982; Smith and Christensen 1982):

- reduced capital costs;
- low maintenance levels;
- minimal energy requirements;
- skilled operators are not required;
- availability of inexpensive land; and
- small populations are adequately served.

2.1.1 Theory of Lagoon Treatment

A waste stabilization pond or lagoon is defined as a shallow earthen basin designed for the biological stabiliztion of wastewater by the use of algae and bacteria (Metcalf and Eddy, Inc. 1979). Lagoons have four basic formats: aerobic (high-rate), facultative (long retention), anaerobic (short retention), and mechanically aerated.

Facultative lagoons (Figure 1) are the most commonly used in Canada and are the subject of this study (Tilsworth and Smith 1982). The water column of a facultative lagoon has three zones: aerobic, anaerobic, and sludge. The aerobic zone is characterized by aerobic heterotrophic and

Table 1 - Lagoon Use in Canada (1980) (Alberta Environment 1980; Canadian Governments 1981)

Location	Number of Lagoons	As a % of Treatment Facilities
Alberta	278	84
British Columbia	34	31
Manitoba	127	85
New Brunswick	58	63
Newfoundland	1	2
Northwest Territories	15	71
Nova Scotia	14	16
Ontario	128	33
Prince Edward Island	17	9
Quebec	59	21
Saskatchewan	129	92
Yukon Territory	8	70
Canada	868	48

Figure 1 - <u>A Schematic Facultative Lagoon</u>

This page contained copyright material from Oswald (1968). It described a cross-section of a facultative lagoon.

autotrophic bacteria which form a symbiotic relationship with algae (Metcalf and Eddy, Inc. 1979). This arrangement arises from the bacterial production of carbon dioxide as a metabolic end-product which algae use as a carbon source to produce algal cells and oxygen under conditions of sufficient light (Gloyna 1971). Photosynthetic oxygen is used by aerobic organisms to continue oxidizing organic matter (Gaudy and Gaudy 1980).

McKinney (1962) described the bacteria genera present in the aerobic zone as similar to those of activated sludge. Pseudonomads predominate due to their ability to use various substrates over a wide range of temperatures (Brock 1979; Henry 1974). Aerobic, biochemical reactions are complex, however simplified conceptual equations were presented by McKinney (1982) and Oswald (1968).

Cyanobacteria (formerly known as blue-green algae) are a special group of photoautotrophs found in lagoons. Periodic, massive blooms can hamper the performance of a lagoon by limiting light penetration (Gloyna 1971). Gloyna (1971) noted the aerobic zone was dominated by the algae *Euglena*, *Chlorella*, *Scenedesmus* and *Chlamydomonas*. He also reported that *Euglena* and *Chlamydomonas* were the hardiest, being the first to recover from anoxic events and remaining active at psychrotrophic temperatures.

The anaerobic zone may be conceived of as a single-stage, anaerobic digestor (Pfeffer 1970). The main difference lies in the lack of control of e vironmental factors. Transformation of organic matter is carried out by obligate anaerobes of the so-called acid forming and methanogenic bacteria (Grady and Lim 1980). Anaerobic, biochemical transformations were described by Oswald (1968).

Two important phenomena of an algal based wastewater treatment system are the diurnal variation of pH and dissolved oxygen (Gloyna 1971). Stumm and Morgan (1981) discussed the effects of aerobic respiration, photosynthesis, and various oxidation-reduction reactions on the alkalinity and pH of an aqueous system. Qualitative effects are summarized in Table 2. They also pointed out that evolution or consumption of carbon dioxide in an aqueous system does not change the alkalinity since no protons are involved. However, when pH exceeds 10, alkalinity will be removed in the form of $CaCO_3$ precipitate.

Lagoon environs are hostile to many enteric organisms, both pathogens and non-pathogens. Mechanisms attributed to natural disinfection include sedimentation, sunlight, adsorption, natural predation, nutrient deficiencies, phages, biotoxins, algae, pH, and specific ion toxicity.

Mitchell and Chamberlin (1978) found that sedimentation and sunlight were significant in the removal of indicator bacteria. Algal toxicity to bacteria was related to sunlight. They discounted any contribution from biotoxins or phages. Parhad and Rao (1974) found *Escherichia coli* to die faster in the presence of algae and at a pH greater than 10. Others reported that algae have no significance in the

Table 2 - Processes Affecting Alkalinity (after Stumm and Morgan 1981)

Process	Alkalinity Change
Aerobic Respiration (no decomposition of NO_3^{2-})	No change
Photosynthesis (no assimilation of NH_4^+)	No change
Nitrification	Decrease
Denitrification	Increase
Sulphide Oxidation	Decrease
Sulphate Reduction	Increase

removal of viruses or bacteria (Davis and Gloyna 1972; Malherbe and Strickland-Cholmley 1966). McGarry and Bouthillier (1966) found the rate of *Salmonella* die-off was related to the organic loading.

Virus removal has been attributed to adsorption of viruses by solids and subsequent sedimentation (Gerba 1981; Sproul 1980).

Bacterial removal kinetics are assumed to be first-order and various authors have applied chemical reactor engineering to model lagoon efficiency (Bowles et al. 1979; Marais 1974). A consistent problem has been the evaluation of the rate constant, k. Polprasert et al. (1983) used a regression technique to evaluate k. The parameters which were incorporated into the k model included organic loading, temperature, bacterium species, algae concentration, and ultraviolet light. They found the main effects were temperature, organic loading, and algae concentration.

Sedimentation is an important physical process in lagoon treatment. Four types of sedimentation have been classified (Fair et al. 1968; Weber 1972):

1. Type I, or discrete particle settling is described by Stoke's Law under laminar conditions. In turbulent flow conditions, Newton's law applies. This type of sedimentation deposits grit and large organic particles in the vicinity of the lagoon inlet where the flow velocity decreases to essentially zero (Middlebrooks et

al. 1982).

2. Type II, or flocculent settling is an important mechanism of suspended solids removal in a sewage lagoon. The dispersed solids of various physical and chemical characteristics gradually flocculate as agglomeration occurs and faster particles overtake slower ones. Type II sedimentation is strongly dependent upon depth and residence time for maximum clarification efficiency (Weber 1972).

3. Type III, or hindered settling is usually found at the bottom of the water column where the suspended solids concentrations are much higher. It is a transition zone between flocculent sedimentation and compression of the sludge zone (Weber 1972).

4. Type IV, or sludge compression is a very slow process which occurs as a result of sludge consolidation as solids accumulate. Middlebrooks et al. (1982) noted that thick sludge zones were anaerobic and gelatinous in nature, being difficult to dewater as a consequence.

Weber (1972) noted that sedimentation typically removed 50 to 60 per cent of suspended solids and 30 to 40 per cent of BOD_5 (5 day biochemical oxygen demand). Feachem et al. (198[illegible]) noted in their review of the effect of various processes on pathogen removal, that 50 per cent of viruses were removed in 3 to 6 hours of sedimentation time through adsorption to larger particles. Bacteria removals were in the order of 50 to 90 per cent. They presented theoretical settling velocities of some pathogens and found only one or two types of helminth and protozoan cysts could be settled.

The hydraulics of a lagoon have generally been assumed to follow either completely mixed flow (CMF) or plug flow (PF) regimes (Middlebrooks et al. 1982).

The kinetics of PF and CMF reactors were described in detail by Grady and Lim (1980). Middlebrooks et al. (1982) provided a very complete and detailed discussion of lagoon hydraulics and various approaches to modelling. Non-ideal flow models have been developed for chemical reactors by Wehner and Wilhelm (1956) and applied to lagoons by Thirumurthi (1969) since ideal CMF or PF conditions rarely occur in lagoons.

All of the foregoing models were deterministic and dealt with continuous flow regimes. Saunders and Minchew (1981) developed a stochastic model for a controlled discharge facility. However the complexity of the model precludes its use on a routine basis.

2.1.2 Factors Affecting Performance

Temperature is the single most important factor affecting lagoon performance (Eckenfelder and Englande 1970). Biological and chemical processes within lagoons are assumed to follow first-order kinetics of the form:

$$r = -k\ X_0$$

where;

r = reaction rate (mass/volume.time)

X_0= initial concentration of substance (mass/volume)

k = reaction rate constant (1/time)

Temperature effects are related to k by the simplified van't Hoff-Arrhenius equation (Metcalf and Eddy, Inc. 1979):

$$k_t = k_{20}\theta^{t-20}$$

where, k_t is the rate constant at any temperature above 4° C and k_{20} is the rate constant determined at 20° C. The coefficient, θ, relates the temperature dependence of mechanisms involved in the process under consideration (Eckenfelder and Englande 1970).

The use of this equation is questionable since: at temperature extremes relative to the reference temperature, θ is very inaccurate; and the microbial population at 20° C is predominatly mesophilic whereas at 0° to 10° C psychrophiles predominate (Gordon 1970; Henry 1974).

Schneiter and Grenney (1983) presented a mathematical model which was purported to overcome the deficiencies of the van't Hoff-Arrhenius approach. This model was recommended for cold water applications.

Temperature greatly influences the properties of water, in particular, density. Cooling and heating of the water surface past the point of maximum density (3.96° C) causes a turnover of the water column (Fair et al. 1968). In general, two turnovers occur per year, one in the spring and one in the autumn. This phenomenon is responsible in part for erratic suspended solids and coliform removal performance (Middlebrooks et al. 1982). The rate of clarification is dependent upon the viscosity and density of water and is inhibited at cold temperatures (Smith et al. 1979)

Other factors which affect the performance of lagoons include wind, thermal stratification, hydraulic short-circuiting, depth, sunlight, water balance, organic loading, pH, ionic strength, toxins, dissolved oxygen, sludge accumulation, flow control devices, and configuration (Gloyna 1971; Middlebrooks et al. 1982; Schneiter 1982).

2.1.3 Process Design

Four approaches are used to design sewage lagoons: retention time, organic loading, empirical models, and analytical models. Middlebrooks et al. (1982) provided an excellent summary of design approaches and examined them in light of reported performances.

McKinney (1982) stated that retention time was the main design criterion for lagoons operated in cold regions. He recommended times ranging from 120 to 150 days. Dawson and Grainge (1969) recommmended 180 to 365 day retention for lagoons in the arctic and sub-arctic. Alberta Environment (1977) required 365 day retention in its quidelines for designing lagoons in new developments.

Organic loading is the oldest criterion for sewage lagoon design. Canter et al. (1969) summarized typical loading rates across the United States, demonstrating a trend towards reduced loading and longer retention times at higher latitudes. They recommended 22 kilograms BOD_5 per hectare per day as a maximum for regions with seasonal ice cover. Dawson and Grainge (1969) supported this value as a criterion for northern Canada. This approach has some problems in cold climates where lagoons have failed to meet effluent standards under continuous flow conditions (Finney and Middlebrooks 1980). The United States EPA (1983) design manual found areal organic loading to be as good as any rational approach for sizing lagoons. It recommended a range of 11 to 22 kilograms BOD_5 per hectare per day depending on the severity of the winter.

Three empirical models for BOD_5 removal in a continuous discharge system have been discussed in the literature. These were:

1. the McGarry and Pescod regression equation (McGarry and Pescod 1970);
2. the Larsen equation (Middlebrooks et al. 1982); and
3. the Gloyna method (Gloyna 1976).

Finney and Middlebrooks (1980) found these to be of little use for designing cold climate lagoons.

Analytical or rational designs have been reported in the literature since the mid-1960's, the major contributors being Marais (1970; 1974) Oswald et al. (1963; 1970) and Thirumurthi (1969). Saunders and Minchew (1981) investigated a model for controlled discharge lagoons which was based on an extension of the model presented by Ferrara and Harleman (1981). It was of use in research and detailed modelling of these lagoons but finds little application on a routine basis.

2.1.4 Microorganism Considerations

2.1.4.1 Lagoon Performance

Performance of lagoons in North America has been studied by many authors over the years but there are few comprehensive reports. Of interest are the studies by Neel et al. (1961), Alberta Department of Public Health (1961), Higo (1965), Miyamoto (1972), and Reynolds et al. (1977). Middlebrooks et al. (1982) published a monograph which compiled performance data for a series of lagoons in the United States which experienced seasonal ice cover. Figure 2 is from this book and summarizes typical effluent values for BOD_5, suspended solids, and fecal coliforms. The Peterborough, New Hampshire lagoon was the only system that disinfected the effluent. These performance curves are typical of other reports in the literature: reduced BOD_5 removals during the winter (Fisher et al. 1968); increased suspended solids during the spring and fall turnovers (Stahl and May 1967); the effects of summer algal blooms on suspended solids (Barsom 1973); and the increased persistence of microorganisms during the winter months (Slanetz et al. 1970).

Feachem et al. (1983) reviewed lagoon performance in terms of pathogen and indicator bacteria removals. They felt that properly designed and operated lagoons could achieve essentially 100 per cent removals without disinfection.

Figure 2 - Seasonal Performance Variation

Pages 19 & 20

This page contained copyright material from Middlebrooks et al. (1982). The curves described typical performance variation in lagoons that suffer seasonal ice cover.

Airaksinen (1978) noted that poor performance of northern lagoons was a result of the fact that many northern lagoons were copies of southern ones and did not account for the effects of cold weather. This fact combined with design for BOD_5 removal rather than microorganism removal, has led to poor microorganism removal during the winter months.

2.1.4.2 Persistence of Microorganisms

Persistence was reported as a function of temperature and time (Feachem et al. 1981; Sattar 1981). Smith and Gerard (1981) speculated that cold climate lagoons allowed microorganisms to escape thermal shock by gradually acclimatizing in the lagoon environment. Slanetz et al. (1970) carried out a study on bacteria and virus survival in a lagoon system in New Hampshire reporting higher persistence during the winter months.

Davenport et al. (1976) reported indicator persistence 2.7 to 5.4 times longer in arctic rivers than previously reported for southern winter conditions. Dahling and Safferman (1979) found enteric viruses had a survival rate of 34 per cent over a 320 kilometre reach of the Tanana River in Alaska. This supported earlier findings by Prier and Riley (1966) who found that Echovirus, Coxackievirus, and Poliovirus were stable in 4° C river and impoundment waters. Katzenelson (1978) stated that viable viruses can be stored indefinitely at deep-freeze temperatures and expected long survival times at 0° to 5° C. Clarke et al. (1962) described results which suggested viruses may remain viable for greater than 200 days at temperatures less than 4° C. They also found that Poliovirus survived three times longer than *E. coli* and *Streptococcus fecalis*. Bitton (1978) reported virus survival of 40 to 90 days at 3° to 5° C under laboratory conditions.

The persistence of microbes in cold water environments, both wastewaters and receiving waters, raises some concern about the potential public health risks to subsequent water users.

2.1.4.3 Public Health

Wastewater is an important source of enteric pathogens. Commonly excreted pathogens and related diseases are summarized in Table 3.

Hrudey and Raniga (1981) summarized selected Canadian health statistics which indicated that incidences of infectious hepatitis, shigellosis, and salmonellosis were above the national average in Alberta and the Northwest Territories. Gastroenteritis was the most common complaint. Yamamoto (1975) demonstrated that large numbers of viruses were endemic in pond effluents in the Northwest Territories and Alberta. Viruses are not enumerated regularly in lagoon effluents since the procedure is difficult and often unreliable (WHO 1979). Studies have shown increased numbers of viruses in sewage during late summer and autumn which raises concern about their fate over the subsequent winter months (Feachem et al. 1983).

Table 3 - Some Commonly Excreted Pathogens (after Feachem et al. 1983)

Group	Agent	Disease
Viruses	Coxsackie virus	Various ailments
	Hepatitis A virus	Infectious hepatitis
	Norwalk agent	Gastroenteritis
	Poliovirus	Poliomyelitis
	Rotavirus	Gastroenteritis
Bacteria	*Campylobacter fetus*	Enteritis
	Escherichia coli	Gastroenteritis
	Leptospira spp.	Leptospirosis
	Salmonella typhi	Typhoid fever
	S. paratyphi	Paratyphoid fever
	Shigella spp.	Bacillary dysentry
	Vibrio cholerae	Cholera
	Yersinia enterocolitica	Yersiniosis
Protozoa	*Entamoeba histolytica*	Amebic dysentry
	Giardia lamblia	Giardiasis
Helminths	*Ascaris lumbricoides*	Ascariosis
	Diphyllobothrium latum	Fish tapeworm

2.1.4.4 Indicator Organisms

Indicator organisms came about because of the difficulty of isolating and identifying various waterborne pathogens from the extensive flora of the human intestinal tract.

An ideal indicator organism should be (Feachem et al. 1981):

1. present only in the intestinal flora of healthy people;
2. absent from non-human animals;
3. present only when pathogens are present;
4. present in larger numbers than pathogens;
5. unable to grow outside of the intestine;
6. dying at the same rate as pathogens or slightly slower;
7. resistant to waste treatment processes to a greater degree than pathogens;
8. easy to detect and enumerate; and
9. non-pathogenic.

There is no indicator which satisfies these criteria.

Geldreich (1978) summarized the flora of human feces of which a few groups were found in large numbers. These included coliforms, fecal coliforms, *Escherichia coli*, fecal streptococci, and enterococci. Olivieri (1982) noted the coliform group had some shortcomings which included:

1. die-off rates which were significantly greater than viruses and protozoa;
2. higher sensitivity to treatment processes than pathogens;

3. aftergrowth potential; and
4. the ability to compete successfully in the environment, altering the ratio between coliforms and pathogens unpredictably.

Despite these deficiencies, no other group satisfies the criteria for indicator organisms as completely as the coliform group.

Splitting the coliform group into total coliforms and fecal coliforms is useful since total coliforms include some soil bacteria which are ubiquitous whereas fecal coliforms are comprised of genera from Enterobacteriaceae which are exclusively of fecal origin (Feachem et al. 1983).

A serious shortfall of indicator bacteria is their inability to exclusively indicate the presence of viruses (WHO 1979). Cabelli (1981) produced epidemiological evidence that Rotavirus and Norwalk agent were responsible for gastroenteritis in swimmmers. He found that both total coliforms and fecal coliforms were inadequate indicators. In another study, Cabelli et al. (1982) found enterococci to have the best correlation with incidences of gastroenteritis in swimmers. They also reported less than 10 *E. coli* per 100 mL were present.

The use of phages – viruses which attack bacteria – has been advocated as viral indicators but published reports on methods of detection and sensitivity for use on a routine basis are rare (Kraus 1977).

2.1.4.5 Bacteria and R Factors

Drug resistance results from a group of plasmids – extrachromosomal genetic structures composed of DNA – called resistance transfer factors (R factors) which are able to travel by cell to cell contact (Brock 1979). They are responsible for acquired resistance to antibiotics, toxins, heavy metals, ultraviolet light, bacteriocins, and phages. Grabow et al. (1974) reviewed the problem in depth and noted that Enterobacteriaceae act as a reservoir for R factors and will transfer them amongst other members of the group including pathogens. They suggested that water quality standards should be reviewed in light of R factors.

Bell et al. (1980; 1981) investigated the occurrence of R factors in the sewage flora of northern Canada and the Prairie Provinces. The remote northern Slave River had low incidences of antibiotic resistant fecal coliforms; 7.1 per cent having multiple drug resistance and 0.79 per cent with tranmissible drug resistance. This differed markedly from the Red River in Manitoba where Winnipeg waste discharges were a major source of drug resistant fecal coliforms; 52.9 per cent having multiple drug resistance and 18.77 per cent with transferable drug resistance. Grabow et al. (1975) warned of lakes, rivers, and impoundments acting as reservoirs for R^+ organisms – those with transferable R factors.

Grabow et al. (1973) found 1.8×10^4 total coliforms per 100 mL had transferable drug resistance in a lagoon in

South Africa. Kish and Lampky (1983) studied a long retention, seasonal discharge lagoon in Michigan and found total coliforms persisted over the winter months with numbers peaking at 10^7 per 100 mL during the spring turnover. Relative numbers of drug resistant organisms increased through the three pond system indicating a resistance to the elements of natural disinfection. Greater than 60 per cent of effluent total coliforms had drug resistance. Bell et al. (1983) recently reported the survival incidence of R^+ fecal coliforms in a continuous discharge, long retention lagoon system at Fort Smith, Northwest Territories. They found the proportion of R^+ organisms increased over the winter. The effluent fecal coliform population contained 58 per cent with resistance to three or more drugs and 4.02 per cent with transferable drug resistance.

The incidence of R^+ coliform organisms play an increasing role in the outbreaks of drug resistant bacterial disease, therefore more attention should be given to the bacteriological quality of wastewater discharges (Grabow et al. 1974).

2.1.5 Upgrading Lagoon Effluents

Methods for upgrading effluents include various filter techniques, dissolved air flotation, land application, coagulation, flocculation, and disinfection (Johnson et al. 1979; Middlebrooks et al. 1978). Middlebrooks et al. (1982)

provided a review of upgrading techniques noting that suspended solids comprised the majority of the problem. Lagoons in regions with seasonal ice cover experience reduced treatment and usually produce primary quality effluent containing unsatisfactory amounts of BOD_5, suspended solids, and indicator bacteria.

Middlebrooks et al. (1982) considered controlled discharge facilities as an upgrading technique. Pierce (1974) reported on the effluent quality of intermittent discharge lagoons in Michigan. Chlorination maintained fecal coliforms to levels less than 200 per 100 mL. Caldwell (1946) discussed chlorine disinfection as a batch treatment of the entire lagoon, though the most common approach today is effluent disinfection. Problems of predicting chlorine demand and maintaining correct doses have made chlorine disinfection ineffective in many cases, killing indicator bacteria but allowing viruses and R^+ bacteria to be discharged (Feachem et al. 1983).

Residual chlorine toxicity and formation of carcinogenic compounds are also serious drawbacks to chlorination, (Metcalf and Eddy, Inc. 1979; Sawyer and McCarty 1978; Venosa 1983) but the health risks are considered significant without disinfection (Haas 1983; Smith and Bell 1981).

Al-Layla and Middlebrooks (1975), McGarry (1970), Tenney et al. (1969), and others studied coagulation of lagoon effluents for algae removal with positive results.

Middlebrooks et al. (1982) felt that coagulation and flocculation of controlled discharge facilities was very cost effective in removing COD (chemical oxygen demand), BOD_5, suspended solids, and phosphorus. The efficacy of microorganism removal by chemical coagulation in a lagoon has not been researched in depth though incidental reductions during phosphorus treatment were reported by some investigators (Graham and Hunsinger 1974; Shindala and Stewart 1971).

2.2 COAGULATION AND FLOCCULATION

The terms coagulation and flocculation are used in many ways by many authors. Bratby (1980) defined coagulation as the destabilization process and flocculation as the transport and agglomeration step. This definition is used here.

Wastewater contains particles composed of organic, inorganic, or biological solids ranging in size from 0.01 to 100 microns (Stumm 1977). Biocolloids include viruses, bacteria, protozoa, and helminth eggs (Tenney and Stumm 1965). Algae range in size from 5 to 50 microns (Ives 1959). Bacteria occur in many different shapes with overall dimensions from 0.1 to 20 microns. Enteroviruses are the smallest, ranging from 0.02 to 0.03 microns (Feachem et al. 1983). Figure 3 illustrates the relative size of biocolloids and various filter materials.

Figure 3 - Relative Dimensions of Colloids

This page contained copyright material from Stumm and Morgan (1981). The figure described the relative sizes of various colloids and filter materials.

Colloid chemists categorize particles as either lyophobic or lyophilic; the latter representing colloids with an affinity for solvent and the former by a lack of this characteristic (Verwey and Overbeek 1948). In the case of water, these are called hydrophobic and hydrophilic colloids respectively.

Colloidal dispersions are termed stable if they tend to remain dispersed indefinitely (Bratby 1980).

2.2.1 Origin of Colloid Stability

Verwey and Overbeek (1948) presented the earliest reliable theory of hydrophobic colloid stability based on the electric double layer concept. The electric double layer results when a charged surface (usually negative) attracts ions of an opposite charge (counterions) to create a second layer (Stumm and Morgan 1981). Stumm and Morgan (1981) presented various electric double layer concepts which are illustrated in Figure 4.

Verwey and Overbeek (1948) demonstrated the inadequacy of the Gouy-Chapman model and preferred the Stern concept since it accounted for specifically adsorbed ions and the hydrated radius of the ions. Lyklema (1978) noted that over-adsorption of counterions led to charge reversal. The point at which the surface charge, σ_0, became zero was called the isoelectric point.

Verwey and Overbeek (1948) analysed the response of hydrophobic colloid pairs in terms of interaction energy, V.

Figure 4 - Electric Double Layer Concepts

This page contained copyright material from Stumm and Morgan (1981). The figure illustrated the three concepts of the electric double layer.

They reasoned that electrostatic repulsion, V^-, interacted with London-van der Waals, V^+, producing the net interaction energy:

$$V = V^- + V^+$$

The repulsive energy was a function of the double layer thickness and the ionic strength of the solution, hence the surface potential, Ψ_0. This has been difficult to measure therefore the zeta potential, ζ, was assumed to be equal to Ψ_0. Overbeek (1977) noted that London-van der Waals forces had an extremely short range and varied inversely to the sixth power of the interparticle distance.

A graphic illustration of interaction energies is presented in Figure 5. Increased ionic strength reduces the double layer thickness which reduces the energy "hump". This model was found to explain the stability of hydrophobic colloids by charge effects alone. Stumm and Morgan (1981), Lyklema (1978), and Bratby (1980) provided excellent summaries of Verwey and Overbeek's work.

The nature of hydrophilic stability is a combination of chemical and physical factors (Stumm and Morgan 1981). Harris and Mitchell (1973) reviewed microbial aggregation and noted that biocolloids were stable due to solvation of cell wall functional groups. O'Melia (1972) observed that solvation effects were largely unknown. Various authors concluded that charge effects were of secondary importance in hydrophilic stability and that chemical interactions between the solvent and the surface chemical groups were the

Figure 5 - Interaction of a Colloidal Pair

This page contained copyright material from Benefield et al. (1982). The figure illustrated the interaction of attractive and repulsive energies at various ionic strengths.

most important (Stumm and Morgan 1962; Tenney and Stumm 1965).

2.2.2 Origin of Surface Charge

Colloidal particles carry a surface charge, often negative, resulting from a number of chemical interactions. Some of the principal ones were described by Stumm and Morgan (1981):

- ionization of ionogenic surface groups;
- specific ion adsorption;
- isomorphic substitution in the crystal lattice of inorganic compounds; and
- preferential adsorption of surfactant ions.

Bacteria have surface charges caused by the protolysis of the carboxyl and amino groups (Figure 6). The value of the charge is highly pH dependent, however for the values found in wastewater, it is usually negative (Stumm and Morgan 1981).

2.2.3 Destabilizing Colloids

Five principal mechanisms have been recognized in destabilizing colloids (Gregory 1978b; O'Melia 1972):

1. double layer compression;
2. charge neutralization;
3. enmeshment in an insoluble metal hydroxide precipitate;
4. polymer bridging; and
5. electrostatic patching.

Figure 6 - Protolysis of a Bacterium

This page contained copyright material from Stumm and Morgan (1981). The figure illustrated the origin of the surface charge on a bacterium.

Double layer compression occurs in the rare case where a dilute, hydrophobic sol is stable from charge effects alone, therefore destabilization is obtained through the addition of an indifferent electrolyte (Van Olphen 1963). The empirical Schulze-Hardy rule (Sawyer and McCarty 1978), corroborated by the theory of Verwey and Overbeek (1948), predicted that the optimum concentration of indifferent electrolyte was a function of z^{-6}, where z is the valence of the counterion, therefore the effectiveness of trivalent, divalent, and monovalent ions varies in the proportion of 3^{-6}: 2^{-6}: 1^{-6} or approximately 1.4: 15.6: 1000.

Hydrophilic dispersions are not directly affected by this mechanism, though some double layer compression occurs (Bratby 1980; Stumm and Morgan 1962). Since Al^{3+} and Fe^{3+} ions are never present in large quantities but occur as charged, metal-hydroxo complexes, double layer compression is not significant when coagulating with trivalent metal salts. Charge neutralization through adsorption of charged, polynuclear, metal-hydroxo complexes is the most important mechanism (Gregory 1978a; Stumm and Morgan 1962).

The hydrated metal species are highly pH dependent following a simple, step-wise reaction series. This was summarized by Stumm and Morgan (1962). Most chemists recognized that these simple reactions did not represent the actual picture and that polynuclear metal complexes were present (AWWA 1971; Gregory 1978a). Figure 7 illustrates the equilibrium of aqueous iron and aluminum systems.

Figure 7 - Equilibrium of Hydrolysed Al^{3+} and Fe^{3+}

This page contained copyright material from Stumm and O'Melia (1968). This figure was an equilibrium diagram of soluble and insoluble species of iron and aluminum.

Stumm and O'Melia (1968) reported that adsorption of hydrolysis products by hydrophobic colloids followed a modified Langmuir isotherm. Gregory (1978a) noted that the hydrolysis products at higher pH levels had a greater affinity for adsorption than those at lower pH values. Adsorption of the charged species leads to charge neutralization and double layer compression. Continued favourable adsorption produces charge reversal and restabilization (Bratby 1980).

Competition for the metal cations by other complexing anions, particularly sulphate and phosphate, alter coagulating characteristics. Bratby (1980) observed that sulphates tended to extend the pH range of destabilization while phosphates shifted the zone to lower pH levels (Figures 8 and 9).

Hydrophilic colloids have charged functional groups which will coordinate with metal cations (Stumm and Morgan 1962). The resulting chemical species include charged polynuclear complexes and insoluble metal precipitates. Sproul (1980) concluded that carboxyl groups in a virus capsid will coordinate with metal cations and become part of the floc matrix.

A third mechanism related to coagulation with metal salts, is sweep flocculation, or enmeshment in an insoluble complexed metal precipitate (Gregory 1978a). The shaded area in Figure 7 is the usual operating region in water and wastewater treatment (O'Melia 1972). An AWWA Committee

Figure 8 - The Effect of Sulphate on Coagulation

This page contained copyright material from Bratby (1980). The figure showed how sulphate broadens the effective pH range of alum.

Figure 9 - The Effect of Phosphate on Coagulation

This page contained copyright material from Bratby (1980). The figure showed how the presence of phosphate tends to depress the effective pH range of alum.

(1971) pointed out that Figure 7 was oversimplified and ignored the presence of other insoluble species originating from other anions in the water which influence the operating region of the process.

The occurrence of charged polymer chains, polyelectrolytes, from both natural and man-made sources provided the fourth mechanism; interparticle bridging introduced by LaMer and Healy (1963). Harris and Mitchell (1973) noted that microorganisms secreted a natural exocellular polymer which promoted flocculation of biocolloids. Man-made polymers have been used extensively in water and wastewater treatment (AWWA 1982; Gutcho 1977). O'Melia (1972) summarized the six steps which occur in polymer coagulation which are presented in Figure 10.

Stumm and O'Melia (1968) pointed out some important aspects of the bridging model which were:

- optimum conditions occurred when only a portion of the available adsorption sites were occupied;
- saturation of adsorption sites caused restabilization to occur;
- extended agitation of the dispersion may break weak adsorption bonds, restabilizing the system; and
- a modified Langmuir isotherm applied.

Gregory (1978b) reviewed the use of polyelectrolytes and noted that adsorption of a polymeric segment to a colloid was only part of the process. Cationic polyelectrolyte coagulation of negatively charged colloids

Figure 10 - Polymer Destabilization of Colloids

This page contained copyright material from O'Melia (1972). The diagram demonstrated the six steps of polymer coagulation.

was effective since double layer compression occurred, enhancing adsorption and bridging bonds. In the case of anionic polyelectrolytes, the presence of Ca^{2+} cations, improved the adsorption characteristics suggesting complex formation (Bratby 1980).

Gregory (1978b) postulated that dispersions with a low surface concentration used an electrostatic patch mechanism. This occurred as a result of non-uniform attachment of polymer segments to the colloid creating regions of positive and negative charge. As colloids presented oppositely charged surfaces, these were attracted to one another and an electrostatic bond formed.

2.2.4 Flocculation

A mathematical description of the transport and agglomeration step was first proposed by Von Smoluchowski (1916). Overbeek (1952) summarized Von Smoluchowski's work on perikinetic and orthokinetic flocculation for laminar flow conditions.

Perikinetic flocculation results from interparticle contacts caused by Brownian motion (O'Melia 1972). The differential equation describing perikinetic flocculation is:

$$\frac{dN}{dt} = \frac{-4\gamma KTN^2}{3\mu}$$

where;

N = total concentration of particles at time, t

γ = dimensionless collision efficiency factor

KT = Boltzman's constant times absolute temperature

μ = absolute viscosity

Perikinetic flocculation is second order with respect to particle concentration and is independent of particle size.

Orthokinetic flocculation describes interparticle contact resulting from bulk fluid motion (Overbeek 1952). The original work of Von Smoluchowski (1916) was modified by Camp and Stein (1943) to account for the turbulent conditions found in water and wastewater treatment. Their differential equation was of the form:

$$\frac{dN}{dt} = -\frac{2}{3}\gamma G d^3 N^2$$

where;

G = mean square velocity gradient (sec^{-1})

d = particle diameter

O'Melia (1972) discussed the volume fraction of colloidal particles, Ω, which characterized the volume of colloidal particles in a unit volume of liquid. It was defined as:

$$\Omega = 0.667\pi d_o{}^3 N_o$$

where;

d_o = diameter of floc particles at time = 0

N_o = number of floc particles at time = 0

Substituting this into the orthokinetic flocculation equation and integrating produces:

$$\ln \frac{N}{N_o} = -\frac{4}{\pi}\gamma\Omega G t$$

Therefore, the number of particles remaining in solution is a function of the mean velocity gradient and flocculation time.

The problem of floc break-up was not accounted for by Camp and Stein (1943). Argaman and Kaufman (1970), LaMer and Healy (1963), and Huck and Murphy (1978) examined complete flocculation models. Huck and Murphy (1978) tested all of the models on simulated mine wastewater using statistical techniques and found none of the models, with and without floc break-up, were adequate. They presented a semi-empirical, second order model with a break-up term which adequately predicted the flocculation of simulated and actual mine wastewater.

Parker et al. (1972) examined turbulent break-up of activated sludge floc and described two likely mechanisms:

1. surface shear; and
2. filament fracture.

Their model was similar to the Argaman and Kaufman (1970) model.

While results are encouraging for simple systems, the complexity of domestic wastewater prevents reliable application of the theory (Bratby 1980). A major problem results from the stochastic nature of wastewater particle size and the large variations in size (Birkner and Morgan 1968; O'Melia 1978). Therefore, jar tests are utilized to determine the flocculation properties and pilot plants are constructed to determine the response of the full scale system (Stevenson 1977).

2.2.5 Factors Affecting the Process

Coagulation and flocculation may be affected by a number of physical and chemical parameters, the magnitude of which may be determined through the use of jar tests.

Bratby (1980) and others noted some physical factors which affected coagulation and flocculation:

- rapid mix intensity and duration;
- flocculation velocity gradient;
- retention time;
- flocculator type;
- geometry of the flocculation basin;
- temperature (Al-Layla and Middlebrooks 1975);
- particle size (O'Melia 1978); and
- surface concentration (Stumm and O'Melia 1968).

Some chemical factors were reported by Bratby (1980) and others to influence coagulation:

- primary coagulant type;
- primary coagulant dose;
- coagulant aid type;
- coagulant aid dose;
- chemical feed concentration;
- sequence of dosing and time lag;
- final pH;
- alkalinity;
- sulphates and phosphates;
- organic matter (Narkis and Rebhun 1983);
- ionic strength (O'Melia 1972); and

- nature of colloidal particles (Stumm and Morgan 1962).

2.2.6 Lagoon Applications

Coagulation followed by flocculation and sedimentation has been used in treating potable water supplies achieving removals of turbidity, colour, taste, viruses, bacteria, and protozoa (AWWA 1971; Feachem et al. 1983; York and Drewry 1974).

A general, lucid account of wastewater coagulation was provided by O'Melia (1978) where he synthesized theories of colloidal stability, destabilization, and flocculation. He reviewed bioflocculation as related to the formation of exocellular polymers by algae and bacteria and its utility in the activated sludge process. Lagoons were not considered.

In the past, chemical treatment of lagoons has been performed for two reasons:

1. reduction of phosphorus levels by precipitation; and
2. removal of suspended solids – mostly algae – from lagoon effluents.

Incidental removals of coliform bacteria, BOD_5, and other water quality parameters have been reported by various investigators (Airaksinen 1978; Balmer and Bjarne 1978; Graham and Hunsinger 1974; Hanaeus 1984). No literature was found which dealt specifically with the removal of indicator organisms from a lagoon effluent using chemical coagulants.

Freedman et al. (1983) studied the removal of purple sulphur bacteria from anaerobic lagoon effluents using alum in jar tests. They found that bacteria reductions increased with decreased final pH over the range 7.0 to 5.5. They also observed that increases were less dramatic at pH's less than 6.5. They used approximately 1390 mg/L of alum ($Al_2(SO_4)_3 \cdot 18H_2O$) and a mixing Gt of 11400. Flocculation used a Gt of 36000 for 60 minutes. Sedimentation was allowed to occur for 4 hours. The estimated chemical cost was $ 0.26 per cubic metre.

Algae are hydrophilic biocolloids, though of somewhat larger size than coliform bacteria, but it is worthwhile to review previous work on algal coagulation. Few methodical studies have been done on coagulating algae. Al-Layla and Middlebrooks (1975) and McGarry (1970) were the only authors to use factorial experimental designs and statistical analysis of the results to determine the important physical and chemical factors. McGarry (1970) found alum ($Al_2(SO_4)_3 \cdot 18\ H_2O$) to be the most effective primary coagulant. He also found cationic polyelectrolytes to be good coagulation aids and of these Dow Chemical's PEI 1090 was the best for the wastewater tested. Anionic and nonionic polyelectrolytes were ineffectual. Al-Layla and Middlebrooks (1975) found alum to be the most effective coagulant, producing the best results at 10° C. Coagulant dose was independent of temperature as was flocculation time. Shindala and Stewart (1971) jar tested alum, ferric chloride, and ferric sulphate and found all to be effective in removing COD and phosphate, though alum was the most economical to use. They noted that the pond effluent contained 5000 coliforms per 100 mL following treatment. No details of the original numbers of coliforms, type of lagoon, or season were given. Van Vuuren and Van Duuren (1965) used alum and lime to treat a pond effluent. They found polyelectrolytes had no effect on alum coagulation nor did pH over the range 7.0 to 11.5. They reported a 0.6 $\log_{10}$ unit reduction of coliforms. The paper used poor coagulation test methodology and the results are not considered important.

Graham and Hunsinger (1974) presented results from the batch treatment of a seasonal discharge lagoon with alum and ferric chloride for phosphorus removal. Fecal coliforms were reduced from 0.4 to 3.2 $\log_{10}$ units (N/N_0 of 0.04 to 0.02) and appeared to be a function of alum dosage. Unfortunately, the alum doses associated with the N/N_0 values were not reported. Jar tests were not conducted to measure bacteria removal, this being ancillary to phosphate removal.

Hanaeus (1984) presented data from the treatment of Swedish lagoons using alum and lime. Lime produced the most consistent removals of bacteria. He reported that alum had inconsistent results for bacteria removals but produced good phosphorus reductions.

Folkman and Wachs (1973), Grabow et al. (1978), and Ronen and Halbertal (1978) examined lime treatment of lagoon effluents for algae removal. They found optimum conditions at pH 10.9 when in the presence of magnesium concentrations of 7 to 45 mg/L. The removal mechanism appeared to be enmeshment in a magnesium hydroxide precipitate. The heavy lime sludge production and a requirement for high doses of lime to adjust pH made this an unattractive method.

Ives (1959) studied algal coagulation in terms of surface charge and zeta potential. He found algae maintained a negative charge over a pH range of 2.5 to 11.5. Though an isoelectric point was not found, minimum values for charge and zeta potential were observed at pH 7 to 8. He postulated that charge neutralization by adsorption of chemical species was the main mechanism followed by enmeshment in a sweep floc.

The role of natural and man-made polyelectrolytes in wastewater coagulation and flocculation was examined by several investigators (Busch and Stumm 1968; Tenney and Stumm 1965; Tenney et al. 1969). Busch and Stumm (1968) found a linear relationship between the amount of polymer present and the number of bacteria. Tilton et al. (1972) reported anionic and nonionic polyelectrolytes were ineffective in coagulating algae. High molecular weight (1.3 x 10‘) cationic polymers produced charge neutralization, followed by charge reversal. They also observed a linear relationship between polymer dose and algae concentration. Tenney et al. (1969) observed a stoichiometric relationship between polymer dose and algal concentration which supported the discussion of Stumm and O'Melia (1968).

Alum sludge generation was considered by Balmer and Bjarne (1978) when precipitating phosphorus in a cold climate lagoon. They noted accumulations in the order of 1.0 to 1.5 cubic metres per year per capita. This was an order of magnitude higher than the 0.09 to 0.15 cubic metres per year per capita reported by Schneiter et al. (1983) for facultative lagoons operated in Alaska and Canada without chemical treatment. James and O'Melia (1982) studied various sludges produced in water treatment plants and found alum coagulation with a cationic polyelectrolyte produced a sludge volume approximately 30 per cent to that of using alum alone.

The literature indicated that removal of microorganisms by coagulation appears feasible provided optimum chemical and physical conditions are determined for the application situation.

2.3 PROCESS SCALE-UP

In 1974, Schmidtke (1974:11) observed:

"at the present time, the scale-up of biological waste treatment systems is poorly defined."

Today this has been marginally corrected, but there is still much information lacking on the scale-up of water and wastewater processes. Stevenson (1977:142), while reviewing the step from jar tests to prototype application, noted:

"This is however an area where experience is necessary as there are factors such as floc volume to take into account."

Attempts at lagoon scale-up have focussed on mathematical models of hydraulic flow based on chemical reactor engineering (Antonini et al. 1983; Polprasert et al. 1983; Thirumurthi 1969).

A workshop on the scale-up of water and wastewater treatment processes yielded a state of the art summary (Schmidtke and Smith 1983a). Schmidtke and Smith (1983b) reviewed elementary scale-up techniques, noting that for performance of the prototype facility to be equal to the pilot plant, hydraulic, chemical, biological, and physical conditions must be the same. They noted modelling was a commonly used approach, where bench scale and pilot scale models determined the overall feasibility of a process and the operational performance respectively. As noted earlier, coagulation and flocculation of wastewater does not lend itself to mathematical modelling due to the complexity of wastewater constituents. However, scale-up factors may still be assessed by other methods. Horvath and Schmidtke (1983) presented scale-up and scale-down criteria on the basis of similarity. They noted six types of similarity:

- geometric;
- kinematic;
- dynamic;
- chemical;
- biological; and

- thermal.

Once an effective scale-up relationship is established, scale-down of processes is possible for laboratory study of operating problems. Horvath and Schmidtke (1983) reflected on the importance of boundary conditions in scale-up. This is of interest in coagulation jar tests where the "wall effects" may potentially influence the mixing and flocculation behaviour in an unpredictable fashion. Cornwell and Bishop (1983) studied velocity gradients in bench scale experiments and in full size flocculation systems. They concluded that G was not necessarily the best parameter to use in measuring flocculator efficiency and that more research was needed in describing the mixing process and scale-up relationships.

3. EXPERIMENTAL INVESTIGATION

3.1 INTRODUCTION TO EXPERIMENTS

An experimental program consisting of seven parts (Table 4), six laboratory trials and one field test, was undertaken to fulfill the following objectives:

1. determine the physical and chemical factors affecting the coagulation and subsequent removal of fecal coliform bacteria from lagoon wastewater;
2. establish the experimental conditions for reducing fecal coliform bacteria to levels less than 10 per 100 mL;
3. assess the scale-up effects on fecal coliform removal efficiencies; and
4. compare the cost of chemical coagulation with that of chlorination, the more traditional approach to biological quality upgrading.

Three different process scales were used throughout the research, ranging from 1.0 litre jar tests to a 135 litre tank and ending with a full scale application in a 35 million litre facultative lagoon.

3.2 LOCATION OF STUDY LAGOON

The lagoon used for the study serves the Town of Gibbons, Alberta, a 3000 person community 40 kilometres northeast of Edmonton, Alberta. The lagoon system consists of four short retention, anaerobic cells followed by three long retention, facultative cells (Figure 11). System

Table 4 - Experimental Program

SERIES	OBJECTIVE
1	Screen eight factors which were reported to affect coagulation and flocculation.
2	Evaluate an expanded experimental region for the significant factors from Series 1.
3	Extend the experimental region of alum dose and pH to establish practical limits.
4	Use of an orthogonal composite design to determine if the response surface approximates a planar or quadratic surface using five factors.
5	Investigate chlorination of the wastewater and compare its effectiveness to that of coagulation.
6	Scale-up experiments from jar tests to a significantly larger container and determine if scale factors exist.
7	Full scale application of the laboratory findings to a seasonal discharge lagoon at Gibbons, Alberta.

storage provides for 180 day retention which requires discharge twice per year; once in the spring and once in the autumn. Only facultative cells 2 and 3 are discharged at these times. Lagoon effluent enters the Sturgeon River at a point approximately 20 kilometres from its confluence with the North Saskatchewan River.

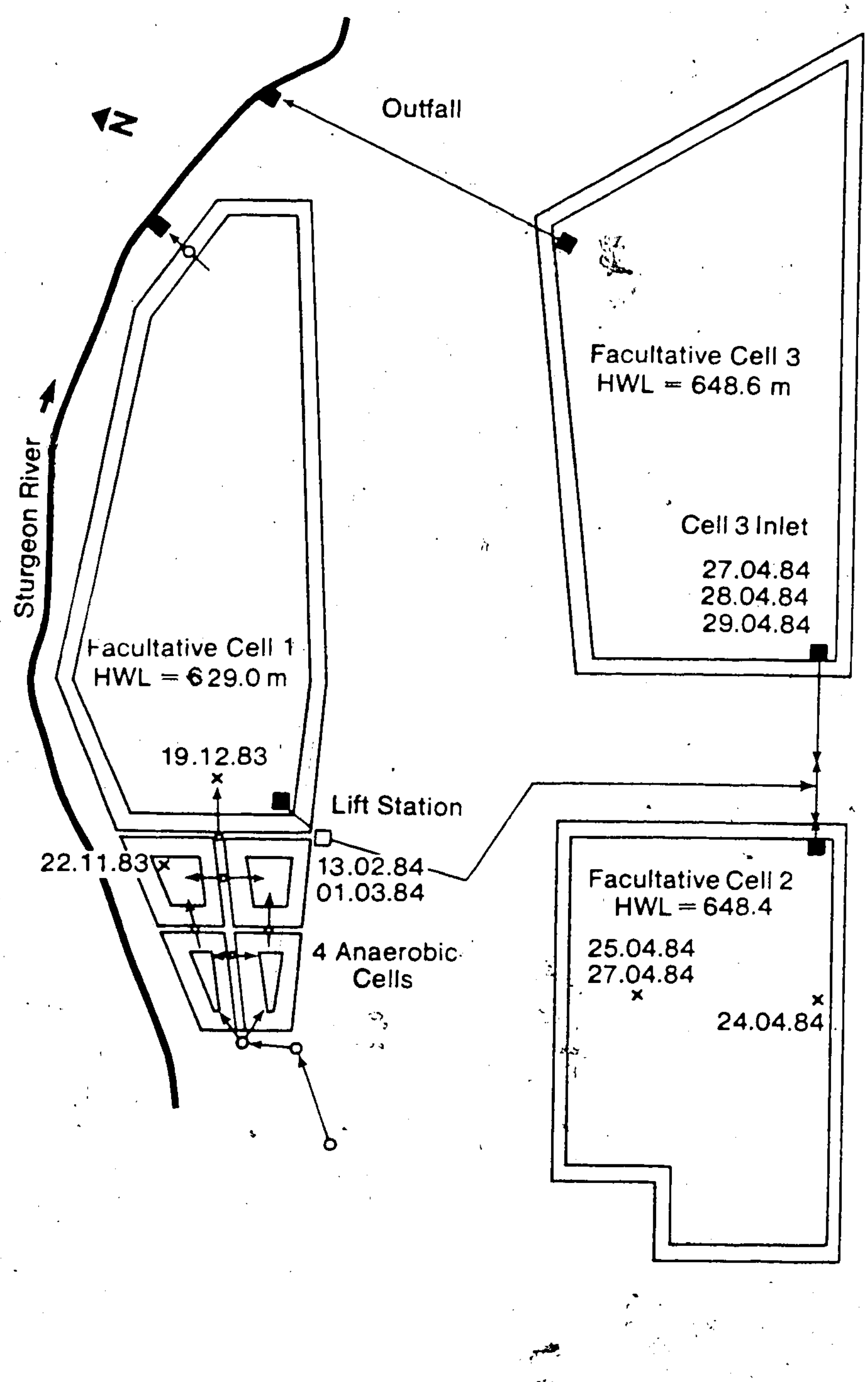
Facultative cell 2 was chosen as the test site since it has the smallest volume, approximately 35 million litres, and offered some flexibility with respect to time of discharge since it could be isolated from the rest of the system. This cell was reconstructed in 1982 following failure of the northern embankment.

3.3 SAMPLING AND WASTEWATER COMPOSITION

Grab sampling was performed at a variety of locations in the lagoon system. One goal of the sampling program was to provide a sample which would best represent the wastewater characteristics expected at the end of the winter. Consequently, samples were collected from anaerobic cell 4, facultative cell 1, facultative cell 2, and the wet well of the lift station (Figure 11). Prior to treatment in April, additional samples were collected from the centre of cell 2 where both a mixed and a stratified vertical sample of the water column were taken.

Samples were stored in the dark at 4 to 6° C and kept closed to the atmosphere. This handling attempted to simulate conditions under 200 mm of ice cover and 250 mm of

Figure 11 - Gibbons Lagoon System and Sampling Locations



snow cover which were observed at the site in January and February.

Table 5 summarizes selected wastewater components over the six month study period from November 1983 to April 1984. These were analysed following procedures established in Standard Methods for the Examination of Water and Wastewater (APHA,AWWA,WPCF 1981).

3.4 BENCH SCALE PROCEDURES AND ANALYSIS

Jar tests using a standard six place stirring apparatus have normally been used when evaluating the coagulation and flocculation processes. Chemical dose and velocity gradients have generally been the only variables considered in this type of experiment. Combining the traditional jar test with a systematic experimental design allows the experimenter to efficiently extract information on the influence of a large number of variables and their interactions on the performance of the processes. The technique of factorial experimental design permits the independent analysis of variables and their interactions using a minimum number of trials or jars. These designs are based on the principle of setting a high and low level for each of the factors and then testing all the possible combinations of these settings with each factor. The outcome of each trial provides information which can be analysed allowing determination of the main effects and interactions. These are a measure of the influence which each factor and interaction have on the

Table 5 - Lagoon Wastewater Composition

Parameter	Units	22.11.83	19.12.83	13.02.84	01.03.84	25.04.84	Standard Method
$CBOD_5$*	mg/L	185	108	-	104	54	507
TSS**	mg/L	98	70	29	31	15	209D
PO_4^{3-}-P	mg/L	9.2	17	9.0	9.2	8.7	424C,D
SO_4^{2-}	mg/L	60	76	60	60	72	426C
Alkalinity (to pH 4.5)	mg $CaCO_3$/L	358	320	321	311	293	403
Temperature	°C	-	1	1	3	10	212
Ionic Strength		0.0140	0.0144	0.0147	0.0147	0.0130	205,(Sawyer & McCarty 1978)
pH		7.5	7.6	6.8	6.7	7.2	423
Fecal Coliforms	No./100 mL	$1.2x10^5$	$3.2x10^5$	$2.8x10^5$	$5.6x10^5$	$4.8x10^3$	909C

* carbonaceous 5 day biochemical oxygen demand

** total suspended solids

response of the experiment.

A short-hand description of factorial designs with two levels and n factors is 2^n. If a large number of factors are involved, the number trials can become very large and impractical to perform. Fortunately, many of the higher order interactions are often negligible and can be confounded with additional factors to reduce the number of trials needed to ascertain the main effects and two factor interactions. Confounding refers to the intentional confusion, or mixing, of two or more effects or interactions with one another in factorial experiments where insufficient trials are available to determine all of the individual effects and interactions. Fractional factorial designs make use of confounding. They are of the general form, 2^{n-m} where m is the fraction, $1/2^m$, of the factorial. The fraction consists of reducing the full design into 2^m blocks. Each block provides information on the main effects and lower order interactions associated with the block. Complete analysis of the confounded, higher order interactions requires additional experimental work using the remaining blocks. Complete descriptions of factorial designs were given by Box and Hunter (1961a; 1961b) and Davies (1979).

3.4.1 Apparatus, Reagents, and Procedure

A Phipps and Bird Inc. six place stirring apparatus was used throughout the bench scale testing. Maximum impeller speed was 100 RPM. The standard 75 mm diameter by 25 mm wide two-bladed paddle was used to mix the 1.0 litre sample in each 95 mm by 95 mm square jar (Plate 1). Cornwell and Bishop (1983) evaluated the velocity gradients for a similar jar at various mixing speeds (Figure 12). Their jar was slightly larger, 115 mm by 115 mm, but their results were assumed to be adequate for the present investigation. A Fisher Accumet 156 pH meter was used to measure wastewater pH before and after treatment. Coagulant and polyelectrolyte feed solutions were prepared from 16 to 24 hours in advance of the experiment. The cationic polyelectrolyte was activated according to the manufacturer's recommendations. Final pH was adjusted using approximately 6 N HNO_3 and 1 N NaOH solutions. Media and dilution water for membrane filtration determination of fecal coliforms was prepared according to Method 909C (APHA,AWWA,WPCF 1981).

Each trial was treated using the combination described by the experimental design matrix determined for each series. The pH was adjusted to the final value during the flocculation period. Following 30 minutes of sedimentation, an aliquot was extracted from the jar using a syringe and was stored at 4 to 6° C until it was analysed for remaining fecal coliforms on the same day.

3.4.2 Analysis

The response of each trial was reported in the design matrix for each series. The statistical package MIDAS (Fox and Guire 1976), was used to perform a multivariate least

Plate 1 - Jar Test Apparatus

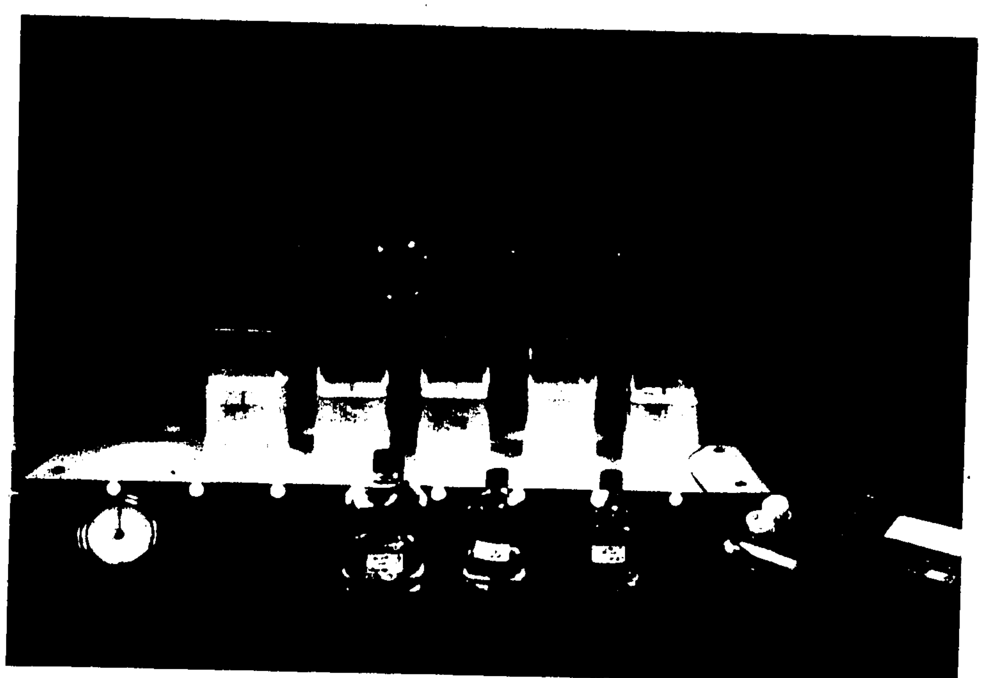


Figure 12 - Velocity Gradients for Apparatus

This page contained copyright material from Cornwell and Bishop (1983). The figure related impeller speed to the mean square velocity gradient at 3° C and 23° C. The jar was a square (gator) jar.

squares regression of the design matrix on the response. The main effects and interactions were calculated from the resulting regression coefficients (β) by the relation:

$$\text{effect} = 2 \times \beta$$

The fecal coliform count per 100 mL was subjected to a $\log_{10}$ transformation to account for the dependence of variance on the number of coliforms counted (Draper and Smith 1981).

Each factor and interaction was evaluated by Yates' (1937) method of analysis of variance (ANOVA) summarized in Davies (1979). The mean square of each term was calculated from the total effect:

$$\text{total effect (TE)} = \beta \times 2 \times 2^{n-m-1}$$

and ;

$$\text{mean square (MS)} = (TE)^2/(\lambda \cdot 2^{n-m})$$

where;

λ = degree of freedom of factor = 1

An estimate of experimental error (error variance) allows differentiation of real effects from null effects. The error mean square (MS_d) is an estimate of the error variance and may be calculated from centre-point replicates or from the mean square of insignificant interactions and effects. Half-normal probability plots of the total effects of each factor and interaction provide a graphical technique for separating real effects from null effects (Daniel 1959). They are constructed using half of the normal probability scale on probability paper. Starting at 50 per cent, the scale is changed by the relationship:

$$P' = 2P - 100$$

where;

P = full-normal probability

P' = half-normal probability

P' value for each factor and interaction is calculated by the rank of the effect, the highest rank being for the largest effect. The equation is:

$$P' = (RANK-1/2)/2^{n-m-1}$$

Null effects will approximate a line through the origin if they are random and normally distributed. The average of the mean square of the factors on this line approximates the error variance. The ratio of MS/MS_e for each term produces a statistic which may be compared to the F-statistic, $F\alpha(\lambda_1, \lambda_2)$, the degrees of freedom being for factor mean square and the error mean square respectively. Those factors and interactions exceeding the tabulated F-statistic at α level (Davies 1979), have statistically significant effects. For these experiments, α was selected as 0.05.

3.5 SERIES 1

Series 1 was conducted to evaluate the impact of various physical and chemical parameters which were reported to influence the processes of coagulation and flocculation. Eight different factors were assessed in the screening process:

1. primary coagulant dose;
2. coagulant aid dose;

3. pH;
4. temperature;
5. rapid mixing velocity gradient;
6. flocculation velocity gradient;
7. rapid mixing duration; and
8. flocculation time.

The most commonly reported coagulants were aluminum sulphate (alum - $Al_2(SO_4)_3 \cdot 18H_2O$), ferric chloride, ferric sulphate, and various man-made polyelectrolytes. Previous work by Al-Layla and Middlebrooks (1975), McGarry (1970), and Shindala and Stewart (1971) used alum as a coagulant because it was economical, easy to use, and produced good results. Ferric chloride and other iron compounds have also been used but produced similar results to alum at significantly higher cost. Also adverse side effects such as coloured effluents were reported (Graham and Hunsinger 1975). On the basis of these past experiences, it was decided to use alum as the primary coagulant.

McGarry (1970) reported enhanced removals of algae using alum in combination with a cationic polyelectrolyte, Dow C-31 Purifloc. This product was difficult to obtain therefore another high molecular weight cationic polymer, Alchem 7122, was used in Series 1.

A full 2^8 factorial design consists of 256 trials which was considered to be impractical. Therefore use was made of a fractional factorial design which allowed a greatly reduced number of trials to be performed, yet providing sufficient information on the main effects and lower order interactions. Information on higher order interactions was confounded with other main effects but since three factor interactions were assumed negligible, this allowed analysis of four additional factors by deliberately confusing them with the interactions.

The design chosen for Series 1 was based on a one sixteenth fractional factorial of a 2^8 design (2^{8-4}). In this design, the three factor interactions were confused with four additional main factors. The resulting design is shown in Table 6. The source of this design was Davies (1979), Appendix 10A and Table M.

Series 1.1 used samples collected on November 22, 1983 (Figure 11) containing 1.2 x 10^5 fecal coliforms per 100 mL. The stock alum and polyelectrolyte solutions were fed as 1 per cent solutions. The different levels of temperature necessitated blocking the experiment based on temperature (factor G). The low temperature block was conducted in the cold room at 5° C and the other at a room temperature of 16° C.

Using the method of Daniel (1959), Figure 13 presents the half-normal plot of the total effects for Series 1.1. It indicates that A, F, and ABCD are probably real effects.

The error mean square for Series 1.1 was estimated from the average of the lower five points on the line in Figure 13. It was 0.0238 and had 5 degrees of freedom. Thus, the F-statistic for comparing MS and MS_d was $F_{0.05}(1,5)$ or 6.61.

Table 6 - Series 1 Experimental Design and Results

(a) **Generating Relations**

I = ABCE — E=ABC
= BCDF — F=BCD
= ABDG — G=ABD
= ACDH — H=ACD

(b) **Design Matrix and Results**

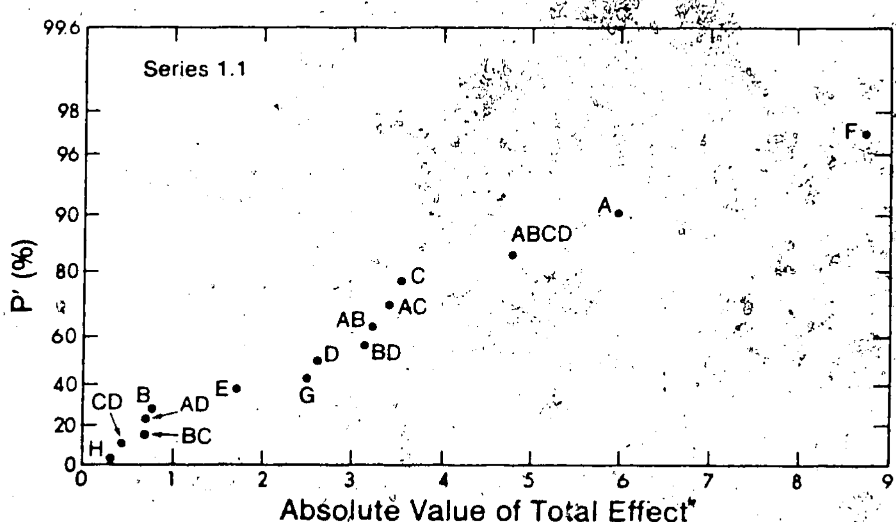
Jar No.	Main Effects and Interactions															Results log FC/100 mL	
	A	B	C	D	E	F	G	H	AB	AC	AD	BC	BD	CD	ABCD	Series 1.1	Series 1.2
1	-	-	-	-	-	-	-	-	+	+	+	+	+	+	+	4.94	4.86
2	+	-	-	-	+	-	+	+	-	-	-	+	+	+	-	3.59	4.20
3	-	+	-	-	+	+	+	-	-	+	+	-	-	+	-	4.73	5.11
4	+	+	-	-	-	+	-	+	+	-	-	-	-	+	+	5.18	4.90
5	-	-	+	-	+	+	-	+	+	-	+	-	+	-	-	5.20	5.08
6	+	-	+	-	-	+	+	-	-	+	-	-	+	-	+	4.70	4.89
7	-	+	+	-	-	-	+	+	-	-	+	+	-	-	+	4.26	4.85
8	+	+	+	-	+	-	-	-	+	+	-	+	-	-	-	2.30	3.75
9	-	-	-	+	-	+	+	+	+	+	-	+	-	-	-	4.85	5.18
10	+	-	-	+	+	+	-	-	-	-	+	+	-	-	+	4.32	4.86
11	-	+	-	+	+	-	-	+	-	+	-	-	+	-	+	3.81	4.32
12	+	+	-	+	-	-	+	-	+	-	+	-	+	-	-	3.95	4.23
13	-	-	+	+	+	-	+	-	+	-	-	-	-	+	+	4.30	4.52
14	+	-	+	+	-	-	-	+	-	+	+	-	-	+	-	2.08	3.72
15	-	+	+	+	-	+	-	-	-	-	-	+	+	+	-	4.51	5.00
16	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	4.48	4.65
17	0	0	0	0	0	+	0	0	0	0	0	0	0	0	0	4.78	--
18	0	0	0	0	0	+	0	0	0	0	0	0	0	0	0	4.90	--
19	0	0	0	0	0	+	0	0	0	0	0	0	0	0	0	4.78	--

Table 6 - continued

(c) **Factors and Levels**

FACTORS	LEVELS (-)	(0)	(+)
A - alum dose (mg/L)	50	75	100
B - polyelectrolyte dose (mg/L)	0	1.75	3.5
C - rapid mix duration (s)	10	25	40
D - flocculation time (s)	60	120	180
E - flocculation G (s^{-1})	10	25	40
F - pH	5	6.5	8
G - temperature (°C)	5	10.5	16
H - rapid mix G (s^{-1})	40	55	100

Figure 13 - Series 1.1 Half-Normal Plot



Therefore, when the ratio of MS/MS_d exceeds 6.61 the factor is significant at the 5 per cent level. It is evident from Table 7 that the single factors A, C, and F were highly significant as were the interactions AB, AC, BD, and ABCD.

Series 1.2 was conducted to confirm the outcome of Series 1.1 and to determine if the new wastewater samples collected on December 19, 1983 (Figure 11) would change the significant factors. This wastewater had 3.6 x 10^3 fecal coliforms per 100 mL. The half-normal plot of Figure 14 clearly illustrates the importance of A and F, alum dose and pH. The error variance was estimated from the lower seven points on the line through the origin resulting in an estimate of 0.0110 with 7 degrees of freedom. The F-statisic for $F_{0.05}(1,7)$ was 5.59. Comparing the MS/MS_d ratio Table 8 reveals that A and F were highly significant followed by C, D, E, G, and AC as marginally significant effects. The ABCD interaction was significant.

The outcome of these two runs confirmed that alum dose and pH effected the most significant removals of fecal coliform bacteria which was consistent with the theory of hydrophilic colloid stability where chemical influences are known to be important. Mixing duration, flocculation time, and flocculation G were found to be marginally significant in both runs suggesting alum dose, pH, mixing time, flocculation time, and flocculation G should be examined further. This conclusion was supported by the effect of the four factor interaction of alum dose, polyelectrolyte dose,

Table 7 - Series 1.1 ANOVA

FACTOR	β	TE	MS	MS/MS_d	SIGNIFICANT $\alpha=0.05$	RANK
A	-0.375	-6.007	2.255	94	Highly	14
B	-0.049	-0.778	0.038	1.5	No	5
C	-0.222	-3.553	0.789	33	Highly	12
D	-0.162	-2.600	0.423	17	Yes	8
E	-0.107	-1.713	0.183	7	Yes	6
F	0.546	8.738	4.772	200	Highly	15
G	0.157	2.516	0.396	16	Yes	7
H	-0.019	-0.303	0.006	0.3	No	1
AB	0.200	3.202	0.641	27	Highly	10
AC	-0.213	-3.412	0.728	30	Highly	11
AD	0.045	0.722	0.033	1.4	No	4
BC	-0.044	-0.711	0.032	1.3	No	3
BD	0.197	3.160	0.624	26	Highly	9
CD	0.025	0.406	0.010	0.4	No	2
ABCD	0.298	4.770	1.422	60	Highly	13

Figure 14 - Series 1.2 Half-Normal Plot

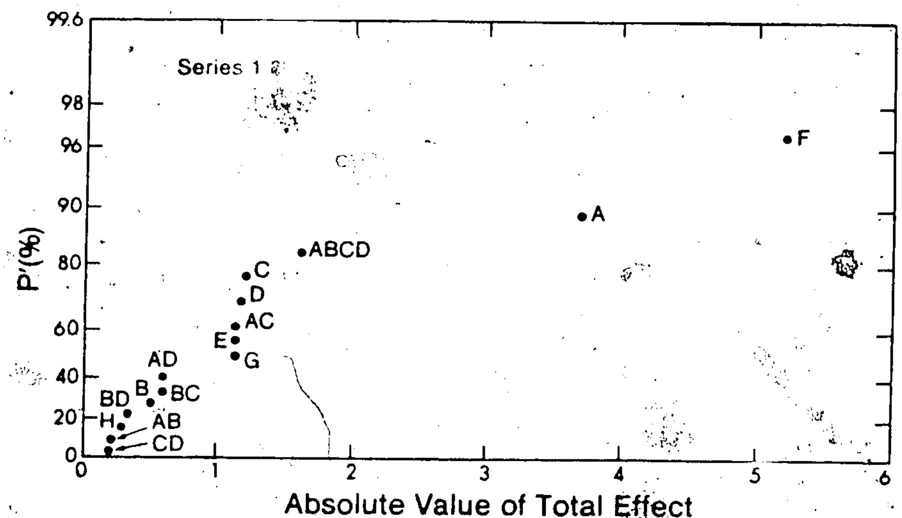


Table 8 - Series 1.2 ANOVA

FACTOR	β	TE	MS	MS/MS_d	SIGNIFICANT $\alpha=0.05$	RANK
A	−0.232	−3.712	0.861	78	Highly	14
B	−0.031	−0.493	0.015	1.4	No	5
C	−0.075	−1.204	0.091	8.2	Yes	12
D	−0.073	−1.173	0.086	7.8	Yes	11
E	−0.071	−1.144	0.082	7.4	Yes	9
F	0.326	5.213	1.698	154	Highly	15
G	0.071	1.142	0.082	7.4	Yes	8
H	−0.019	−0.310	0.006	0.5	No	3
AB	0.012	0.195	0.002	0.2	No	2
AC	−0.073	−1.163	0.084	7.6	Yes	10
AD	−0.039	−0.621	0.024	2.2	No	7
BC	0.036	0.582	0.021	1.9	No	6
BD	0.021	0.340	0.007	0.6	No	4
CD	−0.012	−0.188	0.002	0.2	No	1
ABCD	0.100	1.597	0.159	14	Yes	13

mixing time, and flocculation time. Polyelectrolyte dose was not significant over the experimental region examined, and since it was more difficult to prepare and handle than alum, it was not considered further.

3.6 SERIES 2

The objective of Series 2 was to explore and expand the experimental region of each factor which was found to be significant in Series 1. Davies (1979) noted that three conditions could cause null effects in a factorial design:

1. the base level was near an optimum;
2. the unit change was too small; or
3. the effect was null.

Believing any one of these may have been true for the physical factors, a 2^{5-2} quarter fractional factorial design was used to elicit the real effects of the five factors disclosed in Series 1. The defining relations, design matrix and levels for each factor are given in Table 9. The source of the design was Davies (1979), Appendix 10A. The upper and lower settings for pH were selected within the range of Series 1 to gather information on the reported optimum of alum coagulation in natural waters (O'Melia 1972). All other factors used the upper setting from Series 1 as the lower setting for this series. Suitable upper levels were chosen based on equipment limitations for generating velocity gradients and engineering judgement.

Table 9 - Series 2 Experimental Design and Results

(a) Generating Relations

$I = ABCD - D = ABC$
$= -BCE - E = -BC$

(b) Design Matrix and Results

Jar No.	Main Effects and Interactions							Results log FC/100 mL
	A	B	C	D	E	AB	AC	
1	−	−	−	−	−	+	+	4.27
2	+	−	−	+	−	−	−	3.69
3	−	+	+	−	−	−	−	4.43
4	+	+	+	+	−	+	+	3.88
5	+	+	−	−	+	+	−	4.35
6	−	+	−	+	+	−	+	3.73
7	+	−	+	−	+	−	+	4.39
8	−	−	+	+	+	+	−	3.85

(c) Factors and Levels

FACTORS	LEVELS (−)	(0)	(+)
A - rapid mixing duration (s)	40	80	120
B - flocculation time (s)	180	390	600
C - pH	6	6.5	7
D - alum dose (mg/L)	100	150	200
E - flocculation G (s^{-1})	40	55	100

Series 2 used wastewater collected on December 19, 1983 (Figure 11). This contained 2.4×10^{5} fecal coliforms per 100 mL prior to treatment. The alum feed was a 2 per cent solution, rapid mixing G was 100 s^{-1}, and the temperature was approximately 18° C.

A half-normal plot of the data demonstrates the real effects were alum dose, pH, and flocculation time (Figure 15). All other effect and interactions approximate a line through the origin and were used to estimate the error mean square of this experiment. It was 5.98×10^{-4} with 4 degrees of freedom. The subsequent F-statistic was 7.71 for $F_{0.05}(1,4)$. The ANOVA table for Series 2 is presented in Table 10. Experimental results were consistent with those of Series 1, indicating the highly significant influence of alum dose and pH on fecal coliform removals. Flocculation time was of marginal significance indicating some flocculation was required but the process was relatively insensitive to this physical parameter. However, more information is needed on the boundary conditions of pH and alum dose.

3.7 SERIES 3

Series 3 was intended to help establish the practical limits of pH and alum dose for application in a sewage lagoon. This stage provided an indication of the maximum removals of fecal coliforms using chemical coagulation.

Figure 15 - Series 2 Half-Normal Plot

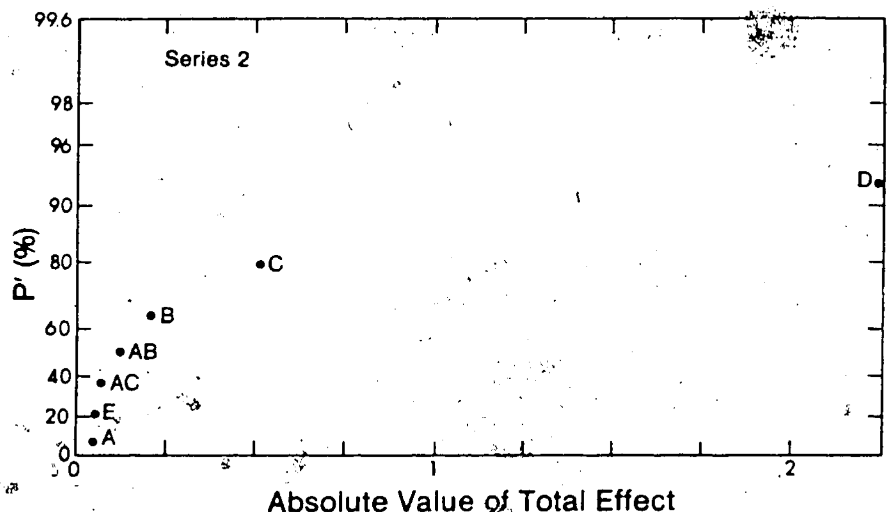


Table 10 - Series 2 ANOVA

FACTOR	β	TE	MS	MS/MS_d	SIGNIFICANT $\alpha=0.05$	RANK
A	0.0050	0.0401	0.0002	0.3	No	1
B	0.0251	0.2008	0.0050	8.4	Yes	5
C	0.0643	0.5145	0.0331	55	Highly	6
D	-0.2862	-2.2900	0.6552	1000	Highly	7
E	0.0057	0.0455	0.0003	0.4	No	2
AB	0.0140	0.1121	0.0016	2.6	No	4
AC	-0.0067	-0.0536	0.0004	0.6	No	3

A full, 2^2 factorial design was used in this investigation, the design of which is shown in Table 11. The same equipment and apparatus was used in this experiment as in the earlier series. A 4 per cent stock alum solution was used with rapid mixing for 60 s at a G of 100 s^{-1} and flocculation for 600 s at a G of 40 s^{-1}. The wastewater was collected on December 19, 1983 (Figure 11) and tested at a temperature of 7° C. The initial number of fecal coliforms present were 1.1×10^5 per 100 mL.

Virtually no fecal coliform bacteria were recovered by the membrane filtration method at all settings of the variables, thus establishing the limits of the experimental region.

To test the possibility of pH alone reducing fecal coliforms, a test was done by adjusting pH from 7.2 to 3.0 without any other treatment. This resulted in essentially no change in the number of fecal coliforms.

No independent estimate of error variance was possible from the centre-point replicates of this experiment. Thus, the average of Series 1.2 and 2 was used based on 12 degrees of freedom. This was 0.006. The F-statistic for $F_{0.05}(1,12)$ was 4.75. Table 12 shows alum dose and pH to be very significant though alum has assumed a more important role at these low pH values.

This experiment indicated that high fecal coliform removals were possible if pH and alum dose were balanced correctly. The probable mechanism of removal was adsorption

Table 11 - Series 3 Experimental Design and Results

(a) Design Matrix and Results

Jar No.	Main Effects and Interactions			Results log FC/100 mL
	A	B	AB	
1	-	-	+	0.301
2	+	-	-	0.000
3	-	+	-	0.778
4	+	+	+	0.000
5	0	0	0	-
6	0	0	0	-
7	0	0	0	0.000

(b) Factors and Levels

FACTORS	LEVELS (-)	(0)	(+)
A - alum dose (mg/L)	300	350	400
B - pH	4	4.5	5

Table 12 - Series 3 ANOVA

FACTOR	β	TE	MS	MS/MS_d	SIGN	RANK
A	-0.270	-1.079	0.291	485		3
B	0.119	0.477	0.057	95	Yes	2
AB	-0.119	-0.477	0.057	95	Yes	1

of polynuclear aluminum complexes by the bacteria followed by enmeshment in a sweep floc of organic matter, aluminum hydroxide, and aluminum phosphate .

3.8 SERIES 4

To this point in the experimental program, it had been assumed that the response surface (remaining fecal coliforms) was planar which may not necessarily be true. Examination of a parabolic surface is not possible using two level factorial designs therefore the experimenter must resort to using three level designs for n factors (3^n). These are satisfactory provided two or three factors are being considered since large numbers of trials are generated otherwise. An alternative experimental design is available which allows examination of a quadratic response surface. This was used in Series 4 to determine if any of the main effects were second order.

Davies (1979) introduced orthogonal composite designs based on full or fractional factorial designs which accounted for main effects and two factor interactions. Additional points were generated by using the centre point and intermediate axial points. To minimize the number of trials in Series 4, it was decided to use a half fractional factorial design to examine four factors; alum dose, pH, rapid mixing duration, and flocculation time. This design left all main effects clear but confounded two factor interactions with one another. This was acceptable since alum dose and pH had consistently been the most important factors and two factor interactions were negligible in Series 2. The number of extra points needed is (2n + 1) where n is the number of factors (Davies 1979). In this case, nine extra points were required. Axial points were calculated from the centre to maintain orthogonality using Table 11.6 from Davies (1979) where a = ±1.414. The complete design is shown in Table 13.

The wastewater collected December 19, 1983 (Figure 11) contained 5.0 x 10^4 fecal coliforms per 100 mL prior to treatment. A 5 per cent solution of alum was used as the feed stock to treat each jar. The 7° C wastewater was mixed at 100 RPM (G = 100 s^{-1}) after the alum was added. As in the earlier experiments, a 30 minute sedimentation period followed flocculation. The results are tabulated in Table 13.

This design did not readily lend itself to analysis by the Yates (1937) method however Draper and Smith (1981) discussed the use of step-wise regression analysis as a method of deciding which variables were significant in describing a response surface. In their opinion, the method of adding factors to the regression equation on a step by step basis was the best approach and recommended its use. The ranking of terms to insert into the model was based on the relative correlation between the dependent variable, fecal coliforms remaining, and the independent variables, summarized in Table 14.

Table 13 - Series 4 Experimental Design and Results

(a) **Generating Relations**

I = ABCD — D = ABC
a = ±1.414

(b) **Design Matrix and Results**

Jar No.	Main Effects and Interactions																Results log FC per 100 mL
	A	B	C	D	A'	B'	C'	D'	AB	AC	AD	BC	BD	CD	ACD	BCD	
1	-	-	-	-	+	+	+	+	+	+	+	+	+	+	-	-	1.11
2	+	-	-	+	+	+	+	+	-	-	-	+	+	+	+	-	1.54
3	-	+	-	+	+	+	+	+	-	+	+	-	-	+	-	+	0.48
4	+	+	-	-	+	+	+	+	+	-	-	-	-	+	+	+	2.90
5	-	-	+	+	+	+	+	+	+	-	+	-	+	-	+	+	0.70
6	+	-	+	-	+	+	+	+	-	+	-	-	+	-	-	+	3.20
7	-	+	+	-	+	+	+	+	-	-	+	+	-	-	+	-	0.95
8	+	+	+	+	+	+	+	+	+	+	-	+	-	-	-	-	1.81
9	-a	O	O	O	a'	O	O	O	O	O	O	O	O	O	O	O	1.30
10	+a	O	O	O	a'	O	O	O	O	O	O	O	O	O	O	O	2.51
11	O	-a	O	O	O	a'	O	O	O	O	O	O	O	O	O	O	1.34
12	O	+a	O	O	O	a'	O	O	O	O	O	O	O	O	O	O	2.00
13	O	O	-a	O	O	O	a'	O	O	O	O	O	O	O	O	O	1.34
14	O	O	+a	O	O	O	a'	O	O	O	O	O	O	O	O	O	1.40
15	O	O	O	-a	O	O	O	a'	O	O	O	O	O	O	O	O	2.59
16	O	O	O	+a	O	O	O	a'	O	O	O	O	O	O	O	O	1.70
17	O	O	O	O	O	O	O	O	O	O	O	O	O	O	O	O	0.70

Table 13 - continued

(c) **Factors and Levels**

FACTORS	LEVELS (-a)	(-)	(0)	(+)	(+a)
A - pH	4.8	5	5.5	6	6.2
B - rapid mix duration (s)	36	60	120	180	204
C - flocculation time[1] (s)	108	180	360	540	612
- alum dose (mg/L)	180	200	250	300	320

Table 14 - Correlation of Main Effects and Interactions

Main Effects and Interactions	Correlation Coefficient of Log FC
A - pH	0.72
B - rapid mixing duration	0.05
C - flocculation time	0.06
D - alum dose	-0.45
A^2	0.10
B^2	-0.01
C^2	-0.15
D^2	0.21
AB	0.04
AC	0.06
AD	-0.66
BC	-0.21
BD	0.10
CD	-0.03
ACD	-0.02
BCD	0.26

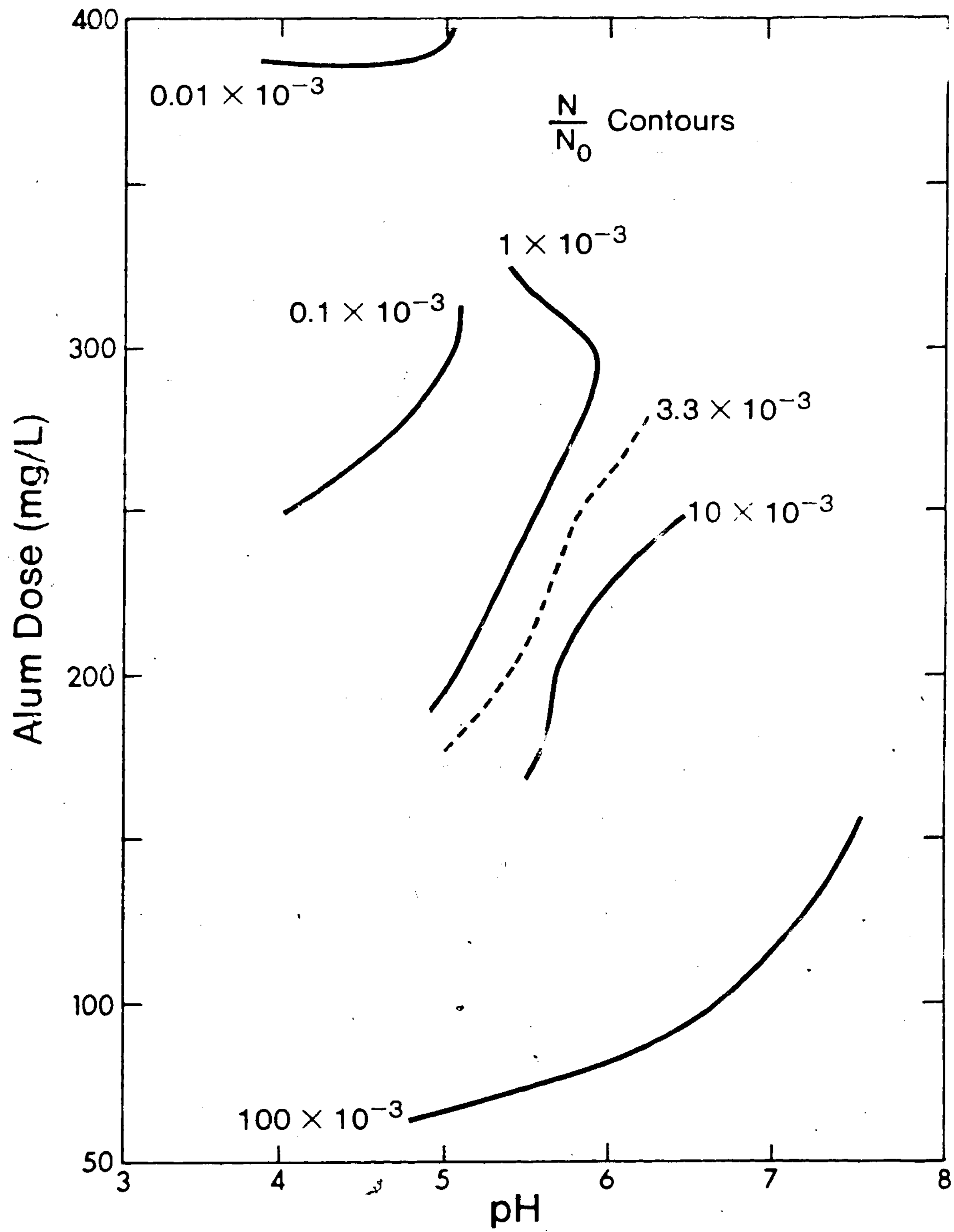
Using MIDAS (Fox and Guire 1976) and the forward step-wise regression routine, SELECT, the first order pH and alum dose terms were found to be the only significant variables. This was consistent with the earlier findings and confirmed that the responses of alum dose and pH were essentially linear.

It was possible to plot the experimental data gathered from Series 1 to 4 on a contour graph with fecal coliform removal as a function of pH and alum dose. The dimensionless number, N/N_0 where N was the number of microorganisms remaining and N_0 was the initial number present, was used in Figure 16 to illustrate the interdependence of pH and alum dose. A sharp improvement in removals was observed when pH dropped below 5.5 and the alum dose was greater than 200 mg/L. This diagram was useful in estimating the alum dose and pH to achieve a desired level of bacteria removal. It should be stressed that if pH control is not practical in a full scale application, jar tests are essential in determining the response of the system to treatment, particularly final pH.

3.9 SERIES 5

The usual method of biological wastewater quality improvement is chlorine disinfection. This practice has been considered to have little positive impact on public health, and the environment in general, since many pathogens are affected less than the commonly used indicator organisms such as fecal coliforms which are very sensitive to chlorine

Figure 6 - Fecal Coliform Removal as a Function of pH and Alum Dose



(Coulter 1982; Feachem et al. 1983). In addition, chlorine residuals have detrimental effects on fish stocks (Osborne et al. 1981).

The purpose of this experimental segment was to determine the chlorine dose required to achieve fecal coliform bacteria reductions similar to the coagulation process. A rough cost comparison could then be made between the two methods.

The experiment consisted of five, 1.0 litre samples of wastewater collected on March 1, 1984 (Figure 11) containing 7.2 x 10^5 fecal coliforms per 100 mL. These were treated with chlorine doses ranging from 1 mg/L to 10 mg/L in the form of a 0.25 per cent calcium hypochlorite solution. Samples were treated at 6° C in jars which were continuously mixed for 30 minutes. At the end of 30 minutes, a sample was extracted from each jar and treated with a sufficient amount of 1 per cent sodium thiosulphate to neutralize the chlorine residual. The samples were then analysed for fecal coliforms using Method 920 (Tentative) (APHA,AWWA,WPCF 1981). No rosalic acid was used in the culture media and the water bath was set at 35° C for 5 hours followed by 18 hours at 44.5° C. Facilities were not available for checking this method against the multiple tube fermentation method. Residual chlorine was measured using method 408A, recognizing the limitations of the method for measuring total residuals less than 1.0 mg/L. The results are tabulated in Table 15. A chlorine dose of 5 mg/L achieved

Table 15 - Series 5 Results

Cl_2 Dose (mg/L)	Cl_2 Residual (mg/L)	FC Remaining (No./100 mL)	N/N_0
1.0	*	6.5×10^5	0.90
3.0	*	3.0×10^5	0.42
5.0	*	4.9×10^3	0.0068
7.0	0.5	0	0
10.0	1.0	0	0

* undetectable using Method 408A

removals similar to about 275 mg/L of alum (Figure 16). It is likely that a greater number of viable viruses and bacteria remain in the chlorinated wastewater than in the coagulated wastewater since chlorine acts chemically to disrupt the cell wall of the organism. The various microorganisms have varying degrees of resistance to chlorine. On the other hand, coagulation only alters the surface charge of the organism allowing biocolloids to flocculate, removing microorganisms by physical means. This is independent of the composition of the cell wall.

The relative cost of the two approaches based on chemical cost alone was:

- liquid alum[1] $ 0.06 per cubic metre
- calcium hypochlorite[2] $ 0.03 per cubic metre

Additional benefits accruing to alum treatment include reductions in BOD_5, suspended solids, and phosphorus. Also toxic by-products are not produced.

3.10 SERIES 6

3.10.1 Introduction to Scale-up

At this point in the study, it is evident that chemical factors play an important role in removing fecal coliforms from the water column. Physical parameters involving fluid motion, particle transport, and duration of mixing were

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[1] $Al_2(SO_4)_3 \cdot 14.2H_2O$ - at $112.00 per tonne for a 48.5% solution
[2] 65% active Cl_2 - at $3.08 per kilogram.

found to play a relatively minor part in reducing fecal coliform numbers. Though the physical parameters were somewhat insignificant, it is desirable to know if this will remain the case when increasing the volume of wastewater treated from a jar test to a larger vessel. The objectives of this experiment were to confirm that chemical factors predominate through a 135 fold scale-up and that scale factors were not present.

3.10.2 Experimental Design

The results from the earlier jar tests indicated that alum dose, pH, and some mixing were important factors influencing fecal coliform removal. A simple factorial design would enable efficient analysis of these factors in a scaled-up experiment. Camp (1955) noted that the dimensionless number, Gt (velocity gradient times duration), was a useful parameter for analysing flocculator design and performance. On this basis it was decided to use a 2^{3-1} half fractional factorial design to evaluate the effects of alum dose, pH, and Gt on the removal of fecal coliforms. The difficulty of controlling the final pH in a full scale application precluded the use of pH as a variable in Series 6. The experimental design, settings of the variables, and results are given in Table 16.

Table 16 - Series 6 Experimental Design and Results

(a) **Generating Relation**

I = ABC — C = AB

(b) **Design Matrix and Results**

Jar No.	Main Effects			Results			
				Series 6.3		Series 6.4	
	A	B	C	log FC/100mL	pH	log FC/100mL	pH
1	-	-	+	3.89	5.8	3.83	5.7
2	+	-	-	4.79	6.3	4.52	6.2
3	-	+	-	4.74	6.3	4.76	6.2
4	+	+	+	3.76	5.9	3.82	5.7

(c) **Factors and Levels**

FACTORS	LEVELS (-)	(0)	(+)
A - rapid mix Gt *	400	11000	18000
B - flocculation Gt **	600	30300	60000
C - alum dose (mg/L)	200	300	400

* rapid mix duration: (-) 40 s (+) 180 s
** flocculation duration: (-) 60 s (+) 600 s

3.10.3 Scale-up Basis

Oldshue (1983) pointed out that chemical reactions involving particles less than 0.5 mm in diameter were independent of impeller size, therefore power input per unit volume was a suitable scale-up parameter. Design of scaled-up mixing processes involved the choice of any two of; power input, impeller speed, or impeller diameter which were interrelated by the Reynolds number, R, and the Power number, Φ, defined by:

$$R = \frac{D^2 S}{\nu}$$

where;

D = diameter of impeller (m)

S = impeller speed (RPS)

ν = kinematic viscosity (m^2/s)

and;

$$\Phi = \frac{P}{\rho S^3 D^5}$$

where;

P = power input (W)

ρ = bulk density (kg/m^3)

A third equation of interest in flocculation was the relationship between G and power developed by Camp and Stein (1943):

$$P = \mu G^2 V$$

where;

G = mean velocity gradient (s^{-1})

μ = absolute viscosity ($N \cdot s/m^2$)

V = volume of flocculation basin (m^3)

Cornwell and Bishop (1983) published calibration curves for determining G from jar tests using square jars and the standard Phipps and Bird Inc. jar test apparatus which enabled the calculation of the power input per litre. Table 17 summarizes the mixing regime for Series 6 at the Gt values listed in Table 16.

Scale-up tank volume was a compromise between a meaningful increase in volume and the logistics of sample collection and storage. A 600 mm square, 135 litre tank was decided upon as a satisfactory compromise which is illustrated in Plate 2. No effort was made to maintain geometric similarity since this does not control any mixing variables (Oldshue 1983).

The 135 fold increase in reactor size necessitated a new impeller, geometrically similar to the standard stirrer, enabling the impeller speed to be maintained in a practical range. Rushton et al. (1950a; 1950b) investigated the power characteristics of various mixing impellers. A two bladed, 100 mm diameter paddle similar to those of Plate 3 had a Power number of 1.77. Cornwell and Bishop (1983) found that the Power number was a function of the Reynolds number over the speed range investigated. They reported for the square

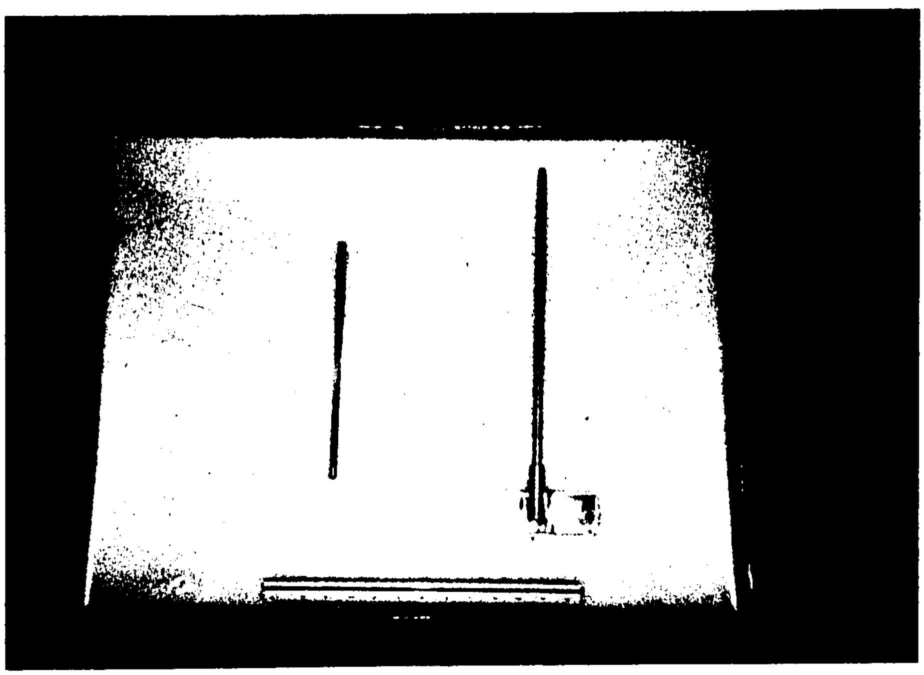
Table 17 - Summary of Series 6 Mixing Characteristics

Parameter	Jar Test Trials - Series 6.3				Tank Test Trials - Series 6.4			
	1	2	3	4	1	2	3	4
G (s^{-1}) (mixing)	40	100	40	100	40	100	40	100
G (s^{-1}) (flocculation)	10	10	100	100	10	10	100	100
Power (W/L) (mixing)	2.2×10^{-3}	1.4×10^{-2}	2.2×10^{-3}	1.4×10^{-2}	2.2×10^{-3}	1.4×10^{-2}	2.2×10^{-3}	1.4×10^{-2}
Power (W/L) (flocculation)	1.4×10^{-4}	1.4×10^{-4}	1.4×10^{-2}	1.4×10^{-2}	1.4×10^{-4}	1.4×10^{-4}	1.4×10^{-2}	1.4×10^{-2}
Total Power (W) (mixing)	2.2×10^{-3}	1.4×10^{-2}	2.2×10^{-3}	1.4×10^{-2}	0.302	1.890	0.302	1.890
Total Power (W) (flocculation)	1.4×10^{-4}	1.4×10^{-4}	1.4×10^{-2}	1.4×10^{-2}	0.019	0.019	1.890	1.890
RPM (mixing)	50	100	50	100	103	191	103	191
RPM (flocculation)	20	20	100	100	41	41	191	191
R (mixing)	3300	6700	3300	6700	20000	36000	20000	36000
R (flocculation)	1340	1340	6700	6700	7900	7900	36000	36000

bulk density = 1000 kg/m^3 at 5° - 10° C
kinematic viscosity = 1.40×10^{-6} m^2/s at 5° - 10° C
absolute viscosity = 1.40×10^{-3} $N.s/m^2$ at 5° - 10° C

Plate 2 - Scaled-up Test Vessel

Plate 3 - Mixing Impellers



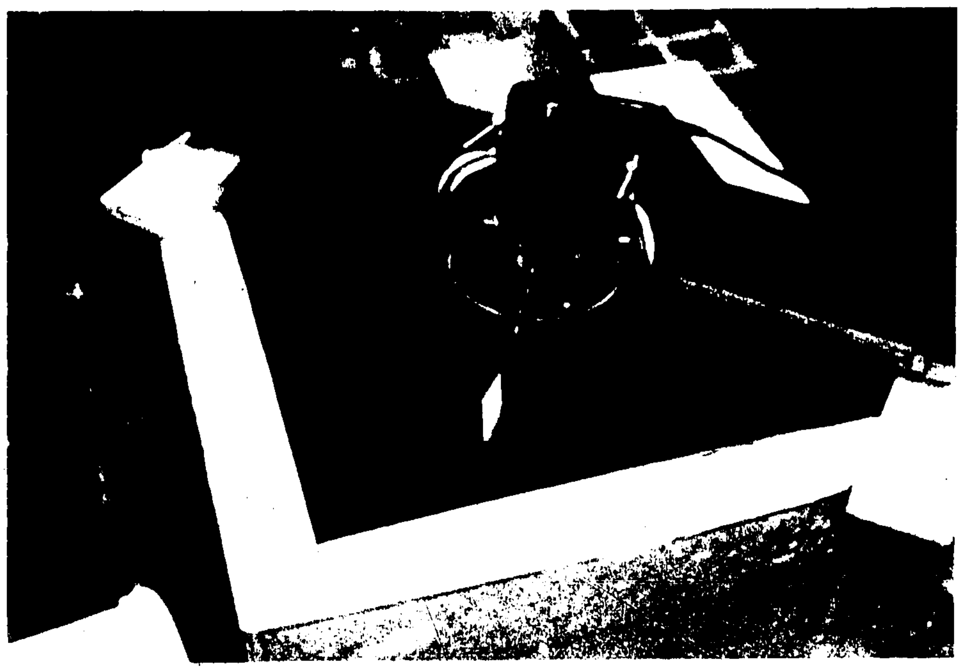
jar and the Reynolds numbers shown in Table 17, a Power number of approximately 3. The value quoted by Rushton et al. (1950b) was used since it correlated much better with the power input calculated by the method of Camp and Stein (1943). The scaled-up impeller had a diameter of 127 mm. The relative sizes of the impellers are shown in Plate 3. The mixing characteristics of the scaled-up flocculation basin are summarized in Table 17.

3.10.4 Apparatus and Procedures

The experiment was carried out using wastewater collected on March 1, 1984 (Figure 11). Jar tests and tank tests were conducted on the same day to ensure comparability of the results. Jar tests were conducted using the Phipps and Bird Inc. apparatus described earlier. The tank was mixed using a variable speed drill and the stirrer shown in Plate 4. The speed was controlled by a variac which was calibrated using a strobe light and hand-held tachometer. Two baffles, 60 mm by 310 mm, were used to prevent vortexing in the tank.

Freshly made alum was fed to the jars as a 5 per cent solution. The tanks used variable concentrations of alum; 5.4 per cent for the 200 mg/L dose and 10.8 per cent for the 400 mg/L dose. Wastewater temperature was approximately 3° C for the tank tests and 14° C for the jar tests. Initial pH was 6.7 and there were initially 5.6 x 10^5 fecal coliforms per 100 mL.

Plate 4 - Tank Apparatus



As in earlier experiments, the vessels were allowed to settle for 30 minutes prior to sampling. Sampling was carried out using a syphon device suspended approximately 75 mm below the surface of the liquid.

3.10.5 Results and Analysis

The results and final pH values are recorded in Table 16. Using the analysis techniques discussed earlier, ANOVA tables were developed for Series 6.3 and 6.4, jar tests and tank tests respectively. These are found in Tables 18 and 19. Using the mean square error value from Series 1.2, 0.011, the F-statistic was 5.59. Clearly alum dose was the only significant factor, which was consistent with earlier findings. It is important to note that the jar tests produced results nearly identical to the larger tanks indicating that scale factors were not present.

One anomoly was observed in these trials. The absolute number of remaining fecal coliforms was approximately ten times higher than could be expected from Figure 16. The explanation for this was not clear since the wastewater characterisics were similar to previous samples. The only difference was that earlier experiments involved a delay of days to weeks between sampling time and treatment time whereas Series 6 came directly from the lagoon to the treatment apparatus. It was suspected that the use of a centrifugal pump to obtain the wastewater sample from the wet well of the sewage lift station, dispersed naturally

Table 18 - Series 6.3 ANOVA

FACTOR	β	TE	MS	MS/MS_d	SIGNIFICANT α=0.05
A	-0.0174	-0.069	0.0012	0.1	No
B	-0.0451	-0.180	0.0081	0.7	No
C	-0.4692	-1.877	0.8808	80	Yes

Table 19 - Series 6.4 ANOVA

FACTOR	β	TE	MS	MS/MS$_d$	SIGNIFICANT α=0.05
A	-0.0637	-0.255	0.016	1.5	No
B	0.0573	0.229	0.014	1.2	No
C	-0.4100	-1.640	0.673	61	Yes

agglomerated bacteria which accounted for the higher numbers detected in the sample of March 1, 1984. This may also account for the poorer overall reductions. The fact that both the jar tests and tank tests had the same results indicated that scaling and temperature were not factors. On this basis, values obtained for Series 6 were not included in the pH - alum dose diagram (Figure 16).

The fourth trial of Series 6.4 was analysed before and after the treatment and the results are summarized in Table 20. Alkalinity consumption was near the theoretical value of 0.50 mg $CaCO_3$/L per mg/L of alum ($Al_2(SO_4)_3 \cdot 14.2H_2O$) added quoted by Clark et al. (1971). Sulphate concentration increased in approximately stoichiometric proportions. Suspended solids increased leading one to suspect that aluminum hydroxide and/or aluminum phosphate precipitates remained in the water column after the 30 minute sedimentation period. Stored, quiescent samples were observed to deposit a whitish sediment after overnight storage. Phosphorus reductions were much lower than expected at the end of the sedimentation period. Analysis of the supernatant after overnight settling revealed a phosphorus content of 0.6 mg/L confirming that some of the suspended solids were indeed aluminum phosphate. BOD_5 reduction was relatively small, thus it was likely that most of the remaining BOD_5 was in the soluble form.

Sludge collection in the tanks was measured. The 400 mg/L alum dose generated approximately 15 mm of sludge in

Table 20 - Series 6.4 Treatment Performance

Parameter	Units	01.03.84 (before)	Series 6.4 Trial 4 (after)
$CBOD_5$	mg/L	104	71
TSS	mg/L	31	48
PO_4^{3-}-P	mg/L	9.2	2.7*
SO_4^{2-}	mg/L	60	260
Alkalinity (to pH 4.5)	mg $CaCO_3$/L	311	134
Temperature	°C	3	3
Ionic Strength		0.0147	0.0144
pH	-	6.7	6.0
Fecal Coliforms	No./100 mL	5.6×10^5	6.6×10^3

* filtered value of 0.6 mg/L

the 135 litre tank after 14 hours. This was approximately equivalent to 0.04 cubic metres of sludge per cubic metre of lagoon contents. The 200 mg/L alum dose produced sludge approximately two thirds of this value after the same time period.

3.11 SERIES 7

3.11.1 Introduction to Full Scale Trial

Conclusions from the laboratory investigation indicated that:

1. alum dose and pH principally controlled the removal of fecal coliform bacteria;
2. some mixing and flocculation was necessary but bacteria removals were insensitive to these parameters; and
3. no factors of scale were present.

Therefore, the field trial was planned on the following basis:

1. reduce fecal coliforms to less than 10 per 100 mL;
2. determine the initial number of bacteria present prior to treatment to enable calculation of the N/N_0 value;
3. estimate the alum dose and final pH level from Figure 16 to achieve the N/N_0 value without adjusting lagoon pH by means other than alum addition; and
4. perform a jar test at this dose to confirm the design based on Figure 16 and to indicate the likely outcome of the full scale trial.

The trial was conducted on April 26, 1984. The weather was generally sunny with a temperature of 8° C, and a light northeasterly breeze.

3.11.2 Jar Test

Wastewater sampled on April 24, 1984 contained approximately 3000 fecal coliforms per 100 mL. Fecal coliform levels less than 10 per 100 mL corresponds to an N/N_0 ratio of 0.0033. From Figure 16, it appears that an alum dose of 300 mg/L and a final pH of 6.4 would provide the desired removal of bacteria. The experience of previous experiments indicated that this alum dose would drop the pH to the desired level.

The jar test was conducted using the same apparatus as in earlier tests. Rapid mixing was carried out for 30 s at a G of 100 s^{-1} and flocculation was done for 600 s at a G of 10 s^{-1}. Undiluted liquid alum[3] was used to treat the jar. Figure 16 was based on technical grade alum ($Al_2(SO_4)_3 \cdot 18H_2O$) therefore a correction must be made for liquid alum on the basis of 1.11 mg/L of technical grade alum per 1.0 mg/L of liquid alum. The final pH was 6.1. Following 30 minutes of sedimentation, a sample was analysed for fecal coliform bacteria. No fecal coliform bacteria were recovered using the membrane filtration method.

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[3]Available as a 48.5 % solution of $Al_2(SO_4)_3 \cdot 14.2H_2O$.

3.11.3 Reagents

Liquid alum was ordered following completion of the jar tests using on an estimated lagoon volume of 34.5 million litres (±10 per cent). The delivered amount of liquid alum was 19430 kilograms resulting in a theoretical dose of 304 mg/L of $Al_2(SO_4)_3 \cdot 18H_2O$.

3.11.4 Equipment and Labour

Two 4.9 metre aluminum boats equipped with 25 HP outboard motors were used to distribute alum, mix the lagoon, and induce flocculation. Each boat had a 200 litre drum for storing alum. The alum was fed over the stern of the boat using a 681 litre per minute gasoline pump (Plate 5). A 4 metre long wharf was placed at the lagoon edge to facilitate the transfer of alum from the shore to the boats (Plate 6). Alum was transferred from the tanker truck to the boats by gravity through an 8 metre long, 75 millimetre diameter hose supplied by the hauler.

Three men were considered to be the minimum number required for an efficient operation. Two were employed operating the boats while the third was responsible for transfering the alum.

3.11.5 Procedure

Each boat was assigned to cover approximately half of the lagoon surface area. Alum was distributed at the operators' discretion using the guiding principle of uniform

Plate 5 - Chemical Feed System

Plate 6-- Loading Apparatus

distribution over the lagoon surface. Rapid mixing was accomplished by directing the liquid alum into the propeller wash of the boat followed by a couple of extra passes with the boat to ensure complete mixing. Plate 5 illustrates a typical pass with the boat. Near the end of the operation, one boat systematically covered the entire lagoon at the optimum boat speed for maintaining bulk fluid motion.

The lagoon was allowed to settle for 27 hours prior to sampling and draining the cell.

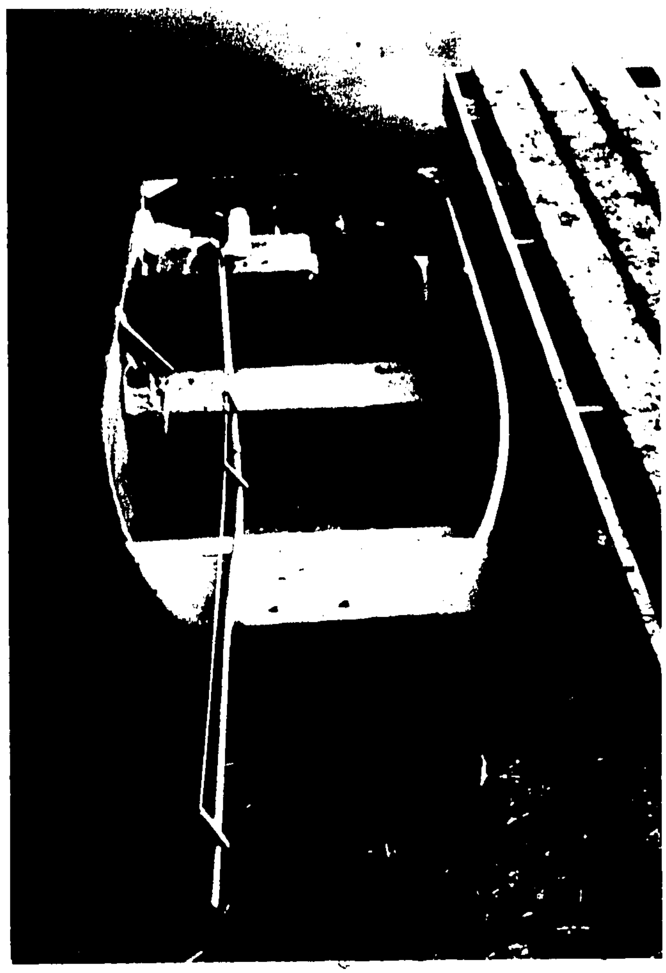
3.11.6 Sampling

Grab samples were obtained at the end of the clarification period at two locations (Figure 11). A vertical profile of the water column was taken in the centre of cell 2 at 0.5, 1.0, and 1.5 metres. The outfall of cell 2 was the inlet to cell 3. This was also sampled (Figure 11).

A small aluminum dinghy was used in collecting samples from the centre of cell 2. A controlled depth sampler was used (Plate 7). Temperature and pH were measured on site. Temperature ranged from 10° C at 0.5 metres to 8° C at 1.5 metres. pH was consistently 6.4. Fecal coliforms were determined the following day after storing samples in the dark at 4° C.

Sampling was continued for the next two days at the inlet to cell 3 since this was felt to be representative of cell 2.

Plate 7 - Sampling Equipment



3.11.7 Observations

Shortly after the operation started at 1100 hours on April 26, floc particles were clearly discernable in the treated water column. As the operation wore on, the entire lagoon contained similar sized floc particles, approximately 6 millimetres in diameter. The particles were observed to be moving constantly which contributed to their agglomeration. The motion was attributed to residual boat action and the light northeasterly wind.

The lagoon colour changed from black to milky white in the immediate vicinity of the alum addition followed by a rapid progression to a pale green which persisted for the duration of the trial. Clear water was observed in the shallows following 60 minutes of sedimentation after alum addition was completed.

Overnight sedimentation had little apparent effect, the lagoon appearing turbid and having a pale green colour. However, examination of a 1.0 litre beaker of supernatant obtained at various depths revealed that the water was very clear suggesting that the turbidity and pale green colour resulted from refraction of sunlight by a finely divided colloidal suspension, likely an aluminum precipitate. Cell 2 commenced being drained 27 hours after the alum application ceased. Considerable foaming was observed at the inlet to cell 3 (Plate 8). Subsequent discharge of cell 3 into the Sturgeon River also produced considerable foaming posing aesthetic problems. The degree of foaming was

Plate 8 - <u>Foaming at the Outlet</u>

associated with the turbulence of the effluent and this suggests that energy dissipation is an important consideration in outfall design to minimize foaming.

Following drawdown of cell 2, the lagoon side slopes and bottom were observed to have accumulated insignificant amounts of sludge. The residual supernatant in cell 2, approximately 450 millimetres, was still clear a week after treatment.

3.11.8 Results and Analysis

Results of the sampling program over the 72 hour period following sedimentation are tabulated in Table 21. The highest value observed was at the cell 3 inlet, 30 minutes after opening the drain valve to cell 2. As cell 2 drained over the ensuing 45 hours, the bacteria count dropped to 1 per 100 mL.

Overall reductions in $CBOD_5$, suspended solids, and phosphorus were good as shown in Table 22. The remaining phosphorus was probably present as insoluble aluminum phosphate which was supported by the suspended solids and aluminum values. Aluminum analysis was done using flame atomic absorption and was not intended to be definitive with respect to aluminum but rather provide a rough indication of aluminum content. Aluminum was undetectable at the final pH. However, subsequent acidification of the sample produced the value shown in Table 22 supporting the hypothesis of some aluminum phosphate precipitate remaining in the water

Table 21 - Fecal Coliform Analysis: Before and After Treatment

Sample Date	Location	FC per 100 mL
25.04.84 (Before)		
1700 hours	Cell 2 centre, mixed	4800
	Cell 2 centre, 0.5 m	3000
	Cell 2 centre, 1.0 m	1800
	Cell 2 centre, 1.5 m	2200
27.04.84 (After)		
1700 hours	Cell 2 centre, 0.5 m	7
	Cell 2 centre, 1.0 m	11
	Cell 2 centre, 1.5 m	23
1800 hours	Cell 3 inlet	33
28.04.84 (After)		
1000 hours	Cell 3 inlet	7
1700 hours		4
29.04.84 (After)		
1500 hours	Cell 3 inlet	1

Table 22 - Full Scale Treatment Performance

Parameter	Units	25.04.84 1700 hours	28.04.84 1700 hours
$CBOD_5$	mg/L	54	35
TSS	mg/L	15	1.5
PO_4^{3-}-P	mg/L	8.7	0.3
SO_4^{2-}	mg/L	72	185
Alkalinity (pH 4.5)	mg $CaCO_3$/L	293	183
Temperature	°C	10	10
Ionic Strength	[illegible]	0.0130	0.0132
pH	-	7.2	6.4
Fecal Coliforms	No./100 mL	4800	4
Al^{3+}	mg/L	*	1

* undetectable using flame atomic absorption

column.

Final pH was slightly higher than the jar test results which may be explained by additional buffering from the lagoon benthic deposits and an alum dose that may have been slightly less than the calculated dose. Alternatively, concentration gradients may have been present in cell 2 which grab samples would not detect.

Sludge accumulation was insignificant compared to Series 6 indications. This probably was a reflection of the considerably larger amount of suspended solids in the Series 6 trials.

3.11.9 Cost Estimate

Treatment of this 34500 cubic metre lagoon had a cost of approximately $ 0.07 per cubic metre for labour and materials. If the boats, motors, and pumps were purchased outright and assuming a 5 year depreciation at 10 per cent per year, an additional annual cost of $ 2220.00 would be involved. Renting these items would likely be the most economical route to follow unless large scale applications of this upgrading technique were anticipated. These cost figures are summarized in Table 23.

Table 23 - Treatment Cost Estimate

Category	Item	Total Cost $	Unit Cost $/m³
Capital Cost	2 - 4.9 m boats	3600.00	
	2 - 25 HP motors	3600.00	
	2 - 681 Lpm pumps	1200.00	
	cost	8400.00	
	annual cost	2220.00	
Operating Cost	0.56 kg liquid alum per m³ @$ 0.112/kg	2200.00	0.06
	24 man hours @ $ 20.00/hr	480.00	0.01
	cost	2680.00	0.07

4. SUMMARY AND CONCLUSIONS

4.1 SUMMARY

Cold climate lagoons provide secondary quality treatment during the summer months but they provide reduced treatment efficiencies during the winter. An upgrading measure has been to use controlled discharge lagoons to store lagoon contents over the winter so that they can be discharged during the spring runoff period to take advantage of the extra dilution. This practice is not desirable since large numbers of microorganisms are released at the same time posing a potential health problem to downstream water users. In addition, high levels of BOD_5, suspended solids, and phosphorus erode the water quality of the receiving water body.

The traditional method of biological water quality control, chlorine disinfection, has numerous drawbacks including lack of effect on viruses, some pathogenic bacteria, and protozoan cysts, residual toxicity in the receiving water, and the formation of carcinogenic chlorinated organics. An alternative to this process is coagulation and flocculation.

An investigation of the effects of eight chemical and physical parameters on the removal of fecal coliform bacteria was carried out using jar tests based on factorial designs and statistical analysis. The jar test results were confirmed using a scaled-up vessel based on power input for mixing and flocculation. The final stage saw an operating seasonal discharge lagoon at Gibbons, Alberta treated with liquid alum for reduction of fecal coliform bacteria prior to the spring discharge.

4.2 CONCLUSIONS

- pH and alum dose were highly significant factors in achieving removals of fecal coliforms from the lagoon wastewater tested.
- Rapid mixing and flocculation were marginally implicated in fecal coliform reductions, suggesting that only a minimum of mixing was required.
- No factors of scale were observed.
- The alum dose obtained from a pH - alum dose diagram which used N/N_0 as a measure of fecal coliform removal efficiency, produced removals in accordance with those predicted by the diagram at the measured final pH value.
- Reduction of fecal coliforms by 99.9 per cent was accompanied by a suspended solids reduction of 90 per cent and a phosphorus reduction of 97 per cent. $CBOD_5$ was reduced by 35 per cent.

- The operating cost of this upgrading technique was \$ 0.07 per cubic metre of wastewater. Boats, motors, and pumps were the only capital investments. The impact of the capital cost on the overall treatment cost depends on the amount wastewater treated annually. In some cases, it may be very economical to rent the equipment for the occasion.
- Future work should focus on commercial application of this technique to existing lagoons which are having operational problems or discharge into water courses with minimal assimilation capacity.
- On the basis of chemical costs alone, alum coagulation is approximately twice the cost of chlorine disinfection for reducing fecal coliforms to similar levels. However, alum coagulation has more treatment benefits than chlorination and does not produce toxic by-products.

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