MANAGEMENT OF
WATER TREATMENT PLANT RESIDUALS
IN NEW ZEALAND

HANDBOOK
September 1998

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a Subgroup of the New Zealand Water and Wastes Association

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Preface

The Water Supply Managers Group is a voluntary group representing many New Zealand water supply authorities. The Group is a subgroup of the New Zealand Water and Wastes Association (NZWWA), a non-profit organisation, which has as a mission: "The advancement and application of fundamental and practical knowledge to natural water resources, water use and wastes."

Members of the Water Supply Managers Group contribute funds for research projects that have a common application to a large proportion of its members. Research projects are approved by the Group at annual meetings.

In the next few years, many operators and owners of water supply utilities will be seeking resource consents relating to the management of wastes produced at water treatment plants. They will be considering similar issues and potential solutions. A common resource handbook and reference guide will assist this work.

The management of water treatment plant wastes is a national issue, so the Water Supply Managers Group approved the preparation of national guidelines, to present handling and disposal methods that meet the requirements of the Resource Management Act.

Use of this Handbook will accelerate the process of reducing waste discharges to natural waters, which is in line with the Government's long-term environmental objectives and priorities. Therefore the project satisfied the requirements of the Ministry for the Environment's Sustainable Management Fund. The Water Supply Managers Group is grateful for the Ministry's assistance.

1. Introduction

1.1 Background

Currently some New Zealand water suppliers deliver groundwater, or even surface water, directly into their distribution system. Some of these supplies may be disinfected, or have the pH corrected or fluoride added, or be aerated to remove gases. Apart from residues resulting from the addition of a chemical solution or powder, such treatment of the water supplies produces no wastes.

Other water supplies include a treatment process to remove some or all of colour and turbidity, iron and manganese, tastes and odours, and micro-organisms. They involve a solids separation process that generates a range of waste products, known as residuals, which must then be disposed of. Many water suppliers still continue the traditional practice of discharging these residuals to natural water without any further processing.

Times are changing. Our society has modified its rules and expectations regarding the use and abuse of natural resources, including its water supplies. Technical, legislative and community standards have changed. In particular the Resource Management Act 1991 requires specific consents to be gained for discharge of residuals to the environment. Response from questionnaires issued by the NZWWA in August 1996 indicated that some 50 new consents were expected nationally over the next few years. Thus many water suppliers will be applying in the near future for consents (or renewals) to discharge residuals to the environment from their water treatment plants.

The degree of protection to the public through further treatment of the raw water is likely to increase in coming years. Thus managers and owners of all water treatment operations will need to reassess their production of wastes and adopt an appropriate long term management plan.

This Handbook has been written to assist in this process.

The Handbook aims to:

- ensure that disposal of water treatment plant residuals is adequately covered within the legislative framework,
- discuss the adverse environmental effects that may result from their disposal,
- offer a consistent, national approach to mitigating these effects,
- describe the options available for handling water treatment plant residuals.

The Handbook addresses each type of water treatment process used in New Zealand that produces a waste, and the further processing and disposal options. Anticipated changes in technology are also covered.

1.2 The Reasons for Change

1.2.1 Future Treated Water Quality Requirements

Over the coming years, treated water quality requirements will change due to factors ranging from ongoing research into the health effects of contaminants to public perceptions of risk from their drinking water supply.

The 'baseline' for considering future requirements is the *Drinking Water Standards for New Zealand (DWSNZ), 1995* (Ministry of Health, 1995).

Whilst the 1995 DWSNZ are not presently enforceable by law, the Ministry of Health grades water supplies *"to provide a public statement of the extent to which a public water supply achieves and can ensure, a consistently safe and wholesome product"*. This grading is published by the Ministry of Health in the *Register of Community Drinking Water Supplies in New Zealand*. This results in considerable public and political pressure to attain an 'A' grading.

In many countries the drinking water standards are enforceable by law. The Ministry of Health has received submissions on whether this should also be implemented in New Zealand. Although there does not appear to be the political will to make this change at present, it should be anticipated that this could happen within the longer term planning period of the water supplier.

The Priority 1 determinands in the 1995 DWSNZ include only the microbial contaminants; faecal coliforms, *Giardia* and *Cryptosporidium*, because they pose an acute health risk. The specific compliance criteria are now based on treatment techniques rather than on actual measured concentrations. The higher water quality goals are driving treatment towards improved coagulation and filtration with tighter control of filtrate turbidity.

This discussion suggests the following trends:

- Some existing water supplies that do not have efficient coagulation/filtration treatment will be or have been downgraded with the change to risk based criteria. Local community/customers are not all tolerant of this diminished status.
- In order to attain a higher grading, more physical barriers to passage of contaminants will be required which will increase the number of coagulation/filtration and/or microfiltration processes.
- The additional water treatment processes will generate an increase of residual solids for disposal.

1.2.2 Resource Management Act

The disposal of water treatment residuals is most likely to require resource consents under the Resource Management Act 1991(RMA).

The RMA empowers regional councils and territorial authorities to grant or refuse consents. Applications for consents related to the fate of solid and liquid residuals are considered by the Regional Council and applications related to land for development of facilities are heard by the Territorial Authorities. Such applications may well be considered in a joint hearing.

The application must be supported by an Assessment of Effects on the Environment (AEE) which should include, in addition to a description of the proposal, a description of the receiving environment, any possible effects, and the means that are proposed to avoid or mitigate these.

It is necessary to identify those persons interested in, or affected by, a proposal, document consultation undertaken, and record the response to the views of those consulted.

An evaluation of different disposal options should also be provided.

The process of preparation of application, notification and hearing can be expected to take at least six months.

1.2.3 Local Government Amendment Act (Nº 4)

The Local Government Act (LGA) has been amended by the Local Government Amendment Act (Nº 4), 1996, to expand the waste management function of territorial authorities to promote waste reduction, reuse, treatment, recycling and resource recovery, as well as refuse collection and disposal. While most territorial authorities focus more on solid refuse, the term 'waste' is deliberately not defined by the LGA and is used in its broadest sense. This definition includes residuals from water treatment operations.

Every territorial authority has a duty to promote effective and efficient waste management within its district and in doing so, shall:

(a) have regard to the environmental and economic costs and benefits to the district

(b) ensure that the management of waste does not cause a nuisance or be injurious to health

1.3 Categories of Residuals

Water treatment plant residuals are typically derived from suspended solids removed from the raw water plus derivatives of chemicals or other introduced compounds added during the treatment processes. There are generally considered to be four major categories of residuals produced from water treatment processes. Not all of these categories are common or likely in New Zealand. Table 1.1 summarises the major categories and their relevance to New Zealand.

Table 1.1: MAJOR RESIDUALS CATEGORIES

Category	Typical Content	Production Processes	New Zealand Relevance
Sludges	raw water suspended solids	sedimentation	Yes
	chemical reaction products	coagulation filter backwashing iron and manganese removal lime softening	Yes Yes Yes No
Concentrates	brines, membrane reject water and spent backwash	IX regeneration membrane filtration	No Yes
Ion exchange (IX) resins, spent GAC, spent filter media	spent resin, activated carbon, sand, anthracite, diatomaceous earth	IX regeneration filter backwashing	No Yes
Air Emissions	off-gases	air stripping odour control ozone destruction	No No Not currently

It can be seen that New Zealand water supplies do not generally require softening, ion exchange or stripping of organic compounds. Thus the main categories of residuals likely for consideration comprise sludges and filter backwash including some spent filter media particles.

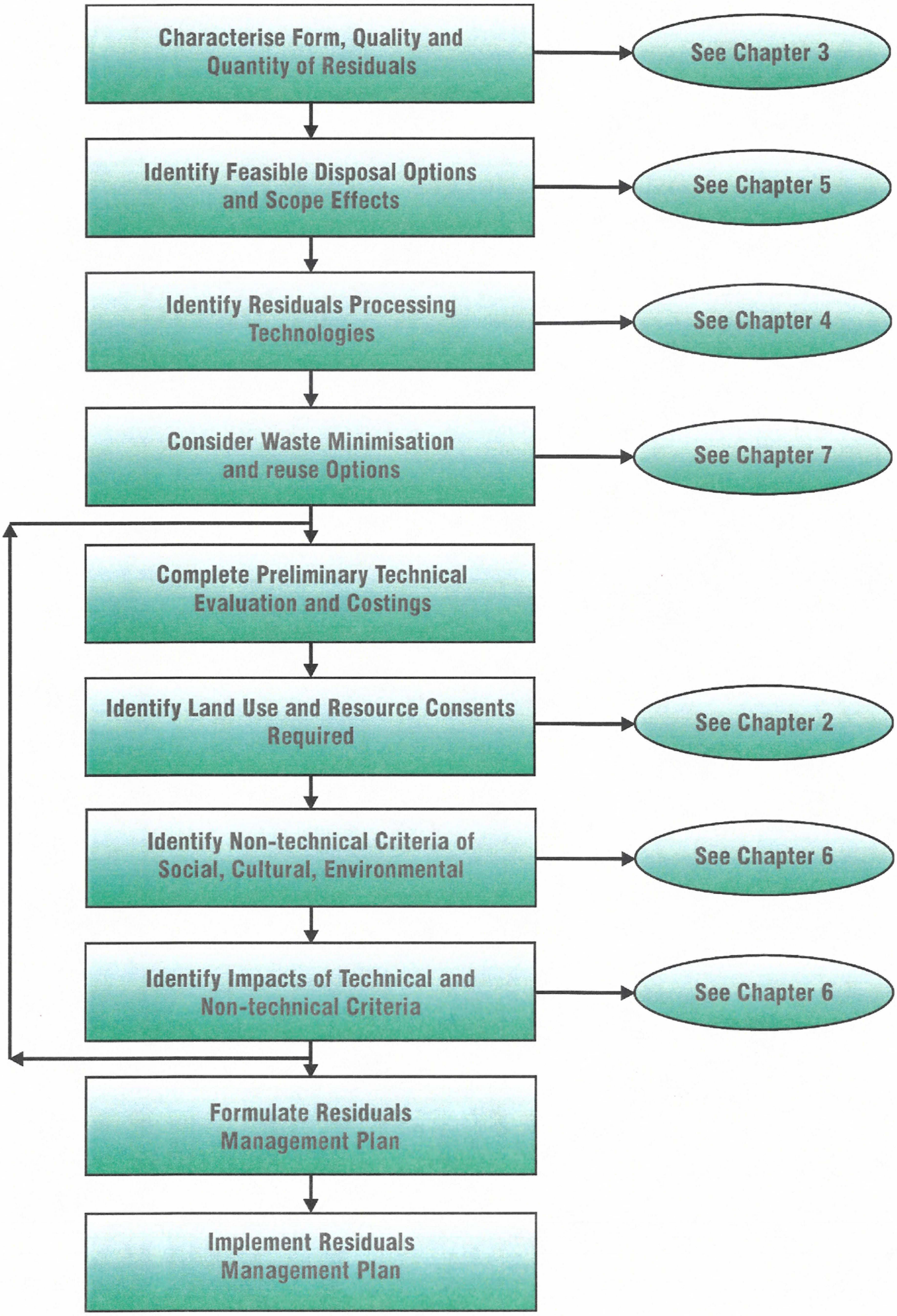
Faced with a wide range of sludge processing technologies and disposal options, the task of producing a residuals management plan can be daunting.

The technical criteria used in the selection of the final management plan will differ from user to user. Economic, cultural, social and environmental factors are also influential and site-specific, and must be included in any final selection.

1.4 How to Use This Handbook

This Handbook offers background, summary information, and references on the components which make up a comprehensive residuals management plan. Developing a successful residuals management plan requires an understanding of the character and production of residuals, options for their processing and disposal, the effects and impacts of such options, and the statutory approvals processes governing consents. Figure 1.2 presents a generic flowchart highlighting key steps necessary in the process of formulating a residuals management plan. Each step in the process is keyed to a section of these guidelines. Figure 1.2 can also be used as a quick reference to find specific information within this handbook.

Figure 1.2: KEY STEPS TO A RESIDUALS MANAGEMENT PLAN



2. Statutory Approvals Process

2.1 Statutory Requirements

The RMA sets the statutory requirements for the management of activities and their effects, and empowers regional councils and district/city councils to formulate management plans and standards, and to manage any consent processes including the granting or refusal of consents.

The responsibilities of regional councils lie principally with discharges to land, water and to air. Those of the district councils are with land use activities.

The disposal of sludge is generally a land based activity (unless it is disposed of directly to water) but which has potential to result in discharge of contaminants into land, water, or air. Contaminants are very broadly defined as including any substance which when discharged onto or into land or into water, or air, changes or is likely to change the physical, chemical or biological condition of the water, land or air into which it is discharged.

Section 15 of the RMA prohibits the discharge of a contaminant into land, water, or air, unless the discharge is expressly allowed by a rule in the regional plan, a resource consent or regulation.

Regional and district councils will each respectively have a regional or district plan. These plans will include objectives and policies addressing the various matters for which they have environmental management responsibilities. They will also include rules for the purpose of implementing the policy.

Regional and district plans can provide for circumstances where consents are not required and activities can be carried out as of right. These are termed permitted activities and are those that can be managed and that will have little if any potential external environmental impacts. They will also provide for circumstances where activities are permitted but at the discretion of the council (discretionary activities). Where activities are not provided for they are termed “non complying”.

There are few regional plans however, which have been prepared under the RMA. At this stage and until the completion of the process of plan development, councils will rely upon a composite of bylaws and notices previously adopted and developed under earlier legislation such as Water and Soils Conservation Act (1967), Soils Conservation and Rivers Control Act (1941) and Clean Air Act (1972). This composite is termed the Transitional Regional Plan.

Transitional Regional Plans are unlikely to have included in them rules expressly allowing discharge of sludge waste, unless in the case of authorised landfills. It must be expected therefore that applications for consent will have to be lodged for a “non complying” activity.

Sludge disposal is most likely to involve both discharges and land use activity, and therefore require consideration and possible consent from both regional and district council authorities. In these circumstances applications, if required, must be made separately to both authorities. The hearings should be held jointly, with the regional council normally being responsible for the management of the hearing process.

In addition to the actual discharge activity, land use consents may well be required for process plant and reticulation.

Table 2.1 is a schedule of activities which may be involved in sludge disposal and for which consents may be required.

Table 2.1: SCHEDULE OF ACTIVITIES REQUIRING CONSENTS

Activity	Council
Pipeline installation	DC
Process plant and facilities	DC
Earthmoving	DC & RC
Discharge to land	DC & RC
Discharge to land in circumstances where contaminants may enter natural waters	RC
Discharge of contaminated air from:	RC
• Plant	
• Land	
• Water	
Discharge to water	RC

DC = District Council; RC = Regional Council

Applications for consents are sometimes considered by local authorities without going through the process of public notification and calling for submissions. However, this is only in circumstances where the effects are considered to be minor. This is unlikely with respect to sludge disposal. Normally it can be expected that an application will be notified for public submission. Figure 2.1 sets out the application process pathway

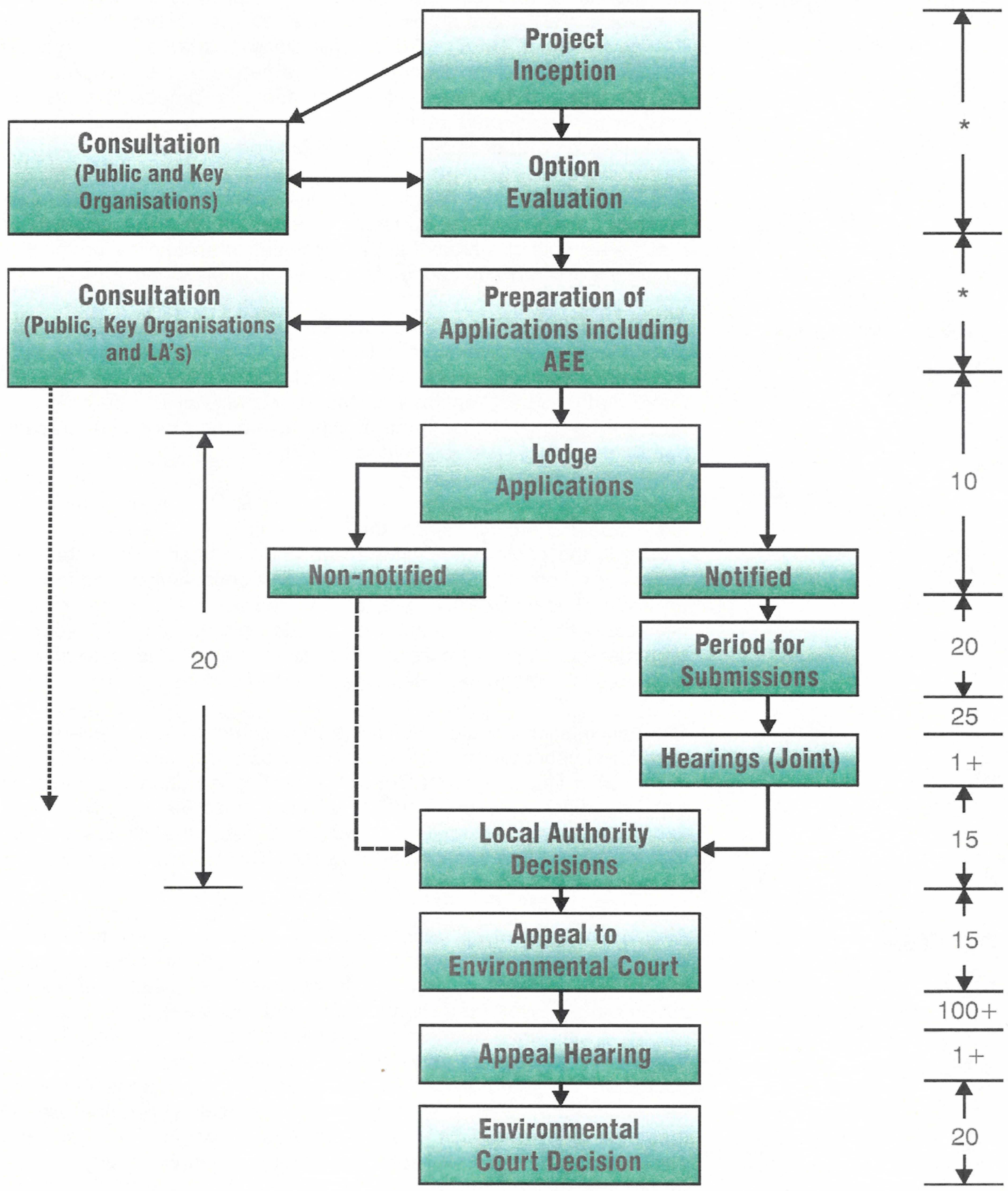
2.2 Public Works Act

If land not already owned is required for the work, then the powers of compulsory acquisition under Sec 22 of the Public Works Act can be used by a council if it becomes necessary.

If the work is not undertaken by the council, but instead by a Network Utility Operator (NUO) as defined by Sec 166 of the RMA, then the Public Works Act can still be used, if necessary, but the NUO must first have been approved by the Minister for the Environment, as a "Requiring Authority". In all cases, before the Public Works Act can be used to compulsorily acquire land, it is necessary to show that adequate consideration has been given to alternative sites and methods of achieving the authority's objectives. This is the same process that is required in pursuing a designation under the RMA for the site that is required.

Thus the council or NUO would complete activities sufficient to obtain their consents, then negotiate for land purchase with the owner. If land purchase negotiations failed, the Public Works Act could then be used for compulsory purchase based on the previously demonstrated consideration of alternatives.

Figure 2.1: APPLICATION PROCESS PATHWAY FOR RESOURCE CONSENTS



NOTES:

(1) The periods expressed are working days. They are Statutory maximums (up to the “Appeal to the Environment Court”) which an efficient local authority may reduce. However, in special circumstances the local authorities have the authority to increase any of the periods by 100%.

(2) These periods marked * will vary according to the scope and scale of the project. Generally a period of several months should be allowed for.

(3) The periods from “Appeal” to “Environmental Court Decision” are not limited, and those given are based on current 1997 experience.

(4) At any stage of the consent application process prior to making a decision the council can request additional information under section 92 of the RMA. This will add additional time (delays) to the process.

2.3 Project Description

An application is required to include a description of the activity for which consent is sought and an assessment of effects the activity may have on the environment (AEE). For sludge disposal that activity assessment should include a clear description of the nature of the sludge discharge, the sensitivity of the receiving environment, the proposals to avoid or at least mitigate adverse effects, and any proposals for monitoring. Any possible site or process alternatives should be described.

It is important that the description of the activities be comprehensive and thorough, and that all the consents required are identified and applied for. If some aspects of a project are not included in an application the consent cannot be granted with consequential delays to the whole project.

2.4 Option Evaluation

The RMA (1991) (Fourth Schedule) suggests that an assessment of effect should include, in addition to a description of the proposal, a description of alternative locations and of methods of discharge. It is not necessary to prove that the "best" option is chosen but it will have to be demonstrated that, for the option chosen, the effects have been properly addressed and detrimental effects will be avoided or mitigated.

2.5 Consultation

Although it is not mandatory to consult, the RMA (1991) suggests that an assessment of effects identify those persons interested in or affected by a proposal, the consultation undertaken and any response to the views of those consulted. It is advisable to undertake consultation, and to do this at an early phase of the project, before decisions are made on site and disposal options. This consultation should be undertaken in order to gain an understanding and appreciation of others values and to be able to give proper consideration to these in the choice of options.

The consultation should involve the local community, key interest groups such as environmental interest groups, Maori, and government agencies, in particular Department of Conservation. Close liaison should be formed with the officers of the consent authorities so as to ensure that the application covers all consents required, and that the officers are satisfied that the information supplied is adequate. This avoids delays later caused by requests for further information.

2.6 Hearings

After the lodging of an application the local authority will consider it and make a decision. Before doing so it will, if deemed necessary, publicly notify the application calling for submissions. After the period for submissions closes (usually a maximum of four weeks) the local authority will set a date for a hearing giving at least two weeks notice. There is an opportunity at this stage to review the submissions and to meet with submitters for the purpose of addressing their concerns and resolving issues raised. The local authorities may themselves arrange pre hearing meetings to facilitate this. If the submitters concerns can be satisfactorily dealt with by, for example, agreement on operational conditions, these can be proffered at the hearing.

Within three weeks of completion of the hearing a decision will be issued. It is open to any party involved to appeal against the decision to the Environmental Court. The Environment Court hearing is a judicial process, with a limited number of designated judges, normally heavily committed to hearing other appeals. This process can be expected to take a further six months at least.

3. Properties and Production of Sludge

3.1 New Zealand Experience

It is important to understand the nature of New Zealand waters and water treatment.

The range in raw waters generally excludes heavy chemical or organic pollution and hard waters. The New Zealand experience is important because it is a little different from overseas:

- the waters are often much softer (there are no water softening plants currently in regular use for public water supply);
- the river sources can contain more clay (runoff from steep hills), and reservoirs contain more natural organic matter (resulting from the bush catchments);
- the raw waters are not as contaminated with chemicals so there is no advanced oxidation or activated carbon treatment;
- the only treatment groundwaters may need is some aeration to remove carbon dioxide, filtration to remove fine sand, iron and manganese removal, and disinfection;
- apart from a few waters containing groundwater with geothermal origins, water sources used for drinking water in New Zealand do not contain hazardous substances.

Table 3.1 summarises the number and type of water supplies in New Zealand, 1997 (National WINZ Database). The following notes apply:

Table 3.1: SUMMARY OF NEW ZEALAND WATER SUPPLIES

	Groundwater source(s) only	**Surface source(s) only**	**Mixed sources**	**Total**
Full Treatment	1	77	2	80
Partial Treatment	29	38	0	67
No Treatment	164	117	8	289
Total	194	232	10	436 plants

Notes for Table 3.1

Criteria:

- "Treatment Plant" refers to either a physical plant or a nominal plant where no actual treatment is performed for a given supply.
- The data set is restricted to those treatment plants that have been formally graded. Usually, but not always, such plants serve communities of 500 or more people.

Table 3.2: SUMMARY OF ALL NEW ZEALAND TREATMENT PLANTS

Treatment plants in database	1632 currently active
Plants meeting above criteria	436 plants supplied by 544 sources
Sources for these plants	256 Groundwaters, 286 Surface waters, 2 Rainwater

Incomplete Definitions

The treatment plant types of Full, Partial and No Treatment are as specifically defined by the Ministry of Health in its 1993 notes for the Grading of Drinking-Water Supplies (Ministry of Health, 1993). The following is a brief summary of these definitions, but may be misleading at the detail level without the full text:

Full: Disinfection with chlorination or equivalent; appropriate filtration with coagulation/flocculation and sedimentation/clarification where appropriate.

Partial: Chlorination or filtration with chlorination; compliance with Appendix III of *Giardia and Giardiasis in New Zealand (Ampofo, 1991)*

None: Neither “full” nor “partial” treatment.

Table 3.3 summarises the responses to the August 1996 NZWWA questionnaire on unit processes, chemicals used and sludge disposal locations. The responses covered 122 water supplies and 76 suppliers.

Table 3.3: NUMERICAL SUMMARY OF 1996 NZWWA QUESTIONNAIRE RESPONSES

Unit Processes	Nº	Chemical Usage	Nº	Sludge Disposal	Nº
Strainer	9	None	26	Freshwater	50
Sedimentation	93	Alum	40	Marine	1
Filtration	86	Ferric Chloride	1	Sewer	26
DE Filters	1	PAC	18	Wetland	3
DAF	1	Polymer	55	Lagoon	12
Carbon Filter	1	Permanganate	1	Land	5
Microfilter *	1	Lime	1	Landfill	15
Oxidation	1				
Totals	193		142		112

* commissioned 1998

3.2 Water Treatment and Residuals Production

Potable water treatment processes are aimed at producing safe drinking water. Water treatment residuals are those materials removed during the treatment process, along with any transport water removed with them. The following general categories of water treatment are common to many countries.

- **Coagulation/Filtration Plant:** Typically used to remove turbidity and pathogenic organisms. May also be used to remove colour, taste, and odour causing compounds from the water supply. May be varied to include aeration and oxidation processes for removal of iron and manganese. Unit processes may include screening, microstraining, aeration, oxidation, sedimentation, coagulation/flocculation, filtration, disinfection, and dissolved air flotation. Other non-chemical variations include direct filtration, diatomaceous, earth filtration, and slow sand filtration.

- **Precipitative softening plant:** A variation of a coagulation/filtration plant using additional processes to reduce water hardness. Additional unit processes include lime softening. Not likely in New Zealand.

- **Membrane filtration:** Typically used to remove turbidity, total dissolved solids, hardness, nitrates and radionuclides from the water supply. Membrane separation may include microfiltration, ultrafiltration, nanofiltration, reverse osmosis and electrodialysis, often in combination with pretreatment practices. Only microfiltration is used in New Zealand for potable water. Limited use of other membranes. Possible for medical and industrial supplies.

- **Ion exchange (IX):** Used to remove inorganic constituents, including hardness, nitrates, arsenics and radionuclides. The process involves the use of IX reactors in combination with pretreatment practices. Not likely in New Zealand.

- **Granular activated carbon (GAC) adsorption:** GAC is generally an additional unit process for the removal of naturally occurring and synthetic organic matter from water. Currently it has little use in New Zealand.

The fundamental differences between the unit processes of water treatment characterise the residuals generated by a particular treatment plant.

As noted, not all of the above water treatment processes and categories of residuals are experienced in New Zealand. Foreseeable developments are unlikely to substantially alter the current range of treatment processes. There will likely be more coagulation/filtration combinations, DAF and microfiltration treatment.

This handbook is consequently focused substantially on sludges from New Zealand water treatment plants and their management.

3.3 Overview of Sludge Production and Processing

Figure 3.1 summarises the unit processes relevant to New Zealand water treatment and the residuals they produce.

Figure 3.1: WATER TREATMENT PROCESSES

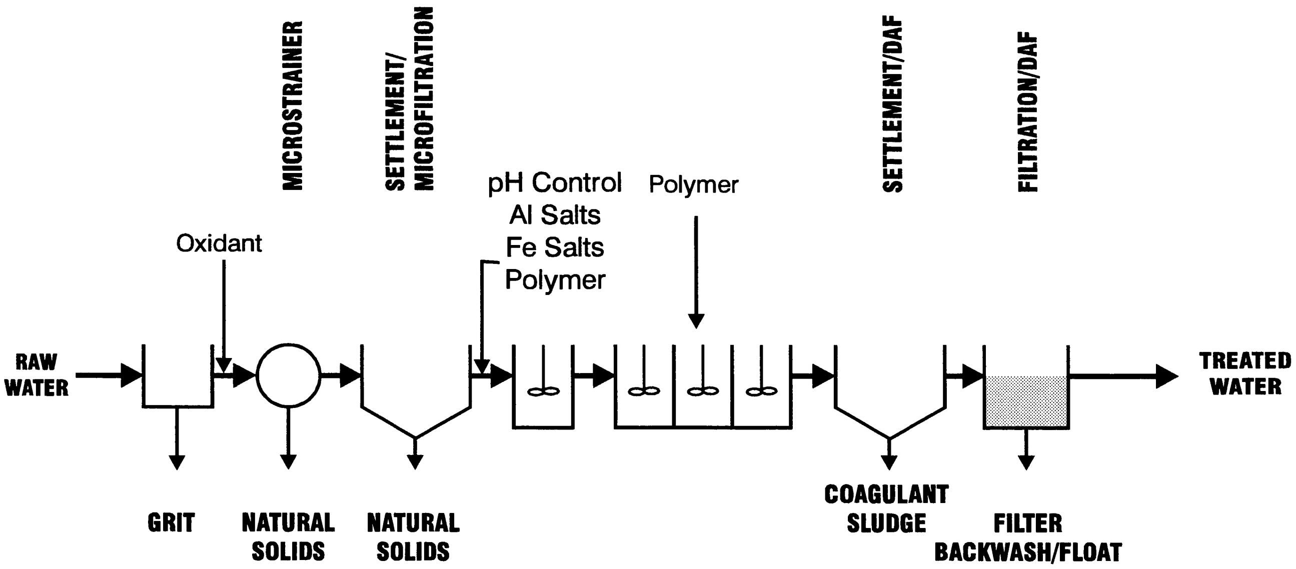


Table 3.4 Summarises the common treatment processes applied to sludges with low, medium and high solids concentrations. These unit processes are often configured to create a combined system suited to giving:

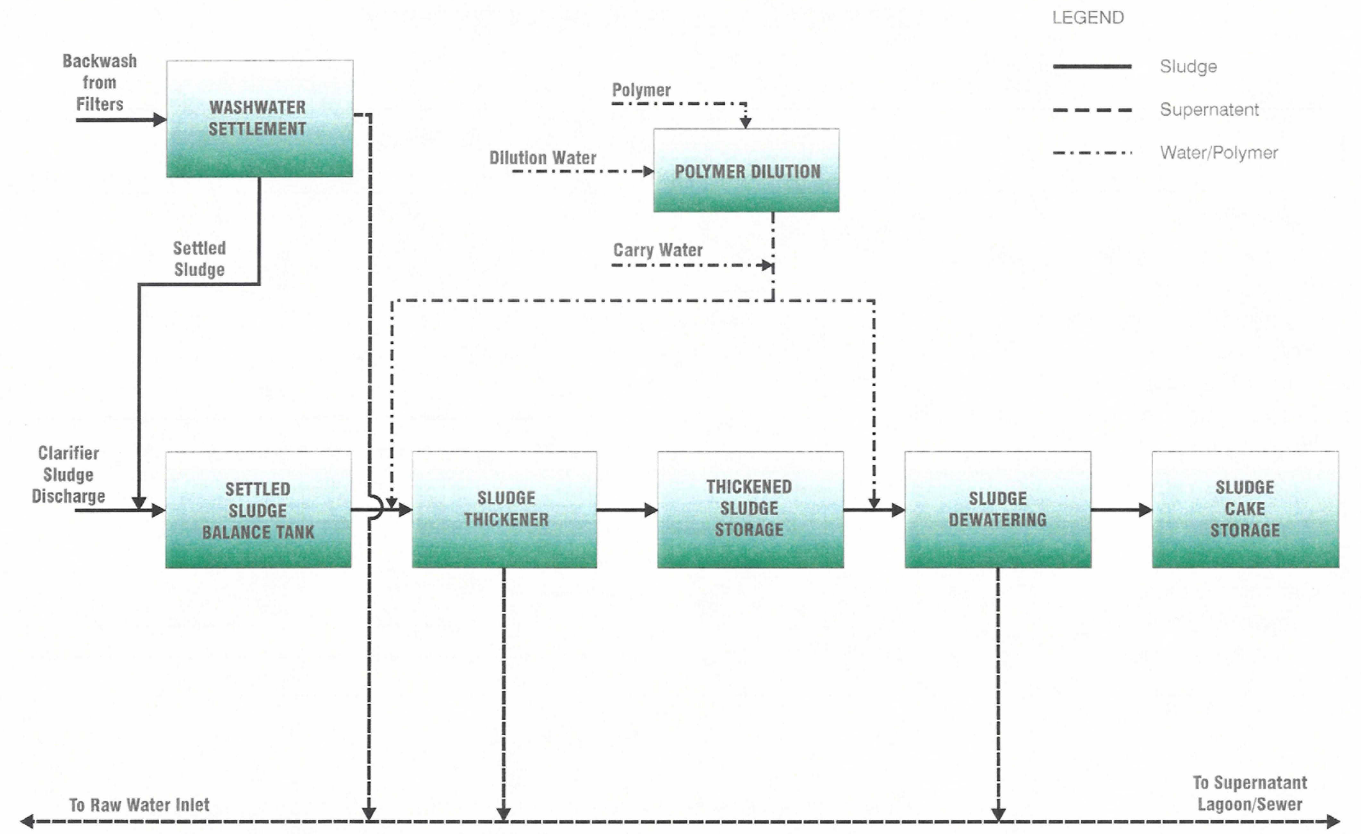
- a limited degree of operational complexity.
- a final sludge product suited to the disposal location
- a high degree of operational flexibility

Table 3.4: SOLIDS CONCENTRATION TREATMENT PROCESSES

Process	Thickening	Dewatering	Drying
Solids Concentration	Low	Medium	High
Equalisation			
Gravity Settling			
Dissolved Air Flotation			
Lagoon			
Mechanical			
Open Air			
Thermal Drying			

Figure 3.2 presents an example flowchart which combines unit processes suited to concentrating sludges into a final solid form. High rate, automated systems for larger facilities may have all processes as mechanised systems. Smaller, simpler plants may have only some of the processes, which themselves are less mechanical and automated. Each plant will require its own unique but comprehensive solution.

Figure 3.2: EXAMPLE OF A WASHWATER PROCESSING FLOWCHART



3.4 Sludge Composition

The majority of solids in water treatment sludges comprise the following:

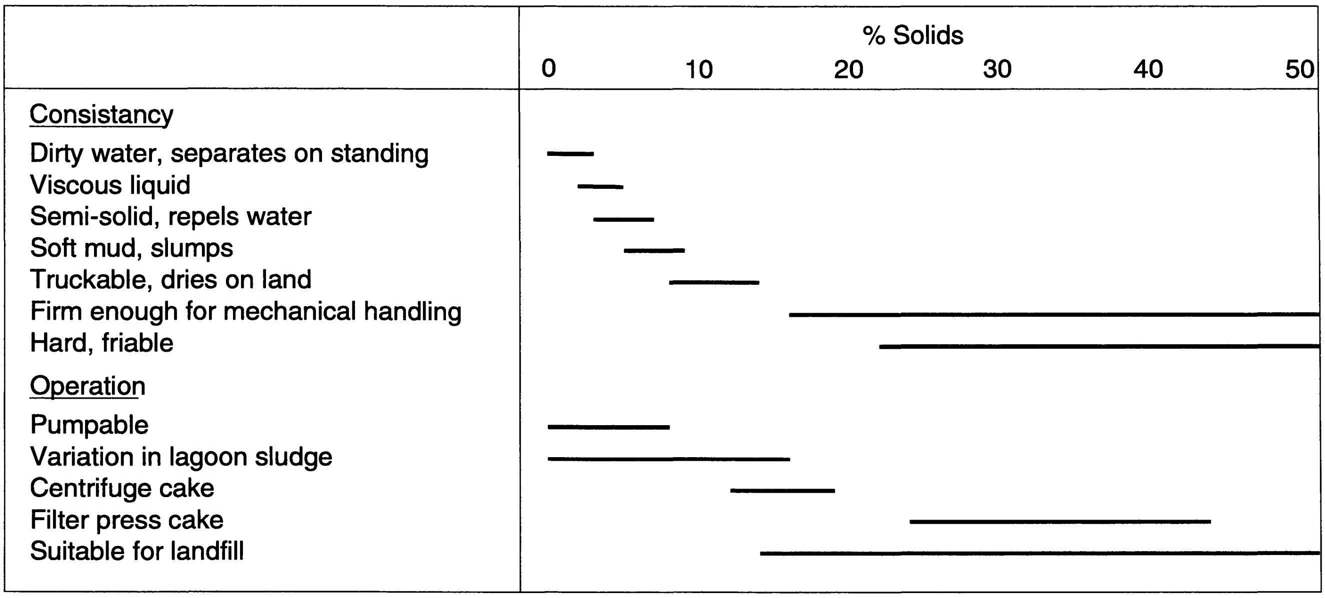
- Naturally occurring colloidal/particulate matter, e.g. silt, clay, algae.
- Naturally occurring soluble substances e.g. iron, manganese which have been converted to their insoluble precipitate form by oxidation.
- Precipitates formed when chemicals are added to water e.g., aluminium hydroxide.
- Small amounts of filter media flushed out during backwash operations.

An understanding of the quantities and properties of sludges is fundamental to determining appropriate management techniques and to designing and consenting facilities to implement those techniques. The sludge characteristics will differ from location to location and even from time to time at the same works.

3.5 Physical Properties

Coagulant sludges from raw water containing low total suspended solids (TSS) will contain a high percentage of gelatinous precipitates, e.g., aluminium hydroxide, and will exhibit the overall characteristics indicated in Table 3.5. As the proportion of natural solids increases the sludge becomes more viscous for the same total solids content. This is why a simple statement of solids concentration will not give a true indication of sludge handleability. Extreme instances can be quoted of a hard filter press cake at 19% solids, a spadeable centrifuge cake at 23% and a continuous thickener underflow which was spadeable at 25%; all waterworks alum sludges (Warden, 1983).

Table 3.5: TYPICAL ALUM SLUDGE CONSISTENCY



Coagulation of waters having substantial algae concentrations will also result in light, low solids concentration sludges. The addition of polymers generally tends to produce higher solids concentrations.

Handleability is best related to the treatment process. All well-pressed cakes handle like fudge, centrifuge cakes like blancmange and polymer - thickened underflows should handle like porridge. The actual solids contents corresponding to these consistencies depend on the balance of constituents in the sludge. All these sludges will dry irreversibly when exposed to the atmosphere. None will absorb moisture in wet weather and revert to slurry, though there may be some erosion in torrential rain.

The choice of which coagulant to use is often directed by cost. However, coagulant cost may be a small part of the total water treatment cost and therefore the benefits of coagulants other than alum, such as ferric salts, should be considered.

Alum, in the presence of natural or added alkalinity (lime, soda ash), forms an insoluble but porous floc to which the colloidal dirt particles adhere. The aluminium hydroxide also binds into its molecules a large water hydration content which results in a material of jelly-like consistency, and traps within its flocculant mass further water which is very difficult to remove.

This loose floc formation is a characteristic particular to coagulant sludges. Under quiescent conditions e.g. settlement, flocs will be seen to develop and grow. If the sludge is stirred, the shear forces will break up the flocs, as they do not have much mechanical strength. If the stirring is vigorous the flocs will disappear but reappear as soon as the turbulence has reduced.

The response of a flocculated sludge to an applied stress is not the same as that of a Newtonian fluid. Their flow properties depend on their strength:weight ratio. At 5% solids concentration, polymer thickened alum sludge has a weight of about 1030 g/l and it pours like porridge. The same sludge under water, allowing for buoyancy, has an effective weight of 30 g/l but the same mechanical strength. Its flow under gravity will now be very much slower; in fact it may not flow at all. This explains the common phenomena of rat-holing when tanks are being desludged. A designer should not expect settled alum sludge to drain from a tank as if it were a dense liquid but provide some positive means of sludge transport.

All waterworks sludges contain hard angular particles ranging from the finest turbidity solids to the occasional sand or anthracite particle washed out of the filters. The particle sizes are generally small and for dilute suspensions, as the particles have little inertia, they move only slowly across streamlines and are not very abrasive.

However, when the sludge thickens, and loses its Newtonian properties, its abrasiveness becomes significant and should be considered when specifying mechanical equipment.

A coagulant sludge will dry when exposed to air to form a hard, dense surface layer which severely restricts further drying. A heap of sludge cake may look like a pile of gravel after six months but at a depth of some 75-100 mm be the same solids content as when it was first tipped.

3.6 Chemical Characteristics

The chemical characteristics of water treatment sludges tend to affect the options for disposal/beneficial reuse more than they affect the ability to handle, thicken, or dewater.

The chemical characteristics of sludges are directly related to the chemical content of the raw water and the coagulant chemicals. The BOD_5, COD and related organic content are representative of the dissolved and suspended organic materials and algae removed from the water. The inorganic solids are derived from the coagulant chemicals and the clay and sediments removed from the raw water. The pH and dissolved solids of the sludge are similar to the raw water being treated.

Thus the majority of solids tend to be inert and the volatile solids proportion is typically less than 30%.

Coagulant sludges will also contain bacteria removed from the raw water. The bacterial count of the sludge will be variable and site specific depending on the quality of the raw water and treatment process employed e.g., prechlorination. Presumably viruses are also present, although no analyses have been found in the literature.

The metals content of sludges is considered important due to potential impacts on the receiving environment, e.g., river or land for disposal. The levels of cadmium, chromium, copper, nickel, lead and zinc in coagulant sludges derive from the levels existing in the raw water sediments plus trace contaminants within each chemical supply. The chemical suppliers will provide specifications for each of their products. Thus metals content can be limited by carefully specifying the chemicals used.

The general levels of metals, other than the coagulant metal e.g., aluminium, are typically 10-30% of the corresponding sewage sludges. The majority of these metals are bound within the oxide or silicate matrix of the sludge and are not available for plant uptake or mobilisation into water. A significant decrease in pH and alternating aerobic/anaerobic activity are required to mobilise metals.

NZWWA published the second edition of "Standard for the Supply of Aluminium Sulphate in Water Treatment" in 1997 (NZWWA, 1997). One of the main reasons for this was the concern at the high concentration of metals in some imported coagulants. The NZWWA standard includes Specific Impurity Limits (SIL) for several contaminants based on the following assumptions:

- the MAV in the 1995 DWSNZ
- all the contaminant enters the drinking water
- a maximum alum dose of 100 g/m^3 alum
- a safety factor of 10

In practice the majority of metal contaminants will be in the particulate form and become a focus and potential concern for the receiving environment of long term disposal.

Table 3.6 presents the results of recent analyses of Fernz solid alum (converted from the analysis of liquid alum) for comparison with the SIL. This indicates that contaminants in the New Zealand product are well within SIL. The table includes the mean concentrations found in three granular alum samples in the USA (Cornwell, 1990).

Table 3.6 also indicates the ranges of contaminants within purchased chemicals and that care should be taken in their specification and purchase. Before importing chemicals their specification should be checked, and perhaps a sample analysed upon arrival.

Table 3.6: SIL VALUES FROM NZ STANDARD FOR ALUM, COMPARED WITH ANALYSIS OF NEW ZEALAND AND USA ALUM

Contaminant	Specific Impurity Limit (mg Contaminant per kg Alum)	NZ Alum (mg Contaminant per kg Alum)	USA Alum (mg Contaminant per kg Alum)
antimony	3	-	
arsenic	10	<0.53	555 - 621
barium	700	-	<50
cadmium	3	<0.11	<5
chromium	50	<1.1	<5
copper	1000	-	<2
iron	200	15-20	
lead	10	<0.21	<5
manganese	500	-	
mercury	2	<0.02	
molybdenum	70	-	
nickel	20	-	<50
selenium	10	<1.1	<5 - 204
silver	-	<1.1	
zinc	3000	-	<5 - 12

3.7 Sludge Quantities

The design load of each effluent stream can be given by a statement of its nature, concentration and flow as a function of time.

The best way to arrive at a realistic design load is to work it out by as many methods as possible and compare the results. If there are discrepancies they should be resolved. It is not reasonable to just take the largest amount.

In many cases the sludge load should be quantified for the range of raw water conditions experienced or anticipated at the plant. For example:

Peak sludge production: Maximum conditions of raw water turbidity/ colour, relevant plant throughput and coagulant dose. (This may give a short duration high solids load and a sludge with low alum content). Sedimentation underflow may comprise the raw water solids only.

Average sludge production: Normal conditions of raw water turbidity/ colour, plant throughput and coagulant dose. (This will give the normal quantities and characteristics to be expected).

Minimum sludge production: Minimum conditions of raw water turbidity/ colour and throughput with relevant coagulant dose (This may give a small quantity of sludge with high proportions of alum).

Residuals processing must cater for the full range of quantity and characteristics. However, lesser efficiency may be acceptable for infrequent events.

3.7.1 Measuring the Sludge Load

For an existing plant it will be possible to measure actual loads by sampling.

Sedimentation sludge should be measured from routine desludging rather than an induced desludging sequence. A representative sample should be taken, recognising the possible change in concentration over the period when the valve is open.

Filter backwash is more difficult to sample given the large volume and continually changing concentration. It is necessary either to take equal volumes over equal times of backwash and making them into a composite sample or to capture the total backwash and resuspend it to give an average concentration sample.

For measurements by sampling it is important that the works is operating at a steady state and details of plant output, coagulant dose and raw water quality are known. It may be possible to confirm these measurements by estimating the solids load going to disposal e.g. the volume and concentration tankered away or lagooned.

Residuals processing must cater for the full range of quantity and characteristics. However, lesser efficiency may be acceptable for infrequent events.

3.7.2 Calculating the Load

The solids produced can be calculated by adding together all the contributions from raw water components and treatment chemicals. It must be remembered that if raw water is stored prior to treatment there will be quality changes and the stored water quality should be considered.

There are a number of researchers who have produced formulae that enable the quantity (mass of suspended solid components) of sludge to be calculated. The formulae are generally of the following form:

$W = S + aT + bH + cD + Y$

Where

W = the weight of sludge solids produced in g/m^3 of treated water.
S = the suspended solids (SS) concentration removed from the raw water.
T = the turbidity of the water after SS removal.
H = the colour of the raw water.
D = the dose of aluminium (Al) or iron (Fe) added in the coagulant used.
a,b,c = factors converting the measured parameters into equivalent SS.
Y = the mass of any other chemicals, e.g. polymer, used.

Each component of the formula are discussed below.

S = Suspended Solids: If sufficient raw water suspended solids measurements are available they can be used directly. It is assumed that all raw water suspended solids goes into the sludge. In many cases only turbidity units (TU) are measured and correlation is required.

Unless turbidity and colour vary together, a correlation between SS and T will not exist and values based on experience and judgement are required.

For example, most of the particles in a river in winter or in flood may be small sand grains or grit and clay and silt; this may give rise to a high SS concentration relative to turbidity. In the summer the lower river flows may produce less grit and more algae; this will give rise to a higher turbidity in relation to SS.

T = Turbidity: If SS measurements are not available turbidity alone can be used to estimate SS values. In this case turbidity is measured on the raw water without any prior treatment. Direct measurement of SS is more accurate and to be recommended.

SS can vary 0.7 - 2.2 x T for low colour, mainly turbidity removal plants. For turbidities less than 100 the suspended solids has been shown to be approximately equal to the turbidity in turbidity units (Nielsen, 1973).

This assumes that the solids are of mineral origin and make no allowance for algal loads which should be noted separately as measured SS.

The UK Water Research Centre (Warden 1980) noted that in the absence of SS measurement use of a = 2 for raw water turbidity while not highly accurate would not be grossly misleading in UK upland waters.

H = Precipitated Colour: Colour is most often measured in Hazen units (ºH). Work by Black and Christman, (1963) suggested a factor of b = 0.2 which is recommended by WRC for UK water.

In Australia, Hartley (1980) suggested that this factor should be 0.3. These relationships for calculating SS assume that the colour measured is the true colour. True colour is measured once the SS have been removed (e.g., by filtration).

TOC: Total Organic Carbon (TOC) removal may provide a better indicator to solids production. A conversion factor of 1 mg/L TOC to 1 mg/L solids would apply.

D = Coagulant: Dosing water with aluminium sulphate (alum), poly aluminium chloride (PAC), or iron salts results in sludges containing hydrous oxides of aluminium or iron. It has been traditional to assume that these hydrous oxides approximate the hydroxide form e.g. $Al(OH)_3$ or $Fe(OH)_3$.

With the increase in coagulant forms available in the market place it is possible to become confused about the amount of hydroxide that will be produced from any quoted dosage. This could result in significant errors when calculating sludge production. The following attempts to clarify this situation:

- Alum (kibbled or solid lumps) is considered to be $Al_2(SO_4)_3.14H_2O$. The Fernz product is 17.0% by weight Al_2O_3 (aluminium oxide) or 9.0% by weight ($^w/_w$) as Al.

- The atomic weight of aluminium is 26.9815. The atomic weight of oxygen is 15.9994 so the molecular weight of Al_2O_3 is 101.9612. The conversion factor from Al_2O_3 to Al is therefore 53.9630 ÷ 101.9612 = 0.529.

- Fernz liquid alum has a specific gravity of 1.32 and it is 8.0% $^w/_w$ Al_2O_3, so it is 4.232% $^w/_w$ as Al and 4.232 x 1.32 = 5.59% $^w/_v$ Al.

- When someone doses at 20 ppm of kibbled alum they are adding 20 g of alum per m^3 of water. That means they are adding 20 x 0.17 = 3.4g Al_2O_3 per m^3, or 20 x 0.09 = 1.8g of Al per m^3, which produces 5.1g $Al(OH)_3$ per m^3.

- PAC can be either a solid (e.g., Fernz Solipac) **or** a liquid (e.g., Fernz Liquipac). PAC is considered to be $[Al_2(OH)_5Cl_{2.6}(SO_4)_{0.2}]_n$. PAC and alum are different chemicals and are not readily converted one to the other by simple arithmetic based on atomic/molecular weights etc, so PAC doses are not converted to alum.

- Solipac is 30% Al_2O_3 or 15.88% Al. Liquipac is 10.3% Al_2O_3 or 5.45% Al. Therefore Liquipac is a 34.3% $^w/_w$ solution of PAC; it has an SG of 1.20.

- If someone is dosing the raw water at 20 ppm PAC (as the solid) they are adding 20 g PAC per m^3 of water. If they are dosing at 20 ppm PAC (as the solid) but are using Liquipac, they are adding 20 x 100/34.3 = 58.3 g Liquipac per m^3 or 48.6 mL of Liquipac per m^3.

- If someone is dosing the raw water at 20 ppm PAC (which = 58.3 ppm Liquipac) they are dosing at 20 x 0.30 ppm Al_2O_3 = 6 ppm Al_2O_3 which produces 9 ppm $Al(OH)_3$. Therefore the conversion factor from PAC dose to $Al(OH)_3$ is 0.45.

- The hydroxide form of coagulant sludges will include some bound water which will not be released in a standard suspended solids test. The bound water should be considered in estimating solids loads that will be later measured by standard laboratory testing of the sludges produced. In the case of alum sludge, an empirical formula for the aluminium hydroxide plus bound water fraction of the sludge (Nielsen 1973) has been determined to be $Al(OH)_{3.1}$-$25H_2O$. If this formula is assumed for both aluminium and ferric coagulants:

Coagulant hydroxide sludge

= $0.34 \times Al_2(SO_4)_3.14H_2O$

or $0.30 \times Al_2(SO4)_3.18H_2O$

or $1.97 \times Al_2O_3$

or $3.72 \times Al$

or $0.59 \times Fe_2(SO_4)_3$

or $0.48 \times FeCl_3$

or $1.62 \times Fe_2O_3$

Y = Other Chemicals: A number of other chemicals can contribute to sludge products. They are discussed below.

Iron Oxidation: Aeration to remove iron by oxidation typically produces 2.9 kg hydroxide sludge for each kg Fe (iron) removed.

Changes in Hardness: Softening is rarely required in New Zealand but dosing of lime for pH control may occur and may be measured as a change in total hardness. Because lime has a low solubility it reacts slowly and this can result in actual lime doses being greater than necessary. The sludge solids formed are given by

sludge solids = (lime added - 0.74 x increase in hardness, as $CaCO_3$).

Ignore the result if it is negative (indicating an inaccuracy in the measurements).

Other additives: Limestone, chalk, bentonite, polyelectrolyte, and other insoluble materials are all assumed to go direct to sludge, i.e., 1 g/g used. Ozone, caustic soda, sulphuric acid and potassium permanganate make no contribution.

3.8 Concentrations and Volumes

The amount of solids in a sludge is the product of volume and concentration. When calculations have been made based on a range of assumptions it is possible to obtain volumes and concentrations which do not have the correct arithmetic relationship. The following comments summarise a range of experience to help decide whether the relationships derived could be realistic.

3.8.1 Microstraining

Most installations have been at water treatment plants that receive algal laden raw water. Raw water enters a rotating cylinder or drum and passes through a stainless steel mesh which is cleaned by high pressure jet mounted above. The washwater is collected in a trough inside the cylinder, above the water line.

Typically the volume of washwater is 1-3% of the raw water throughput.

3.8.2 Sedimentation

Typically the volume of sedimentation sludges are 1-3% of raw water throughput.

Sludges without coagulants typically have a concentration of 1.0% $^w/_v$ for low raw water turbidity and colour, and up to 3.0% $^w/_v$ for high turbidities.

Sludges produced with alum (or iron) coagulants for low to moderate raw water turbidity have solids concentrations ranging 0.1-1.0% $^w/_v$.

In general the higher the coagulant to raw water solids ratio, the lower the sludge concentration.

3.8.3 Filtration

Filter backwash is generally 0.03% w/v solids averaged over a complete cycle, rising to 0.06% where the floc separates readily from the sand. A lower concentration may indicate uneven washwater distribution in the filter. Washwaters tend to settle in 2 to 4 hours and can be recovered at about 1% w/v solids after decantation. The volume of filter washwater is generally between 2 and 5 % of raw water throughput.

Diatomaceous earth filters are fairly common in the swimming pool and food industries but less so in the production of drinking water in New Zealand. These filters typically use less than 2% of raw water as washwater.

The NZWWA questionnaire (1996) responses showed that plants using organic polymers as the primary coagulant use less filter washwater than alum and PAC plants. This is because the dose is lower so the filter blocks (clogs) less quickly, and possibly also because most plants using polymer coagulation have cleaner raw water.

3.8.4 Microfiltration

In some microfiltration plants the washwater is dosed with polymer, settled and the supernatant recycled to the raw water. At others the washwater is passed through a secondary microfiltration unit with the filtrate continuously recycled to the raw water supply. The solids in the final washwater in this latter case will be about 200 times more concentrated than the raw water.

Membranes also require sanitising or Cleaning-in-Place (CIP). This involves a sodium hydroxide solution, usually about 2% strength. The cleaning cycle lasts about 2-3 hours every 2-6 weeks. Sometimes sodium hypochlorite or a detergent are used as well. The sanitising solution can be reused several times before disposal; when spent it is taken-off site by waste disposal specialists. For raw waters that have relatively high iron, manganese or hardness, it may be necessary to clean the membranes occasionally with an acid solution too, usually 2% citric acid.

The backwash water will contain mostly particulate matter that was removed from the raw water and contains no added chemicals so it may be able to be discharged to natural water, or sewer, or the solids may be settled out in a lagoon. The spent sanitising solution will be a 1-2% sodium hydroxide solution and may also contain some sodium hypochlorite and detergent. If this is added to the backwash solids the sludge will be strongly alkaline. The sludge will be less alkaline if the membrane needs to be cleaned occasionally with citric acid.

4. Solids/Liquid Separation Processes

4.1 Collection Processes

Collection processes are the means by which water treatment plant residuals are collected from the process unit in which they were removed from the water. Residuals are removed from the process stream by a combination of physical and chemical methods.

Figure 3.1 summarises the collection processes relevant to New Zealand water treatment and the residuals they produce. Once these processes entrap the solids produced in water treatment, the solids are periodically discharged either as sludges or washwaters for subsequent processing and disposal/reuse.

Figure 3.1 summarises the range of solid/liquid separation processes common for further processing of residuals and typical ranges of performance.

4.2 Settlement

4.2.1 Process Description

Settlement occurs in rectangular or circular tanks which are often equipped with bottom hoppers. The tanks are generally operated in batch mode to separate dilute wastes e.g., filter washwater.

The tanks are filled, allowed to settle for some hours, and then the sludge and supernatant are removed separately. The settled solids can be removed more easily if the supernatant is decanted off first because the relative density of the solids is small compared with its viscosity. Supernatant is typically withdrawn through a submerged but floating bellmouth which maintains its submergence as the water level lowers. After maximum supernatant is withdrawn the settled solids are removed separately through a pipe in the tank base.

4.2.2 Design Outline

The sizing of settlement tanks depends on the continuity of flow of the effluents. In the case of a small filter station where all the filters are washed in a single shift, the tank should be sized to take a whole day's washwater for settlement overnight and decantation in the morning. Where filter washing is more evenly spaced through the day, three tanks may be provided so that there is always one filling, one settling or decanting and one waiting empty. Allowing three hours for settlement and one for decanting, each tank may be sized to take the maximum number of filter washes expected in four hours.

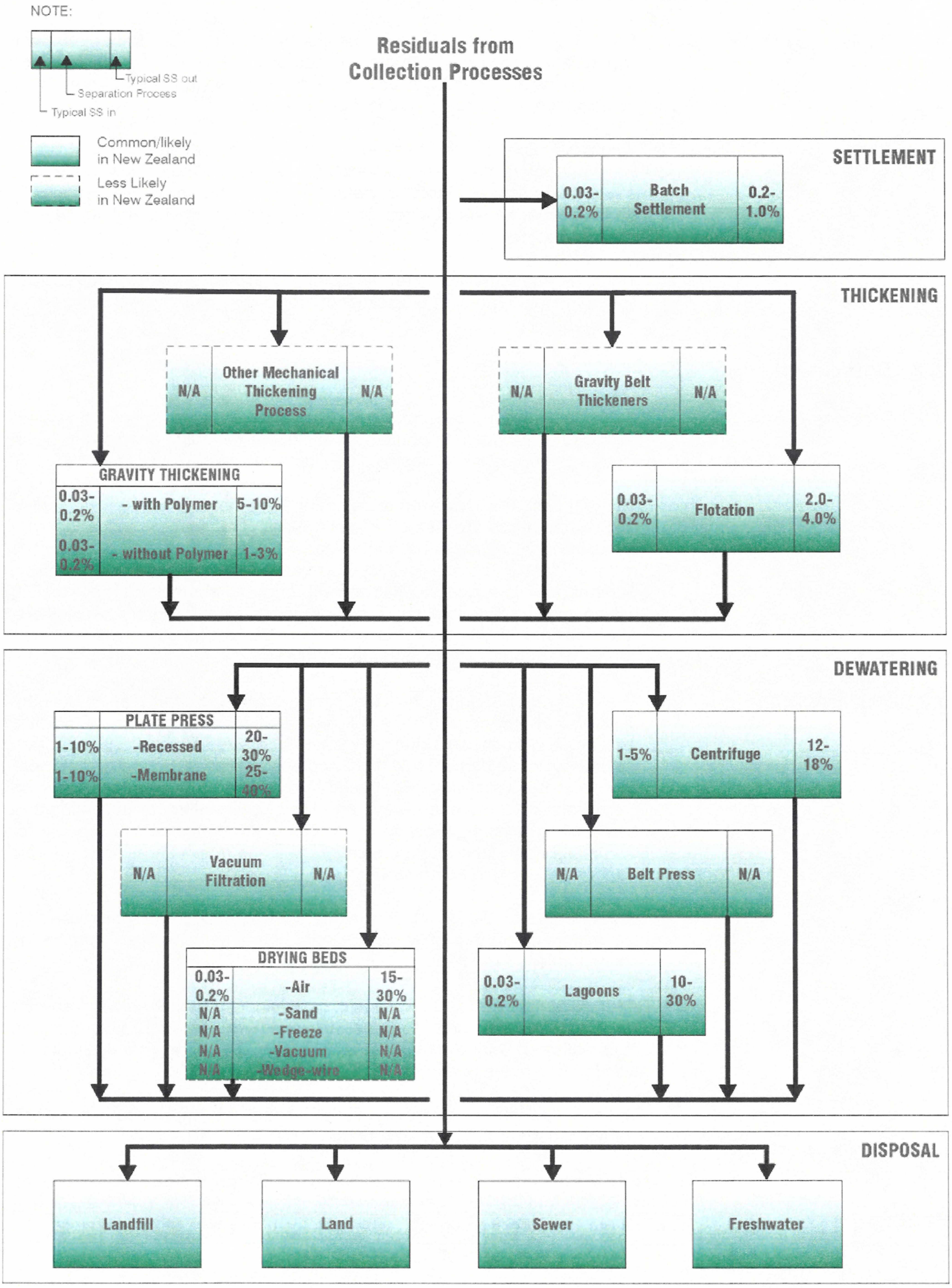
On a works with sedimentation tanks, this capacity may be increased by the volume of four hours' sludge bleed if further settlement will improve its concentration.

After decantation the tanks should be desludged into a holding tank for temporary storage before further treatment or disposal. The common malpractice of letting sludge accumulate in the settling tank should be avoided to prevent deterioration in supernatant quality.

To allow complete emptying, tanks should be hopper-bottomed with surfaces inclined at about 20% or greater.

Batch settlement may be controlled manually on a small works, but the whole sequence of operations may be controlled automatically using timers to control settling time and turbidity monitors to control decantation.

Figure 4.1: SOLID/LIQUID SEPARATION PROCESSES



4.3 Thickening

Thickening processes begin after settlement and filtration processes. They are applied to the metallic hydroxide residuals which come from either clarifier operations or backwashing of filters. They are critical to the economic removal of solids from the treatment process due to their knock-on-effect on downstream processes such as conditioning and dewatering.

The most common thickening process uses gravity thickening.

4.2.2 Gravity Thickening

Process Description

Gravity thickening can either be a batch or continuous process. Residuals thickened in gravity thickeners may require conditioning.

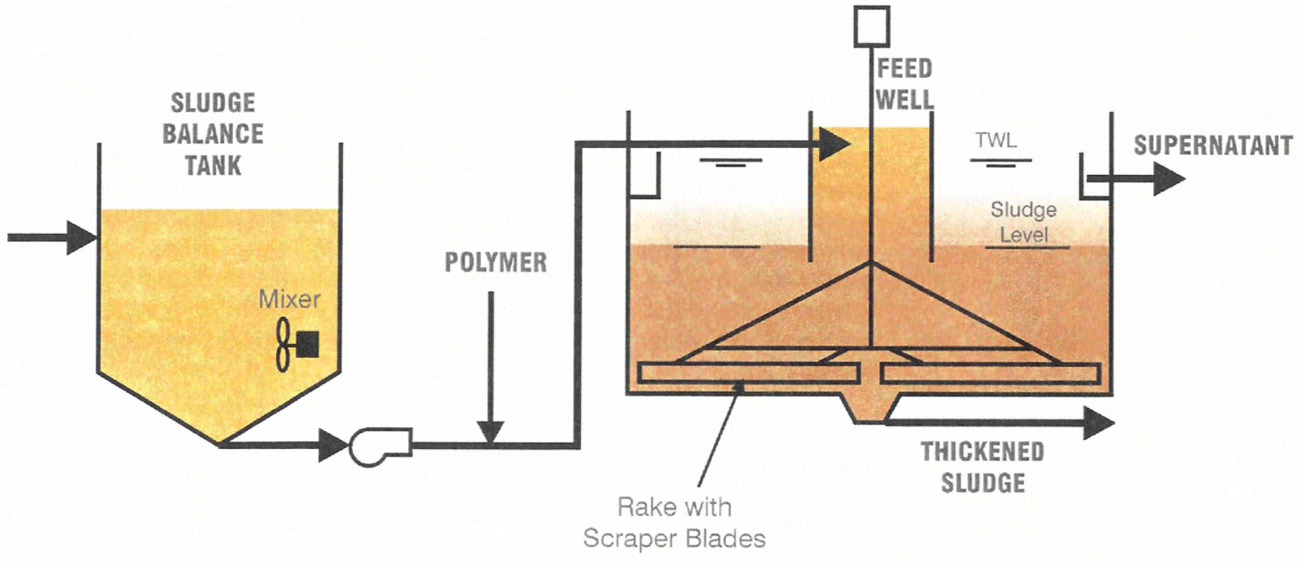
The batch-fed process works on the same principle as the settlement tank. The continuous flow thickeners however are designed for desludging underwater, thus allowing filling, decanting and desludging to proceed simultaneously.

The solids slurry typically enters the thickener through a central feed well although side entry is used in some small tanks. In theory, the solids are distributed equally, both horizontally and vertically. The solids settle to the bottom of the unit and the clarified supernatant flows over discharge weirs located on the periphery of the tank.

The supernatant may be recycled to the works inlet or be discharged. Typically the turbidity would be in the range 4-8 NTU.

The thickener works most efficiently with a feed that is reasonably uniform in flow and quality. This is particularly so if polymer conditioning is used. This can be achieved by collecting the plant sludges in a mixed balancing tank and pumping forward at a steady rate. As with settlement, it is good practice not to store thickened sludge in the thickener, but to provide separate storage of sufficient capacity for this purpose. This may however be too costly for a small works. Figure 4.2 shows a typical arrangement.

Figure 4.2: TYPICAL CONTINUOUS GRAVITY THICKENER ARRANGEMENT



Design Outline

Thickener tanks are generally circular and are usually concrete, although smaller tanks can be made from steel. The diameter is typically based on an upflow rate of about 1.5 m/hour (flowrate/tank plan area) or a solids load of 4 kg/m^2/hour whichever is reached first, and the other constructional dimensions are related to this diameter. Tanks are typically equipped with bottom scraper mechanisms that rotate slowly transporting sludge to the centre draw-off pipe or sump for removal. The floors may be flat or conical in shape with a slope of between 10 and 20 percent.

The slow rotation of the scraper also prevents bridging of the solids and 'kneads' the floc to produce a thicker sludge.

The balancing tank, receiving an intermittent flow of clarifier bleed and surges of filter washwater (possibly settled), is sized to allow a steady forward feed rate. The thickener feed pump works at constant output, switching off at low level in the balancing tank and on at high level. This means that the thickener stops occasionally, but so long as the operation of the rake is not interrupted, performance is quickly re-established when the feed is restarted.

Polymer conditioning should match introduced polymer to the feed solids. Jar tests can confirm the correct ratio which should be maintained. For larger works this can be automated for variable flow and sludge feed concentrations. Use of a streaming current detector on the thickener supernatant has also shown success. If close control of polymer to feed sludge solids ratio is not maintained either poor and variable performance or excess use of costly polymer will result.

Adequate sludge transport capacity to positively move thickened sludge into the hopper is imperative, especially for polymer thickened sludges. There are many unfortunate experiences told of gelatinous hydroxide sludges which had to be manually dug out of the tank.

These issues are dealt with in detail by Warden (1983).

4.3.1 Flotation Thickening

There are no water treatment plants in New Zealand that thicken wastewater using flotation thickening (NZWWA Questionnaire, 1996) but this technology may be used in the future. One plant currently uses DAF technology for water treatment.

The process is usually selected for treatment of surface waters from upland catchments and stored lowland river which produce residuals consisting of low-density particles. DAF processes will normally operate with a lower coagulant dose than most other clarification processes and therefore for a given water it will tend to produce a smaller quantity of sludge. Similarly, the separation and consolidation of the coagulated impurities in the float layer effectively produces a residual sludge that in many cases has a higher concentration of solids than other conventional clarification processes. However, the sludge consists of a mixture of precipitated solids, water and entrained air, which is similar in many respects to the physical format of chocolate mousse.

Thus the formation and removal of the float from DAF units for water treatment is similar to the use of DAF for sludge thickening.

Process Description

Flotation thickening can utilise any of three techniques:

- **Dissolved Air Flotation (DAF).** This is the most common technique.

Small air bubbles 5-100 m in diameter are generated in the incoming feed. Generation is usually by pressurising air and the liquid stream together with subsequent release at the inlet of the flotation tank. The excess air over the saturation value in water emerges as small bubbles which attach themselves to the solid particles.

- **Dispersed Air Flotation.** Gas bubbles are generated through a mixer, cavitation pump or through porous media. Many of these generators produce large bubbles (500-1000 m) which have lesser thickening efficiency. The process can be less mechanically complex and use less electrical power.
- **Vacuum Flotation.** Operates on a similar practice to DAF but with supersaturation of the liquid stream generated by vacuum.

Each of the above techniques uses air bubbles to absorb or attach to solids particles which then float to the surface. The floated solids form a thick (100 -200 mm) layer which is scraped over a weir for subsequent removal. Descriptions of the float and removal characteristics are given by Schofield (Schofield, 1997).

Design Outline

Flotation tanks can be circular or rectangular with mechanical skimmers operating alone or with assistance of hydraulic pulsing. Stabilisation grids can be placed in the tanks to increase the thickened sludge depth and concentration (Schofield, 1997).

Several sources indicate that European facilities have had success in concentrating hydroxide sludge to levels between 3 to 4% solids (ACE/AWWA, 1990; Brown, 1990). These results appear to include facilities that use flotation both as a concentration process(as an alternative to sedimentation) and as a thickening process. Loading rates for hydroxide sludges vary from 2.0-5.0 kg/m^2/hr for facilities achieving from 2 to 4% float solids concentration. Hydraulic loading of DAF units is reported at less than 4.9m^3/hr/m^2 (Cornwell, 1990).

4.3.3 Gravity Belt Thickeners

This equipment is similar to a belt press (see Section 4.4.3) but with large gravity drainage area and less pressure zones. There is little experience internationally on coagulant sludges and gravity belt thickeners do not seem popular in New Zealand.

4.3.4 Other Mechanical Thickening Processes

Examples of these processes include the continuous-feed polymer thickener, drum thickener and centrifuges. They are unlikely to be used in New Zealand, and internationally no full-scale operating data is available (AWWA, 1996).

4.4 Dewatering

4.4.1 Lagoons

Process Description

Lagoons are the commonest and oldest method used to handle water treatment residuals. Lagoons can be used for storage, thickening, dewatering or drying. In some instances, lagoons have also been used for final disposal of residuals.

Solids settle to the bottom of the lagoon and liquid can be decanted from various points and levels in the lagoon after a period of hours or days. Solids are thus concentrated until the lagoon is full of settled sludge and is then left to drain and consolidate over a period of months or years. The lagoon is either dug out or remains as the final disposal location. A complete installation normally requires several lagoons operating in staggered cycles.

A lagoon that is left to “revert to nature” will take a long time to dry out. After the supernatant has been drained away or allowed to evaporate, the sludge at the surface forms a crust, giving the appearance that the lagoon is now filled with dry sludge. This is not so. The crust-covered lagoon can be compared with a frozen pond - the surface may or may not support the weight of a human or stock. Therefore it is important to fence off the lagoon and to ensure that it remains fenced off. It may take many years before the sludge dries out to the base of the lagoon. There will be a large amount of shrinkage during the drying out process.

Design Outline

In recent years lagoons have generally been lined or excavated from impervious materials; the intent being to limit the potential for leakage and subsequent contamination into groundwater. However the potential for contaminants to mobilise is quite limited and popular concerns may be unsubstantiated in practice.

Older lagoons have been constructed deliberately on porous ground to assist drainage. Few environmental problems have been noted in the literature.

Lagoon supernatant quality was measured at Ardmore and Waitakere in the 1980’s. The lagoons at that time were overloaded, awaiting commissioning of the new dewatering equipment. A summary of the results is presented in Table 4.1. Studies were also undertaken by the ESR for Wellington Regional Council (Gregor, 1994) on the lagoons at Te Marua. A summary of the results are presented in Table 4.2. Both sets of results show that lagoon supernatant can have a high quality. Variations in suspended matter primarily relate to turbulence from surges in flowrate and wind induced currents. When it is considered how much natural organic matter is tied up in the lagoon (about a third of the solids), the small soluble component in the supernatant indicates that the sludge is stable and does not break down to release contaminants into the supernatant.

Table 4.1: SUMMARY OF TEST RESULTS OF LAGOON SUPERNATANT IN AUCKLAND 1981-1987

	SS	Aluminium	Fluoride	pH
Number of Samples	353	371	372	370
Maximum, g/m^3	1099	400	4.64	10.4
Minimum, g/m^3	0.1	0.05	0.01	5.6
Mean, g/m^3	29.1	3.3	0.26	7.1
Water Right Condition, g/m^3	30	20, then 1	1	6-9
% Outside Condition	16	32	3	1

Table 4.2: QUALITY OF TE MARUA SLUDGE LAGOON SUPERNATANT

Test	Units	Normal Range of Results
pH		7.2-7.4
Turbidity	NTU	1.5-2

Absorbance at 325 nm	10 mm cell	0.020-0.060
Acid soluble aluminium	g/m^3 Al	1.0-1.5
Soluble aluminium	g/m^3 Al	0.1-0.3

Individual samples or limited lagoons supernatant tests reported (NZWWA, 1996) include mean SS 5.6 g/m^3 during 1996 at Invercargill; pH 7.06, Al 2.3 gm/m^3 and SS 22 g/m^3 at West Taieri; pH 9.8, Al <0.02 g/m^3 and SS 8 g/m^3 at Waikouaiti; pH 6.5, Al 0.5 g/m^3 and SS 10 g/m^3 at Port Chalmers.

Performance

The effectiveness of lagoons in concentrating solids typically depends on the method of operation. For metal hydroxide solids retained in a lagoon for 1 to 3 months, operating the lagoon at full water depth without further air drying of the solids typically results in a solids concentration of 6 to 10%

4.4.2 Air Drying Processes

Air drying refers to those methods of sludge dewatering that remove moisture either by natural evaporation, gravity or induced drainage. Air drying processes are less complex, easier to operate, and require less operational energy than mechanical systems. They are not used frequently, however, as they are land and labour intensive, dependent on climatic conditions and slow to produce end results. The effectiveness of the process is directly related to weather conditions, type of sludge, conditioning chemicals used, and materials used to construct the drying bed.

Drying beds in New Zealand are mainly used for sewage sludge dewatering. As water sludge does not drain or dry rapidly unless it is in very thin layers, drying beds are unlikely to gain popularity.

Air Drying Bed

A drying bed is a rectangular bunded area with a drained porous floor. The bed is operated in a batch mode i.e. one bed would be filled to the required depth after several days and then left to dry. The area required can be based on a cake thickness of 25 mm on lifting, the feed sludge concentration and the desired concentration for the cake. Drying times vary from 2 or 3 weeks in summer to 2 or 3 months in winter depending on weather conditions.

Experience at Waitakere in New Zealand concluded drying beds operated successfully at an average loading of 90 kg dry solids per m^2 of drying bed throughout the year, higher rates may have been possible in wind-exposed areas. (NZWWA, 1996).

Sand Drying Beds

Sand beds are not common in New Zealand. Dewatering on the sand bed

occurs through gravity drainage of free water followed by evaporation to the desired solids concentration level.

Freeze-Assisted Sand Beds

The New Zealand climate makes this process impractical. Overseas it is used on alum residuals which release bond water from cells through the freeze-thaw process changing the consistency of the sludge from gelatinous to granular; which is then easier to dewater.

Vacuum-Assisted Drying Beds

This method is reported as expensive, time consuming and problematic. The technology applies a vacuum to the underside of rigid, porous media plates on which chemically conditioned sludge is placed.

Wedgewire Beds

The wedgewire, or wedgewater, process is physically similar to the vacuum-assisted bed. The base of the bed incorporates a wedgewire screen which holds and drains the sludge.

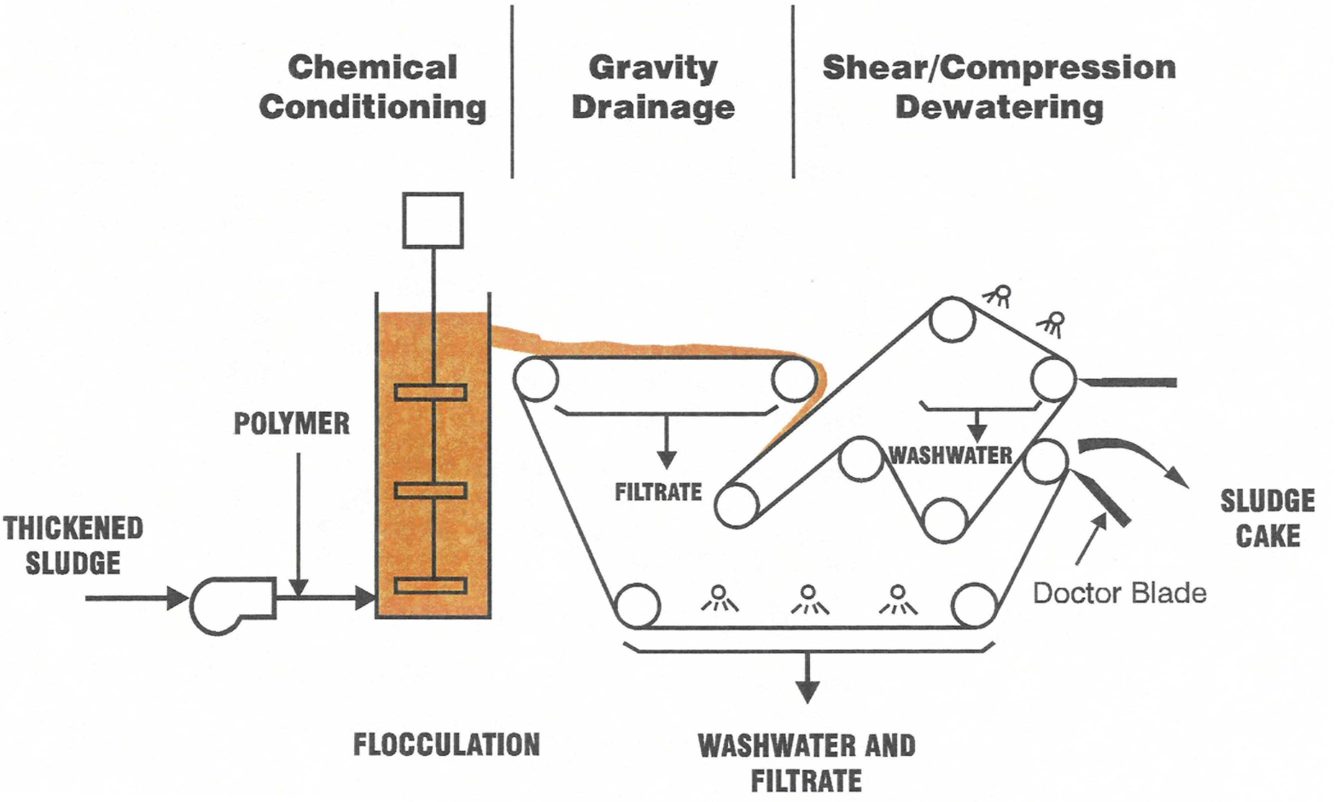
4.4.3 Mechanical Dewatering Equipment

4.4.3.1 Belt Presses

Process Description

The belt press is based on a very simple concept. Sludge sandwiched between two porous belts is passed over and under rollers of various diameters. As the roller diameter decreases, pressure is increasingly exerted on the sludge, squeezing out water. A typical arrangement is shown in Figure 4.3.

Figure 4.3: TYPICAL BELT PROCESS ARRANGEMENT



Design Outline

Although many different belt filter press designs are used, they all incorporate the same basic features - a polymer conditioning zone, a gravity drainage zone, and a low pressure followed by one or more higher pressure zones.

The polymer conditioning zone can be either a small tank with a variable speed mixer located adjacent to the press, or an in-line injector.

The gravity drainage zone, where solids should dewater naturally by gravity, is a flat or slightly inclined belt unique to each press model. Problems can occur if water does not drain readily.

The low pressure or "wedge zone" is where the two belts converge forming a solids "sandwich" - a relatively firm cake able to withstand the forces within the high pressure zone.

In the higher pressure zones forces are exerted on the solids cake by the belts passing over a series of rollers of decreasing diameter. The spacing and tension of the belts are adjustable.

Performance

Belt filter presses can be used to dewater residuals produced from alum coagulation despite their gelatinous nature. Performance can be affected by many variables, including solids type and characteristics, conditioning and pressure requirements, belt speed, tension, type and mesh and raw water source.

Alum residuals must be dewatered at low pressure. A pure alum residual may dewater to 15% or more solids, whereas slurry produced from river water which has silt and sand entrained will more easily dewater producing a drier cake.

To ensure optimum performance, alum solids must be first conditioned with polymer. Polymer produces a larger, stronger floc that allows free water to drain more readily from the solids in the gravity drainage zone of the belt press.

4.4.3.2 Centrifuges

Process Description

This is the most common mechanical dewatering equipment for water treatment residuals in New Zealand.

Centrifugal dewatering of solids is a process that uses the force developed by fast rotation of a cylindrical bowl to separate solids from liquids.

When a mixture of solids and water enters the centrifuge, it is forced against the bowl's interior walls, forming a pool of liquid that separates into two distinct layers. The solids cake and liquid centrate are then separately discharged from the unit.

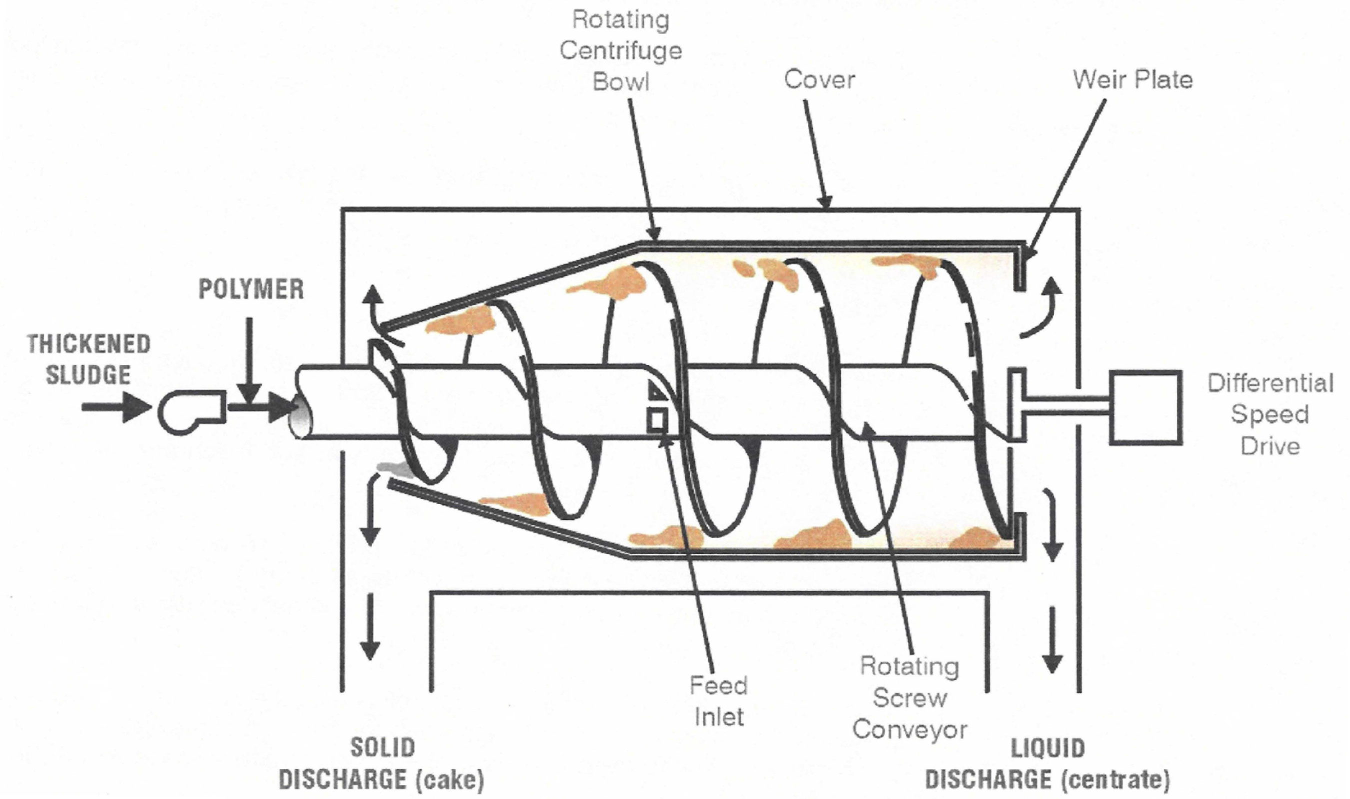
The most common type of centrifuge for this duty is the solids bowl or decanter centrifuge.

Design Outline

The decanter centrifuge is a rotating cylinder with a converging section at one end. A helical screw conveyor fits inside the bowl with a small clearance between its outer edge and the inner surface of the bowl. The conveyor rotates at a lower or higher speed than that at which the bowl is rotating and is analogous to the rake in a continuous thickener. The difference in revolutions per minute (rpm) between the bowl and scroll is known as the differential speed and causes the solids to be conveyed from the zone of the stationary feed pipe, where the sludge enters, to the dewatering edge, where the sludge is discharged.

Figure 4.4 shows a typical decanter centrifuge arrangement.

Figure 4.4: TYPICAL DECANTER CENTRIFUGE



The scroll pushes the collected solids along the bowl wall to the tapered end for final dewatering and discharge. Simultaneously, water flows in the opposite direction and overflows an annular weir. The kneading action of the screw removes all the water possible from the floc at the enhanced gravitational force and the sludge is discharged as a thick paste.

Organic polymers are usually used for flocculation as they improve centrate clarity and increase capacity. Dosages can vary from 3-7 kg polymer/tonne sludge solids.

A centrifuge operates to best effect when the feed is reasonably steady in both flow and solids concentration, as obtained from a well designed balancing tank. If overloaded, the centrifuge still removes the substantial solids content but with reduced quality of centrate recycled to the head of the plant.

4.4.3.3 Plate Press

Plate or filter presses for dewatering were first developed for industrial applications and, until the development of diaphragm presses, were only slightly modified for municipal applications.

Process Description

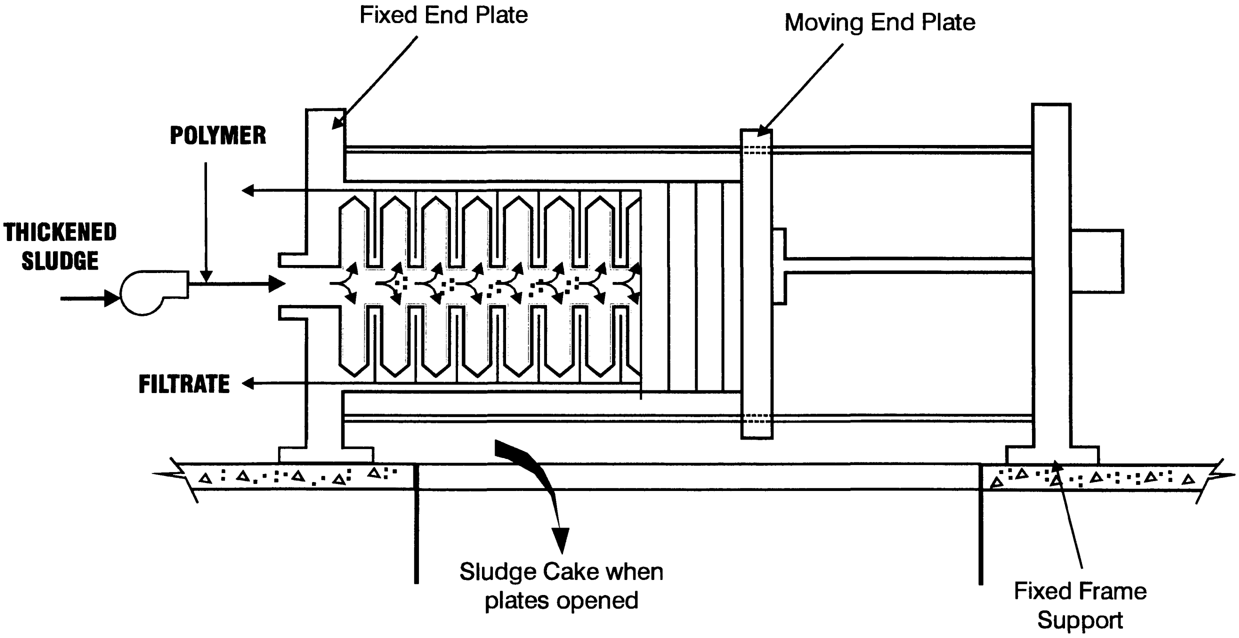
The equipment commonly used to dewater water treatment plant residuals is either the fixed-volume recessed plate filter or the diaphragm filter press. The diaphragm filter press was introduced within the last 10 years (AWWA, 1996).

The plate press converts a liquid sludge into slabs of fudge-like cake. The press consists of a series of plates, each with a fixed volume recessed section that forms the void or chamber into which the solids are pumped for dewatering. Filter media (cloths) are placed against each plate and retain the solids while allowing passing of the filtrate. Figure 4.5 shows a typical arrangement.

The sludge pumps maintain the pressure between the plates for several hours until the desired cake dryness is reached. The pumps are then stopped, the plates opened and sludge cake removed.

The diaphragm filter press is a variation of the plate press which uses a synthetic flexible diaphragm adjacent to each plant. After the initial filling, air or water is used to pressure the reverse side of the diaphragm thus squeezing the cake further and reducing the chamber volume. Pressures of about 1500 kPa are typically used.

Figure 4.5: TYPICAL PLATE PRESS ARRANGEMENT



Design Outline

The size of the press will be determined by the quantity of sludge and the manning level on the plant. Presses with 25 mm thick cake may have pressing times as short as 2-3 hours, but will require high manning levels. If presses are designed to be unloaded every shift, a chamber of 32 or 37 mm may be used. Design of the plant is based on the required frequency of pressing, cake thickness and total solids load. Pilot trials are recommended for determining the required press capacity.

The process can be fully mechanical or automated. Even on a fully automated press, an operator will be required during the discharging period as the cake often sticks to the cloth or membrane.

4.4.3.4 Vacuum Filters

Vacuum filtration involves the use of a cylindrical drum encased in a filter material such as a porous fabric or synthetic cloth containing metal mesh.

The drum is rotated in a vessel containing conditioned sludge and a vacuum is applied to the inside of the drum. The vacuum extracts the water from the sludge and leaves the sludge cake on the filter medium. The equipment is costly and complicated and imposes a heavy structural load on foundations.

This dewatering process is not particularly successful on alum sludges and is unlikely to be used in New Zealand

4.5 *Testing for Performance*

A common way of indicating the likely performance of an intended solid/liquid separation process is to make laboratory tests or pilot tests on representative samples of material to be processed. The closer the test comes to modelling the actual process and the longer the test period, the more accurate will be the prediction. For well established processes plant design may confidently proceed based mainly on experience and a minimum of testing.

Common laboratory tests available include:

- For thickenability
    - physical analysis
    - chemical analysis
    - settling tests in stirred cylinders
    - sludge centrifuge test.
- For dewaterability
    - compressability index
    - specific resistance to filtration
    - capillary suction time (CST)
    - model dewatering equipment; centrifuge, filter press.

Thickenability tests are normally better used to maintain plant control or to investigate operating problems. The tests do not normally yield basic information to the plant designer.

The dewatering tests can yield results applicable to plant design but only if carried out on representative sludge samples. It is very difficult, if not impossible, to produce continuous polymer thickened sludge at laboratory scale as a feed for dewatering tests.

4.6 *Additional Sludge Handling Processes*

Pumping

Positive displacement pumps are usually preferred mainly because they do less damage to floc structures than centrifugal pumps and also because, with the avoidance of high linear speeds, energy losses are minimised. Sludges remain pumpable up to high concentration provided that the plant design ensures the sludge will flow freely into the pump suction at the required rate.

Equalisation Basins

Equalisation basins can be used to balance out flows and concentrations of waste streams. This will increase the efficiency of the following process or allow more certain and closer control of an operation.

The benefit of equalisation basins will generally increase as the size, complexity and degree of automation of the plant increases.

Composting

Alum sludges can be composted with a range of other organic materials. Composting an alum sludge with sewage sludge reduces the concentration of heavy metals in the saleable product. Composting trials with alum sludge and leaves have been successful with a recommended ratio of four parts leaves to one part alum sludge by volume. Growth studies have shown that alum sludge had no properties that inhibited its use as a plant growth medium (Cornwell, 1990).

4.7 Conditioning

Conditioning is used in sludge treatment to optimise the effectiveness of the thickening or dewatering process. Conditioning of water treatment plant residuals is generally done by chemical conditioning although physical conditioning does occur occasionally.

4.7.1 Chemical Conditioning

Chemical conditioning is included in most mechanical thickening or dewatering processes. This conditioning can involve the addition of inorganic coagulants such as ferric chloride and lime, or polymer. In New Zealand polymer is most frequently used as there are few cheap sources of inorganic salts. The type and dosage of chemical conditioners vary widely with raw water quality, chemical coagulants, pre-treatment, desired solids concentration and the thickening or dewatering process used.

A wide variety of polymers are available for use in the dewatering process. The most successful polymers used tend to be charged (anionic), with a high molecular weight (AWWA, 1996). Trials are needed on each sludge to find the most cost effective polymer. Initial screening of a range of polymers should be done by the supplier using quick, simple laboratory tests (jar tests). Two or three of the most suitable polymers should then be trialled on the full-scale process.

Polymers can be obtained in a variety of dry and liquid emulsion forms. Handling should be done with care as they can become a health and safety issue through making surfaces slippery and are difficult to clean up. Polymer handling requires substantial dilution with high quality water and a time delay between dilution and use in sludge conditioning.

Polymers are organic compounds with linear molecules along which ionising groups are attached. These groups may be anionic (have a net negative charge), cationic (have a net positive charge), or non-ionic. Polymer conditioning is not fully understood but it seems that when the polymer is in solution the charged groups repel each other causing the molecules to stretch out. These groups then encounter sludge particles with sites having an opposite charge and become attached. As the polymer sites become full and the charges neutralised the molecule tends to curl up drawing the sludge particles together.

Polymers are added to raw water to assist sedimentation and filtration, usually at doses <0.2 g/m^3. Polymer dose rates for sludge thickening and dewatering tend to be higher giving rise to concerns if the separated liquids are recycled to the raw water. Further discussion on this is included in Section 4.8.

4.7.2 Physical Conditioning

There are a number of physical processes used internationally but unlikely to be used in New Zealand. Such processes use precoat or nonreactive additives, or conditioning by freezing and thermal methods. (Cornwell and Koppers, 1990).

4.8 Recycled Liquors

Historically, filter backwash has been returned to the raw water inlet for further processing. An equalisation basin was regularly used so that the spent backwash water was maintained at less than 10% of the incoming raw water.

Where recycling was not maintained as a regular rated flow difficulties have been experienced in controlling water treatment units.

Concerns have increased in recent years over the recycling of micro-organisms, aggravation of taste and odour problems, increase in disinfection products and increase in polymer monomer concentrations. These concerns have generated a greater interest in improving the understanding of backwash water characteristics. There is increasing use of some settlement and/or disinfection of backwash prior to recycle.

Natural organic matter is returned to the raw water with the recycle too, and this has been seen overseas to cause regrowth in watermains. Many wastewaters and sludges that have been stored for more than a day or two develop anaerobic conditions which release soluble iron and manganese to the supernatant which may not be removed in the water treatment process.

In New Zealand the main concerns are over the return of certain micro-organisms, specifically *Giardia* and *Cryptosporidium*, and monomer concentrations.

There is concern that where relatively large doses of polymer have been used in sludge treatment, the return of supernatants may be inadvisable due to the risk of monomer getting into the supply. An analysis done by Water Research Centre in 1983 showed that provided the total polymer usage on a water treatment plant is less than the maximum permitted relative to the output of drinking water, and provided that the return flow is more or less continuous, there can be no technical objection to recycling. However, achieving this status may impose more effort and cost than is warranted.

The above shows that understanding raw water, coagulant and polymer characteristics and the effect of recycle flows through a water treatment plant is of increasing importance as more processing of residuals is carried out.

5. Disposal Options

Historically, direct discharge of water treatment plant wastewaters to surface waters has been the most common method of disposal. The wastewaters originally comprised mainly natural solids. Over time there has been increasing use and awareness of the waterway and increasing chemical content in many wastewaters.

Many wastewaters were subsequently settled prior to discharge and some removed for discharging elsewhere, with or without further treatment.

The predominant disposal options from New Zealand water treatment plants based on the 1996 NZWWA survey were:

Discharge to freshwater	45%
Discharge to sewer	23%
Discharge to landfill	14%
Discharge to lagoon	11%
Discharge to land	5%
Other discharges	2%
	100%

It is likely that the lagoons are periodically emptied with contents discharged to either landfill or land. Regional Plans include guidance and compliance levels for concentrations of contaminants in the receiving environment. These should be considered together with a literature search for any particular discharge application.

Characteristics of each disposal option are discussed below.

5.1 Discharge to Natural Water

5.1.1 Unmodified Wastewaters

The disposal of water treatment plant wastewaters containing sludges to a body of water is the most common method and may involve discharge to sea, river, lake, or seepage into groundwater. The primary requirement is that the receiving water has the capacity to cope with the sludge. It is by far the most economic method available as historically very little or no pre-treatment or handling has been required. However, legislative requirements have almost eliminated this method of disposal in most developed countries.

Discharge to natural water may be allowed in circumstances where the regional council is convinced that the discharge of unmodified wastewater will cause little adverse environmental effects. One such condition may be where the wastewater flow is very small compared with the flow of the receiving water, for example, some of the smaller water treatment plants discharging to the Waikato River. Unmodified wastewater may also be permitted to be discharged when the receiving water is particularly turbid, for example at the Waipaoa River water treatment plant at Gisborne. Some regional councils allow washwater from filtration systems where chemicals are not used to be returned to the natural water, for example at Ashburton, and perhaps in future at membrane filter plants.

5.1.2 Discharge of Settled Wastewaters

Wastewater from settling tanks and/or filters is frequently discharged to a sludge lagoon where the solids settle out, producing a much clearer supernatant which may then be permitted to be discharged to natural water. The quality of the lagoon supernatant may be affected by:

- the volume of the lagoon in relation to the rate that it receives wastewater
- the composition of the wastewater, mainly the solids loading
- whether the lagoon system receives any other inputs such as stormwater and runoff
- the position of the outlet in relation to the inlet
- whether there is more than one lagoon, in series
- whether there is any baffling to prevent the rush of fresh wastewater stirring up the lagoon
- the shape and dimensions of the lagoon, i.e. whether there is any short-circuiting
- the depth of the lagoon, and more particularly, the depth of water above the sludge level
- whether weeds and “sludge islands” reduce the effective flow path and thus retention time
- the degree of exposure to strong winds that can resuspend the sludge
- whether the lagoon is desludged while resting or while still in use
- whether any chemicals affect the quality

Guidelines on use of lagoons are discussed in Section 4.4.1.

5.2 Discharge to Sewer

Co-disposal of water treatment plant wastes with sewage has been widely practised overseas, often where the utility managing the sewage treatment plant also manages the water treatment plant. However there is in general a reluctance for sewage treatment plant managers to accept alum wastes.

Water treatment sludges can be discharged direct to sewer or be thickened first to reduce the volume. For thickened sludge discharge, care should be taken to ensure settlement is not a problem in the sewer.

5.3 Discharge to Land

Sludges can be applied to land in either unmodified or concentrated forms. For either application the solids characteristics are the dominant focus. Alum sludges generally have similar organic carbon and organic nitrogen content, and organic material mineralisation rates to many soils. Water treatment plant sludges are therefore more like soil than sewage sludge (biosolids). Trace metal contents are strongly adsorbed to the aluminium hydroxide precipitates and as a result are generally not transported into plants or groundwater as easily as from biosolids (Cornwell, 1990).

Water treatment plant sludges contain very little phosphorus but large amounts of aluminium or iron hydroxide which are strong adsorbents of phosphorus. The decreased phosphorus availability means extra fertiliser may be needed to obtain the desired crop yields.

Applying sludge to forested land is an attractive option for forests which have problems with fertiliser runoff. The phosphate binding capacity of the sludge-amended soils may offer slow release fertilisation while minimising fertiliser loss to runoff. Generally, forested land has slow demands for phosphate and studies have indicated that sludge application is a viable option (Grabarek, 1987; Geertsema 1994).

The agronomic application of sludge to land presents an attractive and potentially sustainable method of disposal for many water treatment plants. Management strategies need to be implemented and parameters such as pH control, crop selection and availability, application rates, and fertiliser requirements must be included in the management planning (Herschel et al 1990).

5.4 Discharge to Landfill

Sludges are regularly codisposed in municipal landfills and this practice appears likely to continue for plants sited close to existing landfills. Currently most New Zealand landfill operators are only interested in the sludge being 'spadeable' or firmer which for many, implies a dewatered cake of 15% solids or greater. However landfill operators are going through the process of developing 'waste acceptance criteria' and more specific criteria may be utilised.

Traditional physical characteristics for landfill design based on soil mechanics include the following:

- plasticity - effects sludge handleability
- compaction data - this provides moisture/density curves and relates to landfill construction control.
- compressibility - effects landfill settlement
- shear strength - effects landfill stability i.e. controls the maximum height and slope of the landfill and ability to support heavy equipment.

For monofills, i.e. landfills containing a single material, all the above parameters could be considered and utilised with some accuracy. For codisposal, where a large range of different materials are mixed and compacted, the practical use of this data is limited. In general alum sludges have a high plasticity index and are relatively difficult to handle.

An effort has begun in Europe to define the handleability and stability of water plant sludge in terms of underdrained shear stress, as measured with a torque vane. The torque vane is commonly used in soil mechanics, and measures the torque required to turn the vane through a particular substance. The sludge shear stress is obtained by determining the maximum torque and relating this data to the calibration tables of the motor vane.

Although no standards currently exist for minimum shear stress of water plant sludges, Germany and The Netherlands have adopted a preliminary standard of 10 kN/m^2. To investigate whether 10 kN/m^2 is appropriate as a general standard for minimum shear stress, various types of dewatered sludges have been tested (Cornwell, 1990). The results of these tests show that the sludge shear stress depends on various factors, including the following:

- sludge type as a result of the composition of the raw water and treatment process applied;
- sludge conditioning;
- dewatering method; and
- any post-treatment with bulking agents.

Thus generalised relationships of physical properties with sludge moisture content have not yet been defined with any exactitude and further research is required.

American research described in Cornwell, 1992 proposed a shear strength of 2.87-3.83 kN/m^2, about a third of the European standard. The European standard was adopted to define a handleable sludge that would also support heavy equipment, whereas the US proposal only considered handleability. Landfill owners/operators in New Zealand will need to consider this when defining their acceptance criteria.

Monofills are possible with alum sludge but care is needed in their design. Specialised geotechnical advice should be sought for any specific application. In general it is likely that the design will include a structural retaining bund and deposition will benefit from compaction of thin layers of dewatered sludge cake with air drying allowed between successive layers.

5.5 Option Evaluation

Choosing the most appropriate disposal option can be aided by use of an evaluation matrix such as that shown in Table 5.1. Each criteria is assessed for the given situation and given a numeric value. The values are totalled for each disposal option, and compared.

Table 5.1: EXAMPLE OF AN OPTIONS EVALUATION MATRIX

RESIDUALS DISPOSAL OPTIONS PERFORMANCE MATRIX				
Evaluation Criteria		**Disposal Options**		
	Option A	**Option B**	**Option C**	**Option D**
Natural Environment Factors				
Habitat values				
Flora and fauna				
RMA matters				
Landscape values				
Subtotal				
Physical Environment Factors				
Soil suitability				
Air and water quality				
Noise				
Land use				
Subtotal				
Social Environment Factors				
Family resettlement				
Community disruption				
Health & Safety perceptions				
Lifestyle/quality of life				
Subtotal				
Cultural Environment Factors				
Waahi tapu/ancestral sites and areas				
Archaeological/historical sites				
Treaty of Waitangi issues				
Amenity values				
Subtotal				
Technical Factors				
Land area				
Effluent quality				
Conveyance				
Flexibility/complexity				
Subtotal				
Financial Factors				
Construction				
Operation				
Maintenance				
Property acquisition costs				
Subtotal				
RAW SCORE TOTAL				

6. Environmental Effects

The effects on the environment of disposal of sludge must be assessed in the consideration of options for disposal and in agreeing on the final option. The effects must also be considered in devising measures to avoid or mitigate effects, which may be required in order to obtain consents, and which should be included in any sludge management plan.

The environmental effects and their significance will vary according to the circumstances and there is little in the way of national criteria on standards to offer guidance in this consideration

A number of studies have been taken of the different effects of discharges on particular environments. These are referred to below.

6.1 Impacts on Receiving Waters

6.1.1 Toxicity Effects

Aluminium

Aluminium is amphoteric i.e. soluble in acidic and basic solutions but quite insoluble at neutrality. The trivalent state, Al^{3+} is the only naturally occurring oxidation state found in solutions and solids. The aqueous chemistry of aluminium is extremely complex and is not considered here in detail. In general when mixed with turbid water at pH 4.0 to 8.5, aluminium sulphate forms new compounds especially with phosphates and organics, which are insoluble and precipitate. The major factor controlling aluminium solubility is pH. Within the pH range of 5 to 6 aluminium bonds with phosphate and the resulting compound is removed from solution and can cause nutrient depletion of the water.

When aluminium is mobilised in lakes and streams it may be toxic to aquatic life. It is difficult to generalise about the environmental impacts of coagulant sludges on receiving waters because such impacts are inherently dependant on an array of physical, chemical and biological stream parameters.

The form in which aluminium appears varies with pH so its toxicity is also pH dependent. This is acknowledged in Table 2.1 of ANZECC (1992) which includes a water quality guideline for protection of aquatic ecosystems for total aluminium of 0.1 mg/L Al at pH >6.5, and < 0.005 mg/L Al for water with pH <6.5; they do not offer a guideline in marine waters. They do not suggest how to measure total aluminium at the 0.005 mg/L level. Calcium has been shown to ameliorate aluminium toxicity to fish by reducing stress induced by ion loss across the gills (O'Donnell et al 1984).

It should be noted that guideline levels are not set for specific environments and therefore may not be applicable to all situations.

Most aluminium toxicity research in the Northern Hemisphere in recent years has been related to acid rain studies. O'Donnell, Mance, and Norton (WRC Technical Report TR 197, 1984) stated:

"The toxicity of aluminium has been shown to be drastically reduced at pH values over 6.0. Experiments by Freeman and Everhart using rainbow trout fingerlings showed that aluminium toxicity under basic conditions is not entirely due to soluble species, but that suspended aluminium was also involved. The toxicity of dissolved aluminium was shown to be more acute than that of suspended species at similar concentrations. No toxic effects were exhibited by the fish at a dissolved concentration of 0.05 mg/L Al. For similar concentrations of total aluminium, the proportion of dissolved species increased as the pH rose from 7.0 to 9.0 and this was reflected by an

increase in the toxicity of the aluminium to the fish. Analysis of the mortality rates in these experiments showed that toxic effects may begin at lower concentrations of suspended aluminium than dissolved aluminium.

Apart from the acute toxicity data reported, chronic toxic effects shown by the fish included lack of appetite, darkening of colour, gill hyperplasia and loss of fright reaction. In the natural environment such chronic reactions might predispose fish to increased predation, loss of territory, and inability to maintain position in a flowing stream. Freeman and Everhart suggested a safe concentration for the protection of rainbow trout of 0.1 mg/L total aluminium in natural waters with a pH greater than 5.5.

Data reported in the only other recent study of aluminium toxicity in basic conditions also suggest that the element's toxicity increased with rising pH and was in proportion to the concentration of dissolved aluminium. However, no toxic effects were noted at high aluminium concentrations of 0.5 to 200 mg/L total aluminium at pH 7.0 after 10 days' exposure. This conflicts with Freeman and Everhart's data in which a concentration of 0.52 mg/L total Al produced 44% mortality after 45 days. This suggests that lower aluminium concentrations may still be chronically toxic over a longer time period.

Discharge of aluminium-rich sludge from filter backwashes has been associated with fish mortalities. In such streams concentrations of 1 mg/L Al and above may be necessary before fish mortalities result from short-duration exposure under basic conditions.

The toxicity of aluminium at low pH is complex, and interpretation is not assisted by the variety of fish species used in determining its toxicity. The toxic effects observed have varied with pH, species, life-stage, hardness and composition of the test media. These variations result from the complex chemistry of aluminium over a range of low pH values. Physiological differences between test species may also interact with the aluminium present, thus accentuating or reducing its toxicity.

The single most important factor affecting the toxicity of aluminium in acidic environments is the pH. At pH values of 4.0 and below, the toxic effects of the elevated hydrogen ion concentration are more important than the presence of low concentrations of aluminium. However, at pH values of 5.5 to 6.9 aluminium is considerably less toxic and there are no reports of any substantial sub-lethal effects at test concentrations. Between pH values of 4.0 and 5.2 aluminium can be acutely toxic to fish at concentrations as low as 0.1 mg/L."

Aluminium forms complexes readily with organic matter which can modify its toxicity. For example, trout lived for 10 days in water at pH 4.7 with 0.18 mg/L Al plus humic substances (74-80% of the Al was organically bound), whereas trout died within 2-3 days in the absence of humic substances - 98% of the Al was inorganic (Wilters et al 1990).

Generally it is considered that aluminium is more toxic to fish than to smaller animals. It is believed that aluminium coagulates the mucus on their gills, causing osmoregulatory and respiratory problems. Although aluminium may not be toxic to some plankton and small invertebrates, these species can be coagulated with alum and be removed from the water column in slow moving waters.

Potential threats to benthic communities also occur when direct discharge occurs into quiescent receiving waters. Under this circumstance little dilution or sediment transport occurs and the wastewater solids cover the bottom sediments, damaging the periphyton community. Also the high level of suspended solids associated with sludge discharge can depress productivity of phytoplankton.

The USEPA (1988) has developed an ambient water quality criterion for aluminium requiring that the instream and soluble aluminium level not exceed 0.087 mg/L on a 4-day average. The 1-hour average cannot exceed 0.75 mg/L Al.

In summary, for natural waters with pH greater than 6.5 (most New Zealand waters), the above discussion refers to "control levels" of:

total Al	0.10 mg/L	protection of aquatic ecosystems
total Al	0.10 mg/L	safe for trout
total Al (mean)	0.5 mg/L	no significant effects on fish observed
dissolved Al	0.05 mg/L	no effect on fish
soluble Al	0.09 mg/L	4 day average (ambient water quality criterion)
soluble Al	0.75 mg/L	1 hour average (ambient water quality criterion)

Aluminium levels in livestock drinking water and irrigation water commonly have guideline limits of 5 g/m^3 or about 50-60 ppm as alum (Ayers, 1985; Victoria EPA, 1983; ANZECC, 1992).

Heavy Metals

Water treatment plant wastewater would not normally be expected to contain high levels of heavy metals because careful source selection should avoid contaminated raw water. Apart from raw waters containing a geothermal input, the main source of some heavy metals could be the chemicals used in the treatment process. These are discussed in Section 3.

Carbon Dioxide

Carbon dioxide is used in the treatment process for corrosion control and as a buffering agent against pH change. The following excerpt from Alabaster (1980, p23) discusses the toxicity effects of CO_2 and its relationship with pH.

> *"The discharge of acid wastes into a water containing bicarbonate alkalinity will result in the formation of free carbon dioxide. If the water is hard, sufficient free carbon dioxide may be liberated to be toxic to fish, even though the pH value does not fall to a level normally considered to be lethal (Doudoroff and Katz, 1950). In well aerated waters the toxic levels of free carbon dioxide are usually above 100 mg/l for rainbow trout (Salmo gairdneri) (Alabaster, Herbert and Hemens, 1957). However, Lloyd and Jordan (1964) found that much lower levels can considerably reduce the survival times of fish within a range of low pH values which would not otherwise be lethal. In water containing 10 mg/l free carbon dioxide or less, the median lethal pH value for fingerling rainbow trout was 4.5 after 15 days exposure, but where the water contained more than 20 mg/l free carbon dioxide, the median lethal pH value rose to 5.7; this increased toxicity was apparent only after a day's exposure to the test conditions. It is, therefore, difficult to interpret some published data where the level of free carbon dioxide in the test conditions is either not given or cannot be calculated".*

Polymers and Monomers

There is some concern that where relatively large doses of polymer have been used in sludge treatment the discharge of supernatants may be detrimental to receiving waters. There is little toxicity data on this issue in the literature; the main reference found is summarised below.

The polymers (or polyelectrolytes) commonly used are polyacrylamides or polyamides. Stitting Toegepast Onderzoek Waterbeheer (STOWA), (1995) reported investigation into the environmental impact of polyelectrolytes and their by-products used in sewage treatment plants in The Netherlands. They only investigated cationic polyacrylamides, and only the effect of liquid effluents on receiving waters. They found no methods of analysis for polyacrylamides in sludge or in water so they found no data about their concentration in natural waters.

They considered that most cationic polyacrylamides would be associated with the treatment plant sludges and that any which escaped to natural water would be adsorbed strongly to the humic substances in the water (anionic polyacrylamides are not adsorbed so may be toxic). Cationic polyacrylamides are generally poorly biodegraded. The side chain of some polyacrylamides can hydrolyse forming choline which is biodegradable, and an anionic polyacrylamide which is not. If the main chain of the polymer degrades it usually only breaks down into smaller polymer units (oligomers) which are not biodegradable - note that this can also result from over-vigorous stirring or pumping of the polyacrylamide solution.

STOWA summarised the toxicity data (96 hour LC_{50} or EC_{50}) they found for cationic polyacrylamides; this is reproduced in Table 6.1.

Table 6.1: SOME LC_{50} OR EC_{50} (96 h) FOR CATIONIC POLYACRYLAMIDES

Organism	LC_{50} or EC_{50} (mg/L
fish	0.06-1000 (most <100)
algae	0.2-7500
bacteria	0.9-7500
crustacea	<0.06-1000
insects	<6.25->100

Values of Cationic Polyacrylamides

STOWA derived a NEC (No Effect Concentration) of 0.0012 g/m^3 in freshwater for cationic polyacrylamides. They developed a model to estimate the contribution of cationic polyacrylamides to natural water from sewage treatment plants in The Netherlands and concluded that the environmental risk was negligible.

STOWA found references to several potential by-products of cationic polyacrylamides but considered the only one of any significance was acrylamide (up to 0.1% of the polymer). They found hydroxyproprionitrile (up to 0.05%) to be a significant by-product of anionic polyacrylamides. They found that acrylamide in natural water is fairly quickly biodegraded to below its detection limit.

STOWA summarised toxicity data (96 h LC_{50} or EC_{50}) they found for by-products of cationic polyacrylamides; this is reproduced in Table 6.2.

Table 6.2: SOME TOXICITY DATA (96 h EXPOSURE) FOR POLYACRYLAMIDE BY-PRODUCTS

Organism	LC_{50} or EC_{50} - acrylamide	LC_{50} or EC_{50} - hydroxypropionitrile
fish	8-460 mg/L	0.215-1.37 mg/L
bacteria	13500	
algae	72	
crustacea	2.0-160	
insects	410	

STOWA derived a NEC (No Effect Concentration) of 0.2 mg/L in freshwater for acrylamide and 0.000215 mg/L for hydroxyproprionitrile. They developed a model to estimate the contribution of polyacrylamide by-products to natural water from sewage treatment plants in The Netherlands and concluded that neither substance presented an environmental risk.

6.1.1 Physical Effects

Colour, Turbidity and Suspended Solids

USEPA (1976) developed a criterion that stated that the increase in colour (in combination with turbidity) should not reduce the depth of the compensation point for photosynthetic activity by more than 10% from the seasonally established norm for aquatic life.

ANZECC (1992) includes a water quality guideline for colour and clarity for the protection of aquatic ecosystems. It states that there should be less than a 10% change in the euphotic depth in freshwater and marine waters.

D Rowe and T Dean of NIWA updated delegates at the 1996 New Zealand Limnological Society annual conference about their study on the effects of increased levels of suspended solids on fish feeding ability. They found in tank experiments that although most native species were primarily visual feeders, several could feed in the absence of light cues, and use other sensory systems (e.g. lateral line) for prey location and capture. Despite this ability, feeding rates of most species were reduced by turbidities above 320 NTU. Banded kokopu was the most sensitive species with feeding rates being depressed above 20 NTU. Juvenile rainbow trout and adult common bullies were relatively adept at feeding in the dark; while their feeding rates were relatively unaffected by SS, selectivity for larger prey was reduced as SS levels increased.

The effects of discharge fluctuation and the addition of fine sediment on stream fish and macroinvertebrates below a direct-feed water filtration facility was studied by Erman (1988). The major effect was found to be caused by the fluctuating backwash flows, containing fine sediment, that displaced small fish downstream and created unstable benthic substrates for invertebrates.

6.2 Impacts of Sewer Discharge

Co-disposal of water treatment plant wastes with sewage has been widely practised overseas, often where the utility managing the sewage treatment plant also manages the water treatment plant. However there is in general a reluctance for sewage treatment plant managers to accept alum wastes.

Concerns expressed include potential for increased final effluent suspended solids, decreased effective sludge digester capacity, overloading of primary clarifier and sludge removal systems, and overloading of dewatering operations (van Nieuwenhuyze, 1990).

Disposal to a sewer increases both the volumetric and solids loading on the sewage treatment plant, the latter without the benefit of any gas production (AWWA, 1986; Bishop, 1978). Improved BOD_5/COD removal in the settling tanks has been reported although generally to a minor extent (Cornwell, 1990).

A potential beneficial effect of codisposal is the ability of the aluminium in the sludge to bind phosphorus. The alum sludge also has a strong ion-exchange capacity and may fix heavy metals that are present in the sewage sludge. Phosphorus is generally present in high concentrations in sewage and can be a cause of eutrophication of estuaries, lakes and rivers. Studies dosing alum and iron salts at the sewage plant have achieved in excess of 90% phosphorus removal, but there do not appear to have been many phosphorus removal studies when adding water treatment plant wastewater to the sewer.

A number of US studies are summarised by AWWA (1996). The findings from these studies confirmed that addition of coagulant sludge to sewer:

(1) increased the volume of sludges to be treated

(2) improved phosphorus removal from the sewage

(3) may decrease sludge concentration in the primary clarifier underflow.

The following comments summarise known effects on wastewater treatment processes. For further information refer to AWWA (1996).

- If the dosing of coagulant sludge is equalised so that surges do not occur, and the dose is kept below 200 g/m^3, no direct effect on the activated sludge is likely to take place although downstream process effects or solids handling process effects are possible (van Nieuwenhuyze, 1990).

- Unless alum residuals are introduced at a very high rate, anaerobic digesters are unlikely to be significantly effected, provided the digester has adequate capacity to accommodate the increased solids loading. In assessing digester capacity, the reduced volatile solids content of the sludge feed must be taken into account. This was confirmed by bench scale trials at Palmerston North where no toxic effects were noted at loadings up to 70 g Al/m^3 of digester contents. (J Anderson internal reports).

- Sludge thickening and dewatering operations may show little effect from addition of waterworks sludge to sewage sludge provided their total solids capacity are not exceeded. Many studies, both pilot and full scale have been reported indicating minor variations in dewatering characteristics, some negative and some positive. It appears that thickening and dewatering effects cannot be predicted accurately without actual testing. If the impacts of dewatering sludge are critical to subsequent operations, the effects of water treatment plant sludge should be fully investigated before full scale implementation is committed.

Edwards (1997) investigated the effects of discharging iron coagulation sludges to sewers. A number of benefits found included sulphide removal, via direct reaction with the sludge, and struvite formation was prevented throughout the solids handling facilities. Disadvantages included elevated trace metal concentrations in the wastewater and increased solids loading.

6.3 Impacts of Land Discharge

Water treatment plant sludges contain very little phosphorus but large amounts of aluminium or iron hydroxide which are strong adsorbents of phosphorus. The decreased phosphorus availability means extra fertiliser may be needed to obtain the desired crop yields. Trace metal contents are strongly adsorbed to the aluminium hydroxide precipitates and as a result are generally not transported into plants or groundwater as easily as from biosolids (Cornwell, 1990).

Applying sludge to forested land is an attractive option for forests which have problems with fertiliser runoff. The phosphate binding capacity of the sludge-amended soils may offer slow release fertilisation while minimising fertiliser loss to runoff. Generally, forested land has slow demands for phosphate and studies have indicated that sludge application is a viable option (Grabarek, 1987; Geertsema, 1994).

Elliott and Singer (1988) investigated the impact of an iron sludge on the fertility of a silt loam soil by looking at the effect sludge incorporation had on the growth of tomato shoots. They concluded that land application is a feasible option for disposal and, provided that a suitable crop is chosen and the sludge loading is monitored, there should be no detrimental effects to plant growth.

Although alum sludge contains few if any plant nutrients (usually particularly low in nitrogen), it may contribute other beneficial properties such as improving soil structure, increasing water retention, and minimising fertiliser run-off. The incorporation of sludge into soil will improve the soil's cation exchange capacity and enhance the soil's ability to retain fertiliser.

Ahmed et al (1998) reviewed the impact of alum sludge on changing soluble aluminium concentrations in soils, and plant growth. They found that, due to its inherently high pH and buffering capacity, alum sludges were found to possess minimal soluble aluminium concentrations by comparison with naturally occurring soils. When mixed with an acidic soil of low buffering capacity, the pH increased into the neutral-to-alkaline range, the bulk density decreased, the infiltration rate increased, and the plant available nitrogen and plant yield increased. The main drawback of the sludge was its large phosphorus fixing capacity, which generated phosphorus deficiencies in plants and necessitated large applications of fertiliser.

Although aluminium is not listed, ANZECC (1992) does note that crop toxicity effects due to aluminium in the soil solution water can be seen at 0.1-0.5 mg/L Al, and that aluminium leaches into the soil water as the soil becomes increasingly acidic.

Water treatment plant sludges are generally not contaminated with pathogens and public health safety is unlikely to be an issue. Application of water treatment plant sludge is likely to improve soil water retention and physical structure.

Heavy metals in the sludge derive from previous contamination of the raw water solids prior to water treatment and contaminants in the purchased water treatment chemicals. Careful management of the raw water catchment and chemicals specifications will reduce heavy metal concentrations to low levels. These levels can be monitored against existing guidelines e.g. ANZECC 1992, DOH 1992, for any particular application. For most water treatment plant sludge applications to land, heavy metal contamination should not be a dominant issue.

When applying water treatment plant sludge to land the solids loading must allow for the sludge to crack and fully dry out, otherwise the roots of any vegetation will become waterlogged and starved of air. As the sludge is chemically neutral it will not produce a polluting leachate.

The agronomic application of sludge to land presents an attractive and potentially sustainable method of disposal for many water treatment plants. Management strategies need to be implemented and parameters such as pH control, crop selection and availability, application rates, and fertiliser requirements must be included in the management planning.

6.4 *Impacts of Disposal to Landfill*

In a monofill situation, environmental impacts will the same as land disposal, as discussed in Section 6.3. For codisposal, any impact will become part of the overall impact of the landfill. It should be noted that in either case the toxicity of landfilled alum sludge is pH dependent.

6.5 *Other Environmental Impacts*

Table 6.3 summarises other possible environmental impacts and possible mitigation measures.

Table 6.3: SUMMARY OF POSSIBLE ENVIRONMENTAL IMPACTS

Impact	Mitigation Measure	Comments
Contamination in surface waters	Reduce solids content of discharge Reduce contaminants in sludge Discharge at high receiving water flows	Possibly most sensitive impact Severity will depend on use of and fauna in receiving waters
Contaminant in soil	Reduce contaminants in sludge Limit application loading	
Contaminant in groundwater	Reduce contaminants in sludge Limit application loading, control soil pH	Unlikely to be a major concern
Access restrictions	Limit application loadings and timing	Can be managed to avoid common usage periods
Noise	Acoustic control of plant and operators, specified operating hours	Relates to vehicle movements and disposal operations
Dust	Use water cart, lower speed of movement	Relates to vehicle movements and soil operations
Odour	Operational changes, chemical addition	Few odours are associated with Water Treatment Plant operations. Localised odours are possible at open air operations.
Insects and other pests	Operational changes, chemical addition	Possible but uncommon for open air processes to be breeding grounds
Space requirements		Limited space tends to drive process selection to mechanical options and away from open air processes
Public safety	Health and Safety procedures	Mechanical handling involves highest concerns with use of heavy equipment. Exclude public from lagoons.
Traffic/transportation	Minimise truck movements and/or axle loads. Alter timing of movements. Seal tailgates to minimise spillage.	Tends to affect neighbours and other route users.

7. Further Considerations

7.1 Waste Minimisation

Over the years, there has been an increased emphasis on minimising negative impacts on the environment and on-site disposal is generally no longer permissible, requiring the utility to construct a monofill, discharge to the sewerage system or to agricultural land, or transport the material to a nearby landfill.

While all of these are viable options, implementation is becoming more problematic. Construction of an onsite monofill requires available land and is costly both to construct and to monitor. Discharge to a sewer may be limited by a wastewater treatment plant's capacity to accept more solids, both of which might affect the treatment process. The wastewater treatment plant is also subject to tight regulatory restrictions requiring higher and more expensive levels of treatment which in turn have a cost implication for the water treatment plant if they use this disposal route for their residuals. The landfill option will also become increasingly restricted and costly.

The net result of all these changes is that utilities need to develop long-term flexible management plans. Innovative techniques for reducing waste need to be considered and cost-effective reuse options adopted. The benefits to the plant would be reduction in costs and improved public acceptance. The dilemma, however, is the lack of experience in the treatment techniques for reducing residual waste stream volume or recovering chemical coagulants, which may be costly, at present. The markets for beneficial reuse of residuals at this time are limited.

Some of the available waste minimisation, recovery, and reuse options that apply primarily to residuals generated from the coagulation/filtration processes are discussed below.

7.1.1 Process Modifications

The best means of reducing waste is to minimise its production and optimise the types and quantities of coagulants used with associated mixing conditions. Control at the source is the most effective first step and should be evaluated in all facilities. For example, where raw water is of relatively good quality, the selection of direct filtration over conventional treatment can result in significant savings in chemical and residual handling costs (Monk, 1987). Process optimisation allows facilities to achieve a balance between water quality, chemical addition and physical parameters.

To achieve optimal chemical coagulation, the following points should be considered:

- Better control of coagulant dose and pH: in some cases it might be beneficial to use a higher than optimum coagulant dose, as this can reduce pH sensitivity. Also optimisation of mixing energies and contact times can improve performance from the same doses.
- Choice of coagulant: performance may be improved by changing to a more suitable coagulant. Available are: alum, iron salts, synthetic polymers and polyaluminium chloride (PAC). Many utilities have continued the use of alum, but decreased its dosage through the use of a coagulant-aid polymer.
- Simultaneous use of additional chemicals: partial substitution of metal ion coagulants with cationic polyelectrolytes will reduce sludge volumes and may improve treated water quality. Actual data from any operational water treatment plant demonstrate that if 0.1-1.5 mg/L of cationic polymer are applied in conjunction with alum, the alum dosage can be reduced by 30-50%. The combined application of 0.2-0.3 mg/L of anionic or nonionic polymer and alum often improves flocculation dramatically.

7.1.2 Volume Reduction

Various dewatering and drying methods were discussed in Chapter 4 and their use is an important step in minimising waste.

Several mechanical dewatering techniques are available and in use throughout the country. Typically these use filter presses and centrifuges. Mechanical dewatering significantly reduces the residuals volume by removing some of the water, thereby increasing the % solids content.

The applicability of any process depends on raw water quality, coagulant practices, and desired dewatered cake characteristics. The need for chemical conditioning or pretreatment is also site specific. Most of these techniques were developed for application in the wastewater industry. As a result, little information is available relative to performance in the drinking water industry.

As discussed in Chapter 4, a drying step would further reduce the quantities of solids generated for reuse or disposal. Use of such processes may achieve a dewatered cake with a greater than 90% solids content. However, applicability to the potable water industry is still unclear and in need of further investigation (AWWA, 1996).

7.1.3 Chemical Recovery

Coagulant recovery not only recovers a resource (i.e., coagulant) but also minimises waste by extracting alum or iron coagulants from the waste stream. Extraction is achieved by acidification, which puts the metals back into solution. Critical design and operational factors include extraction pH and acid contact time. Extraction pH is typically in the range of 1.8 to 3.0. Acid contact time of 10-20 minutes appears to be reasonable based on full-scale operations data and laboratory testing (Saunders, 1991).

Factors that must be taken into account when considering the feasibility of coagulant recovery are:

- The quality of the recovered coagulant
- The impact of coagulant reuse on treatment plant operation and resulting final water quality.

While the acidification process is beneficial for achieving dissolution of the coagulant, concentrations of coagulant impurities and raw water contaminants may also become dissolved and get recycled to the head of the plant. Although these contaminants may again be removed from the water supply by the coagulant, settling, and filtration processes, the net result is to increase the concentration of contaminants. This impact must be carefully considered as the regulatory climate becomes increasingly stringent.

7.2 Reuse Options

Several alternative uses and disposal methods for residual solids are presented in *Slib, Schlamm, Sludge,* (Cornwell, 1990) and in other recent articles (Copeland 1994, 1995). These alternatives include options such as land application and reclamation, turf farming, blending with compost material, and cement production. Implementation of these alternative methods may prove to be more cost-effective than traditional disposal methods and will likely achieve a high degree of public acceptance.

Potential concerns have been raised about aluminium levels in alum coagulant residuals and their tending to build up available phosphorus in the soil. Phosphorus is needed as a nutrient of vegetation. To overcome this potential deficiency, supplemental fertilisation may be necessary.

A number of commercial products are discussed below:

Turf Farming: use as new soil base. Turf grass has a relatively low nutrient demand but requires significant moisture levels, particularly for initial growth phases. Dewatered water treatment plant solids applied to a turf farm at the beginning of the seeding process can provide excellent water retention capabilities.

Top Soil Blending: commercial producers of top soil utilise a variety of soil products to develop a marketable product for nurseries, home owner, professional landscapers etc. In this process the raw soils are screened and blended with some organic material before being sold as a product. Water treatment plant solids can be blended during the top soil production process to increase the water retention capabilities and aeration. The amount of water treatment plant solids added is typically 10% or less and is a function of consistency, quality and availability. The acceptable quality of water treatment plant solids is specific to individual top soil producers.

Eutrophication Control: A novel use of thickened sludge was reported by Cornwell, (1990). Alum sludge (11% dry solids) from Tampa, USA, was applied across a eutrophic lake so it would settle over the bottom sediments. A surface application of sludge at 200 mg/L was shown to result in up to an 81% decrease in chlorophyll a after 72 hours, with significant removals of dissolved orthophosphate, total phosphorus, total organic carbon and turbidity, as well as more than 90% removal of several heavy metals. A dose of 1000 mg/L was found to be excessive.

7.3 Risk Management

Risk management is recognised as an integral part of good management practice and can be used to minimise risks and their impacts. It is an iterative process consisting of well defined steps which can be applied to any situation where an undesired or unexpected outcome could be significant or where opportunities are identified. Risk Management Standard AS/NZS 4360 can be used to provide a generic framework for identification, evaluation, treatment and monitoring of risk.

Risk management can be applied to selective studies, e.g. waste minimisation at the water treatment plant, or the possibility of specific environmental damage, or it can be applied to more complex issues such as the process of formulating a complete residuals management plan.

The main elements of the process are:

- Establish the context - the boundaries of the system, the nature of the risks and the stakeholders involved.
- Identify the risks - what, why and how things can arise.
- Analyse the risks - what are their controls, likelihood and consequence, use an initial screening/matrix to develop a hierarchy for focusing further effort.
- Assess and prioritise risks - evaluate against established objectives, use risk analysis simulations for quantifiable events e.g., Monte Carlo simulations.
- Treat risks - develop a risk management strategy i.e., accept, avoid, prevent, mitigate or transfer to others, e.g. insurance.
- Monitor and review - ensure implementation is according to plan, have any changes occurred, is performance up to expectations.

The entire process is iterative. The most important benefits from using this systematic approach are a greater level of safety, reduced losses e.g., production, chemicals and equipment, and improved environmental responsibility.

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