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The Safe Disposal of Hazardous Wastes

The Special Needs and Problems of Developing Countries

Volume II

Roger Batstone, James E. Smith, Jr., and David Wilson, editors

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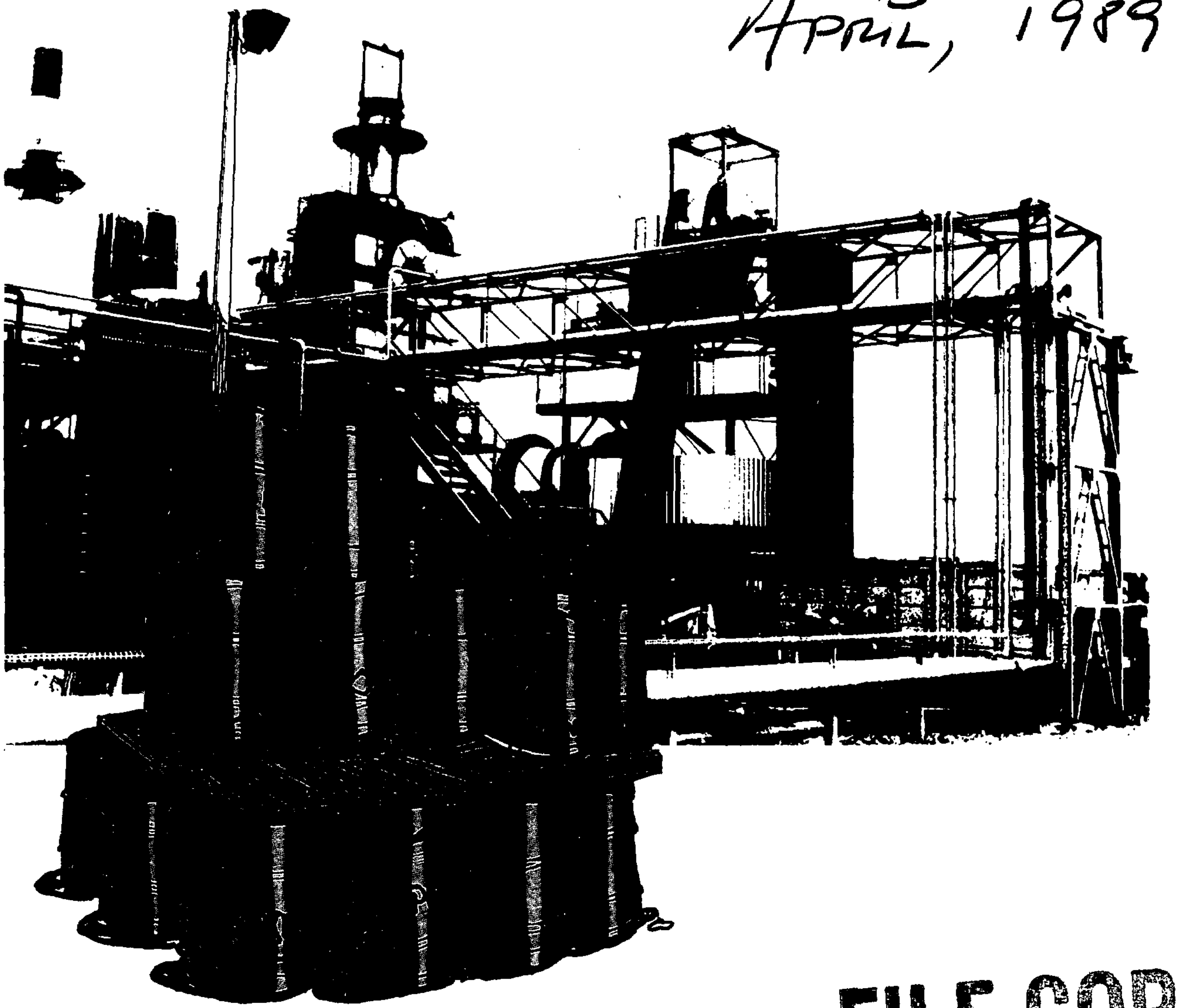

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Roger Batstone is an industrial risk management specialist in the World Bank's Environment Department. James E. Smith, Jr., a technical adviser to the World Health Organization, is a senior environmental engineer at the Center for Environmental Research Information, U.S. Environmental Protection Agency. David Wilson, a consultant to the World Bank and the United Nations Environment Programme, is the director of Environmental Resources, Ltd.

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Volume II

CHAPTER 6 - Hazardous Waste Treatment Technologies

6.1 Introduction

Many different hazardous waste treatment technologies can be used prior to ultimate disposal. Their aims are modification of the physical and/or chemical properties of the waste. They reduce volume, immobilize toxic components or detoxify. The choice of the best practicable way of treating a given waste depends on many factors, including the availability and suitability of disposal or treatment facilities, safety standards, and cost considerations. No disposal route offers absolute safety, and any waste treatment or disposal technology has an associated level of risk.

It is not the intent of this chapter to provide a detailed discussion of industrial wastewater treatment. Rather the treatment of the hazardous wastes generated by these processes will receive emphasis, as will other hazardous residuals handling.

The array of possible processes for treating hazardous waste is large. A study was done in 1976 (DeRenzo 1978) of 47 engineering unit processes for their applicability to the task of treating hazardous industrial wastes. Of these processes, 10 could be applied for phase separation, 3 were basically for pre-processing of bulk solids or tars, 12 reacted chemically to destroy or detoxify the hazardous components, while 25 could be used to separate specific components within the waste stream. Some of these unit processes were commonly used for industrial waste treatment, while others required further R&D efforts before they would become commercially attractive. Four (dialysis, electrophoresis, freeze drying and zone refining) were immediately found not to be applicable to waste treatment. The treatment processes studied are shown in Table 6-1 and fell naturally into four classes: (1) phase separation processes, potentially useful in volume reduction or resource recovery; (2) component separation processes, capable of physically segregating particular ionic or molecular species from multicomponent, single-phase waste streams; (3) chemical transformation processes, which promote chemical reactions to detoxify, recover, or reduce the volume of specific components in waste streams; and (4) biological treatment methods, which involve chemical transformations brought about by the action of living organisms.

Selection of a treatment process for a given waste stream is not easy and involves consideration of: the nature of the waste, the desired characteristics of the output stream, the technical adequacy of the treatment alternatives, economic and financial considerations, environmental considerations, energy considerations, operations and maintenance considerations and the making of an overall evaluation.

6.2 Discussion of Typical Treatment Technologies

Physical, chemical and biological treatment systems that have been selected and employed successfully are discussed below together with their applications.

TABLE 6-1

Treatment Processes Identified

Physical

- Air Stripping
- Suspension Freezing
- Carbon Adsorption
- Centrifugation
- Dialysis
- Distillation
- Electrodialysis
- Electrophoresis
- Evaporation
- Filtration
- Flocculation
- Flotation
- Freeze Crystallization
- Freeze Drying
- High Gradient Magnetic Separation
- Ion Exchange
- Liquid Ion Exchange
- Steam Distillation
- Resin Adsorption
- Reverse Osmosis
- Sedimentation
- Liquid-Liquid Extracting of Organics
- Steam Stripping
- Ultrafiltration
- Zone Refining

Chemical

- Calcination and Sintering
- Catalysis
- Chlorinolysis
- Electrolysis
- Hydrolysis
- Microwave Discharge
- Neutralization
- Oxidation
- Ozonolysis
- Photolysis
- Precipitation
- Reduction

Biological

- Activated Sludge
- Aerated Lagoon
- Anaerobic Digestion
- Composting
- Enzyme Treatment
- Trickling Filter
- Waste Stabilization Pond

Pretreatment of Bulk Solids of Tars

- Crushing and Grinding
- Cryogenics
- Dissolution

6.2.1 Physical Treatment

These processes include various methods of phase separation and solidification. At the most basic level, phase separation encompasses lagooning, sludge drying in beds, and prolonged storage in tank processes. All three depend on gravitational settlement, and the first two also allow the removal of liquid by decanting, drainage and evaporation. Lagooning and tank storage are widely used to separate oil and water from mixed wastes, sometimes following preliminary treatment with emulsion breaking agents and occasionally, in the case of tank storage, combined with heating. A development of this last method is the burning of the recovered oil to raise process steam for tank heating.

(i) Solidification or Fixation Processes

These processes convert the waste into an insoluble, rock-hard material, and they are generally used as pretreatment prior to landfill disposal. The conversion is achieved by blending the waste with various reactants to produce a cement-like product.

Asbestos forms part of a class of natural fibrous hydrated silicates which are very widely used and well known for the occupational hazards (e.g., asbestosis and lung cancer) they present. Large quantities arise as waste from industry (e.g., more than 10,000 tonnes of asbestos waste arise in various forms each year in the London area alone), some of it as a dust, which is probably the most dangerous physical form. Because of its occupational hazards, strict regulations apply to its use, and waste should be prepared for disposal in a sealed condition, e.g., in sealed bags of polyethylene. Consolidation with waste cement, where possible, can be a useful precautionary measure (Fish 1977).

There are a number of sources of arsenical waste, including glass manufacture, timber preservation, treatment of hides and furs, agriculture and the smelting of copper, tin, zinc and lead. Arsenic is a well known poison and is usually considered carcinogenic to man; hence its industrial handling and disposal are generally controlled. Recovery is the most desirable method of disposal, but where this is not possible, disposal of arsenical waste is often by tipping under the most stringent safeguards. Casing in concrete would seem to be highly desirable for substantial amounts, but slurrying with cement or concrete presents difficulty, due to the formation of sodium arsenite which can inhibit the setting of cement (Fish 1977).

6.2.2 Sludge Processing

A substantial portion of industrial waste streams is water. Accordingly, the mass of residue requiring ultimate disposal can be substantially reduced by efficient dewatering. Most commonly this can be achieved by lagoons, drying beds, vacuum filters, belt-filter presses, filter presses and centrifuges. Often, too, the dewatering process is preceded by some kind of thickening. This may be a gravity process or some other type. Occasionally stabilization of sludge is practiced. This may be by a biological process or through the use of chemicals like lime.

6.2.3 Chemical Treatment

Chemical treatment methods are used both to facilitate the complete breakdown of hazardous waste into non-toxic gases and, more usually, to modify the chemical properties of the waste (e.g., to reduce water solubility or to neutralize acidity or alkalinity).

(i) Chemical Oxidation

Cyanide is a common poisonous waste which may occur in the liquid (aqueous solution) or solid form. Landfilling has been widely practiced in the past as a cheap disposal method, but its continued use on a wide scale is undesirable. Since cyanide can be rendered non-toxic by relatively simple methods, there is little need for its dumping. Aqueous cyanide wastes, including sludges, are readily treated by chemical oxidation. Most of this waste arises from plating shops (either as rinse water or sludge from the plating tanks) and therefore will often also contain toxic metals. The usual method of treatment is to oxidize the cyanide waste in alkaline solution with chlorine or hypochlorite. This process may be shown as:

$$CN^- + 2OH^- + Cl_2 = CNO^- + 2Cl^- + H_2O$$

The cyanate can be further oxidized by excess chlorine:

$$2CNO^- + 3Cl_2 + 4OH^- = 2CO_2 + N_2 + 6Cl^- + 2H_2O$$

While the quantities of residue produced by this process are variable, they are generally not significant. The major factor controlling the amount of residue generated is the concentration of metals which precipitate at the pH (about 8.5) of the process.

Where chromate waste is available, this may be used as the oxidizing agent and this also serves the purpose of reducing the hexavalent chromium ion to the much less toxic trivalent form. When the reaction is complete, the solution or sludge must be disposed of in a safe manner. Although if recoverable amounts of metals are present, it may be subjected to a metal recovery process.

Although the preceding method is effective, other methods are more likely to be used where large quantities are to be dealt with. One such method is an electrolytic process in which the electric current is passed through an aqueous cyanide solution containing chloride ions. Much solid cyanide waste arises from "bale-out" from the salts used in case-hardening steel, which may contain 12%-18% of cyanide. An obvious method of treating such salts is to crush them, dissolve them in water and then proceed as for aqueous cyanides. However, with large quantities of fused cyanide waste, high-temperature treatment methods may be preferable (Fish 1977; Suess 1983).

(ii) Heavy Metal Precipitation

Plating effluents, however, often contain in solution various heavy metals such as copper, nickel or zinc. These are subsequently removed by the addition of an excess of slaked lime or sodium hydroxide to precipitate them as water-insoluble compounds. Wastes processed by hazardous waste plants are most commonly precipitated as hydroxides. Hydroxide sludges have been found to be relatively resistant to metals leaching except for low pH extraction solutions. Alternative precipitants for heavy metal solutions include sodium sulfide, thiourea and dithiocarbonates, all of which produce insoluble sulfide precipitates. With certain exceptions sulfide precipitation is generally used as a final process after initial precipitation with lime or caustic soda. Some sulfide sludges are susceptible to atmospheric oxidation, making toxic metals potentially available in a leachable form. As noted previously often when treating for the removal of cyanide, metals are also precipitated.

(iii) Chemical Reduction

Chromic acid is a corrosive and highly toxic material widely used in the surface treatment of metals and in chrome plating. It is reduced chemically to the relatively non-toxic chromium (III) state. Various chemicals function effectively as reducing reagents, including; sulphur dioxide (SO_2); sulphite salts ($SO_3{}^{-2}$), bisulphite salts (HSO^{3-}) and ferrous salts (Fe^{2+}). A typical process is shown below:

$$2Na_2CrO_4 + 6FeSO_4 + 8H_2SO_4 = Cr_2(SO_4)_3 + 3Fe_2(SO_4)_3 + 2Na_2SO_4 + 8H_2O$$

This process is conducted at a pH of 2.5 to 3.0. The soluble chromium (Cr^{3+}) is then usually removed through alkaline precipitation according to:

$$Cr_2(SO_4)_3 + 3Ca(OH)_2 = 2Cr(OH)_3 + 3CaSO_4$$

Chemical reduction of Cr^{+6} followed by alkaline precipitation produces significant quantities of residue. The stoichiometry of $Cr(OH)_3$ production predicts 2 kg of sludge per kg of Cr^{+6} treated. Cr^{+3} can be precipitated with NaOH rather than line to produce less sludge.

Chromic acid as previously noted is a powerful oxidizing agent and may also be used to destroy cyanide waste, the chromic acid being simultaneously reduced to chromium (III). Although this process has been used by the waste disposal industry, it is unsuited to routine, in-house cyanide treatment.

(iv) Neutralization

Aqueous solutions of mineral acids arise in large quantities from engineering firms and chemical industries. Much arises from various metal treatments and contains metals such as iron, zinc, copper, barium, nickel, chromium, cadmium, tin and lead. They are extremely corrosive materials, but where their corrosiveness is a problem, they may be neutralized. Slaked lime is the cheapest alkali available and is therefore usually chosen for large-scale acid neutralization. The gypsum produced may present a problem in which case it can be landfilled after filtration.

Alkaline waste also arises mainly from engineering and chemical works, but its composition varies much more than that of acid waste and recovery of useful materials at disposal sites is even more difficult. Alkaline waste too comes from petroleum refining, paint factories and specialist cleaning firms. Apart from such solids as spent clay, catalysts, metallic hydroxides, there may be present phenolates, naphthenates, sulfonates, cyanides, heavy metals, fats, oils, tarry substances, natural or synthetic resins, etc. At present only the metals seem to be recoverable. For needed acid addition, sulphuric (H_2SO_4) and hydrochloric (HCl) acids are most commonly used. Sulphuric acid will form more insoluble precipitates and therefore more residue than hydrochloric acid (Fish 1977; Suess 1983).

(v) Oil/Water Separation

A large amount of this is tipped at present. This practice is certainly not recommended, however, since much of the organic matter is combustible (after separation of water, if necessary) in suitable incinerators. Some (e.g., waste lubricating oils) contains potential carcinogens and can foul water supplies. Nevertheless, treating waste oils is not always easy. Some can be skimmed off the top of an insoluble mixture mechanically as was noted previously under the Physical Treatment Section. Emulsions may, however, need considerable treatment to break them down (an example is treatment of the emulsion with aluminium hydrogen sulfate, followed by lime slurry to form a floc which absorbs the oil and which may subsequently be burned). In many cases, however, recovery of oils for reuse is possible. Fatty emulsions from the food industry can usually be broken down easily and the liberated fat either burned or, possibly, recovered for reuse (Fish 1977; Suess 1983).

However, some soil organisms can break down oily materials to harmless substances and it is not intended to rule out the possibility of biological treatment. This might eventually be developed to become a means of disposing of some oily waste, provided that toxic metals are absent. Further discussion occurs under the Biological Treatment Section in this chapter and in another chapter on land treatment.

(vi) Solvents/Fuels Recovery

Combustible organic solvents are frequently toxic and their vapours when mixed with air can be explosive. Most of this waste is recoverable and, indeed, often is recovered at the source. Where this is not possible, combustion would usually be the most appropriate method of disposal.

Non-combustible organic solvents include oily, greasy sludge composed of degreasing agents and paint strippers of the chlorinated hydrocarbon class which are highly toxic. Although described as incombustible, they can be, and indeed are, best disposed of in special high-temperature incinerators using diesel oil or other suitable auxiliary fuel and fitted with scrubbing equipment to remove the hydrochloric acid gas formed.

6.2.4 Biological Treatment

Many industrial wastes are treated by biological methods similar to those used for sewage treatment. Hazardous waste is occasionally amenable to such treatment, even though the concentrations of toxic materials present are often lethal to microorganisms. Major industrial users of land treatment have included petroleum refining, industrial organic chemicals, wood preserving, petroleum production, plastics, materials, residues and paints and allied products. The in-plant biological treatment of dilute aqueous effluents is well established, and microorganisms have been developed to selectively degrade specific toxic chemicals.

Successful co-treatment of industrial waste and domestic waste with addition of nutrients in biological systems is often practical. It is proved to be very economical and effective method in India today and many systems are in operation with good results and cheaper costs than with chemical treatments.

For example:

(i) Laminate factory effluent Hydrabad Phonol - 20,000 to 40,000 mg/1 is mixed with domestic waste (BOD 250-300) in ratio of 1:3 and treated with Activated sludge (Aeration 12 hrs) with 95 to 98% removal of phenol.

(ii) Trichlorophenol (2000 - 3000 mg/1) containing liquid effluent) from Drug Plant at Pimpri, Poona - volume 150-200 m^3/day, is treated entirely by itself with addition of N & P in activated sludge plant (Aeration 24 hrs) with removal efficiency of 95 percent plus

(iii) Indian Drug and Pharmaceutical Ltd. Hyderabad - effluent containing 2000 to 35000 mg/1 of phenol is treated with addition of nutrients N & P in trickling filter with recirculation ratio of 2.5 and removal efficiency of 90 per cent.

(iv) Liquid effluents containing Monochlorobenzene (MCB), Dichlorobenzene (DCB), D.D.T., chloral, etc. with concentration up to 50 mg/1 is treated with activated sludge at Hindustan Insecticide Ltd. Panvel with no trace of DDT or organics in effluents.

The natural microbiological activity in topsoil is also used in farming for degrading some organic chemicals, notably oily waste. Composting may also be useful for certain organic chemical products.

6.3 Treatment Technology Considerations

Hrudey (1985) reviewed the situation in Europe and North America. In terms of wastes to be handled, he found that heavy metal solutions and residuals are the dominant category on a weight basis with waste oils a distant second. Twelve major categories of process technology were encountered, and residue generation information was available for eight. These were neutralization, oxidation, reduction, precipitation, oil/water separation, solvents/fuels recovery, activated sludge and incineration.

Considerable and detailed information is presented in the following sections for processes thought to have the widest applicability to users of this manual. Included for each technology are: a general description, its applications/limitations for hazardous waste treatment, design considerations, design criteria, technology selection/evaluation and costs.

Rarely will any one unit treatment process be sufficient for aqueous waste treatment. Therefore, the discussions which follow include information on unit treatment processes which are frequently used in combination and any pretreatment requirements which are a prerequisite to effective use of each treatment process.

6.4 Precipitation/Flocculation

Precipitation is a physicochemical process whereby some or all of a substance in solution is transformed into a solid phase. It is based on alteration of the chemical equilibrium relationships affecting the solubility of inorganic species. Removal of metals as hydroxides or sulfides is the most common precipitation application in wastewater treatment. Generally, lime or sodium sulfide is added to the wastewater in a rapid mixing tank along with flocculating agents (described below). A simple form of the hydroxide precipitation reaction may be written as:

$$M{+}{+} + 2(OH)_2 = M(OH)_2 \text{ (Precipitate)}$$

Precipitation of the pollutant metal (M) in the waste stream by an excess of sodium sulfide may be written as:

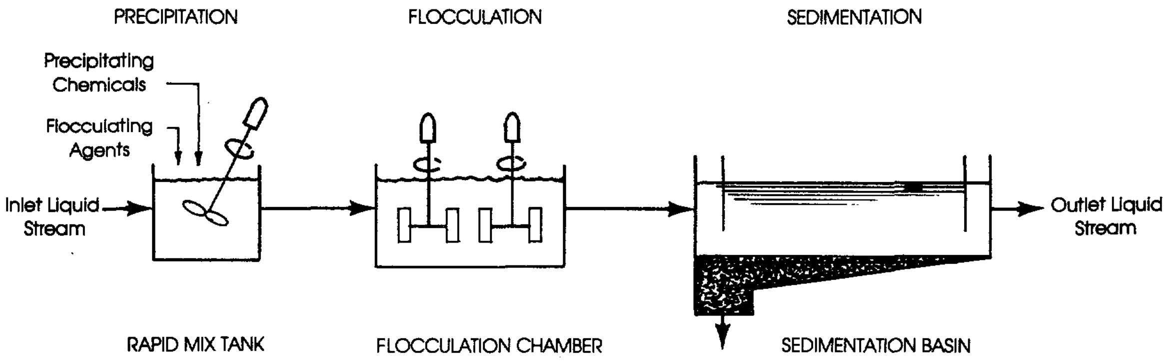
$$Na_2S + MSO_4 = MS \text{ (precipitate)} + Na_2SO_4$$

Following addition of the flocculants, the wastewater flows to a flocculation chamber in which adequate mixing and retention time is provided for agglomeration of precipitated particles. Agglomerated particles are separated from the liquid phase by settling in a sedimentation chamber, and/or by other physical processes such as filtration. Figure 6-1 illustrates a typical configuration for precipitation, flocculation and sedimentation.

Although precipitation of metals is governed by the solubility product of ionic species, in actual practice, effluent concentrations equal to the solubility product are rarely achieved. Usually, the amount of lime added is about three times the stoichiometric amount that would be added to reduce solubility due to the common ion effect. Figure 6-2 gives solubilities of various metal hydroxides and sulfides at various pH levels. The metal sulfides have significantly lower solubility than their hydroxide counterparts and more complete precipitation is achieved. Metal sulfides are also stable over a broad pH range. Many metal hydroxides, on the other hand, are stable only over a narrow pH range; metals reach a minimum solubility at a specific pH, but further addition of lime causes the metal to become soluble again. Therefore, dosages of lime need to be accurately controlled. This may be particularly challenging when working with aqueous wastes from waste disposal sites where wide variations in flow rates and quantities of metals are to be expected. The stabilities of metal carbonates are also quite dependent on pH (USEPA 1985).

FIGURE 6-1

Representative Configuration Employing Precipitation, Flocculation, and Sedimentation



After the addition of precipitating chemicals the precipitation reaction commences to form very small particles called precipitation nuclei. The flocculating agents allow these particles to agglomerate.

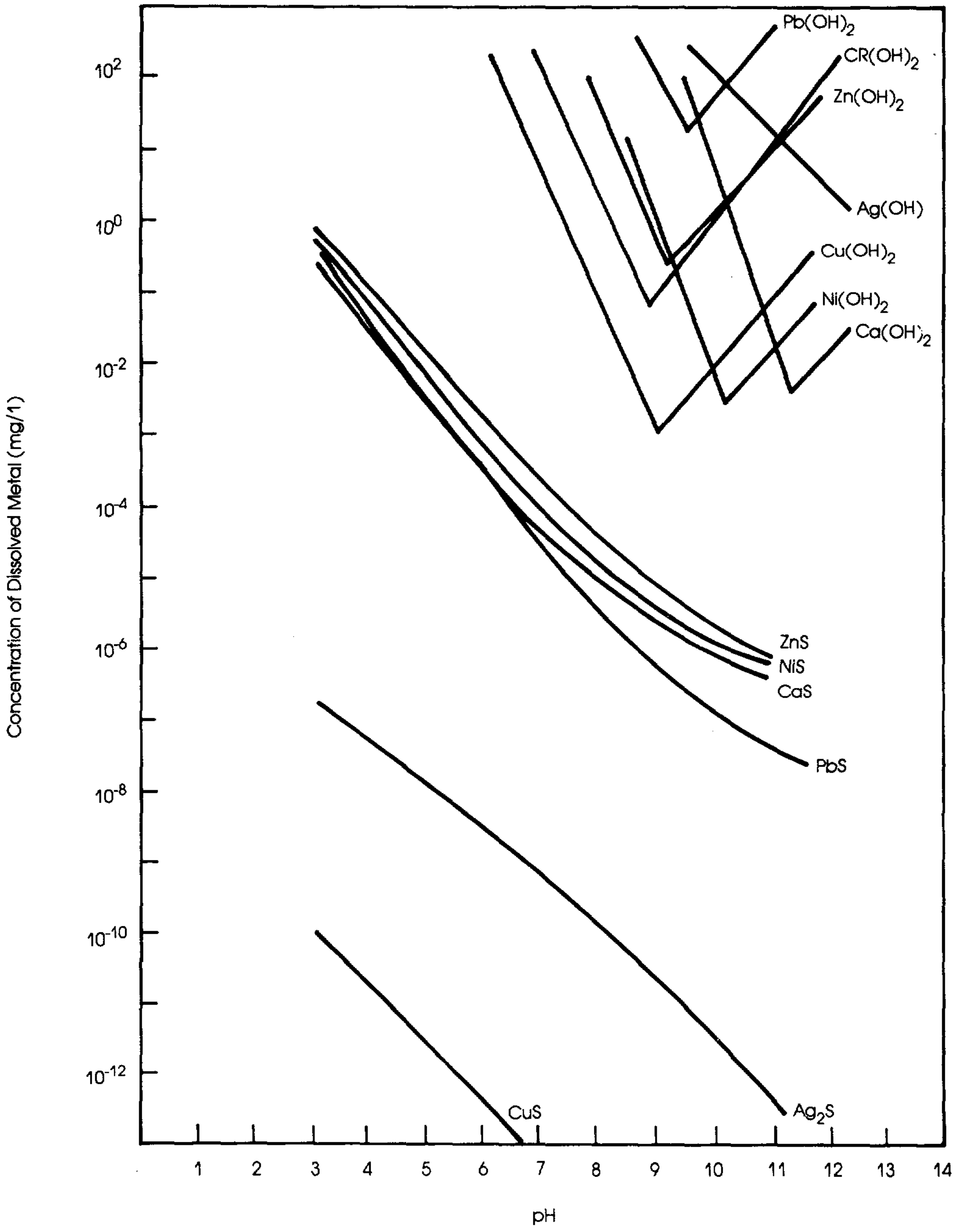
By slow and gentle mixing the precipitated particles aided by the flocculating agents collide, agglomerate, and grow into larger settleable particles.

The settleable particles produced by the flocculation step are settled, collected, and periodically removed.

Source: De Renzo, D.J. ed. 1978. Unit Operators for Treatment of Hazardous Industrial Wastes. Park Ridge, New Jersey: Noyes Data Corporation.

FIGURE 6-2

Solubility of Metal Hydroxides and Sulfides



Flocculation is used to describe the process by which small, unsettleable particles suspended in a liquid medium are made to agglomerate into larger, more settleable particles. The mechanisms by which flocculation occurs involve surface chemistry and particle change phenomena. In simple terms, these various phenomena can be grouped into two sequential mechanisms (Kiang and Metry 1982). The first is chemically induced destabilization of the requisite surface-related forces, which allows particles to stick together when they touch. The second is chemical bridging and physical enmeshment between the now nonrepelling particles, which allows for the formation of large particles. Flocculation involves: (1) addition of flocculating agent to the waste stream, (2) rapid mixing to disperse the flocculating agent and (3) slow and gently mixing to allow for contact between small particles.

Typically, chemicals used to cause flocculation include alum, lime, various iron salts (ferric chloride, ferrous sulfate) and organic flocculating agents, often referred to as "polyelectrolytes." These materials generally consist of long-chain, water-soluble polymers such as polyacrylamides. They are used either in conjunction with the inorganic flocculants, such as alum, or as the primary flocculating agent. A polyelectrolyte may be termed cationic, anionic or ampholytic depending upon the type of ionizable groups; or non-ionic if it contains no ionizable groups. The range of physical/chemical characteristics (e.g., density, viscosity, toxicity and molecular weight) of the several thousand available polymers is extremely broad.

The inorganic flocculants, such as alum, lime or iron salts, make use of precipitation reactions. Alum (hydrated aluminum sulfate) is typically added to aqueous waste streams as a solution. Upon mixing, the slightly higher pH of the water causes the alum to hydrolyze and form fluffy, gelatinous precipitates of aluminum hydroxide. These precipitates, partially due to their large surface area, enmesh small particles and thereby create larger particles. Lime and iron salts also have a tendency to form large fluffy precipitates or "floc" particles. Many precipitation reactions, such as the precipitation of metals from solution by the addition of sulfide ions, do not readily form floc particles, but rather precipitate as very fine and relatively stable colloidal particles. In such cases, flocculating agents such as alum and/or polyelectrolytes must be added to cause flocculation of the metal sulfide precipitates (Canter and Knox 1985).

Once suspended particles have been flocculated into larger particles, they usually can be removed from the liquid by sedimentation, provided that a sufficient density difference exists between the suspended matter and the liquid.

6.4.1 Applications/Limitations

Precipitation is applicable to the removal of most metals from wastewater including zinc, cadmium, chromium, copper, fluoride, lead, manganese, and mercury. Also, certain anionic species can be removed by precipitation, such as phosphate, sulfate and fluoride. Precipitation is useful for most aqueous hazardous waste streams. However, limitations may be imposed by certain physical or chemical characteristics. In some cases, organic compounds may form organometallic complexes with metals, which could inhibit precipitation. Cyanide and other ions in the wastewater may also complex with metals, making treatment by precipitation less efficient.

Flocculation is applicable to any aqueous waste stream where particles must be agglomerated into larger more settleable particles prior to sedimentation or other types of treatment. There is no concentration limit for precipitation or flocculation. Highly viscous waste streams will inhibit settling of solids.

There is a lower concentration limit beyond which precipitation and flocculation are not technically feasible. For precipitation it is the limit of the solubility product. Flocculation is impractical for very low turbidities. Although the upper limit for precipitation is 100% concentration of the waste, other techniques become economically attractive at high concentrations because of the potential for material recovery. For very high concentrations of suspended solids, plain sedimentation may be more appropriate.

In addition to being used to treat waste streams, precipitation can also be used as an in-situ process to treat aqueous wastes in surface impoundments. In an in-situ application, lime and flocculants are added directly to the lagoon, and mixing, flocculation and sedimentation are allowed to occur within the lagoon. In some cases, wind and pumping action can provide the energy for mixing. Industries which have successfully used hydroxide precipitation include (USEPA 1985):

- Inorganic Chemicals Manufacturing
- Metal Finishing
- Coil Coating
- Copper Forming
- Aluminium Forming
- Foundries
- Explosives Manufacturing
- Steam Electric Power Plants
- Photographic Equipment and Supplies
- Pharmaceutical Manufacturing
- Rubber Processing
- Porcelain Enameling
- Battery Manufacturing
- Iron and Steel
- Nonferrous Metals Manufacturing
- Coal Mining
- Electrical and Electronic Components
- Ore Mining and Dressing

Sulfide precipitation is being successfully practiced in the following industries:

- Photographic Equipment and Supplies
- Inorganic Chemicals Manufacturing
- Coal Mining
- Textile Mills
- Nonferrous Metals Manufacturing
- Ore Mining and Dressing

6.4.2 Design Considerations

Selection of the most suitable precipitate or flocculant and their optimum dosages is determined through laboratory jar test studies. In addition to determining the appropriate chemicals and optimum chemical dosages, other important parameters which need to be determined as part of the overall design include (Canter and Knox, 1985): (1) the most suitable chemical addition system, (2) optimum pH requirement, (3) rapid mix requirements, (4) sludge production and (5) sludge flocculation, settling and dewatering characteristics. Chemicals typically required and utilized are:

- Hydroxide precipitation: quicklime (CaO), hydrated lime ($Ca(OH)_2$), and liquid caustic soda ($NaOH$).
- Sulfide precipitation: sodium sulfate (Na_2SO_4), sodium sulfide (Na_2S), and ferrous sulfate ($FeSO_4$).

6.4.3 Design Criteria

Chemical precipitation treatment can either be a batch or continuous operation, with batch treatment being favored when waste flows are small. In batch treatment, the equipment usually consists of two tanks, each with a capacity to treat the total wastewater volume expected during the treatment period. These systems can be economically designed for flows up to 190,000 liters per day (50,000 gallons per day), (USEPA 1982a).

The batch treatment tanks serve the multiple functions of equalizing the flow, acting as a reactor, and acting as a settler. For a typical treatment operation, the wastewater is stirred, and a homogeneous sample is taken and analyzed to determine the chemical dosage requirements. The chemicals are then added, mixed, and stirred for about 10 minutes. After the reaction is complete, the solids are allowed to settle for a few hours. The clear liquid is then decanted and discharged. Settled sludge is retained to serve as a seed for crystal growth for the next batch, but must be drawn off periodically for disposal. For larger daily flows, a typical continuous flow treatment scheme consists of a chemical feed system, flash mixer, flocculator, settling unit, and in some cases a filtration system. In a continuous system, a control system is used to regulate the chemical feed to the process. For high-speed mixing, residence times of 10 to 30 seconds have been reported as satisfactory and a mix time of as much as two minutes has been recommended for two parallel units (Catalytic, Inc. 1981). For development of good floc characteristics, residence times of 15 to 30 minutes have been suggested (Catalytic, Inc. 1981).

The chemical dosage for precipitation can be determined on the basis of wastewater alkalinity and acidity, desired pH level to be maintained for the process, and stoichiometric requirements. An alternative is to estimate the chemical dosage on the basis of jar tests. These jar tests react the wastewater with a series of chemical doses, with the optimum dose selected on the basis of observed and measured removal effectiveness. A typical jar test is run as follows. Wastewater samples are placed in containers such as 1 or 2 liter beakers and rapid mix is started at 100 rpm. Selected dosages of coagulant covering the expected range of the optimum concentration are rapidly added to the containers and mixed for approximately 1 minute.

If a polymer is to be used as a coagulant aid, it is usually added to each jar at or just before the end of the rapid mix. The paddles are then slowed to 30 rpm and mixing continues for 20 minutes. The paddles are then stopped and the sampling apparatus previously described is placed in position. At settling times of 1, 3,5, 10 and possibly 20 minutes samples of supernatant are drawn for turbidity measurement. After the final turbidity sample is drawn, a larger volume of supernatant may be decanted for more complete analysis. Results are plotted as illustrated in Figure 6-3 for selection of the optimum coagulant dosage. Once an approximate optimum coagulant concentration has been determined, it may be desirable to repeat the jar test using that optimum with varying quantities of added alkalinity to give different pH values. Experience in coagulating a given wastewater provides the best guide as to methods for controlling the process.

Dosages determined on the basis of stoichiometric requirements may have to be increased by up to four times the stoichiometric amount as a result of chemical interactions, solubility variances, mixing effects and multivalent competition (Catalytic, Inc. 1981).

Lime [$(Ca(OH)_2$] requirements in neutralization and metal precipitation processes together with the process costs are given in Figure 6-4 (USEPA 1985a).

Storage of bagged lime should be in a dry place, and preferably elevated on pallets to avoid absorption of moisture. System capacities often make the use of bagged quicklime impractical. Maximum storage period is about 60 days.

Quicklime feeders are usually limited to the belt or loss-in weight gravimetric types because of the wide variation of the bulk density. Feed equipment should have an adjustable feed range of at least 20:1 to match the operating range of the associated slaker. The feeders should have an over-under feed rate alarm to immediately warn of operation beyond set limits of control. The feeder drive should be instrumented to be interrupted in the event of excessive temperature in the slaker compartment. Lime slakers for wastewater treatment should be of the continuous type, and the major components should include one or more slaking compartments, a dilution compartment, a grit separation compartment and a continuous grit remover. Commercial designs vary in regard to the combination of water to lime, slaking temperature, and slaking time, in obtaining the "milk of lime" suspensions.

Chemical mixing facilities should be designed to provide a thorough and complete dispersal of chemical throughout the wastewater being treated to insure uniform exposure to pollutants which are to be removed. The intensity and duration of mixing of coagulants with wastewater must be

FIGURE 6-3

Jar Test Results

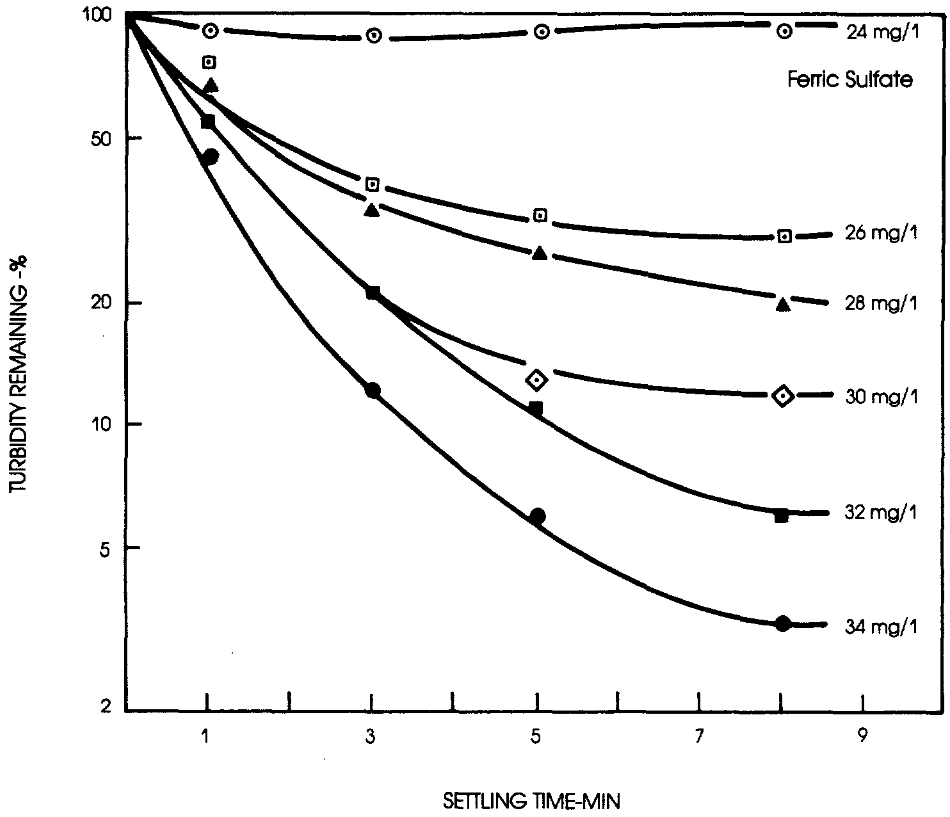
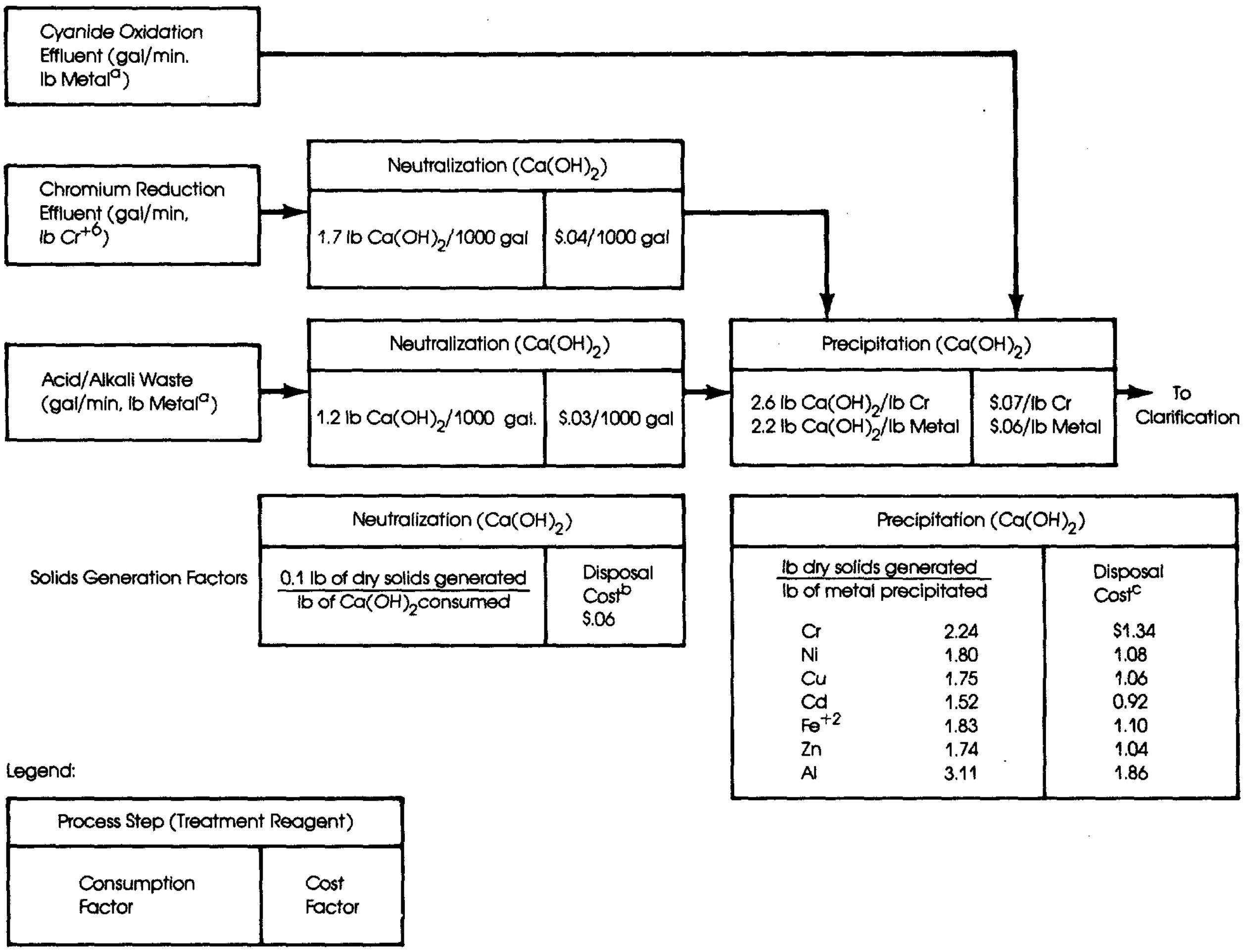


FIGURE 6-4

Consumption and Cost Factors Using Lime for Wastewater Treatment



Precipitation ($Ca(OH)_2$)

	lb dry solids generated / lb of metal precipitated	Disposal Cost[c]
Cr	2.24	$1.34
Ni	1.80	1.08
Cu	1.75	1.06
Cd	1.52	0.92
Fe^{+2}	1.83	1.10
Zn	1.74	1.04
Al	3.11	1.86

Basis:

Lime consumption of 20% in excess of neutralization requirements.
10% of lime feed is insoluble.
1 kg = 2.20 lb
1 lb= 0.22 gal

[a] *lb metals expressed as lb of metal ions.*
[b] per lb $Ca(OH)_2$ consumed (a 20% solids and $1.00/gal disposal fee.
[c] per lb metal precipitated (a 20% solids and $1.00/gal disposal fee.

controlled to avoid overmixing or undermixing. Several types of chemical mixing units have been used. These include high speed mixers, in-line blenders and pumps, and baffled mixing compartments or static in-line mixers (baffled piping sections). The high-speed mixer is a common choice for water treatment. Designs usually call for a 10 to 30 second detention time and approximately 300 m/s/m velocity gradient.

The proper measure of flocculation effectiveness is the performance of subsequent solids separation units in terms of both effluent quality and operating requirements. Effluent quality depends greatly on the reduction of residual primary size particles during flocculation, while operating requirements relate more to the floc volume applied to separation units. A maximum of 15 minutes detention for wastewater coagulation is recommended using paddle speeds of 2.5 to 5.1 mm/s.

6.4.4 Technology/Selection/Evaluation

Precipitation and flocculation are well established technologies and the operating parameters are well defined. The processes require only chemical pumps, metering devices, and mixing and settling tanks. The equipment is readily available and easy to operate. Precipitation and flocculation can be easily integrated into more complex treatment systems. The performance and reliability of precipitation and flocculation depends greatly on the following:

- Variability of the composition of the waste being treated.
- Maintenance of an alkaline pH throughout the precipitation reaction and subsequent settling.
- Addition of a sufficient excess of treatment ions to drive the precipitation reaction to completion.
- Addition of an adequate supply of sacrificial ions (such as iron or aluminum) to ensure precipitation and removal of specific target ions.
- Effective removal of precipitated solids.

Chemical addition must be determined using laboratory tests and must be adjusted with compositional changes of the waste being treated or poor performance will result.

Precipitation is non-selective in that compounds other than those targeted may be removed. Both precipitation and flocculation are nondestructive and generate a large volume of sludge which must be disposed.

Precipitation and flocculation pose minimal safety and health hazards to field workers. The entire system is operated at near ambient conditions, eliminating the danger of high pressure/high temperature operation with other systems. While the chemicals employed are often skin irritants, they can easily be handled in a safe manner.

6.4.5 Costs

Table 6-2 shows a breakdown of costs for the 2.5 l/s Sulfex heavy metal removal system illustrated in Figure 6-5.

The precipitator is sized to operate at a surface rate of 1.1 l/s/m2 and the filter at a surface rate of 2.2 l/s/m2. Chemical costs for the Sulfex process and a hydroxide precipitation process are shown in Table 6-3. These costs were estimated for treatment of an influent containing 4 mg/l Cu, Cd, Cr+3, Ni, and Zn at pH 6.0. Hydroxide precipitation has been shown to be cheaper than ISP. (Davis et al. 1987).

Figure 6-6 shows capital and operating costs for a flocculation system including chemical storage, chemical feeding and rapid mix. A polymer dosage of 1 mg/l at 0.25 percent solution is assumed. Construction costs also include piping and building to house the feeding equipment and bag storage. For a 2.6 m^3/min plant size and smaller manual feed procedures are used, and two systems of tanks and feeders are included. For a 26 m^3/min plant size the cost of feeders and mixing tanks are included as well as one day tank and 2 solution feeders. For a 263 m^3/min plant size costs for 4 feeders and mixing tanks, 2 holding tanks and 10 solution feeders are included. The rapid mix tank is concrete, equipped with stainless steel mixer and handrails. For a 0.3 m^3/min plant size no separate building is required. Manual operation of feeder, mix tank solution feeder and holding tank are also included.

6.5 Sedimentation

Sedimentation is a process that relies upon gravity to remove suspended solids from an aqueous waste stream. The fundamentals of a sedimentation process include (Kiang and Metry 1982):

- A basin or container of sufficient size to maintain the liquid to be treated in a relatively quiescent state for a specified period of time.
- A means of directing the liquid to be treated into the above basin in a manner conducive to settling.
- A means of physically removing the settled particles from the liquid (or liquid from the settled particles).

Sedimentation can be carried out as either a batch or continuous process in lined impoundments, conventional settling basins, clarifiers, and high rate gravity settlers. Modified and above ground swimming pools have been used many times for sedimentation in temporary, short-term treatment systems at hazardous waste sites. Figure 6-7 illustrates three different design configurations for sedimentation. In sedimentation ponds the liquid is merely decanted as the particles accumulate on the bottom of the pond. Backhoes, draglines, or siphons are used periodically to remove settled solids.

TABLE 6-2

1985 Capital Costs* for Sulfex Heavy Metal Precipitation System

Selling Price

Equipment	Price
1. Precipitator with clear well, centrifuge for dewatering, chemical feeds and agitators, and engineering drawings	$68,768
2. Filter with transfer pump and engineering drawings	7,623
3. Neutralization system including agitators, chemical feeds, pH controls, sump pump, and engineering drawings	86,126
Total selling price	$162,517
Installation Cost (estimated by outside contractor)	66,657
Total	$229,174

*Costs were updated to $1985 using the 1983 and 1985 ENR Construction Cost Index

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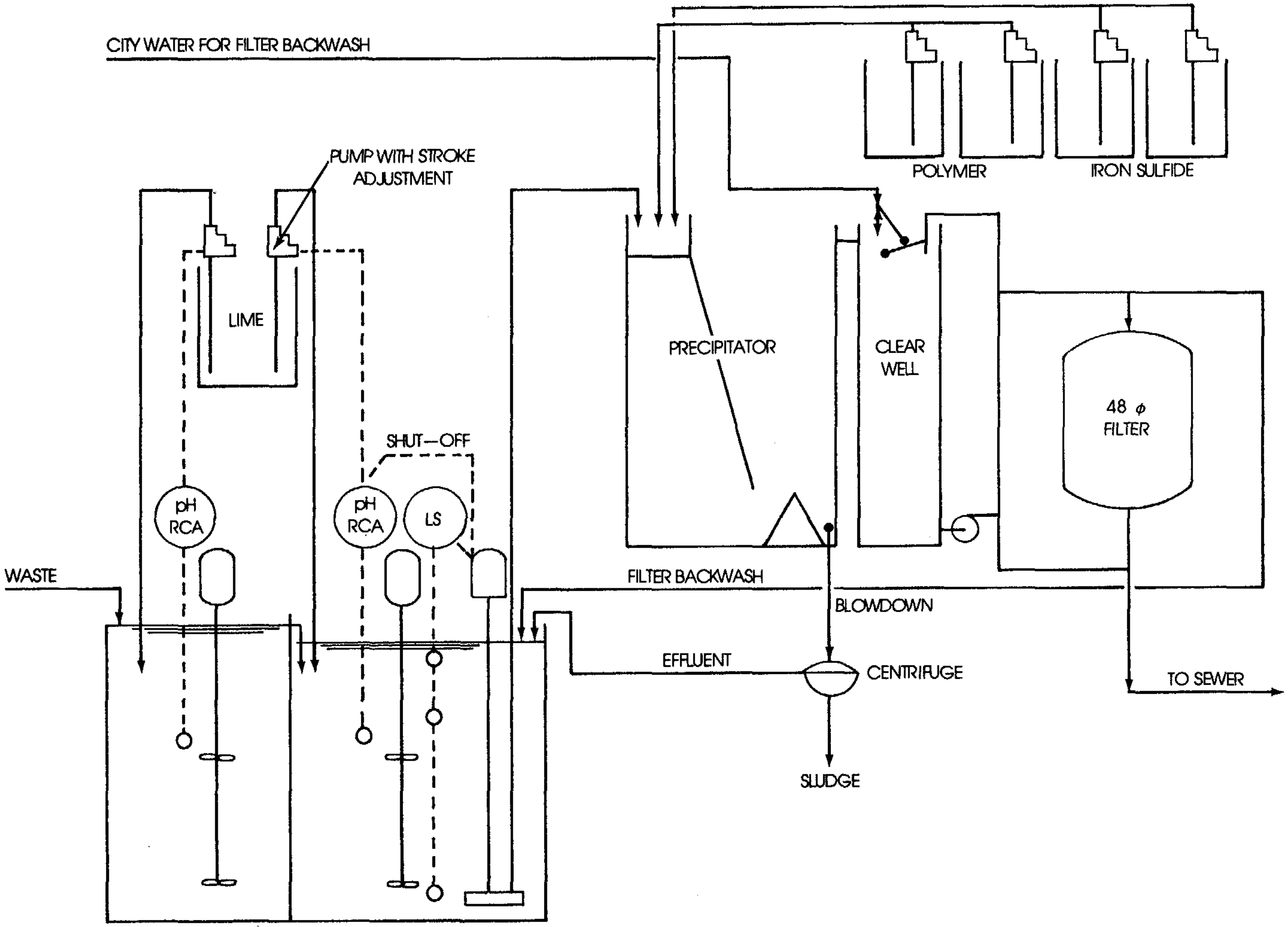
Source: Metal Finisher's Foundation. 1977. Treatment of metal finishing waste by sulfide precipitation. Prepared for United States Environmental Protection Agency's Industrial Environmental Research Laboratory in Cincinnati, Ohio. #PB-267-284. Springfield, Virginia: United States National Technical Information Service.

FIGURE 6-5

Treatment for Combined Removal of Cr^{+6}, Zn, Cu, Cd, Ni and Fe
(1 in = 25.4 mm)



Treatment for Combined Removal of Cr^{+6}, Zn, Cu, Cd, Ni and Fe 1 in = 25.4 mm.

Source: Metal Finisher's Foundation. 1977. Treatment of metal finishing waste by sulfide precipitation. Prepared for United States Environmental Protection Agency's Industrial Environmental Research Laboratory in Cincinnati, Ohio, # PB-267-284. Springfield, Virginia. United States National Technical Information Service.

TABLE 6-3

Comparison of Chemical Costs* of Hydroxide and Sulfide Precipitation Process

(1) Hydroxide Process

	Eff. Qyal. (mg/l)		Chemical	Dosage lb/1000 gal.		Cost cent/1000 gal.		
	pH 7.5	pH 10		pH 7.5	pH 10	cent/lb	pH 7.5	pH 10
Cu	0.1	<0.1	$CA(OH)_2$	0.33	0.92	3.16	1.05	2.99
Cd	3.8	<0.1	Polymer	0.03	0.03	105.4	3.16	4.22
Cr	<0.5	<0.1	H_2SO_4	-	0.61	8.8	-	5.45
Ni	2.3	<0.1						
Zn	1.3	<0.1				Total	4.21	12.66

(2) Sulfex Process

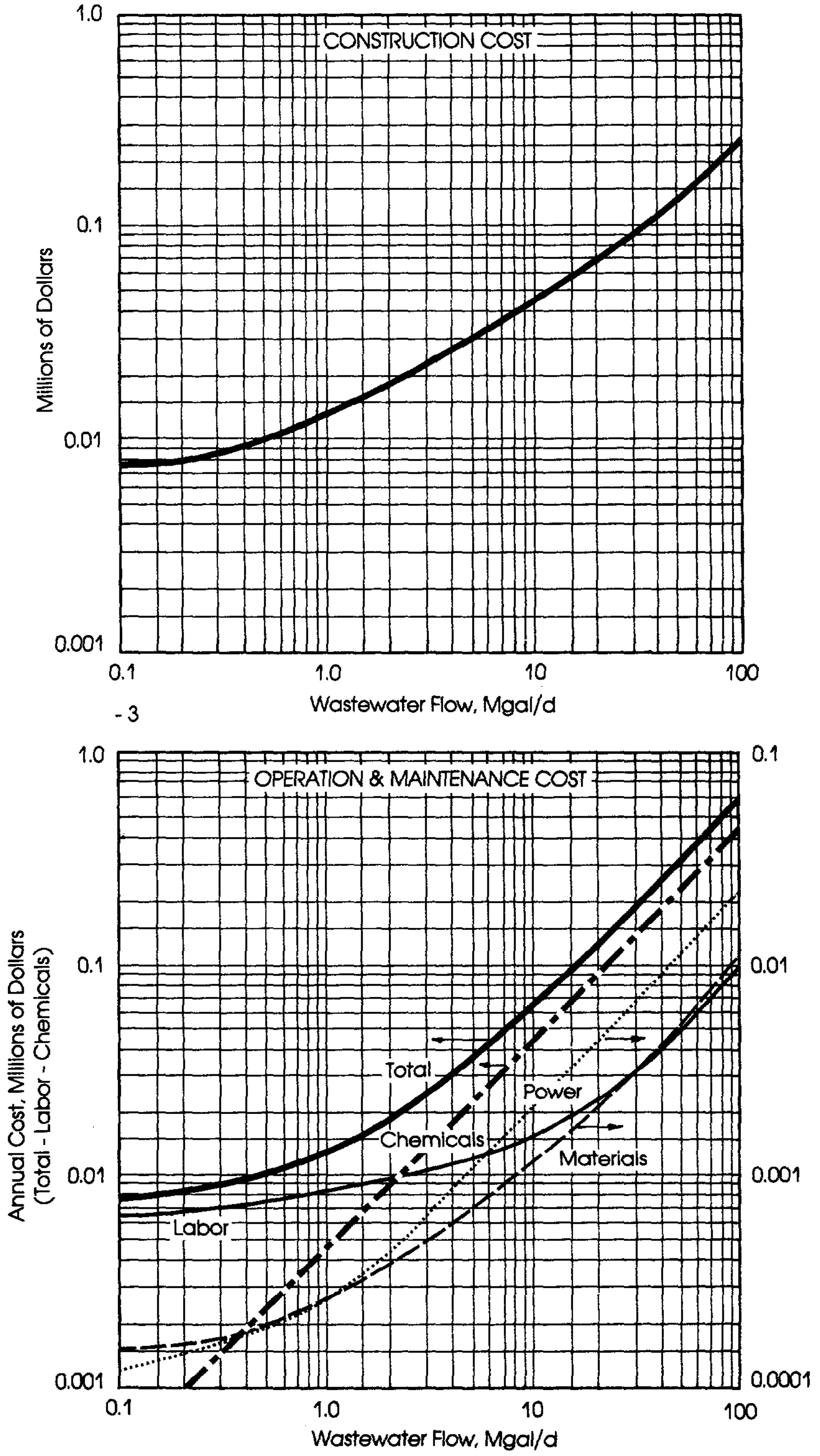
	Eff. Qyal. (mg/l)	Chemical	Dosage lb/1000 gal.	Cost cent/1000 gal.	
	pH 8.5		pH 8.5	.lb	pH 8.5
Cu	0.01	71% NaHs	0.09	19.77	1.76
Cd	0.1	$FeSO_4.7H_2O$	0.77	3.95	2.99
Cr	<0.05	Polymer	0.03	105.4	3.16
Ni	0.05	$Ca(OH)_2$	1.13	3.16	3.51
				Total	11.42

*Costs were updated to $1985 using the 1977 and 1985 ENR Construction Cost Index.
1 gal = 3.785 l
1 lb/1000 gal = 0.1198 g/m^3

Source: Metal Finishers's Foundation. 1977. Treatment of metal finishing waste by sulfide precipitation. Prepared for United States Environmental Protection Agency's Industrial Environmental Research Laboratory in Cincinnati, Ohio. #PB-267-284. Springfield, Virginia: United States National Technical Information Service.

FIGURE 6-6

Polymer Addition Costs*

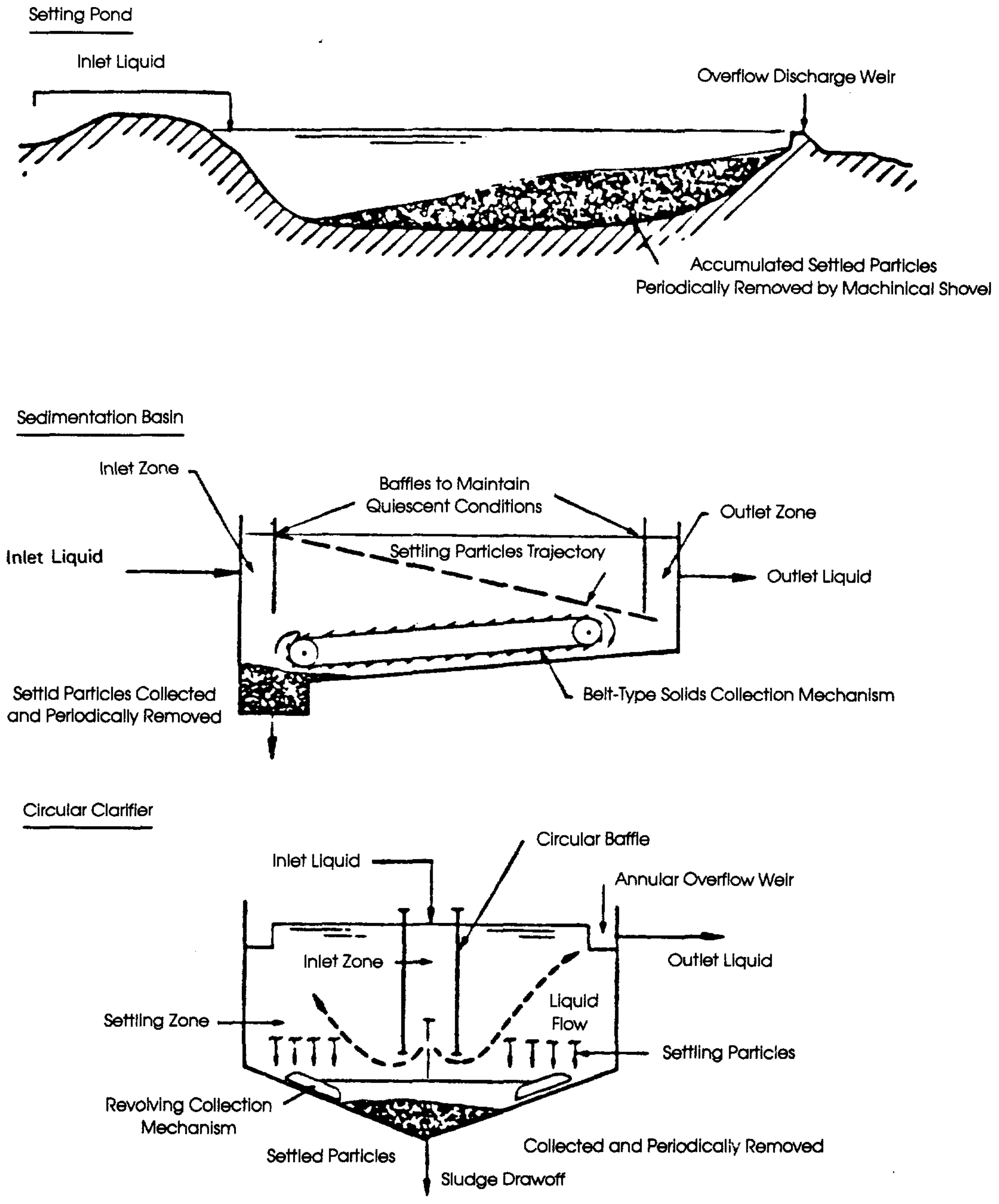


*Costs can be updated to $1985 using ENR Construction Cost Indices for 1982 and 1985 (multiply value shown on this figure by 1.303)

Source: USEPA 1982a. Treatability Manual. EPA-600/2-82-001. Stock No. 055-000-00215-1 (in 4 volumes). Washington, D.C.: U.S. Government Printing Office.

FIGURE 6-7

Representative Types of Sedimentation



Source: De Renzo, D. ed. 1978. Unit Operation for Treatment of Hazardous Wastes. Park Ridge, New Jersey. Noyes Data Corporation.

Sedimentation basins and clarifiers usually employ a built-in solids collection and removal device such as a sludge scraper and draw-off mechanism. Sedimentation basins are generally rectangular, usually employ a belt-like collection mechanism, and are mainly used for removal of truly settleable particles from liquid.

Clarifiers are usually circular and are used in applications involving precipitation and flocculation as well as sedimentation. Many clarifiers are equipped with separate zones for chemical mixing and precipitation, flocculation and sedimentation (Kiang and Metry 1982).

6.5.1 Applications/Limitations

Sedimentation is commonly applied to aqueous wastes with high suspended solid loadings. This may include surface run-off, collected leachate or landfill toe seepage, dredge slurries, and effluents from biological treatment and precipitation/flocculation. Sedimentation is also required as a pretreatment step for many chemical processes, including carbon adsorption, ion exchange, stripping, reverse osmosis and filtration.

This technology is applicable to the removal of suspended solids heavier than water. Suspended oil droplets or oil-soaked particles may not settle out and may have to be removed by some other means. Some sedimentation units are fitted with skimmers to remove oil and grease that float to the water surface. However, these would not be effective in removing emulsified oils.

6.5.2 Design Considerations

Sedimentation is frequently considered in terms of ideal settling. The ideal settling theory results in the following equation for surface loading or overflow rate.

$$V_O = Q/A$$

where: V_O = settling velocity
Q = flow through the basin
A = surface area of the basin

Sedimentation basin loadings (Q/A) are often expressed in terms of gallons per day per square foot. Thus under ideal settling conditions, sedimentation is independent of basin depth and detention time, and depends only on the flow rate, basin surface area and properties of the particle.

However, sedimentation does not perform according to ideal settling conditions, since settling is affected by such conditions as turbulence, and bottom scour. Therefore removal of particles is dependent on basin depth and detention time as well as flow rate surface area and particle size. The performance of a sedimentation basin on a suspension of discrete particles can be calculated, but it is not possible to calculate sedimentation basin performance for a suspension of flocculating particles, such as a wastewater, because settling velocities change continually. Laboratory settling tests, however, may be performed to predict sedimentation basin performance.

6.5.3 Design Criteria

Sedimentation tank performance is related to the surface hydraulic loading (the overflow rate). This is the inflow (or overflow) divided by the surface area of the basin and is commonly expressed in units of volume per day per unit area (l/day/sq. m. or gpd/sq. ft.). Typical hydraulic rates for numerous wastewater treatment cases are shown in Table 6-4.

The surface loading rates for sedimentation following chemical treatment vary considerably from one application to another. This wide variation emphasizes the importance of testing and pilot work in designing sedimentation facilities. In addition to the hydraulic loading rate, other design criteria include: solid loading rate, depth, detention time, weir loading rate, and length/width ratio for rectangular basins.

6.5.4 Technology Selection/Evaluation

Sedimentation provides a reliable means to remove suspended matter from a waste stream, provided the suspended matter is settleable and the treatment process including the use of flocculants/coagulants has been appropriately designed from laboratory settling tests. Clarifiers are capable of removing 90 to 99% of the suspended solids. Efficiencies normally, however, range from 50 to 70%. With some precipitation sludges the efficiency may be higher but seldom reaches 99%.

Sedimentation employs readily available equipment and is relatively easy to operate. The process is versatile in that it can be applied to almost any liquid waste stream containing suspended solids. It can also be easily integrated into a more complex treatment system as a pre- or post treatment method. Sedimentation is non-selective and nondestructive, resulting in a large volume of potentially contaminated sludge that may require further treatment and disposal.

6.5.5 Costs

The cost of a system which includes chemical clarification, rapid mixing, flocculation with alum and polymer and sedimentation is shown in Figure 6-8. The cost estimate assumed alum and polymer dosages of 200 mg/l and L mg/l respectively, and a flow rate to the clarifier of 22.6 $l/min/m^2$. The costs of chemical sludge processing and disposal are not included in the capital costs. O & M costs include costs of chemical purchase (Culp, Wesner and Culp 1978).

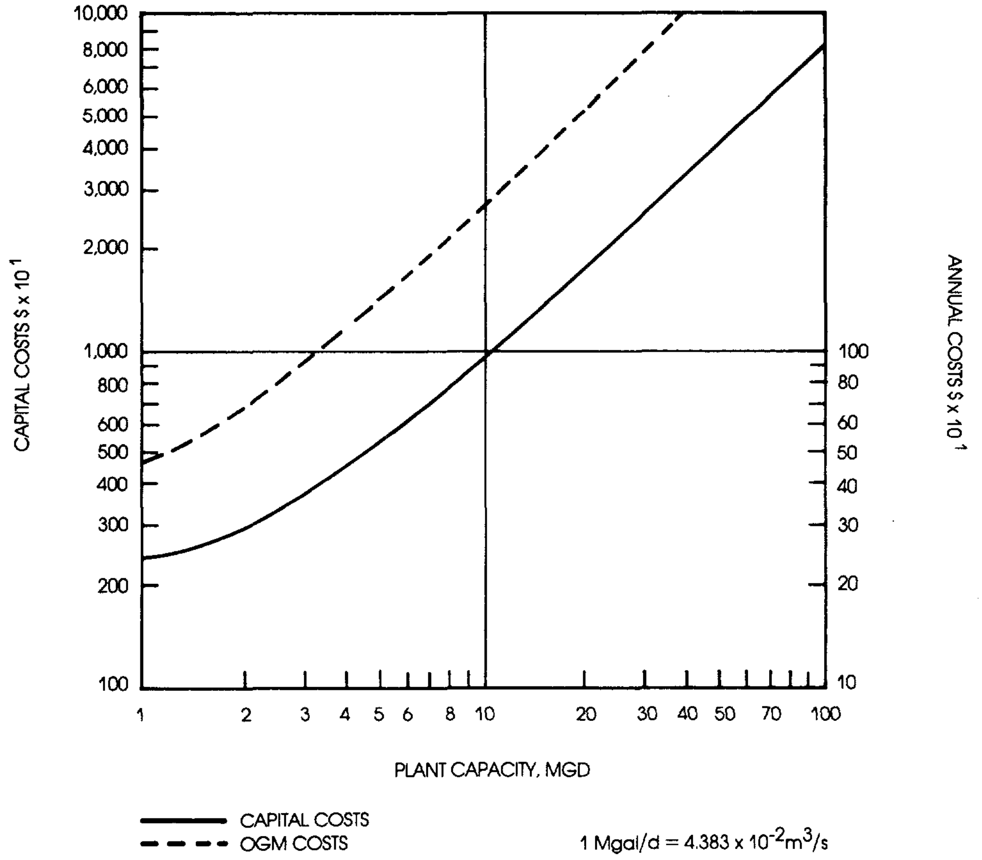
TABLE 6-4

Typical Hydraulic Loading

Type of treatment	Typical Overflow Rates Average	 Peak
	L/day/sq. m. (gpd/sq. ft.)	L/day/sq. m. (gpd/sq. ft.)
Primary settling followed by secondary treatment	33,000 - 49,000 (800 - 1,200)	81,000 - 120,000 (2,000 - 3,000)
Primary settling with waste activated sludge return	24,000 - 33,000 (600 - 800)	49,000 - 61,000 (1,200 - 1,500)
Settling following trickling filtration	16,000 - 24,000 (400 - 600)	41,000 - 49,000 (1,000 - 1,200)
Settling following air activated sludge (excluding extended aeration)	16,000 - 30,000 (400 - 800)	41,000 - 49,000 (1,000 - 1,200)
Settling following extended aeration	8,000 - 16,000 (200 - 400)	33,000 (800)
Settling following chemical treatment		
Alum	-	20,000 - 24,000 (500 - 600)
Iron	-	29,000 - 33,000 (700 - 800)
Lime	-	57,000 - 65,000 (1,400 - 1,600)

FIGURE 6-8

A Cost of chemical Clarification with Alum



1 Mgal/d = 4.383 x $10^{-2} m^3/s$

Source: Culp, R.L., G.M. Wesner, and G.L. Culp. 1978. Handbook of Advanced Wastewater Treatment. New York: Van Nostrand Reinhold Environmental Engineering Series. Copyright © 1978 by Van Nostrand Reinhold. All rights reserved.

6.6 Oily Wastes

Techniques commonly employed to remove oil include skimming, coalescing, emulsion breaking, flotation, centrifugation and removal by contractor hauling. Oil removal techniques may also afford additional removal of toxic organics.

6.6.1 Applications/Limitations

Oily wastes include process coolants and lubricants, wastes from cleaning operations directly following many other unit operations, wastes from painting processes and machinery lubricants. Oily wastes generally are of three types: free oils, emulsified or water soluble oils and greases. Oil separation is a widely used process in the following industries: iron and steel manufacturing, auto and other laundries, metal finishing, aluminum forming, battery manufacturing, gum and wood chemicals, oil palm and rubber processing, timber products processing, coil coating and soap and detergent manufacturing (USEPA 1982a).

Segregated oily wastes originate in the manufacturing areas, are collected in holding tanks and sumps and can have oil and grease concentrations as high as 400,000 mg/l. Combined oily wastes are those generated from washing or rinsing of oily parts, spills and leakages. They generally have lower oil and grease concentrations than segregated oily wastes by several orders of magnitude. Furthermore, oily wastes in combined wastewater streams, such as common metals wastewaters, require larger and thus more costly treatment systems for oil removal than do segregated oily waste-waters, because the combined wastewaters have significantly greater flow rates.

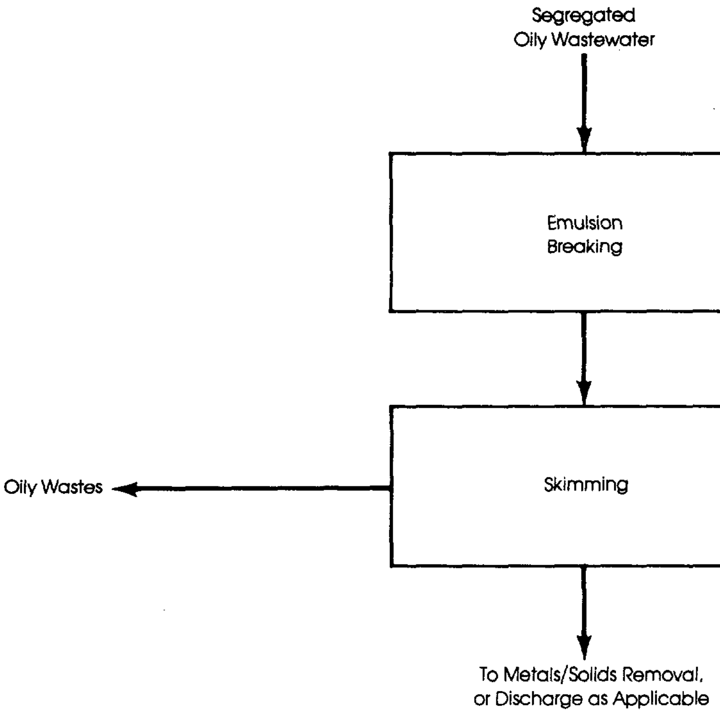
Treatment of segregated oily wastes consists of their separation from the associated water. This separation can require several different steps depending on the character of the oily wastes. If the oils are all of a free or floating variety, physical means such as decantation or the use of a gravity oil separator should be used to remove the oils. If the oily wastes are emulsified, techniques such as emulsion breaking or dissolved air flotation with the addition of chemicals are necessary to accomplish removal of the oils. Once the oil-water emulsion is broken, the oily waste is physically separated from the water by decantation or skimming. After the oil-water separation has been accomplished the water can either be sent to a precipitation/sedimentation unit for removal of metals or to some other treatment process.

6.6.2 Design Considerations - Emulsion Breaking

Treatment of segregated oily wastewater usually consists of emulsion breaking followed by skimming, as is illustrated in Figure 6-9. The emulsion breaking is effected by the addition of chemicals (such as alum or polymers) to accomplish coagulation and flocculation of the oily wastes. These floating oily wastes are then removed via skimming. Alternatives to the approach just described include ultrafiltration, dissolved air flotation, coalescing gravity separators, thermal emulsion

FIGURE 6-9

Treatment of Segregated Oily Wastes



breaking and the use of centrifugation. Concern over the price of oil has caused some manufacturing plants to tend toward the use of treatment techniques such as ultrafiltration, reverse osmosis, or centrifugation for the recovery and direct reuse of oils.

Emulsion breaking is a process by which emulsified oils are removed from oil/water mixtures. Emulsified oils are commonly used as coolants, lubricants, and antioxidants for many unit operations. Methods of emulsion breaking include a variety of chemical processes, thermal processes, and combinations of the two processes.

Chemical emulsion breaking can be accomplished either as a batch process or a continuous process. A typical system (with skimming incorporated) is illustrated in Figure 6-10. The mixture of emulsified oils and water is initially treated by the addition of chemicals to the wastewater. A means of agitation (either mechanical or by increasing the turbulence of the wastewater stream) is provided to ensure that the chemical added and the emulsified oils are adequately mixed to break the oil/water emulsion bond.

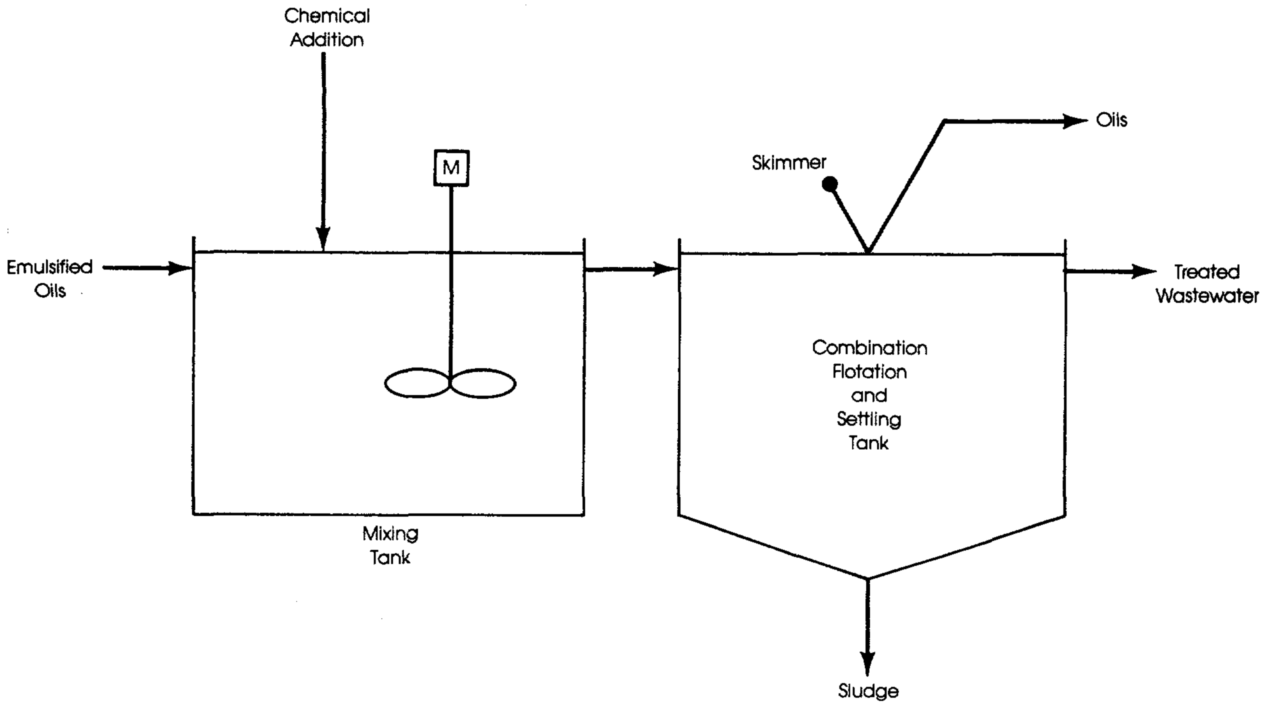
Finally the oily residue (commonly called scum) that results rises to the surface and is separated from the remaining wastewater by a skimming or decanting process. The skimming process can be accomplished by any of the many types of mechanical surface skimmers that are presently in use. Decanting methods include removal of the oily surface residue via a technique such as controlled tank overflow or by removal of the demulsified wastewater from the bottom of the tank. Decanting can be accomplished with a series of tap-off lines at various levels which allow the separated oils to be drawn off the top or the wastewater to be drawn off the bottom until oil appears in the wastewater line. With any of these arrangements, the oil is usually diverted to storage tanks for further processing or hauling by a licensed contractor.

Chemical emulsion breaking can be accomplished by a large variety of chemicals which include acids, salts, or polymers. These chemicals are sometimes used separately, but often are required in combination to break the various emulsions that are common in wastewater. Acids are used to lower the pH to 3 or 4 and can cleave the ion bond between the oil and water, but they can be very expensive unless acid rich wastewaters, such as pickling wastes, can be used. Acids are more commonly employed in oil recovery systems than in oily waste removal systems. Iron or aluminum salts such as ferrous sulfate, ferric chloride,or aluminum sulfate are more commonly used because they are less expensive. These salts combine with the wastewater to form acids which in turn lower the pH and break the oil/water bond. The salts have the additional benefit of aiding in agglomeration of the oil droplets. The use of these salts, however, produces more sludge because of the addition of iron or aluminum. Polymers, such as polyamines or polyacrylates and their copolymers, have been demonstrated to be effective emulsion breakers and generate less sludge than do metal salts.

After chemical addition, the mixture is agitated to ensure complete contact of the emulsified oils with the demulsifying agent. With the

FIGURE 6-10

Typical Emulsion Breaking/Skimming System



addition of the proper amount of chemical and thorough agitation, emulsions of 5% to 10% oil can be reduced to approximately 0.01% remaining emulsified oil. The third step in the emulsion breaking process is to allow sufficient time for the oil/water mixture to separate. Differences in specific gravity permits the oil to rise to the surface in approximately two hours. Heat can be added to decrease the separation time. After separation, the normal procedure involves skimming or decanting the oil from the tank.

6.6.3 Technology/Selection/Evaluation

Emulsion breaking technology can be applied to the treatment of emulsified oil/water mixtures such as occur in the metal finishing industry wherever it is necessary to separate oils, fats, soaps, etc. from wastewaters. Certain machining coolant emulsion cannot be chemically or thermally broken and must be treated by ultrafiltration.

The main advantage of the chemical emulsion breaking process is the high percentage of oil removal possible. For proper and economical application of this process, the oily wastes (oil/water mixture) should be segregated from other wastewaters either by storage in a holding tank prior to treatment or by being fed directly into the oily waste removal system from major collection points. Further, if a significant quantity of free oils are present, it is economically advantageous to precede the emulsion breaking with a gravity separator. Chemical and energy costs can be high, especially if heat is used to accelerate the process.

Chemical emulsion breaking can be highly reliable if adequate analysis is performed prior to the selection of chemicals, and proper operator training is provided to ensure that established procedures are followed. For chemical emulsion breaking, routine maintenance is required on pumps, motors, and valves as well as periodic cleaning of the treatment tank to remove any sediment which may accumulate in the tank. The use of acid or acidic conditions will require a lined tank, and the lining should be checked periodically. Emulsion breaking generates sludge which requires proper disposal.

6.6.4 Performance

The performance attainable by a chemical emulsion breaking process is dependent on addition of the proper amount of de-emulsifying agent, good mixing agitation and sufficient retention time for complete emulsion breaking. Since there are several types of emulsified oils, a detailed study should be conducted to determine the most effective treatment techniques and chemicals for a particular application.

6.6.5 Design Considerations - Skimming

Skimming is used to remove floating wastes and normally takes place in a tank designed to allow the debris (with a specific gravity less than water) to rise and remain on the surface. Skimming devices are therefore suited to the removal of oily wastes from raw waste streams after any necessary demulsification. Common skimming mechanisms include the rotating drum type, which picks up oil from the surface of the water as it rotates. A knife edge scrapes oil from the drum and collects it in a through for disposal or reuse. The water portion is then allowed to flow under the rotating drum. Occasionally, an underflow baffle is installed after the drum; this has the advantage of retaining any floating oil which escapes the drum skimmer. The belt type skimmer is pulled vertically through the water, collecting oil from surface which is again scraped off and collected in a tank. System design and operational controls are important in drum and belt type skimmers in order to ensure uniform flow through the system and avoid oil bypassing the skimmer mechanism.

Gravity separators, such as the American Petroleum Institute's (API) Type, utilize overflow and underflow baffles to skim a floating oil layer from the surface of the wastewater. An overflow-underflow baffle allows a small amount of wastewater (the oil portion) to flow over into a trough for disposition or reuse while the majority of the water flows underneath the baffle. This is followed by an overflow baffle, which is set at a height relative to the first baffle such that only the oil bearing portion will flow over the first baffle during normal plant operation. An inlet diffusion device, such as a vertical slit baffle, aids in creating a uniform flow through the system and increasing oil removal efficiency (American Petroleum Institute [API] 1969).

The modern trend is to keep separator design small and simple to reduce costs. Typical design configurations include: horizontal cylindrical decanters, vertical cylindrical decanters, and cone bottomed settlers in addition to the classical API type. Baffles help to provide additional surface area, which promotes oil droplet coalescence. The cone-bottomed design is particularly useful if heavy solids are to be settled out of the waste while oil separation is proceeding (USEPA 1985b).

6.6.6 Design Criteria

In designing a separator such as the API one, which is the most common gravity oil separator; it is helpful to have data on the separator influent temperature variance, the specific gravity of water in the separator, the specific gravity of oil in the separator and the flow rate into the separator. The basic design is a long rectangular basin that provides enough detention time for most of the oil to float to the surface and be removed. It has been found that two or more parallel channels are required to meet the following practical limitations: (a) separator channels equipped with flight scrapers normally are limited to a width of 6.1 m, (b) the disturbing effect of operating flight scrapers in shallow water has led to the recommendation that the minimum depth of water shall be 0.9 m and (c) it is recommended that a minimum of two parallel channels be provided so that one is available for use when it becomes necessary for the other to be removed from service for repair or cleaning (API 1969).

Other criteria to be followed include: (a) horizontal velocity is a maximum of 15 mm/s, (b) depth is a 0.9 m minimum to 2 m maximum, (c) depth to width ratio is 0.3 minimum to 0.5 maximum and d) width is 2 m minimum to 6.1 m maximum (API 1969). Table 6-5 below gives the minimum surface areas for API rectangular separators.

6.6.7 Technical Selection/Evaluation

Oil skimming is used in industries like the metal finishing one to remove oily wastes from many different process wastewater streams. Skimming is applicable to any waste stream containing pollutants which float to the surface. It is used in conjunction with emulsion breaking, dissolved air flotation, clarifiers, and other sedimentation type devices.

API or other gravity-type separators are more suitable for use where the amount of surface oil flowing through the system is consistently significant as with free oils. Drum, belt, or rotary type skimmers are applicable to waste streams which carry smaller amounts of floating oily waste and where surges of floating oil are not a problem. The use of a gravity separator system preceding emulsion breaking is a very effective method of removing free oil constituents from oily waste streams.

Skimming as a pretreatment is effective in removing naturally floating waste materials, such as free oils, and improves the performance of subsequent downstream treatments. Many pollutants, particularly dispersed or emulsified oil, will not float "naturally" but require additional treatments. Therefore, skimming alone will not remove all the pollutants capable of being removed by more sophisticated technologies.

Because of its simplicity, skimming is a very reliable technique, however, a mechanical skimming mechanism requires periodic lubrication, adjustment, and replacement of worn parts. The collected layer of debris (scum) must be disposed of in an approved manner. Because relatively large quantities of water are present in the collected wastes, direct combustion or incineration is not always possible (USEPA 1983).

6.6.8 Performance

The performance attainable by skimming is dependent on proper mechanical operation of the skimmer and on the separation rate of the oil/water mixture which is affected by such factors as the size and specific gravity of the oil globules. Examples of performance of skimmer systems for oil and grease are presented in Table 6-6.

6.6.9 Costs

Oily waste separation with chemical emulsion breaking can be performed in either a continuous or a batch mode. Each operating mode, the equipment associated with each mode, and the design and operating assumptions incorporated are discussed in the following paragraphs.

Investment costs associated with continuous and batch operating modes for chemical emulsion breaking are shown in Figure 6-11 as a function of the waste stream's flow rate. For the continuous operating mode, the cost

TABLE 6-5

Minimum Surface Areas for Rectangular Separators (Fresh Water)

Influent temperature:	50 F			70 F			105 F		
Influent oil gravity, deg API	19.70	26.00	31.00	19.70	26.00	31.00	19.70	26.00	31.00
Specific gravity	0.94	0.90	0.87	0.94	0.90	0.87	0.94	0.90	0.87
Oil globule rise rate, v_t, fmp	0.11	0.18	0.23	0.16	0.25	0.32	0.28	0.41	0.50
Horizontal flow velocity, V_H, fpm*	1.70	2.70	3.00	2.40	3.00	3.00	3.00	3.00	3.00
V_H/V_t, fpm	15.00	15.00	13.00	15.00	11.80	9.40	10.50	7.40	6.00
Turbulence--short-circuiting factor, F	1.64	1.64	1.59	1.64	1.56	1.50	1.52	1.44	1.38
Overflow rate: gpm per sq ft area	0.50	0.80	1.07	0.73	1.21	1.60	1.39	2.15	2.75
gpd per sq ft area	724	1,154	1,540	1,056	1,744	2,314	2,000	3,100	3,960
Area per 1,000 gpm flow, sq ft	1,985	1,246	934	1,362	824	624	720	465	364

1 gal/d/ft^2 = 4.715 x 10^{-7}m/s

1 fpm = 5.1 mm/s

^{o}C = 1.8(F - 32)

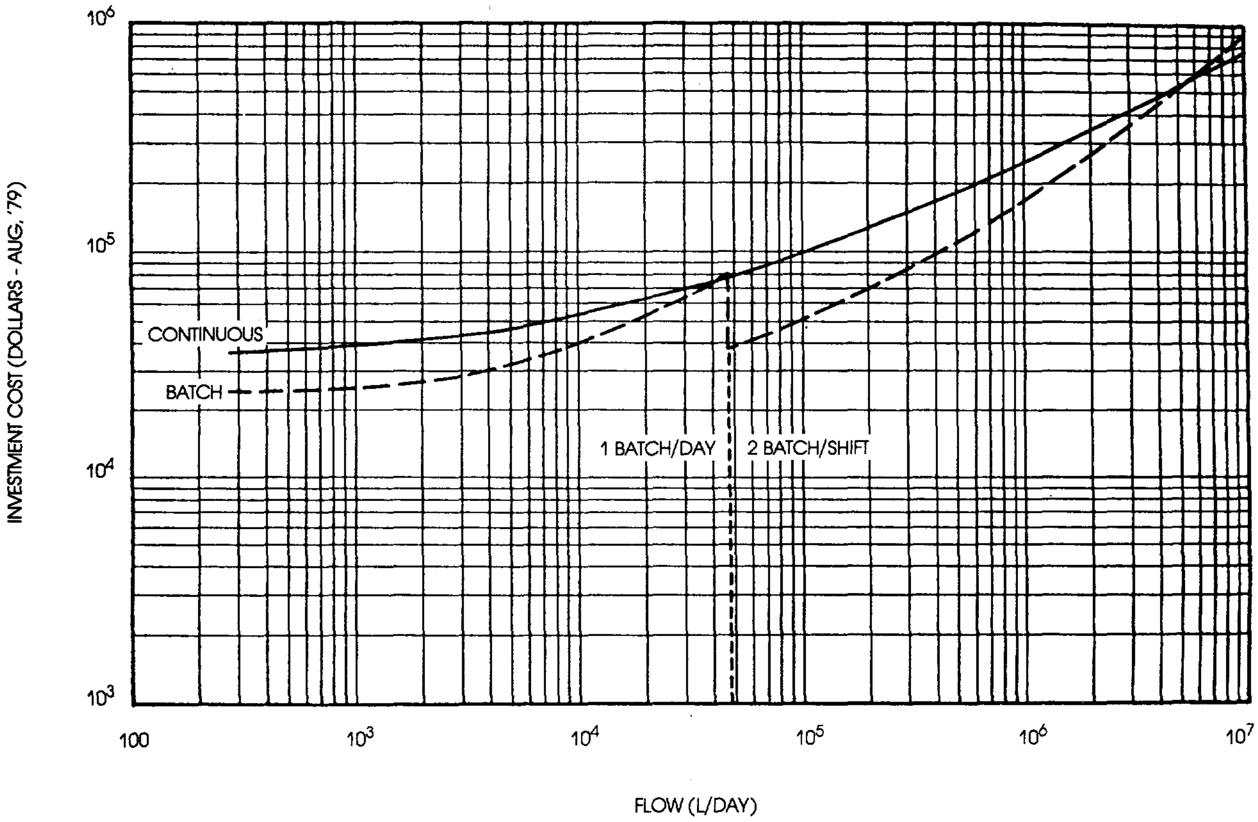
TABLE 6-6

Skimming Performance Data for Oil and Grease (mg/l)

Plant ID	Oil and Grease Influent (mg/l)	Oil and Grease Influent (mg/l)	Type of Skimmer
6058-14-0	395,538	13.3	API
6058-15-5	53,800	16	API
6058-14-0	19.4	8.3	Belt
11477	61	14	Belt

FIGURE 6-11

Emulsion Breaking Investment Costs



curve is based upon the purchase and installation of the following equipment:

- 2 -946 liter alum dilution tanks
- 2 -Variable speed alum feed pumps
- 2 -Polymer dilution tank mixers
- 2 -Variable speed polymer feed pumps
- 1 -Steel gravity separation tank with liner, weirs, and baffles (1 hour retention)
- 1 -Skimmed oil transfer pump
- 1 -Waste oil storage tank (20 day retention)
- 2 -Alum dilution tank mixers
- 2 -946 liter polymer dilution tanks
- 1 -Steel mixing tank with liner for chemical addition (15 minute detention)
- 1 -Separation tank surface oil skimming mechanism
- 1 -Separation tank effluent transfer pump

For the chemical emulsion breaking unit operated in the batch mode, the cost curve is based upon the following purchases:

- 1 -946 liter alum dilution tank
- 1 -Alum feed pump with pump stand
- 1 -Polymer dilution tank mixer
- 2 -Steel gravity separation tanks with liners
- 2 -Tank mixers
- 1 -Alum dilution tank mixer
- 1 -946 liter polymer dilution tank
- 1 -Polymer feed pump with pump stand
- 1 -Separation tank effluent transfer pump

The chemical emulsion breaking systems have been sized for a 20 % excess capacity factor. Selection of the operating mode is based on a least cost basis.

Operation and maintenance costs associated with the chemical emulsion breaking unit consist of labor and material expenses. Annual labor expenses for both continuous and batch operating modes for the chemical emulsion breaking unit are shown in Figure 6-12. For both operating modes, labor requirements are based on estimated manhours required for diluting and mixing the polymer and alum solutions and operating the unit. General operation labor has been estimated at 0.75 manhours per 8 hour shift or batch processed. General maintenance of the entire system has been estimated at 2 manhours per week.

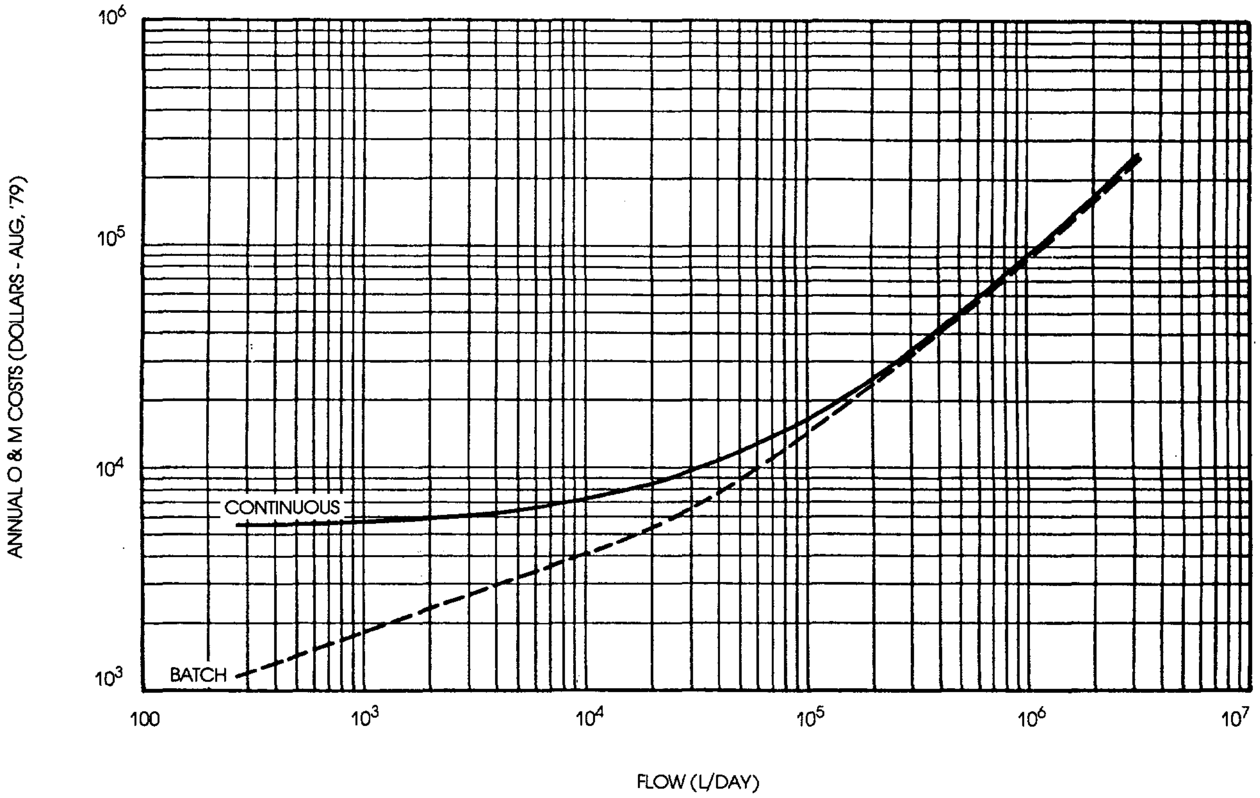
Labor expenses have been calculated using a labor rate of $6.71 per manhour plus an additional 15% to cover indirect labor costs. Material costs are associated with the alum and polymer chemical addition requirements. Polymer is added to the wastewater until a concentration of 150 mg/l is attained. Alum is added to the wastewater until a concentration of 25 mg/l is attained. Chemical costs have been based upon the following unit prices:

$0.38 per kg of alum
$1.55 per kg of polymer

The assumption has been made that the unit operates 24 hours per day, 5 days per week, 52 weeks per year (USEPA 1983).

FIGURE 6-12

Annual O & M Costs vs. Flow Rate for Chemical Emulsion Breaking



6.6.10 Case History

Ponamarev and Zakharina. 1975 described a USSR installation for waste emulsion regeneration, and this is shown in Figure 6-13. According to this scheme oily waste is fed to a gravity separator (1), where suspended matter and oil are separated. Floated oil is fed to an oil collector (5), and suspended matter collected in the conic part is placed in a sludge-holding tank (4). Filters are used to remove emulsion from fine suspended solids (2). After filtration the emulsion is pumped over to a flotation process (3) to which if necessary and if there is odor of hydrogen sulphide, 15 mg/l of KMnO4, 1 g/l of sodium nitrate and 0.2 g/l of hexachlorophenol* in an alkali solution are added. In addition to renew emulsion properties cutting emulsion and water are added to it. After all components are added to the emulsion, it is mixed with compressed air, and the non- emulsified oil is removed by flotation. Oil is isolated in the flotation- reaction tank is pumped over to the oil collector (5) and then is fed to regeneration. Sludge from the sludge-holding tank is dewatered by filter presses (6).

If waste emulsions can not be regenerated, they must be broken. Due to the high stability of emulsions, they must be treated by such methods as coagulation, flotation and/or centrifugation.

In the coagulation process ferric, aluminum, calcium and/or magnesium salts, which at hydrolysis give positively charged colloids of hydroxides, are used as reagents. Upon their interaction with the negatively charged micelles of the emulsion mutual coagulation takes place with the formation of suspensions, containing particles of metal hydroxides, sorbing at their surfaces and drops of oil and of emulsifying agents.

Application of ferric sulfate with the addition of lime, ferric chloride with sulfuric acid and aluminum sulfate with lime or sodium hydroxide have appeared most effective for emulsion breaking. The optimum dosage of a coagulant varies within the range of 3-7 g/l depending on the emulsified oil content.

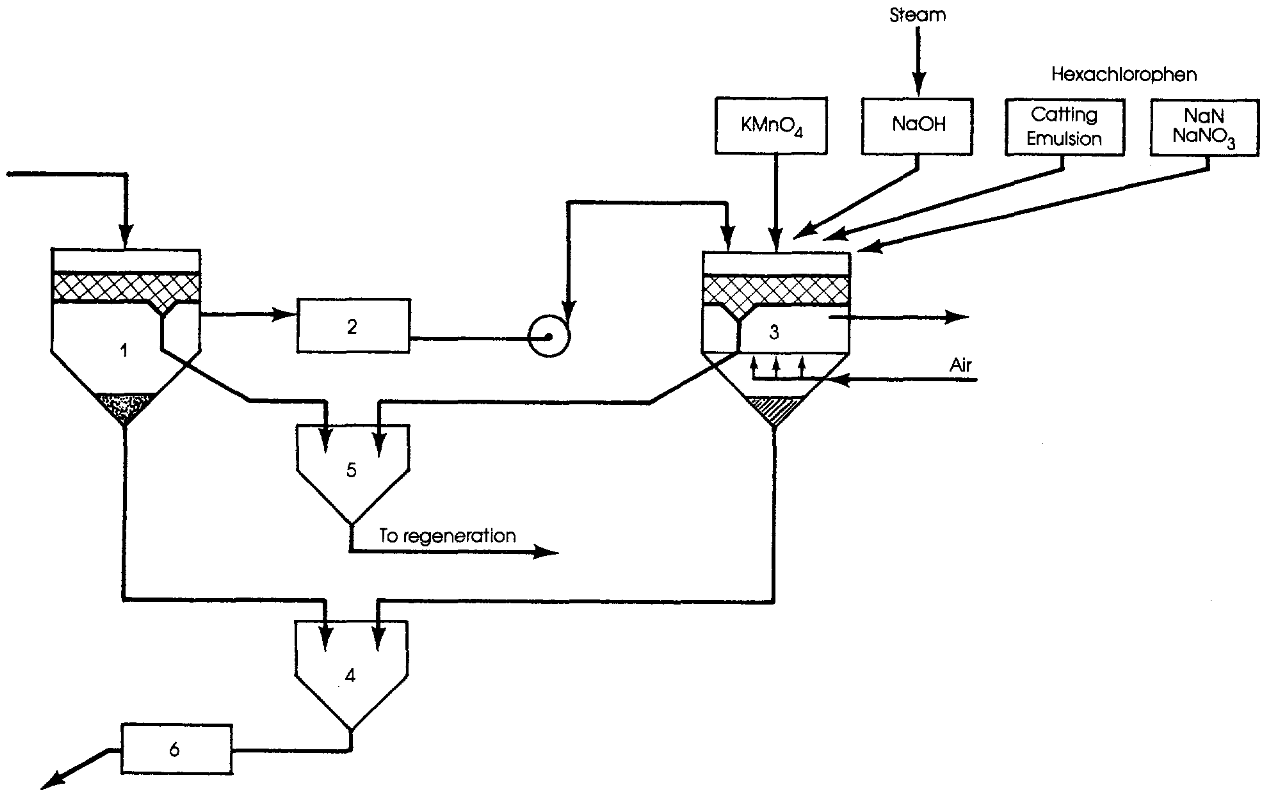
An analyses indicated the volume of the sludge separated averages 17-20 % of the total effluent. Depending on the type oil and associated concentration the sludge can take different positions in a settling tank. That is it may either settle to the bottom of the tank or float to its surface. In a number of cases the sludge formed is located in the form of flocs along the whole volume of the settling tank. That is why the determination of the method for sludge separation from the waste emulsion is one of the important problems.

The method of flotation was investigated on a pilot plant scale, and the investigation showed that saturation of the emulsion with the air should be conducted at a pressure of 3 atm; the saturation duration should be 5 minutes and the flotation period should be about 15-20 minutes. Reagents

*Note: Hexachloraphenol should be added sparingly because of its solubility.

FIGURE 6-13

Scheme of Emulsion Regeneration



should be added to the emulsion immediately before the saturator. Optimum doses of coagulating agents at an emulsion oil content of 30 g/l are: aluminum sulfate - 6 g/l; ferrous sulfate - 7 g/ l; ferrous chloride - 4 g/l; with addition of sulfuric acid - 6 g/l. The volume of the foam formed is equal to 20-25 % of the effluent being treated volume.

The content of ether soluble matter in the treated waste is equal to 0.15 - 0.3 g/l. The analysis showed that mineral oil concentration was equal to 50-70 mg/l, and the residual ether soluble matter are emulsifying agents.

With the aim towards decreasing foam volume, tests were carried out with flocculating agents such as the "komet" flocculating agent and polyacrylamide. It was established that with these flocculating agents at doses of 10-100 mg/l the foam volume could be decreased by 30-50 %.

On the basis of the investigations made, a fullscale installation for waste oil emulsion degradation was designed and built at several industrial plants (See Figure 6-14). According to this scheme waste emulsion is directed from the shop to the settling tank (1) where sulfuric acid is added. Mixing with the acid is done by means of compressed air. By this process pH of the emulsion is reduced to 6-7. After the waste emulsion is retained in a settling tank for an hour and the oil is allowed to rise to the surface, the floated oil is fed to the oil collector (2), and the sludge is introduced to the sludge-holding tank (3). Emulsion, which was not broken in the settling tank is fed to the flotation unit, where the second stage of its treatment occurs. Coagulating agent is mixed with the effluent before the saturator (4). The saturator should be designed in such a way, that all discharge of the emulsive effluent passes through it. Emulsion saturated with the air is introduced to the flotation compartment (5). The condensed foam is skimmed off the surface of the flotation compartment and is fed to the foam collector (6). Clarified liquid is neutralized by means of NaOH, water or lime to pH = 7, and then is discharged to the sewerage system where it is mixed with the wastewaters from the enterprises and goes to the full scale treatment facilities for processing. Foam from the flocculation unit is either fed directly to an incinerator or is mixed with sludge and they are treated together.

6.7 Chemical Oxidation

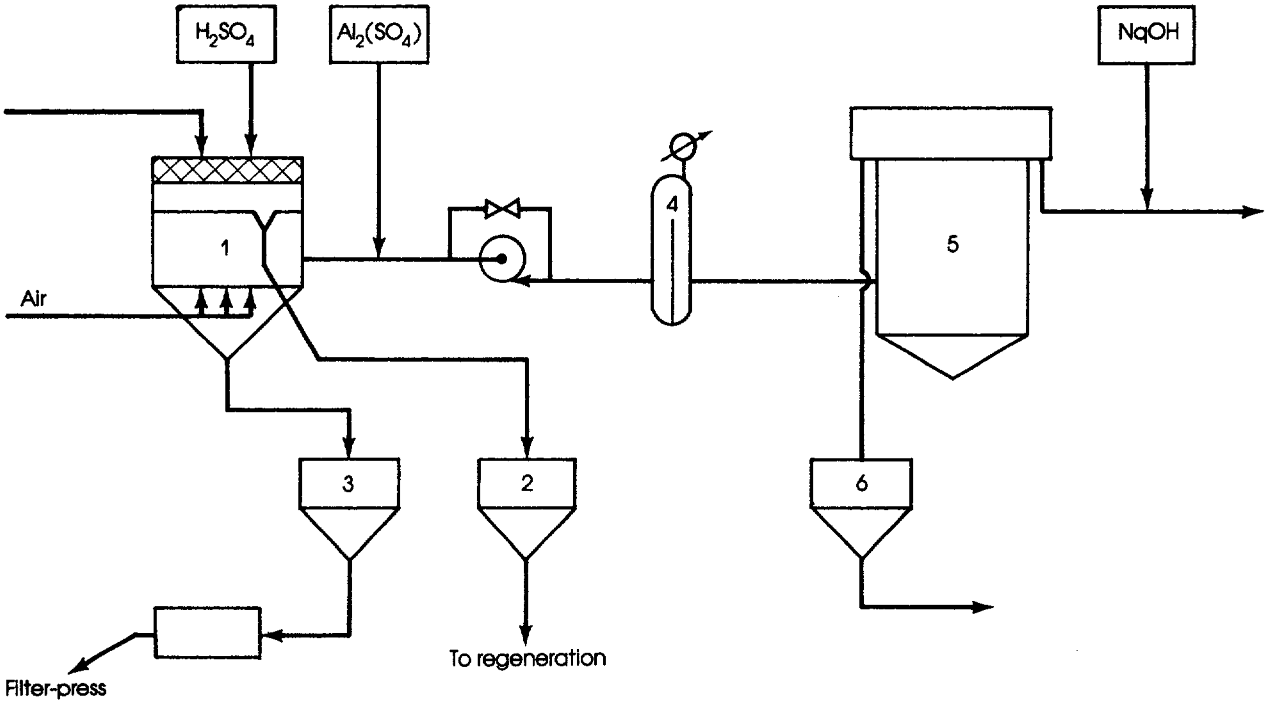
6.7.1 Description

Oxidation is a chemical reaction process in which one or more electrons are transferred from the chemical being oxidized to the chemical initiating the transfer (the oxidizing agent). In a typical oxidation reaction shown below, the oxidation state of the cyanide ion is raised from -1 to +1; the cyanide is oxidized as it combines with the atom of oxygen to form cyanate. The oxidation state of the permanganate decreases from -1 to -2 (permanganate is reduced to manganate).

$$2\ MnO_4^- + CN^- + 2OH^- = 2\ MnO_4^{-2} + CNO^- + H_2O$$

FIGURE 6-14

Scheme of Emulsion Breaking by Flotation



permanganate + cyanide + hydroxyl ion = manganate + cyanate + water

The change in oxidation state implies that an electron was transferred from the cyanide ion to the permanganate. The increase in the positive valence or decrease in the negative valence with oxidation takes place simultaneously with reduction in chemically equivalent ratios. Some oxidation reactions proceed readily to carbon dioxide (CO_2). In other cases, the oxidation is not carried as far, perhaps because of the dosage of the oxidant, the pH of the reaction medium, the oxidation potential of the oxidant, or the formation of stable intermediates. The primary function performed by oxidation in the treatment of hazardous wastes is essentially detoxification. For instance, oxidants are used to convert cyanide to the less toxic cyanate or completely to carbon dioxide and nitrogen. A secondary function is to assure complete precipitation, as in the oxidation of Fe|++| to Fe|+++| and similar reactions, where the more oxidized material has a lower solubility under the precipitation reaction conditions.

6.7.2 Representative Types and Modifications

There are many oxidizing agents; however, their application in wastewater treatment requires that a specific determination be made of their effectiveness in removing the pollutants, and in particular, to determine if the reaction products are innocuous. The oxidizing agents commonly used in waste treatment are described below.

(i) Oxidation by Chlorine

Chlorine in elemental or hypochlorite salt form is a strong oxidizing agent in aqueous solution and is used in industrial waste treatment facilities primarily to oxidize cyanide. Chlorine and hypochlorites can also be used to oxidize phenol based chemicals, but their use is limited because of the formation of toxic chlorophenols if the process is not properly controlled.

The cyanide alkaline chlorination process uses chlorine and caustic to oxidize cyanides to cyanates and ultimately to carbon dioxide and nitrogen. The oxidation reaction between chlorine and cyanide is believed to proceed in two steps as follows:

(1) $CN^- + Cl_2 = CNCl + Cl^-$
cyanide ion + chlorine = cyanate choride + chloride ion

(2) $CNCl + 2OH^- = CNO^- + Cl^- + 2H_2O$
cyanogen chloride + hydroxyl ion = cyanate ion + chloride ion + water

The formation of cyanogen chloride (CNCl) is essentially instantaneous. The second reaction, the formation of cyanate, is accomplished most rapidly and completely at a pH of 10 or higher. A detention time of 30 minutes to two hours is usually allowed (USEPA 1982a).

The cyanates can be further decomposed into nitrogen and carbon dioxide by excess chlorination:

$2CNO^- + 4OH^- + 3Cl_2 = 6Cl^- + 2CO_2 + N_2 + 2H_2O$
cyanate + hydroxyl ion + chlorine = chloride ion + carbon dioxide + nitrogen + water

An alternate approach to be avoided uses acid hydrolysis:

$CNO^- + 2H_2O = CO_2 + NH_3 + OH^-$
cyanate ion + water = carbon dioxide + ammonia + hydroxyl ion

Decomposition by excess chlorination can be accomplished in about one hour if the pH is adjusted to 8.0-8.5. Acid hydrolysis usually takes place at pH 2-3. Since care must be taken to avoid the liberation of the toxic cyanogen chloride as a gas, acid hydrolysis is not usually the chosen option (USEPA 1982a).

Chemical oxidation equipment often consists of an equalization tank followed by two reaction tanks, although the reaction can be carried out in a single tank. Each tank has an electronic recorder-controller to maintain required conditions with respect to pH and oxidation reduction potential (ORP). In the first reaction tank, conditions are adjusted to oxidize cyanides to cyanates. To effect the reaction, chlorine is metered to the reaction tank as required to maintain the ORP in the range of 350 to 400 millivolts, and 50 percent aqueous caustic soda is added to maintain a pH range of 9.5 to 10. In the second reaction tank, conditions are maintained to oxidize cyanate to carbon dioxide and nitrogen. The desirable ORP and pH for this reaction are 600 millivolts and a pH of 8.0. Each of the reaction tanks is equipped with a propeller agitator designed to provide approximately one turnover per minute. Treatment by the batch process is accomplished by using two tank, one for collection of water over a specified time period, and one for the treatment of an accumulated batch. If dumps of concentrated wastes are frequent, another tank may be required to equalize the flow to the treatment tank. When the holding tank is full, the liquid is transferred to the reaction tank for treatment. After treatment, the supernatant is discharged and the sludges are collected for removal and ultimate disposal.

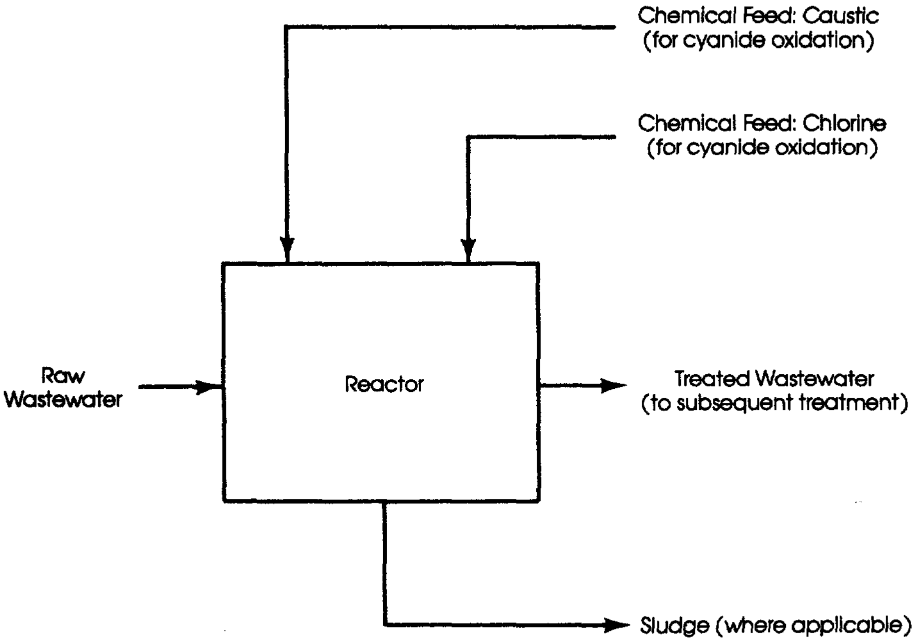
An alternative chlorination technique involves the use of sodium hypochlorite (NaOCl) as the oxidant. Reactions with sodium hypochlorite are similar to those of chlorine except that there is no caustic requirement for destruction of free cyanide in the oxidation stages. However, alkali is required to precipitate metal cyanide complexes as hydroxides. A typical oxidation system using chlorine and caustic is shown in Figure 6-15.

(ii) Oxidation by Ozone

Ozone, an unstable molecule (O_3), is a highly reactive oxidizing agent that is approximately ten times more soluble than oxygen on a weight basis. Ozone is used in industrial waste treatment to oxidize cyanide to

FIGURE 6-15

Process Flow Sheet - Chemical Oxidation



cyanate and to oxidize phenols and dyes to a variety of colorless non-toxic compounds. Ozonation is best suited for wastewaters that contain low levels of oxidizable material. There are no inherent restrictions on levels of dissolved or suspended solids in the wastewater, so long as they do not contain oxidizable material that can compete for ozone with the pollutant component to be removed.

The cyanide oxidation can be illustrated by the following ionic equation:

$CN^- + O_3 = CNO^- + O_2$
cyanide ion + ozone = cyanate + oxygen

The reaction indicated by the above equation represents the oxidation of cyanides to cyanates. Continued exposure to ozone will convert the cyanate to carbon dioxide; however, this is not economically practical. Thus, further breakdown of the cyanate waste is dependent on processes such as hydrolysis and bio-oxidation.

Ozone is also effective in the treatment of phenols. It is about twice as powerful as hydrogen peroxide and is not as selective; thus it oxidizes a wide range of materials. For low concentration phenolic wastes, the usual practice is to oxidize the phenolic compound to intermediate organic compounds that are toxic but readily biodegradable. For concentrated or intermediate level phenolic wastes, oxidation by ozone may not be as economical as compared to biological oxidation; however, it is useful as a polishing process following a biological system.

Ozone is more soluble and more stable in acidic solutions than in basic solutions. However, the rate of ozonation reaction is relatively insensitive to pH. Therefore, it is rarely worthwhile to adjust pH prior to ozonation, since the cost of the neutralization process will frequently offset any gains in ozonation efficiency. One exception to this generalization is cyanide ozonation. The cyanate formed initially hydrolyzes more rapidly in alkaline media. If complete conversion of cyanide to carbon dioxide is required, acidic streams should be adjusted to a pH of about 9 before ozonation.

Because ozone is unstable, it must be generated on site. Ozone generators utilize a corona discharge that occurs when a high- voltage alternating current is imposed across a discharge gap. Ozone is generated either from air or from an air stream. Two to two and a half times as much ozone will be produced from a stream of 100% oxygen as from an air stream. The ozonized air from the generator is introduced into a contact chamber designed to ensure good mixing with the waste streams. The two ozone/water mixing systems in most widespread use are venturi mixers and porous diffusers. With the venturi mixer, ozonified gas and waste flow co-currently, and ozonized gas flow is limited to 30-60% of the liquid volume flow. In a porous diffuser system, a counter-current flow is usual, and gas flow may be up to twenty times the liquid flow. In some systems the contact column is a packed bed. This increases surface area and increases the rate of mass transfer of ozone into solution. One equipment manufacturer has been using ultrasonics in conjunction with ozonation, which also increases the surface area available for mass transfer. Depending on the extent of treatment required, it may be necessary to incorporate two or more contact stages, which can be of different types. If oxygen is used as the feed gas instead of air, the closed-loop ozonation method is utilized. In this system off-gas from the contactor is recycled back to the ozone generator and enriched with ozone.

Modern ozone systems are completely automated. An ozone monitor provides continuous on-line monitoring of the ozone concentration in the gaseous effluent from the contactor. If the concentration of ozone exceeds a preset level (usually 0.05 ppm), the voltage or frequency of the ozone generator is reduced.

(iii) Oxidation by Ozone with Ultraviolet (UV) Radiation

One of the modifications of the ozonation process is the simultaneous application of ultraviolet light and ozone for oxidation of cyanides, phenolic chemicals, and halogenated organics. Some compounds that are highly resistant to ozone alone can be ozonated in the presence of UV light. UV radiation catalyzes the ozonation process and the mode of action seems to be activation of ozone itself. Application of ultraviolet light reduces the amount of ozone required compared with a system using ozone alone.

(iv) Oxidation by Hydrogen Peroxide

Hydrogen peroxide (H_2O_2) is a powerful oxidizing agent and has been used to oxidize phenols, cyanides, sulfur compounds, and metal ions. Hydrogen peroxide in the presence of a metal catalyst (e.g., Fe++, Fe+++, Al+++, Cu++, and Cr++) effectively oxidizes phenols over a wide range of temperature and concentrations. The process is sensitive to pH, with an optimum pH range of 3 to 4 and efficiency decreasing rapidly at both higher and lower values (USEPA 1982a).

The oxidation of cyanide with hydrogen peroxide can be carried out by two processes. The first involves the reaction of hydrogen peroxide with cyanide at alkaline pH and in the presence of a copper catalyst. The second process, known as the Kastone process, uses a formulation containing 41% hydrogen peroxide, with trace amounts of catalyst and stabilizers, in conjunction with formaldehyde. The cyanide wastes are heated to 120°C (248°F), treated with oxidizing solution and formaldehyde, and then agitated for one hour. Principle products from the reaction are cyanates, ammonia, and glycolic acid amide. Complete destruction of cyanates requires acid hydrolysis subsequent to this reaction.

Hydrogen peroxide has been used for the separation of metalons by selective oxidation. In this way, it helps remove iron from combined streams by oxidizing the ferrousion to ferric, which is then precipitated by the addition of the appropriate base. In dilute solution (<30%), the decomposition of hydrogen peroxide is accelerated by the presence of metal ion contaminants. At higher concentrations of hydrogen peroxide, these contaminants can catalyze violent decomposition. Therefore, hydrogen peroxides should be added slowly to the solution with good mixing. Sulfides, sulfites, and sulfur dioxide can also be readily oxidized with hydrogen peroxide. Under alkaline conditions, sulfates are usually produced.

(v) Oxidation by Potassium Permanganate ($KMnO_4$)

Potassium permanganate has been used for destruction of organic residues in wastewater and in potable water. Potassium permanganate reacts with aldehydes, mercaptans, phenols, and unsaturated acids. It is considered a relatively powerful oxidizing agent. The reduced form of permanganate is manganese dioxide (MnO_2), which can be removed by filtration.

Potassium permanganate is mainly used to destroy phenolic compounds in industrial wastewater. It reacts by cleaving the aromatic ring structure of phenol to produce a straight chain aliphatic molecule. The aliphatic is then further oxidized to carbon dioxide and water. The initial reaction takes place almost immediately, and almost 90% of the phenol is oxidized in the first ten minutes. A retention time from one to three hours is sufficient to insure complete oxidation of the phenol (USEPA 1982a). The process is affected by pH; the higher the pH (up to 9.5), the faster is the reaction time.

6.7.3 Technology Status

The technology for large-scale application of chemical oxidation is well developed. Industrial wastes such as cyanides and other hazardous species in dilute waste streams (e.g., phenols and organic sulfur compounds) are commonly treated by chemical oxidation. The oxidation of cyanide wastes by chlorine is a classic process and is found in most plants using cyanides in electroplating operations.

6.7.4 Applications

Chemical oxidation can be used to treat both organic and inorganic waste components. The main applications of the process are as follows:

(i) Oxidation of Cyanide Effluent

Numerous plating and metal finishing plants use chemical oxidation methods to treat their cyanide wastes. Cyanides and heavy metals are often present together in plating industry wastes. Their concentration and their value influence the selection of the treatment process. If the cyanide and heavy metals are not economically recoverable by a method such as ion exchange, the cyanide radical is converted either to the less toxic cyanate or to carbon dioxide and nitrogen by oxidation, while the heavy metals are precipitated and removed as a sludge. In treating cyanide waste by oxidation, hypochlorite or caustic plus chlorine (alkaline chlorination) or ozone, or ozone with UV radiation may be used to oxidize the cyanide. The treatment of cyanides by chemical oxidation is currently practiced in the following industries on a widespread basis:

- Inorganic Chemicals Manufacturing,
- Metal Finishing, and
- Textile Mills.

Chemical oxidation is also used on a limited basis in the following industries:

- Iron and Steel Manufacturing,
- Coil Coating,
- Photographic Equipment and Supplies,
- Nonferrous Metals Manufacturing,
- Ore Mining and Dressing, and
- Petroleum Refining.

(ii) Oxidation of Phenol

Chemical oxidation of phenols has found application to date only on dilute waste streams. Potassium permanganate, one of the oxidants used, is reduced to manganese dioxide ($MnO2$) which is a filterable solid. Chlorine gas is not frequently used because of the high potential of formation of chlorophenols. An ozonation process for oxidation of phenols is currently practiced in the Iron and Steel Manufacturing industry.

(iii) Oxidation of Other Organics

Chemical oxidizing agents have been used for the control of organic residues in wastewaters and in potable water treatment. Among the organics for which oxidative treatment has been reported are aldehydes, mercaptans, benzidine, and unsaturated acids. For these applications, sodium hypochlorite, calcium hypochlorite, potassium permanganate, and hydrogen peroxide have been reported as oxidants. In one application nitrous acid was used.

In addition to the above major applications, chemical oxidation using chlorine and calcium hypochlorite has been used to prevent accumulation of soluble sulfides in sewer lines. Oxidation of pesticides has also been investigated to remove residual Diquat and Paraquat from water.

6.7.5 Advantages and Limitations

Some advantages of chlorine oxidation are operation at ambient temperature, suitability for automatic control, and low cost. Limitations of the process include the need for careful pH control, possible chemical interference in the treatment of mixed wastes, and the potential hazard of storing and handling chlorine gas. In addition, oxidation of phenols using chlorine may result in the formation of dangerous and more toxic chlorophenols if the wastewater is underchlorinated.

Some advantages of ozone oxidation are that it is well suited to automatic control, on-site generation eliminates treatment chemical procurement and storage problems, reaction products do not include chlorinated organics, and no dissolved solids are added in the treatment step. Some limitations of the process are high capital expense, possible chemical interference in the treatment of mixed wastes, and an energy requirement of 25 kwh/kg of ozone generated.

The principal disadvantage of chemical oxidation is that the process is not selective. Since priority pollutants are often a small percentage of total organic material, high dosages may be required to achieve removal of desired pollutants. For these reasons, its used is generally restricted to specific product waste streams where the target pollutant is a major factor, or where no reasonable alternative exists.

6.7.6 Reliability

The oxidation process is highly reliable with proper monitoring and control and proper pretreatment to control interfering substances.

6.7.7 Chemicals Required

Common chemicals used as oxidizing agents include chlorine (Cl_2), sodium hypochlorite ($NaOCl$), calcium hypochlorite ($Ca[OCl]_2$), potassium permanganate ($KMnO_4$), hydrogen peroxide (H_2O_2), and ozone (O_3).

6.7.8 Residuals Generated

Most chemical oxidations will generate a residue which is removed by subsequent treatment operations. The most significant generation of residue occurs from the use of caustic or lime slurry with chlorine gas in alkaline chlorination. Smaller amounts of residue result from oxidations using hypochlorites.

6.7.9 Design Criteria

Either batch or continuous operations may be employed for oxidation. Batch treatment is preferred for flows less than 190,000 to 380,000 litres per day (50,000 to 100,000 gallons per day). Oxidizing reagent consumption and choice of reagent will depend upon process efficiency, presence of competing oxidizable material, and temperature, and should be determined by pilot- scale testing. Very simple equipment is required for chemical oxidation. This includes storage vessels for the oxidizing agents and perhaps for the wastes, metering equipment for both streams, and vessels with agitators to provide suitable contact of oxidant and waste. Some instrumentation is required to determine the concentrations of pollutants, pH of the water, and the degree of completion of the oxidation reaction. The process may be monitored by an oxidation-reduction potential (ORP) electrode. This electrode is generally a piece of noble metal (often platinum) which is exposed to the reaction medium. The electrode produces an electromotive force (EMF) output that is empirically related to the ratio of oxidized to reduced constituents in the solution.

A typical reaction condition for the alkaline chlorination of one kilogram (2.2 pounds) of cyanide to cyanate requires six kilograms (13.2 pounds) each of sodium hydroxide and chlorine. The reaction is carried out at pH 10, and at least 15 minutes contact time is required to drive the reaction to completion. If metal cyanide complexes are present, extended chlorination for longer periods may be necessary. Complete destruction of cyanate requires a second oxidation stage with approximately 45 minutes retention at a pH below 8.5. The theoretical reagent requirement for this second stage is 4.1 kilograms (9.0 pounds) of chlorine and 1.1 kilograms (2.4 pounds) of caustic per kilogram (2.2 pounds) of cyanide (USEPA 1982a).

Oxidation of cyanide to cyanate with ozone requires approximately 2 kilograms (4.4 pounds) of ozone per kilogram (2.2 pounds) of cyanide, and complete oxidation requires 5 kilograms (11 pounds) of ozone per kilogram (2.2 pounds) of cyanide. Cyanide oxidation to cyanate is very rapid (10 to 15 minutes) at pH 9 to 12 and practically instantaneous in the presence of trace amounts of copper (USEPA 1982a).

Ozone requirements for partial destruction of phenols range from one to five parts per part of phenol. The actual ozone demand will be a function of phenol concentration, pH, and retention time.

6.7.10 Performance

Chemical oxidation is very effective in destroying free cyanide as well as cadmium, copper, and zinc cyanide complexes. Important to note is that chemical oxidation only destroys the cyanide part of the complex. However, nickel cyanide is incompletely destroyed, and iron cyanide complexes are apparently unaffected by chlorine or ozone (USEPA 1982a). The ozone-UV radiation process is effective for treatment of complexed cyanides such as ferric cyanide, copper cyanide, and nickel cyanide. Performance data of oxidation processes from the following industries and/or waste streams are presented in the data sheets:

- Inorganic Chemicals Manufacturing,
- Ore Mining and Dressing,
- Organic and Inorganic Wastes,
- Textile Mills,
- Organic Chemicals, and
- Adhesives and Sealants.

Tables 6-7 and 6-8 present removal efficiency data for chlorine and ozone oxidation respectively.

6.7.11 Costs for Chlorine Oxidation of Cyanide

(A) Basis of Design

This presentation is for estimating costs for oxidation of cyanide, by the alkaline chlorination process. A system of the type considered is represented in Figure 6-16. The capital cost factor is the volume of the two stage reactor vessel. The principal design factors for cyanide oxidation systems are waste-water flow and influent cyanide concentration. Influent oil and grease and TSS are checked to determine if pretreatment

TABLE 6-7

Control Technology Summary for Chemical Oxidation-Chlorine

Pollutant	Data Points Pilot Scale	 Full Scale	Effluent Concentration Range	 Median	Remove Efficiency, % Range	 Median
Classical pollutants, mg/L:						
COD		1		980		35
TSS		2	33-160	97		97
Ammonium nitrate		1		120		37
Toxic pollutants, ug/l:						
Copper		1		320		14
Cyanide	1	2	<2-130	35	82->99	>99
Lead		1		2,500		0

Blanks indicate data not available.

TABLE 6-8

Control Technology Summary for Chemical Oxidation-Ozone

	Data Points		Effluent Concentration		Remove Efficiency	
Pollutant	Pilot Scale	Full Scale	Range	Median	Range	Median
Classical pollutants, mg/L:						
BOD (5)	2		610 - 5,200	2,900		10
COD	2		76 - 12,000	6,000		84
TSS	2		12 - 140	76		NM
TOC	3		200 - 980	540	5 - 52	11
Oil and grease	1			4		97
Total phosphorus	2		0.16 - 1.1	0.63		0
Total phenols	2		0.01 - 0.13	.07	50 - >99	>74
Aluminum	2		0.13 - 7	3.6		NM
Manganese	2		0.07 - 0.09	0.08		12
Vanadium	2		0.02 - 0.54	0.28		NM
Barium	2		0.01 - 0.12	0.06		0
Iron	2		0.25 - 2.3	1.3		4
Silicon	2		2.6 - 4.7	3.6	2 - 4	3
Titanium	2		0.002 - 0.18	0.09		NM
Boron	2		0.74 - 1.0	0.87		0
Calcium	2		4.8 - 30	17		9
Magnesium	2		2.2 - 6.1	4.2	0 - 8	4
Sodium	2		53 - 56	54	2 - 8	5
Molydenum	2		<0.01 - <0.4	<0.2		NM
Cobalt	2		0.008 - 0.38	0.19		NM
Strontium	2		0.002 - 0.03	0.016	0 - 99	50
Ammonia	2		0.26 - 5.5	2.9		NM
Nitrate	2		1.8 - 8.8	5.3		47
Tin	1			<0.02		NM
Toxic pollutants, ug/L:						
Antimony	2		25 - 1,200	610		NM
Arsenic	2		4 - 43	24	0 - 48	24
Beryllium	2		<0.04 - <4	<2.0		NM
Cadmium	2		<2 - 250	<130		NM
Chromium	2		6.3 - <200	<100		NM
Copper	2		89 - 590	340		NM
Cyanide	4		<2 - 1,500	<24	>33 - >98	78
Lead	2		<22 - <900	<460		>29
Mercury	1			<1.1		NM
Nickel	2		66 - 5,000	2,500		NM
Selenium	1			<1		NM
Silver	2		16 - 1,300	660		NM
Thallium	1			<50		NM
Zinc	3		90 - 460	240		96

TABLE 6-8 (continued)

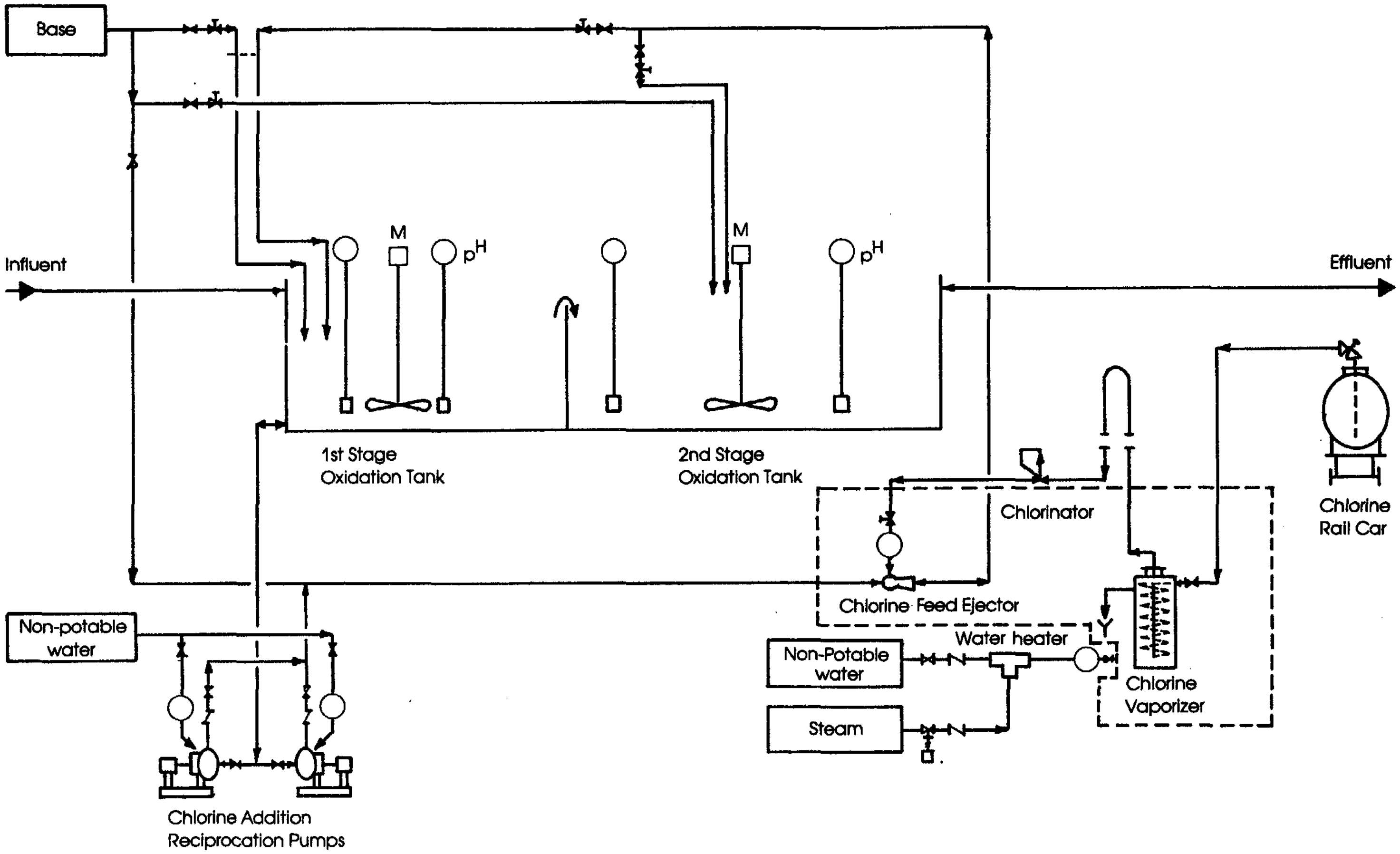
Control Technology Summary for Chemical Oxidation-Ozone

	Data Points		Effluent Concentration		Remove Efficiency	
Pollutant	Pilot Scale	Full Scale	Range	Median	Range	Median
Bis (2-ethyhexyl) phthalate	2		90 - 110	100		NM
Butyl benzyl phthalate	1			BDL		98*
Di-n-butyl phthalate	1			2.7		77
Benzene	1			BDL		80*
Ethylbenzene	2		BDL - 0.1	0.05		NM
Toluene	2		0.9 - 1.2	1.0		31
Anthracene	2		BDL - 0.4	0.2		98*
Benzo (a) pyrene	1			BDL		95*
Benzo (k) fluoranthene	1			BDL		90*
Fluoranthene	1			0.1		50
Pyrene	1			0.1		67
Chloroform	1			BDL		NM
1, 1 - Dichloroethane	1			BDL		NM
1, 2 - Trans-dichloroethylene	1			2.1		NM
Methylene chloride	2		15 - 61	38		NM
Trichloroethylene	1			0.9		NM

Blanks indicate data not available.
BDL = below detection limit.
NM = not meaningful.
*Approximate value.

FIGURE 6-16

Process Flow Diagram for Chemical Oxidation



is necessary. Chlorine is supplied to the system at a mass ration of 15 parts chlorine to 1 part cyanide and caustic is added to control pH between 8.0 and 9.5 and to subsequently neutralize any excess chlorine. The reaction vessel is sized for 10 minutes residence time in the first stage and 30 minutes in the second.

(1) Source

The unit cost information in this section was derived from the BAT Effluent Limitation Guidelines engineering study for the Organic Chemicals/Plastics and Synthetic Fibers Industries (USEPA 1982a). The method for developing the design factor is based on assumptions and procedures in the Contractor Developed Design and Cost Model (USEPA 1982a).

(2) Required Input Data

Wastewater flow l/s (mgd)
Wastewater characteristics
cyanide (mg/l)
oil and grease (mg/l)
TSS (mg/l)
pH

(3) Limitations

Chlorine oxidation is not considered applicable if cyanide is present at less than 10 mg/l.

(4) Pretreatment

Pretreatment should be provided as indicated for the following conditions:

(a) Equalization if necessary, due to flow variations.

(b) If influent oil and grease >50 mg/l, use oil removal process.

(c) If influent TSS >50 mg/l, use multi-media filtration.

(5) Design Equation

The primary cost factor for alkaline chlorination of cyanide is the volume of the reaction vessel. The required basin volume for a chemical oxidation system is calculated based on a standard hydraulic detention time of 40 minutes (10 min. first stage and 30 min. second stage).

Metric

$$VOL = (FLOW \times 40 \times 60) / 1000$$

where: VOL = basin volume, m3
FLOW = average influent flow, l/s
40 = detention time, min.

60 = seconds/minute
1000 = conversion factor, l/s to m3/s

English

VOL = (FLOW x 40) / 1.44

where: VOL = basin volume, thousand gallons
FLOW = average influent flow, mgd
40 = detention time, min.
1.44 = conversion factor, mgd to thousand gallons/min

(6) Subsequent Treatment

None specified

(7) Chlorinated Organics

Possible formation of chlorinated organics from alkaline chlorination of cyanide should be carefully considered.

(B) Capital Costs

The volume of the two stage oxidation tank is the primary factor for estimation of capital cost using the capital cost curve (Figure 6-17). Costs estimated using this curve must be adjusted to a current value using an appropriate current cost index.

(1) Cost Data

Items included in the capital cost estimate for alkaline chlorination of cyanide are as follows (USEPA 1982).

Two stage concrete reaction vessel
Agitators (2)
Chlorine feed systems
chlorine vaporizer
chlorinator
circulator pumps (2)
Piping, instrumentation, electrical
Metal shed
pH control
ORP (oxidation/reduction potential) control

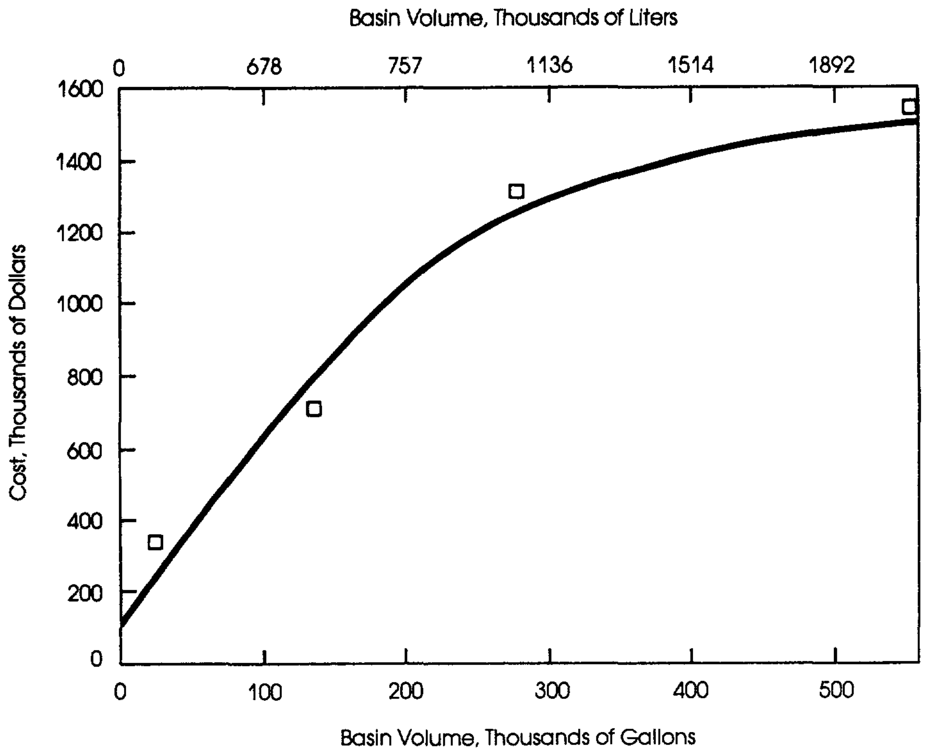
(2) Capital Cost Curves

Curve - see Figure 6-17.

- Cost (thousands of dollars) vs. basin volume (cubic meters or thousand gallons).

FIGURE 6-17

Capital Cost Estimate for Chemical Oxidation



- o Curve basis, cost estimates on four volumes: 105, 526, 1050, and 2100 m3 (27.8, 139, 278, and 556 thousand gallons) 43.8, 219, 438, and 8761/s (1, 5, 10, and 20 mgd) at 40 minute detention.

(3) Cost Index

Base period, July 1977, St. Louis Chemical Engineering (CE) Plant Index = 204.7

(C) Operation and Maintenance Costs

Operating costs include both fixed and variable components. Variable operating costs include power, chlorine, and caustic. Fixed operating costs include labor, supervision, overhead, laboratory labor, maintenance, services, insurance and taxes, and service water. All fixed and variable operating costs should be adjusted to current levels using an appropriate index or unit cost factor.

(1) Variable Costs

(a) Power Requirements

Includes agitators, recirculation pumps. The following equation was developed using regression analysis procedures (USEPA 1982a).

Metric

$$KW = (0.166 \times VOL) + 23.6$$

where: KW = power, kilowatts
VOL = basin volume, m3

English

$$HP = (0.845 \times VOL) + 31.6$$

where: HP = power, Hp
VOL = basin volume, thousand gallons

(b) Power Cost

Metric

$$PC = KW \times 24 \times EC$$

where: PC = power cost, \$/day
KW = power, kilowatts
24 = hr/day
EC = electricity cost, \$/Kw-hr

English

$$PC = HP \times 24 \times 0.746 \times ECo$$

where: PC = power cost, \$/day
HP = power, Hp
24 = hr/day
746 = Kw-hr/Hp-hr
EC = electricity cost, \$/Kw-hr

(c) <u>Chemical Requirements</u>

- **Chlorine**

Metric

$$CL = FLOW \times 0.086 \times CN \times 15$$

where: CL = chlorine requirement, Kg/day
FLOW = influent flow, l/S
.086 = conversion factor
CN = influent cyanide (as NaCN), mg/l
15 = ratio, kg chlorine/kg influent cyanide

English

$$CL = FLOW \times 8.34 \times CN \times 15$$

where: CL = chlorine requirement, lb/day
FLOW = influent flow, mgd
8.34 = conversion factor
CN = influent cyanide, (as NaCN), mg/l
15 = ratio, lb chlorine/lb influent cyanide

- **Caustic** (needed to maintain pH between 8.0 and 9.5)

- **If influent pH <8.0:**

Metric

$$CR = [(CN \times 17) + (8 - pH)3 \times 15] \times 0.086 \times FLOW$$

English

$$CR = [(CN \times 17) + (8 - pH)3 \times 15] \times 8.34 \times FLOW$$

- **If influent pH >8.0:**

Metric

$$CR = CN \times 17 \times 0.086 \times FLOW$$

where: CR = required amount of caustic, Kg/day
17 = Kg caustic/Kg CN-

English

CR = CN x 17 x 8.34 x FLOW

where: CR = required amount of caustic, lb/day
17 = lb caustic/lb CN-

(d) Chemical Costs (except caustic):

Once the total requirements for chlorine has been established, the associated cost may be estimated as follows:

CC = CL x N

where: CC = chlorine cost ($/day)
CL = calculated requirement for chlorine Kg/day or lb/day
N = unit cost of chlorine, $/Kg or $/lb

(2) Fixed Costs

The fixed O & M components for this technology are listed in Table 6-9, including values for the cost basis and the unit costs (USEPA 1982a).

(a) Miscellaneous Costs

Costs for engineering, and other common plant items such as land, piping, and buildings, are calculated after completion of costing for individual units.

(b) Modifications

None.

6.8 Stabilization/Solidification

6.8.1 Introduction

Stabilization/solidification is a technology where a waste material is mixed with materials that tend to set into a solid, thus capturing or fixing the waste within the solid structure.

Although the terms solidification and stabilization are usually used interchangeably, they represent different concepts for the control of waste materials.

- **Solidification** suggests the production of a solid, monolithic mass with sufficient structural integrity to be transported in conveniently-sized pieces without requiring any secondary container.
- **Chemical stabilization** suggests immobilization of toxic substances by reacting them chemically to form insoluble compounds in a stable crystal lattice.

TABLE 6-9

Fixed O & M Cost Basis and Unit Cost Factors for Chemical Oxidation (4-11)

Element	Cost Basis (Equivalent Unit Quantity)	Base Unit Cost (July 1977)
Labor (1,2)	0.20 Weeks (4.80 hrs/day)	$ 9.80/hr
Supervision (1)	10% Labor (0.48 hrs/day)	$11.76/hr
Overhead (1)	75% Labor Cost	NA
Laboratory (3)	0.15 Shifts (0.86 hrs/day)	$10.70/hr
Maintenance	3.93% Capital	NA
Services	0.04% Capital	NA
Insurance and Taxes	2.50% Capital	NA
Service Water	0.075 L/s (1.72 Thou gpd)	$ 0.13/thou L ($.50/thou gal)

NA - not applicable

(1) Labor may vary from 0.7 to 1.2 times the standard amount indicated depending on the overall scale of the plant. Labor, Supervision, and Overhead may be adjusted for the scale of the plant as indicated in Miscellaneous Costs (Section IV.3.5)

(2) One week = 7 days = 168 hours = 4.2 shifts

(3) One shift = 40 hours

- **Physical stabilization** involves blending the sludge or semi-liquid with a bulking agent such as pulverized fly ash (pfa) to produce solids of a coarse-grained, soil-like consistency that can be readily transported by truck, conveyor or railcar to a disposal site. The primary objective of this straight-forward blending process is the production of a dry, transportable waste product with acceptable environmental properties.

Other fixation techniques include clay-based processes, and various forms of encapsulation using organic polymers that harden during treatment.

The prime objective in all these processes is to convert toxic waste streams into an inert, physically stable mass, having very low leachability and with sufficient mechanical strength to allow for land reclamation or landfilling. The mobilization of wastes into water has always been a major consideration in containing toxic waste, and so much of the emphasis on fixation processes has been placed on preventing the waste from coming into contact with water and controlling the chemical conditions of the aqueous environment in order to minimize solubility. Unfortunately, other serious long-term human-health and environmental impacts cannot be completely ruled out because there is little or no information about long-term environmental degradation other than water pollution.

6.8.2 Processes

Present stabilization/solidification or fixation systems can be grouped into the following seven classes of processes (Poon et al. 1983).

(i) Solidification through cement addition.

(ii) Solidification through the addition of lime or other pozzolanic materials.

(iii) Embedding wastes in thermoplastic materials such as bitumen, paraffin, or polyethylene.

(iv) Thermosetting micro-encapsulation.

(v) Macro-encapsulation of wastes in an inert coating.

(vi) Treatment of the wastes to produce a cementitious product with major additions of other constituents.

(vii) Formation of a glass by fusion of wastes with silica (classification).

The first two methods are the most commonly used, being suitable for the large majority of inorganic process wastes. Treatment costs for the remaining processes are generally higher; these latter techniques are reserved for problematic wastes such as radioactives or those with a high organic content.

(i) Cement-based Processes

Cement-based techniques generally use Ordinary Portland Cement (OPC) materials with additives to improve the physical characteristics and decrease the leaching losses from the resulting solidified waste. Hazardous waste is mixed in a slurry with water and anhydrous cement powder. The mechanism of stabilization is the formation of hydration products from silicate compounds and water. A calcium silicate hydrate gel forms. This gel then swells and forms the cement matrix composed of interlocking silicate fibrils and hydration products (Birchall, Howard, and Bailey 1978).

A number of compounds can interfere with the solidification process (USEPA 1981). Some, such as soluble salts of manganese, tin, copper and lead, can increase setting times and greatly decrease physical strengths. Impurities such as organic matter, silts and some clays, can cause significant delays in setting. These impurities and other insoluble materials, fine enough to pass a No. 200 mesh sieve, can coat larger particles and weaken the waste/cement bond.

Variations of the basic process can accommodate atypical physical and chemical characteristics of the waste. For example, while water is essential for the hydration and setting of the cement, an excess of water will hinder the curing. For liquid or slurry-type wastes, a bulking agent, generally PFA, is added to arrive at a consistency suitable for cement addition. Another problem frequently encountered with acid wastes is that cement, being alkaline, first mops up free acidity, before commencing to set at about pH 11. Addition of OPC to such wastes is therefore wasteful, and again lime or PFA is first introduced to thicken the waste and raise its pH above 7. A minimum of about 10% cement on wt/wt basis is generally required to produce a cured solid product with the necessary mechanical strength.

Clay minerals, such as bentonite, with their cation-exchange capacity and extensive specific surface, will react with organic compounds and have therefore been used to extend the range of wastes suitable for fixation to inorganic wastes containing up to 5% of certain, generally basic, organics (Kupiec 1980). There are many types of clay minerals but only those with sufficiently high cation-exchange capacity are suitable.

Examples include vermiculite and montmorillonite clays which have cation-exchange capacities in the range of 130 to 150 meq per 100g (Grim 1962, Theng 1974). The setting agent, usually OPC, reinforces the gelling action of the clay and forms a physically stable material which has the ability to entrap and bind the contained wastes. Thus the waste material is converted into a chemically and physically stable solid simulating soil which is claimed to be substantially insoluble in water but is spongy so as to reabsorb water without leaching out to any appreciable extent.

(ii) Lime-based Process

Lime and pozzolanic materials in combination produce cementitious matrices which chemically and physically entrap the waste.

Pozzolanas are defined as materials which, though not cementitious in themselves, contain constituents which will combine with lime at ordinary temperature in the presence of water to form stable insoluble compounds having such properties (Lea 1970).

Both natural and artificial materials may be used, including volcanic dust and ash as well as artificial pozzolanas such as burnt clays and shales, spent oil shales, burnt gauze, burnt mortar and PFA. The most commonly used pozzolanas in chemical fixation processes are fly ash and cement-kiln dust (Pojasek 1979). Both are wastes themselves and the process has the obvious advantage of co-disposal. Included in the mechanistic interpretation of the lime-pozzolana reaction is the theory that pozzolanic materials undergo similar types of reaction to zeolitic compounds with respect to exchange of their base ion constituents. It is suggested that this base exchange capacity may enhance the combination with lime and other toxic metal ions in the fixation process (Pojasek 1979).

An alternative interpretation is that the major pozzolanic reactions are due to the formation of new hydrated compounds, namely the tricalcium silicate hydrate as in the hydration of cement. The net effect, it is claimed, is that the waste particles are micro-encapsulated within a largely gel matrix.

Table 6-10 shows the influence of lime addition to a fibre-gas desulphurization (FGD) sludge physically stabilized with PFA (Pojasek 1979). The strength of the final solid product is enhanced with greater lime concentrations. In this instance, PFA is used both as a bulking agent and as the pozzolanic material - an excess of water is as undesirable as in cement-based processes. In order to produce a mechanically strong solid, lime additions of up to 30% may be required, depending on the waste type and the chemical composition of the pozzolana.

(iii) Thermoplastic Encapsulation

Thermoplastic materials are organic plastics capable of reversibly softening and hardening upon heating and cooling. These polymeric materials are usually linear with few, if any, cross-links and either swell appreciably, or are soluble, in organic solvents. Thermo-plastic materials include asphalt, bitumen, polyethylene, polypropylene and nylon. The most frequently used materials in this group are asphalt and bitumen.

In the encapsulation technique, the waste needs to be either dry or de-watered to a high solids content and is then mixed with the polymer at an elevated temperature (usually 100°C). Some processes utilize liquid slurries and mix them with the thermoplastic materials while raising the temperature to evaporate excess water. The process has been used in both continuous and batch mixing operations but is clearly energy intensive in operation (Pojasek 1979). Materials such as sulphur have been incorporated in the encapsulating matrix to improve physical strength and structural integrity. Sulphur/asphalt blends containing up to 60% of

TABLE 6-10

Effect of Chemical Fixation on Product Strength

Additive	Wt % Additive	Ash/Sludge Wt Ratio	Unconfined Compressive Strength (PSI)
Lime	0	1/1	85
Lime	1	1/1	250
Lime	3	1/1	600
Lime	5	1/1	950
Lime	5	1/2	360

Source: Pojasek, R. B. 1979. Toxic and Hazardous Waste Disposal. Vol. 1. Ann Arbor: Ann Arbor Science Publishers, Inc.

sulphur can offer a more cost-effective technique subject to the availability of large quantities of sulphur at low cost from secondary sources such as stack gas wastes, oil refineries and coal processing (Brenner and Rugg 1982).

(iv) Thermosetting Micro-encapsulation

Thermosets are materials which become solid and harden when heated. They remain solid even after subsequent heating and cooling cycles.

Micro-encapsulation processes involve in-situ polymerization forming an impervious coating surrounding the waste particles. Organic monomers are thoroughly mixed with the waste and promoter/catalyst is added to initiate polymerization. Cross-linking is influenced by such parameters as pH, water content and ionic constituents in the feed streams with the hardened cross-linked polymer produced being stable to heat and soluble in organic solvents (Subramanian and Wakalingan 1977, USEPA 1977)

The major organic polymer types currently in use for waste treatment are the ureaformaldehyde, polyester and polybutadiene processes with phenolformaldehyde and epoxide resins being used to a small extent.

(v) Macro-encapsulation

Macro-encapsulation basically consists of sealing the waste materials in a treated or untreated form with an impervious and inert covering.

For containerized waste, it is usually assumed that the container wall will eventually corrode and subsequently fail to exhibit load-bearing characteristics and therefore a load-bearing jacket can be designed to reinforce the container. In one process, the reagent used is a mixture of fiberglass, reinforced thermosetting resins (epoxides) and waterbased polyurethane resins. The mixture is sprayed or brushed onto the container wall and it is claimed to provide a tough exterior jacket which will protect the contents from leaching and mechanical stress (Lubowitz and Telles 1981).

Mention must also be made of a very simple macro-encapsulation technique involving the use of OPC, typically employed when materials such as laboratory smalls (e.g., bottles of chemicals, etc.) are to be disposed of. The material, already contained in packaging, is transferred to a steel drum, and the interstices filled with an OPC slurry. The hardened cement encases the chemicals and is in turn supported by the load-bearing drum. The container can then be disposed of in a landfill site. A similar technique can be used to encapsulate radioactive materials, prior to disposal.

(vi) Cementitious Techniques

Self-cementation can be applied to wastes containing large amounts of calcium sulphate or calcium sulphite, such as flue-gas cleaning sludges or desulphurization sludges. A portion of the waste, usually 8 to 10 percent by weight, is calcined and then re-mixed with the waste along with proprietary additives. Fly-ash may be used to absorb excess moisture. The resultant product is an easily handled stable solid (Pojasek 1979).

(vii) Vitrification (Classification)

Wastes are mixed with silica and heated to extremely high temperatures, and allowed to cool into a glass-like solid. A variation of this technique, using graphite electrodes driven into buried wastes, allows in-situ vitrification. All vitrification systems employ some type of hood to capture and treat the fumes and vapours given off during operation.

Because these systems are very energy-intensive, thus costly, they have generally been considered only for high level radioactive or other extremely dangerous wastes (Pojasek 1979).

However, a new process developed in the UK is designed to vitrify asbestos by roasting in a specially adapted high-temperature furnace (Environmental News Data Service 1985). It is claimed that the volume of the waste is reduced by a factor of up to 10 and that the non-fibrous, harmless product can be used as hardcore.

6.8.3 Applicability

A wide variety of wastes have been treated by these processes including metal finishing waste, plating and lead smelting acid-waste, mine tailings, effluent treatment sludges, incineration ash, food production sludges and flue gas emission control waste. Available data suggest that the cementation processes are more favorable for inorganic wastes, especially those containing cations. Organic wastes and inorganic anionic wastes are more suitable for organic encapsulation processes.

Tables 6-11, 6-12 and 6-13 summarize the available information on the relative merits and disadvantages of the various processes.

While many materials are not totally retained by these processes, their release to the environment by leaching is considerably retarded.

A large number of proprietary processes, based on the principles discussed above, have been patented (Pojasaek 1979; Poon et al. 1983). In general, the bulking (physical stabilization) of sludge with PFA and/or the addition of OPC to a waste to promote hardening are not believed to infringe existing patents. However, use of other waste materials such as cement kiln flue dust or blast furnace slag to reduce costs, or of additives such as clays, could infringe existing patents. Thus, users of this manual are cautioned to conduct a thorough survey of the patent literature before embarking on ventures involving significant financial investment.

TABLE 6-11

Summary of Advantages and Disadvantages of Inorganic Fixation Processes

Advantages	Disadvantages
Low capital investment of equipment and running costs	Large amount of raw materials are required
Materials required are relatively cheap and plentiful	Raw materials (especially cement are energy intensive
Techniques for processing are relatively well established	Certain wastes, such as those containing organic compounds may cause some difficulties in curing
Natural alkalinity of materials helps to neutralize acidity in waste solution	Weight and volume of treated product is considerably increased
Extensive dewatering is not required because materials incorporate water and can be adapted to wide range of water content	Treated wastes are vulnerable to leaching, especially mild acids and therefore additional sealant may be required
Physical properties of treated waste can be varied from soft clay to a monolithic material by selectively varying the ratio of reagent	Mechanism of stabilization not well established
Lime based process can dispose two wastes in a single process	
Clay based process can be used for treatment of some organic waste	

Source: Poon, C. S., C. J. Peters, and R. Perry. 1983. Youth of stabilization processes in the control of toxic wastes. Effluent and Water Treatment Journal 23 (11) (November): 451-453 and 459.

TABLE 6-12

Summary of Advantages and Disadvantages of Organic Encapsulation Processes

Advantages	Disadvantages
Contamination migration rates are generally lower than with inorganic processes	Materials used are expensive
Less fixative is required when compared with inorganic processes	For thermoplastic and thermo-setting encapsulation, large expenditure of energy in drying, fusing and polymerization
Treated materials are of lower density thus reducing transportation costs	Some organic polymers are flammable
Organic materials form an impermeable boundary between waste and leaching solution	Except for macro-encapsulation, processes require skilled labour and expensive equipment
Processes can encapsulate wider range of waste	Materials are biodegradable and liable to attack by organic solvents
For macro-encapsulation, state-of-the-art of spray on equipment for resin application may be employed directly and does not require extensive expenditure of energy.	Incomplete polymerization of some of these materials could itself cause pollution

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Source: Poon, C. S., C. J. Peters, and R. Perry, 1983. Youth of stabilization processes in the control of toxic wastes. Effluent and Water Treatment Journal 23 (11) (November): 451-453 and 459.

TABLE 6-13

Compatibility of Selected Waste Categories with Different Waste Solidification/Stabilization Techniques

Waste Component	Cement Based	Lime Based	Thermoplastic Encapsulation	Thermosetting Micro-Encapsulation	Macro-Encapsulation
Organics					
Organic solvents and oils	Many impede setting, may escape as vapour	Many impede setting may escape as vapour	Organics may vaporize on heating	May retard set of polymers	Must first be absorbed on solid matrix
Solid organics (e.g., plastics resins)	Good-often increases durability	Good-often increases durability	Possible use as binding agent	May retard set of polymers	Compatible-many encapsulation materials are plastics
Inorganics					
Acid wastes	Cement will neutralize acids	Compatible	Can be neutralized before incorporation	Compatible	Can be neutralized before incorporation
Oxidizers	Compatible	Compatible	May cause matrix breakdown fire	May cause matrix breakdown	May cause deterioration of encapsulating materials
Sulphates	May retard setting and cause spailing unless special cement is used	Compatible	May dehydrate and rehydrate causing splitting	Compatible	Compatible

TABLE 6-13 (continued)

Waste Component	Cement Based	Lime Based	Thermoplastic Encapsulation	Thermosetting Micro-Encapsulation	Macro-Encapsulation
Halides	Easily leached from cement, may retard setting	May retard set most are easily leached	May dehydrate	Compatible	Compatible
Heavy metals	Compatible	Compatible	Compatible	Acid pH solubilises metal hydroxides	Compatible
Radio-active materials	Compatible	Compatible	Compatible	Compatible	Compatible

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Source: USEPA. 1981. Report No. SW-873 (PB-81-181-505)

6.8.4 Process Evaluation

The relative proportions of waste and other constituents in a satisfactory stabilization/solidification "mix" vary considerably depending on the exact composition and nature of the waste and on the details of the process. If one is selecting the most appropriate process for a particular waste stream, published information on proprietary processes is unlikely to be very helpful. There is no substitute for a detailed process evaluation in the laboratory. In addition, it is necessary to carry out specific laboratory tests on each waste stream which is to be solidified in a particular facility.

The effectiveness of a stabilization/solidification process for a particular waste can be evaluated in terms of three main properties of the treated waste:

- salting or curing time;
- physical properties; and
- resistance to leaching (solubilization) of the hazardous components.

As shown in Table 6-13, a number of materials retard the **setting** of the mix, particularly with cement or lime-based processes. Setting time is easy to test in the laboratory, for example by periodic use of a penetrometer, which is a cone of standard size and weight, dropped from a standard height.

Physical properties such as density and compressive strength of the treated waste are of importance when determining its suitability as a material for land reclamation. Typical compressive strengths, as claimed by the vendor of a proprietary cement-based process, are shown in Tables 6-14 and 6-15.

Resistance to leaching of the hazardous components is affected both by the permiability of the treated waste, which controls the rate of penetration of leaching fluid, and by the rate of leaching of the contaminants into the fluid.

Table 6-16 compares typical permeability values for various materials with that claimed by the vendor for a cement-based fixation product.

A variety of leaching tests have been devised which claim to simulate ground leaching processes which occur in landfill. Factors which affect the leaching potential of a waste include ions present, pH, stoichiometry, temperature, electrostatic charge (surface effect), presence of ligand or chelating compounds, oxidation-reduction (redox) potential, and many other parameters. The leaching process is affected by many complex reactions taking place, so that the leaching rate cannot usually be expressed in a conventional mathematical model, although this has been attempted (Poon et al. 1983).

TABLE 6-14

Compressive Strength of a Cement-Based Fixation Product (psi) (As reported by the Vendor)

	After 3 Days	After 7 Days	After 28 Days
An Arsenical Waste		390	750
An Aqueous Effluent	193	330	610
A Chromium Waste		108	220
A Chromium Waste		155	310

Source: Pojasek, R. B. 1979. Toxic and Hazardous Waste Disposal. Vol. I. Ann Arbor, Michigan: Ann Arbor Science Publishers, Inc.

TABLE 6-15

Typical Compressive Strengths of Materials (psi)

Concrete (B.S.12) consisting of a standard mix of Portland cement, sand and gravel, after 28 days.	4,500 ± 1,000
Mortar (B.S.12) consisting of a standard mix of Portland cement and sand, after 3 days.	2,100
Industrial grouts used for void-filling, soil stabilization, mud jacking and general site work, after 28 days.	77 to 616
Cement-based fixation products, after 28 days. (as claimed by vendor)	200 to 800

Source: Pojasek, R. B. 1979. Toxic and Hazardous Waste Disposal. Vol. I. Ann Arbor, Michigan: Ann Arbor Science Publishers, Inc.

TABLE 6-16

Typical Values of Coefficient of Permeability (cm/sec)

Clean Gravel	1.0 and greater
Clean Coarse Sand	1.0 to 0.01
Sand Mixtures	0.01 to 0.005
Fine Sand	0.005 to 0.001
Silty Sand	0.002 to 0.0001
Silt	0.0005 to 0.00001
Clay	0.000001 and smaller
Typical Concrete	0.000001
Cement Based Fixation Product (as claimed by a vendor)	0.0000001 to 0.0000002

Source: Pojasek, R. B. 1979. Toxic and Hazardous Waste Disposal. Vol. I. Ann Arbor, Michigan: Ann Arbor Science Publishers, Inc.

The two main methods for leach testing are the column and batch (shake) tests. Currently many variations in procedure are in use in different laboratories (Young and Wilson 1982) and it is therefore often impossible to correlate published results from these tests. The advantages and disadvantages of column and shake tests are compared in Table 6-17. The basic design of these and other leaching tests is shown schematically in Figure 6-18.

Table 6-18 illustrates the results of leaching tests on cement-based fixation products, again using figures provided by the vendor of a proprietary process. The United States EPA has published independent reports assessing a selection of proprietary stabilization/solidification processes. (USEPA 1978).

6.8.5 Design Criteria

Selection of any particular technique for waste fixation must include careful consideration of the containment required, the cost of processing, the increase in bulk of material, and the changes in handling characteristics. The design and location of any landfills that will eventually receive the fixed waste are also major considerations in deciding on the degree of containment and the physical properties which will be required.

The inorganic stabilization/solidification process typically will include the following operations.

- **Collection of Waste:** Some plants may segregate the wastes to be solidified. The blending of the wastes will affect the process chosen for pre-treatment and solidification.

- **Pretreatment of Waste:** Physical pre-treatment required may include decanting, de-watering, evaporation or calcination, cyanide destruction, neutralization, or chromium reduction. Chemical treatment prior to solidification may be required for some processes.

- **Addition of Bulking and Solidification Agents:** Each stabilization/solidification technique will have a requirement for adding the agents. The amount of additives probably will be dictated on the lower end by the specifications required to pass the structural integrity procedure. Other specifications may be required depending upon the end use of the material.

- **Mixing/Packaging Systems:** Solidification is often conducted either in a container or by in-line mixing with containerized or uncontainerized discharge. In-container mixing of the waste and solidification agent can be accomplished with roller mixers, tumbler mixers or kettle mixers (e.g., Figure 6-19. In-line mixing is accomplished either by dynamic or batch mixing. (e.g., Figure 6-20. If containers are used, they must still be inspected, monitored and labelled.

- **Setting and Curing Area:** Uncontainerized releases while still

TABLE 6-17

Advantages and Disadvantages of Column and Batch Tests

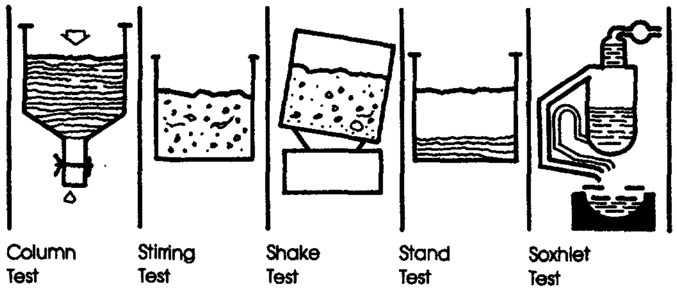
Shake Test

Advantages	Disadvantages
Edge effect in column test is avoided	Do not simulate the leading environment in a landfill site
Time required for test is generally less than column test	Do not measure true leachate but equilibrium concentration
Results are more reproducible	A standard filtering procedure is required
Column Test	
It simulates the waste-leachate content (except around the column) and the slow leachate migration found in landfill	Chanelling and non-uniform packing
	Clogging
Good prediction of the release pattern with time	Biological growth
	Edge effects
	Time requirement
	Reproducibility

Source: Poon, C. S., C. J. Peters, R. Perry. 1983. Youth of stabilization processes in the control of toxic wastes. Effluent and Water Treatment Journal 23 (11) (November): 451-453 and 459.

FIGURE 6-18

Laboratory Leaching Tests



- OBJECTIVE:
  Leachability -Dumping
  -Application in bricks/
  road construction

- PROCEDURE:
  -Pretreatment of waste
  Drying, Milling, Sieving, Homogenising
  -Tests
  with water, synthetic leachate, etc.
  -Analysis
  pH., COD, TOC, Metals, Anions
  (Hard) Organics

- EVALUATION:
  -Concentration
  -Relative leaching
  -Mutual comparison

Source: Stichting Verwijdening Afvalstoffen. 1979. Standard Leaching Tests. SVA/3014/151. Amersfoot.

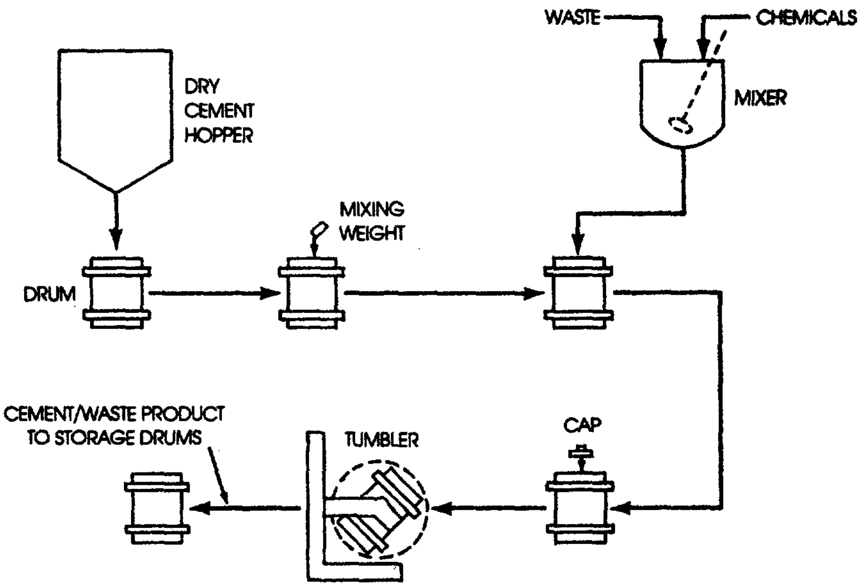
TABLE 6-18

Equilibrium Leaching Tests. Results for a Cement-Based Fixation Process as Claimed by the Process Vendor

Pollutant	Concentration of Pollutant in Waste (ppm)	Concentration of Pollutant in Leachate (ppm)	% Leached
Chromium (Hexavalent)	25,000	0.2	0.03
Copper	25,000	0.26	0.03
Nickel	25,000	0.50	0.055
Zinc	55,150	0.15	0.009
Lead	101,150	0.5	0.016
Cadmium	78,090	0.1	0.042
Manganese	45,590	0.08	0.057
Sulphide	14,660	0.1	0.05
Tin	9,740	0.1	0.03
Arsenic	12,500	0.16	0.03

FIGURE 6-19

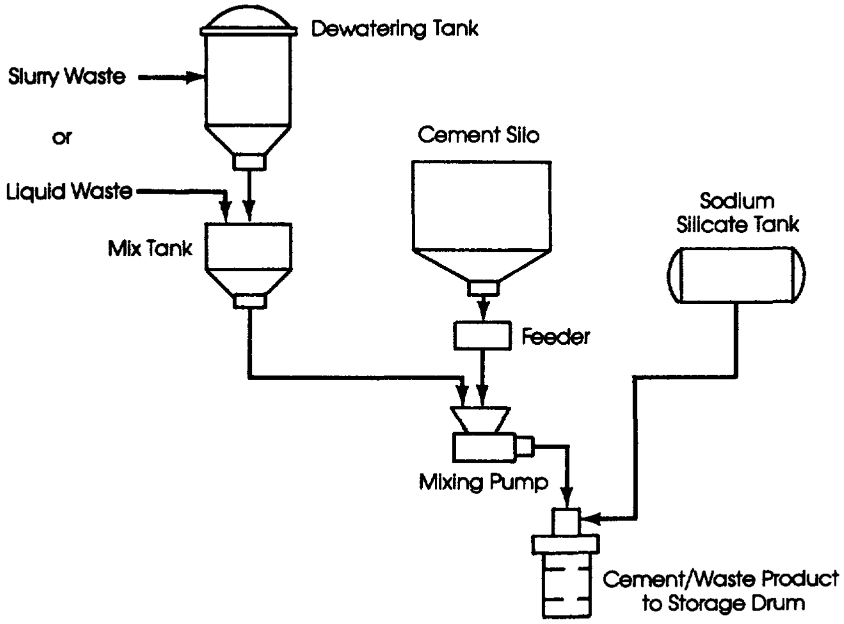
In-Drum Cement Incorporation Process



Source: Pojasek, R.B. 1979. Toxic and Hazardous Waste Disposal Vol. I. Ann Arbor, Michigan: Ann Arbor Science Publishers, Inc.

FIGURE 6-20

External Cement Incorporation Process



Source: Pojasek, R. B. 1979. Toxic and Hazardous Waste Disposal Vol. I. Ann Arbor, Michigan: Ann Arbor Science Publishers, Inc.

pumpable, may be directed to a series of setting and curing lagoons, or may be discharged directly to the final disposal site for in-situ curing. Lagoon areas should be sufficient to accommodate treated waste for at least one week, during which time the hardening process should be complete.

- **Deposition of Cured Waste:** Proximity of the treatment plant to a repository for the fixated waste is a key factor in determining the commercial viability of the process. The final product is bulkier than the original waste; if intermediate lagoons are used, removal of the fixated product by excavators and transportation to the repository can add significantly to the total treatment costs.

6.8.6 Process Considerations

The discussion below is limited to inorganic fixation processes.

(i) Characterization of Waste

The chemical and physical characteristics of the waste must be established. Acidity/alkalinity, metal content, organic contaminants, etc. are all essential information required for a process evaluation.

(ii) Process Evaluation/Quality Control

Laboratory control over all stages of the fixation process is vital. It must be stressed that amounts of chemicals and bulking agents to be used, and the integrity of the final product are empirically determined, and beaker or small-scale tests in the laboratory must be performed on a waste to establish these parameters. Large-scale fixation processes involving discharges into setting lagoons offer no opportunity for remedial action should a waste be incorrectly processed. The most common fault is a product that does not set to the desired level of mechanical strength, resulting in the collapse of lagoon retaining bunds due to the pressure of the slurry.

Most additives used by proprietary processes are patented, and because of commercial competition, companies are reluctant to discuss their techniques and rates and ratios for mixing. Because of this fact, very little data of scientific value is available to the waste producer. A forthcoming document prepared for the UK Department of the Environment will give guidance for a basic fixation process involving the addition of PFA and OPC to the waste.

(iii) Storage/Blending

Storage facilities are required for the raw waste(s) and the additives. Dry materials are stored in silos equipped with transfer gear, weighing facilities and controls. All inorganic fixation processes, whether cement-based or lime-based, require alkaline ($pH>10$) conditions for the final setting and curing stage. There is considerable scope for improvisation during pre-treatment, for example by blending two waste streams to raise pH, thus saving on additives such as lime.

(iv) Mixing

Many forms of mixing can be used, depending on the scale of the process and the physical nature of the additives. For small to medium sized operations, cement mixers are most suitable. Large paddle-stirred cylindrical tanks may also be used. Mixing pumps and other forms of static mixers are suitable for the contacting of slurries or in drum-scale processes (Pojasek 1979).

(v) Setting Lagoons

Treated waste can either be gravity - or screw-discharged into setting lagoons, or pumped while still in slurry form. Lagoons should be sized to provide at least one week's residence time for the waste. Typically, the liquid slurry begins to set within 24 hours. After three days the product is sufficiently solidified to walk on and after about one month the product should support vehicular traffic. The ultimate strength of the product is reached after a period of six months. A mechanical digger removes solidified waste from the lagoons.

Lagoons are generally well drained, clay-lined enclosures, and are provided with leachate collection ditches feeding into a break tank where final chemical adjustments can be made prior to discharge, should the need arise. Leachate monitoring is essential as an indication of the efficacy of the process.

(vi) Complete Systems

Two examples of a drum-scale process for fixating wastes are shown in Figures 6-19 and 6-20.

For large-scale processes treating, say, about 30 tons/day of waste, the basic requirements would be:

- two $10m^3$ stainless steel cement mixers;
- three silos, each of about $20m^3$ capacity, for PFA, cement and lime and equipped with transfer gear, batch weighing mechanisms and controls;
- two $500m^3$ lagoons, each with capacity for one week's supply of treated product;
- two sludge transfer pumps;
- tanks for storing, blending and pre-treating raw wastes and a mechanical digger.

Manning would be at a level of two operators per shift (Pojasek 1979).

A typical sequence of operations is as follows:

- 25 ml of the batch of waste is titrated to pH 7 with pfa and possibly other waste materials; Pfa requirements are noted.

- $3m^3$ - $5m^3$ of the waste is then pumped into the mixer, together with the necessary quantity of pfa. After a mixing period of 10 - 15 minutes, the pH of the mixture is checked for neutrality.

- Cement is added in a quantity equal to about 10% by weight of the amount of waste, and thoroughly mixed in. The alkalinity of the cement raises the pH further. If necessary, cement or lime is added to maintain the final pH above 10.

- After a further 15 minutes of mixing, the pH and consistency are checked. If, from experience, it is deemed too wet, more pfa is added to bulk up the mixture.

- The product is discharged and allowed to harden. Provided that the mixture has been suitably bulked, no water is released during drying, other than a small amount immediately after discharge.

(vii) Maintenance

The corrosive nature of some wastes and the abrasion of the mixing drum from solid additives necessitates periodic drum replacement, the frequency of which is dependent on the extent of use. Typically, a six-monthly replacement program is necessary for a 7 day/week operation.

6.8.7 Economic Considerations

One of the most important advantages of stabilization/solidification techniques is that disposal of the treated waste can take place at a greater number of sites than would otherwise have been the case. This is an obvious advantage in regions where hazardous waste producers are at a financial and practical disadvantage by not being able to dispose of certain wastes through lack of landfill space locally (Poon et al. 1983).

A complete analysis of the costs of treatment must include transportation costs of both the raw and stabilized wastes, the equipment required and labor costs. Costs of the various different systems are given in Table 6-19 (Hill 1981).

Capital and installation costs for inorganic fixation processes depend on the scale and level of sophistication required on the site. For instance, pfa, lime and cement can be stored in bags or 200 litre drums, and the cement mixer fed with the aid of a fork-lift truck. For small-scale operations employing, for example, a single mixer of $10m^3$ total volume (i.e., $6m^3$ working volume), such storage facilities are adequate. Similarly, the treated product can be screw or gravity discharged into well drained skips rather than into lagoons, thus dispensing with the need for a digger and saving on space requirements. Mixers can be driven electrically or with diesel engines; slurry pumps can also be operated electrically or with a diesel-powered air compressor.

For the example of a 30 ton/day facility provided in the previous section, the installed cost is estimated to be in the region of $500,000, with

TABLE 6-19

Economic Considerations for Waste Stabilization/Solidification Systems

Type of Treatment System	Major Materials Required	Unit Cost of Materials (1980)	Amount of Material Required to Treat 100 lbs of Raw Waste	Trends in Price	Equipment Costs	Energy Requirements
Cement-based	Portland cement	\$0.03/lb	50lb-100lb	Stable	Low	Low
Pozzolanic	Lime flyash	\$0.03/lb	50lb-100lb	Stable	Low	Low
Thermoplastic (bitumen-based)	Bitumen drums	\$0.05/lb \$27/drum	100lb 0.8/drum	Related to oil prices	Very high	High
Organic polymer (polyester system)	Polymer catalyst drums	\$0.45/lb \$1.11/lb \$17/drum	43lb of polyester-catalyst mix	Related to oil prices	Very high	High

Source: Hill, R. D., et al. 1981. Four options for hazardous waste disposal. Civil Engineering. ASCE. New York. 51(9) (September): 82-85.

civils and installation costs accounting for 10% of the total. Maintenance is estimated at $50,000/year.

6.8.8 Encapsulation

Certain chemical substances require destruction by incineration at temperatures in excess of 1200°C, to render them non-hazardous. If the volumes for disposal are insufficient to ensure the viable operation of an incinerator operating at these high temperatures, and if alternatives such as cement or lime Kiln are not available.

An ENCAPSULATING process for permanently storing these chemicals in specially strengthened concrete cells may prove suitable.

Chemicals that have been encapsulated include:

PCB (polychlorinated biphenyls)
OCB (orthochlorophenols)
Mercuric compounds
Poison Group 1 Pesticides (DDT, Wafarin)
Capacitors (filled with PCB)

The encapsulation is done on Class 1 landfill sites. Clay is the geologic formation at the base of the site.

The blocks, size 3,2 x 4,6m, are cast on to the clay base. The floor, reinforced with mesh (which forms a complete cage inside the concrete), is cast first with a thickness of 25 - 30 cm. The concrete is allowed to cure for 3 - 4 days and then the block is loaded with the drums/capacitors for encapsulation. Each drum/capacitor has a 100mm space between it and the next unit so that the concrete completely surrounds each unit.

Once the block is loaded and a diagram drawn up of the contents and their relative positions, the wire cage is bent over the top and 28 Map concrete is poured into the cell, filling all the gaps between the drums/capacitors.

When the block is complete it is sealed in the trench with clay.

6.9 Neutralization

6.9.1 Description

Neutralization is the process of adjusting either an acidic or a basic waste stream to a pH near neutrality. Many industries produce effluents that are acidic or alkaline in nature. Neutralization of an excessively acidic or basic waste stream is necessary in a variety of situations, for example:

- Precipitation of dissolved heavy metals.
- Preventing metal corrosion and/or damage to other construction materials.
- Preliminary treatment, allowing effective operation of biological treatment processes.
- Providing neutral pH water for recycle uses and reduce detrimental effects in the receiving water.

Neutralization is also used in oil emulsion breaking and in the control of chemical reaction rates (e.g., chlorination).

Simply, the process of neutralization is the interaction of an acid with a base. The typical properties exhibited by acids in solution are a result of the hydrogen ion concentration, (H^+). Similarly, alkaline (or basic) properties are a result of the hydroxyl ion concentration, (OH^-). In aqueous solutions, acidity and alkalinity are defined with respect to pH, where $pH = - \log (H^+)$, and $pH = 14 - \log (OH^-)$ (at room temperature), respectively. In the strict sense, neutralization is the adjustment of pH to 7, the level at which the concentrations of hydroxyl ion and hydrogen ion are equal. Solutions with excessive hydroxyl ion concentration ($pH>7$) are said to be basic; solutions with excess hydrogen ions ($pH<7$) are acidic. Since adjustment of pH to 7 is not often practical or even desirable in waste treatment, the term "neutralization" is sometimes used to describe adjustment of pH to values near neutrality, usually in the pH range 6-9. A typical neutralization system is shown in Figure 6-21.

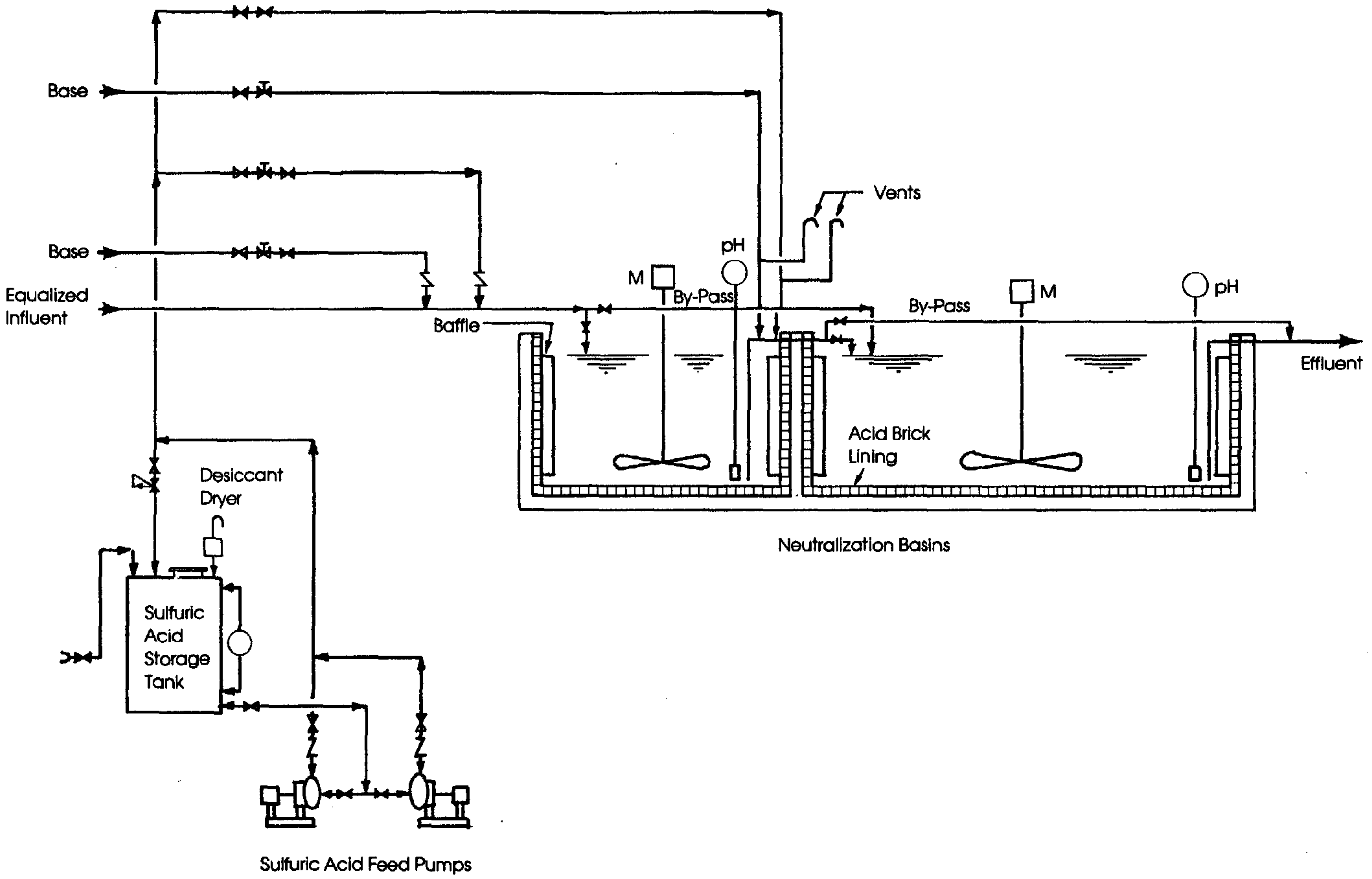
6.9.2 Representative Types and Modifications

There are many acceptable methods of neutralizing overacidity or overalkalinity of wastewaters, such as:

- Mixing acidic and alkaline wastes so the net effect is near-neutral pH;
- Passing acid wastes through beds of limestone;
- Mixing acid wastes with lime slurries;
- Adding concentrated solutions of alkalies (e.g., caustic soda [NaOH] or soda ash [Na_2CO_3]) to acid wastes;

FIGURE 6-21

Process Flow Diagram for Neutralization



- Blowing waste boiler flue gas through alkaline wastes;
- Adding compressed carbon dioxide (CO_2) to alkaline wastes; and
- Adding acid (e.g., sulfuric or hydrochloric) to alkaline wastes.

The method chosen depends upon the wastewater characteristics and subsequent handling or use. For example, mixing of various streams is often insufficient as a preliminary step to biological treatment or sanitary sewer discharge. In this case, supplemental chemical addition is generally required to obtain the proper pH.

The most commonly used chemicals are lime (to raise the pH) and sulfuric acid (to lower the pH). Limestone is the cheapest reagent for acidic wastes and is easy to apply but is ineffective in neutralizing sulfate-bearing wastes because calcium sulfate will precipitate, thus coating the limestone and rendering it inactive. Caustic soda or soda ash are more expensive to use but can be substituted for lime in treating wastes containing sulfates.

If the waste stream is nutrient deficient in either nitrogen or phosphorus, ammonia or trisodium phosphate addition serves the dual purpose of providing both alkalinity and the deficient nutrient (USEPA 1982a).

Mixing of waste streams can be performed in a collection tank, rapid mix tank (where treatment chemicals may also be added), or in an equalization tank. Chemicals can be added in a mix tank or directly to a clarifier. Final pH adjustment in preparation for discharge can be done in a small neutralization tank at the end of the treatment process.

6.9.3 Technology Status

Neutralization is considered to be a demonstrated technology and is widely used in industrial waste treatment.

6.9.4 Applications

Neutralization is widely used as a preliminary treatment or in preparation for discharge, in the following industries:

- Battery Manufacturing,
- Aluminum Forming,
- Coal Mining,
- Inorganic Chemicals Manufacturing,
- Iron and Steel Manufacturing,
- Photographic Equipment and Supplies,
- Explosives Manufacturing,
- Nonferrous Metals Manufacturing,
- Soap and Detergent Manufacturing,
- Pharmaceutical Manufacturing,
- Ore Mining and Dressing,

- Steam Electric Power Plants, and
- Textile Mills.

It is also used on a limited basis in the following industries:

- Auto and Other Laundries,
- Rubber Processing,
- Porcelain Enameling,
- Gum and Wood Chemicals, and
- Paint and Ink Formulation.

In the steelmaking sub-category of the Iron and Steel industry, acid is added to the recycle system blowdown from the basic oxygen furnace (BOF) wet air pollution control system to neutralize the pH of the typically alkaline wastewaters. Facilities in this sub-category also use lime addition to neutralize the typically acidic wastewaters from open hearth furnace operations. If central treatment is practiced, neutralization sometimes is achieved by mixing the acidic scale removal wastes with alkaline wastes from other sources that are compatible for treatment.

In the Gum and Wood Chemicals industry, neutralization may be required to adjust the pH of the waste streams before biological or other treatment can be accomplished. The pH of the effluent ranges from 3 to 9 and must be lowered to less than 3 for oil emulsion breaking and, raised to approximately 9 for metals precipitation.

6.9.5 Advantages and Limitations

Neutralization is a technology with proven effectiveness. Other advantages include automatic control of the process and operation at ambient conditions (15-32°C or 60-90°F).

The major limitation of neutralization is that it is subject to the influence of temperature and the resulting heat effects common to most chemical reactions. In neutralization, the reaction between acid and alkali normally is exothermic (evolves heat), which will raise the temperature of the wastewater stream and may create an undesirable condition. An average value for heat released during neutralization of dilute solutions of strong acids and bases is 13,400 cal/g mole of water formed. By controlling the rate of addition of neutralizing reagent, the heat produced may be dissipated and the temperature increase minimized. For each reaction the final temperature depends on initial reactant temperatures, chemical species participating in the reaction (and their heats of solution and reaction), concentrations of the reactants, and relative quantities of reactants approaching stoichiometric proportions. This can result in boiling and splashing of the solution, and accelerated chemical attack on materials. In most cases, proper planning of the neutralization scheme with respect to concentration of neutralizing agent, rate of addition, reaction time, and equipment design can alleviate the heating problem.

Neutralization will usually show an increased total dissolved solids content due to addition of chemical agents. Anions resulting from neutralization of sulfuric and hydrochloric acids are sulfate and chloride, respectively, which are not considered hazardous, but recommended discharge limits exist based primarily on problems in drinking water. Common cations present after neutralization involving caustic soda and lime (or limestone) are sodium and calcium (possibly magnesium), respectively, which are not toxic and have no recommended discharge limits. However, calcium and magnesium are responsible for water hardness and accompanying scaling problems.

Acidification of streams containing salts, such as sulfide, will produce toxic gases. If there is no satisfactory alternative, the gas must be removed through scrubbing or some other treatment.

6.9.6 Reliability

Neutralization is highly reliable with proper monitoring, control, and proper pre-treatment to control interfering substances.

6.9.7 Chemicals Required

Chemicals used in neutralization are specific to the wastewater being treated. The following chemicals are frequent used:

- lime [CaO or $Ca(OH)_2$],
- limestone ($CaCO_3$),
- caustic soda (Na_2CO_3),
- carbon dioxide (CO_2),
- sulfuric acid (H_2SO_4), and
- hydrochloric acid (HCl).

When treating nutrient deficient wastewaters, the following chemicals can be used:

- ammonium phosphate ($Na_4H_2PO_4$), and
- trisodium phosphate (Na_3PO_4).

The selection of a neutralization chemical depends on such factors as price, availability, and process compatibility. Sulfuric acid is the most common acid used for the neutralization of alkaline waste. It is less costly than hydrochloric acid, but tends to form precipitates with calcium-containing alkaline wastewater. When hydrochloric acid is used for neutralization, the compound formed is soluble. An important consideration in the use of alkaline reagents for neutralization of acidic wastewaters in the "basicity factor," which is the number of grams of calcium oxide equivalent in the neutralization capacity of a particular alkali. Caustic soda has a high basicity factor and high solubility, but is expensive. Slaked lime is less costly but has low to moderate basicity and solubility and forms precipitates with acidic wastewaters containing sulfuric acid, causing disposal and scaling problems. Limestone and soda ash have low to moderate basicity and soda ash has higher solubility than slaked lime. Limestone is the least soluble of these materials.

6.9.8 Residuals Generated

Neutralization may be accompanied by reduction in the concentration of heavy metals if the treatment proceeds to alkaline pH's. This may result in the generation of residuals that can be removed in subsequent operations. Where solid products are formed (as in precipitation of calcium sulfate or heavy metal hydroxides), clarifier/thickeners and filters must be provided; if the precipitate is of sufficient purity, it would be a salable product; otherwise, a disposal scheme must be devised.

6.9.9 Design Criteria

Depending on the volumes of the wastewater, either batch treatment or continuous treatment is used. A continuous system will employ automated control systems to reduce pH fluctuations and increase reaction effectiveness. A control system measures the pH of the solution and controls the addition of a neutralizing agent to maintain the effluent within the acceptable pH limits. The operation of the control system is based on such factors as flow, acid or base strength, and method of adding the neutralizing agent.

The neutralization system can be designed as a single or multiple stage. As a general rule, one stage can be used if the pH of the raw wastewater is between four and ten. Two stages are often required if the pH is as low as two or higher than ten. More than two stages are generally required if the pH is less than two or greater than twelve.

The size of the neutralizing vessel depends on the wastewater volume or flow, reaction time, solubility of the reagent, and the insoluble precipitates formed from the reaction.

6.9.10 Performance

One data sheet from the following industry provides performance data on neutralization:

- Iron and Steel Manufacturing

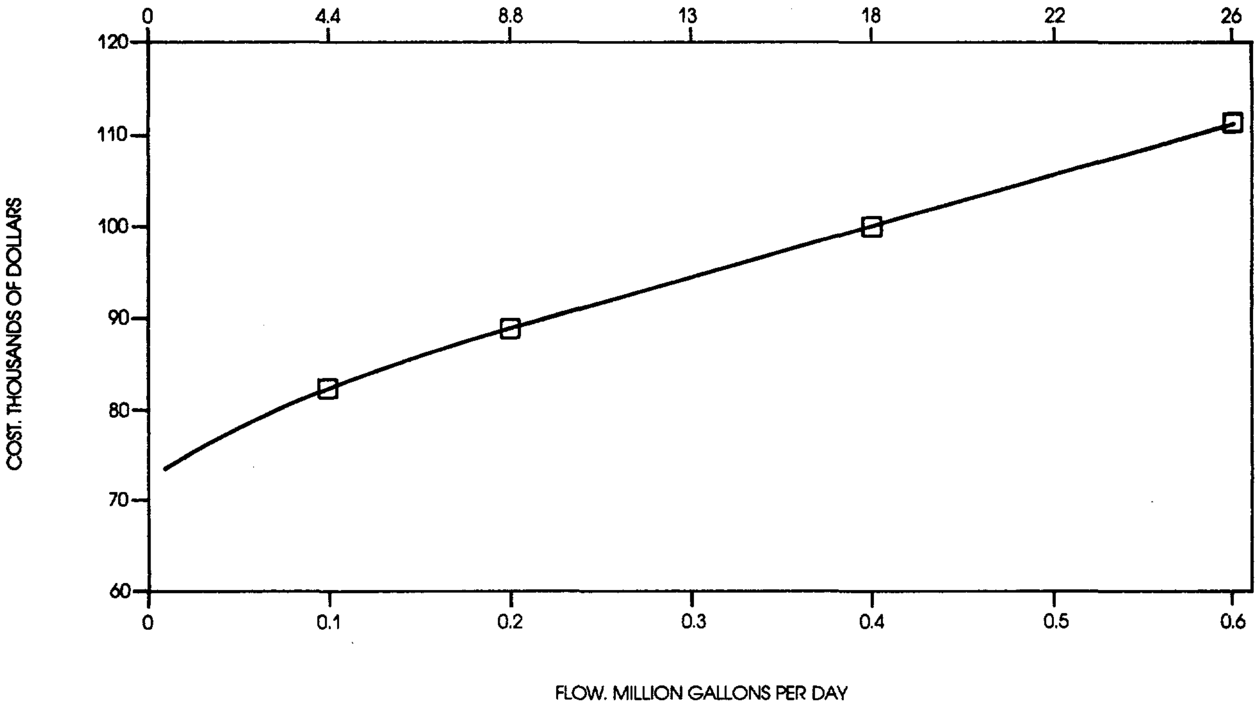
6.9.11 Cost of Neutralization

(A) Basis of Design

This presentation is for the neutralization of acidic or basic wastewater streams by base or acid addition. The system as represented in Figure 6-22 consists of a chemical addition system and a two stage neutralization tank with a design detection time of 5 minutes in the first chamber and 20 minutes in the second. The principal design and cost factor for this technology is wastewater flow. A scale factor is used to adjust for the presence or lack of flow equalization upstream of the unit. Other important factors include influent acidity, alkalinity, pH, TDS, and TSS. Three alternative methods of estimating the neutralization chemical requirements are provided corresponding to the types of information typically available. The preferable method is to base the design dosage

FIGURE 6-22

Capital Cost Estimate for Neutralization (Low Order)



of sulfuric acid or base (slaked lime or caustic soda) required to neutralize the wastewater stream on influent acidity or alkalinity data (in mg/l $CaCO_3$ equivalents). If these data are not available, the required reagent additions may be approximated based on pH data. For streams where no alkalinity, acidity, or pH data are available a standard chemical dose estimate may be used based on best engineering judgment. However, it should be kept in mind that use of these last two methods can introduce considerable error. The neutralization process is assumed to achieve a control to an average pH of 7.0 with a pH range of 6.5 to 8.0.

(1) Source

The unit cost information in this section was derived from the BAT Effluent Limitations Guidelines engineering study for the Organic Chemicals/Plastics and Synthetic Fibers Industries (USEPA 1982a). The method for developing the design factor is based on assumptions and procedures in the Contractor Developed Design and Cost Model (USEPA 1982a).

(2) Required Input Data

Wastewater Flow l/s (mgd)
Alkalinity, acidity (in mg/l $CaCO_3$ equivalents) pH
TDS, TSS (mg/l)

(3) Limitations

None specified.

(4) Pre-treatment

Neutralization is usually preceded by flow equalization except when neutralization is needed first to avoid severe corrosion of downstream units.

(5) Design Equation

Average daily wastewater flow in l/s (mgd) is the primary design and capital cost factor for neutralization systems. The design residence times of the reaction and attenuation chambers are five and 20 minutes, respectively, at 120% of average daily flow. A scale factor is applied to the capital cost estimate if the neutralization unit precedes flow equalization to account for sizing the units for 200% of average daily flow instead of 120%.

(6) Subsequent Treatment

None specified.

(B) Capital Costs

Influent flow is the primary capital cost factor for this unit process. Capital costs can be estimated for neutralization systems less than or equal to 8.76 l/sec (0.2 mgd) in capacity using the low order cost curve (Figure 6-22) and for systems between 8.76 and 876 l/s (0.2 and 20 mgd) in capacity using the high order cost curve (Figure 6-23). A scale factor of 1.67 is applied to the capital cost if the neutralization unit is not preceded by an equalization unit. Costs estimated using these curves must be adjusted to current values using an appropriate current cost index.

(1) Cost Data

Items* included in the capital cost curve estimates are as follows (USEPA 1982a):

(a) Low Order <8.76 l/s, (0.2 mgd)

Mixing tank, fiberglass
Attenuation tank, fiberglass
Acid storage and feed
Agitators (2)
Piping, electrical
Instrumentation

(b) High Order, 8.76 to 876 l/s (0.2 to 20 mgd)

Mixing tank, concrete, acid brick lined
Attenuation tank, acid brick lined
Acid storage and feed
Agitators (2)
Piping, electrical
Instrumentation

*Note that the lime or caustic handling and feed equipment is designed to serve the entire plant's needs and is sized and costed separately.

(2) Capital Cost Curves

(a) Low Order Curve - See Figure 6-22

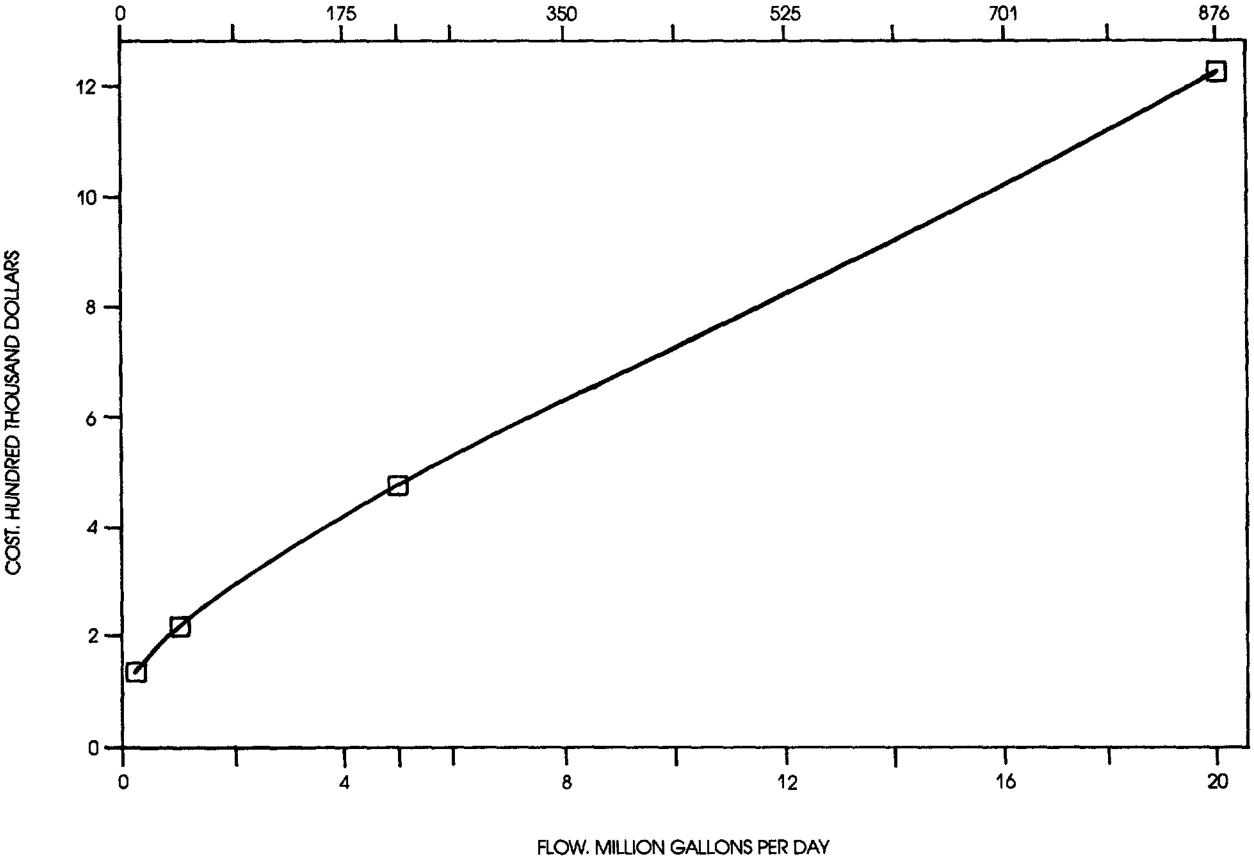
- Cost (thousands of dollars) vs. flow (liters per second or million gallons per day)
- Curve basis, cost estimates on four systems with flow rates of 4.38, 8.76, 17.5, and 26.3 l/s (0.1, 0.2, 0.4, and 0.6 mgd)

(b) High Order Curve - See Figure 6-23

- Cost (hundred thousand dollars) vs. flow (liters per second or million gallons per day)
- Curve basis, cost estimates on four systems with flow rates of 8.76, 43.8, 219, and 876 l/s (0.2, 1.0, 5.0, and 20.0 mgd)

FIGURE 6-23

Capital Cost Estimate for Neutralization (High Order)



(c) Scale Factor - If neutralization is not preceded by equalization, a scale factor of 1.67 is applied to standard capital cost.

(3) Cost Index

Base Period, July 1977, St. Louis
Chemical Engineering (CE) Plant Index = 204.7

(C) Operation and Maintenance Costs

Operating costs include both fixed and variable components. The variable components of operating cost are power and chemical costs. Fixed operating costs include labor, supervision, overhead, laboratory labor, maintenance, services, insurance and taxes, and service water. All fixed and variable operating costs should be adjusted to current levels using an appropriate index or unit cost factor.

(1) Variable Costs

(a) Power Requirements, Low Order (Flow <8.76 l/s (0.2 mgd)

- pumps, agitators (USEPA 1982a). These equations were developed using regression analysis procedures.

Metric

$$KW = (0.55 \times FLOW) + 0.657$$

where: KW = power requirement, kilowatts
FLOW = influent flow, l/s

English

$$HP = (32.3 \times FLOW) + 0.881$$

where: HP = power requirement, Hp
FLOW = influent flow, mgd

(b) Power requirements, High Order (Flow 8.76 to 876 l/s (0.2 to 20 mgd) (USEPA 1982a). These equations were developed using regression analysis procedures.

Metric

$$KW = (0.266 \times FLOW) + 6.49$$

where: KW = power requirement, kilowatts
FLOW = influent flow, l/s

English

$$HP = (15.6 \times FLOW) + 8.70$$

where: HP = power requirement, Hp
FLOW = influent flow, mgd

(c) Power Cost

Metric

$$PC = KW \times 24 \times EC$$

where: PC = power cost, $/day
KW = power required, kilowatts
24 = hours/day
EC = electricity cost, $/Kw-hr

English

$$PC = HP \times 24 \times 0.746 \times EC$$

where: PC = power cost, $/day
HP = horsepower required, Hp
24 = hours/day
0.746 = Kw-hr/Hp-hr
EC = electricity cost, $/Kw-hr

(d) Chemical Requirements

The chemical requirements for neutralization may be estimated in one of three ways depending on the influent waste water quality data available. The preferred method is to use acidity/alkalinity data (Case I), but methods using only pH data (Case II) or a standard dose (Case III) can be used.

o CASE I - Influent Acidity and Alkalinity Data Available:

- If both acidity and alkalinity are present in the influent, determine the dominant characteristic.

$$A = AP - (a - 2)$$

where: A = modified dominant acidity or alkalinity, mg/l ($CaCO_3$ equivalents)
AP = influent measured dominant acidity or alkalinity factor, mg/l
a = influent concentration of other factor, mg/l

This modified alkalinity or acidity should be used in subsequent calculations where applicable in place of the dominant influent value.

1) Lime and topping acid requirements based on acidity

If the influent wastewater dominant characteristic is acidic, lime is added to neutralize the acid and topping acid is added to cover minor acidity fluctuations.

Metric

LIME = 0.74 x AC x FLOW x 0.086

where: LIME = lime requirements, Kg/day
0.74 = stoichiometric ratio of $Ca(OH)_2$ to $CaCO_3$
AC = modified influent acidity, mg/l ($CaCO_3$ equivalents)
FLOW = influent flow, l/s
0.086 = conversion factor

English

LIME = 0.74 x AC x FLOW x 8.34

where: LIME = lime requirements, lb/day
FLOW = influent flow, mgd
8.34 = conversion factor

Topping acid requirements are based on modified influent acidity as follows:

Metric

TA = ADOSE x FLOW x 0.086

where: TA = topping acid, lb/day
ADOSE = acid dose, mg/l (from above table)
FLOW = influent flow, mgd

2) Acid and topping lime requirements based on alkalinity

If the influent wastewater is predominantly alkaline, sulfuric acid is added to neutralize the waste and topping lime is added to cover minor alkalinity fluctuations.

Metric

ACID = 0.98 x ALK x FLOW x 0.086

where: ACID = acid (H_2SO_4) requirements, Kg/day
0.98 = stoichiometric ratio of H_2SO_4 to $CaCO_3$ equivalents
ALK = modified influent alkalinity, mg/l, ($CaCO_3$ equivalents)
FLOW = influent flow, l/s
0.086 = conversion factor

English

ACID = 0.98 x ALK x FLOW x 8.34

where: ACID = acid (H_2SO_4) requirement, lb/day
FLOW = influent flow, mgd
8.34 = conversion factor

Topping lime requirements are based on modified influent alkalinity as follows:

Metric

TL = LDOSE x FLOW x 0.086

where: TL = topping lime requirement, Kg/day
FLOW = influent flow, l/s
ADOSE = lime dose, mg/l
(LDOSE determined from the following table)

Alkalinity (mg/l)	Topping Lime Dose (LDOSE) (mg/l)
ALK > 150	50
100 < ALK > 150	200 - ALK
ALK < 100	100

English

TL = LDOSE x FLOW x 8.34

where: TL = topping lime requirement, lb/day
LDOSE = lime dose, mg/l (from above table)
FLOW = influent flow, mgd

o CASE II - Only pH Data Available

If influent alkalinity and acidity data are not available, the lime and acid requirements for a neutralization system may be estimated based on the following influent pH ranges. Estimates derived using this method should be scrutinized for reasonableness; particularly when dealing with highly buffered wastewaters.

1) If (low pH) >7.0, then acid and topping lime are required:

ACIDC = [(low pH) - 7.0]2 x 20 or
50 mg/l whichever is larger and
TLC = 50 mg/l

where:
ACIDC = acid (H2SO4) requirement, mg/l
(low pH) = minimum influent pH value

TLC = topping lime requirement, mg/l

2) If (low pH) <7.0 and (avg pH) >7.0, then lime and topping acid are required:

LIMEC = [7.0 - (low pH)]3 x 20 or
50 mg/l whichever is larger and
TAC = [[((avg pH) + (high pH)) - 2] - 7]2 x 20 or
50 mg/l whichever is greater

where: LIMEC = lime requirement, mg/l
(avg pH) = average influent pH value
high pH) = highest influent pH value
TAC = topping acid requirement, mg/l

3) If (low pH) <7.0 and (avg pH) <7.0 and (high pH) >7.0, then lime and topping acid are required:

LIMEC = [7.0 - [((avg pH) + (low pH)) - 2]]3 x 20 or
50 mg/l whichever is greater; and
TAC = [(high pH) - 7.0]2 x 20 or
50 mg/l whichever is greater

4) If (low pH) <7.0 and (avg pH) <7.0 and (high pH) <7.0, then lime only is required:

LIMEC = [7.0 - (avg pH)]3 x 20 or
100 mg/l whichever is greater; and
TAC = 0

5) To convert chemical requirements to daily weight basis;

Metric

ACID = ACIDC x FLOW x 0.086
TL = TLC x FLOW x 0.086
LIME = LIMEC x FLOW x 0.086

where: ACID = acid required, Kg/day
TL = topping lime required, Kg/day
LIME = lime required, Kg/day
TA = topping acid required, Kg/day
FLOW = influent flow, L/s
0.086 = conversion factor

English

ACID = ACIDC x FLOW x 8.34
TL = TLC x FLOW x 8.34
LIME = LIMEC x FLOW x 8.34
TA = TAC x FLOW x 8.34

where: ACID = acid required, lb/day
TL = topping lime, lb/day
LIME = Lime required, lb/day
TA = topping and, lb/day
FLOW = influent flow, mgd
8.34 = conversion factor

- CASE III - No Data Available

For streams where no pH, acidity, or alkalinity data are available, a standard dose of 100 mg/l of acid and 100 mg/l of lime may be assumed. These additions are considered suitable to neutralize occasional pH swings (USEPA 1982a). For streams of an essentially neutral pH, a minimum standard dose of 50 mg/l of acid and 50 mg/l of lime may be used.

Metric

LIME = SDL x FLOW x 0.086

where: LIME = lime required, Kg/day
SDL = standard dose of lime, mg/l
(100 mg/l or 50 mg/l minimum)
FLOW = influent flow, l/s
0.086 = conversion factor
ACID = SDA x FLOW x 0.086

where: ACID = acid required, Kg/day
SDA = standard dose of acid, mg/l
(100 mg/l or 50 mg/l minimum)

English

LIME = SDL x FLOW x 8.34

where: LIME = lime required, lb/day
FLOW = influent flow, mgd
8.34 = conversion factor
ACID = SDA x FLOW x 8.34

(e) Chemical Costs (except lime*)

AC = ACID x N

where: AC = acid cost, $/day
ACID = acid requirement, lb/day
N = unit cost of sulfuric acid, $/lb

*Cost for lime is based on total plant needs rather than on the needs of an individual unit process. Lime requirements should be accounted for but costs for handling systems and lime should be estimated separately after design of all unit processes requiring lime.

(2) Fixed Costs

The fixed O & M components of this technology are listed in Table 6-20 including the cost basis and the unit costs the Model (USEPA 1982a).

(D) Miscellaneous Costs

Costs for engineering, and common plant items such as land, piping, and buildings, are calculated after completion of costing for individual units.

(E) Modifications

The effluent stream may be adjusted to account for changes in total dissolved solids (TDS) and TSS which result from the neutralization process. TDS is expected to change as a result of additions of acid and lime. If both sulfate and calcium are present in the wastewater and additional amounts are added during neutralization, additional TSS may be formed as the solution reaches the solubility limit for calcium and sulfate. The formation of TSS from the wastewater is of some interest for cost considerations since it could affect the volume of sludge which would eventually be collected and disposed of in subsequent unit processes.

(1) TDS Increase due to Neutralization

Metric

$$\text{TDSE} = \text{TDSI} + [[\text{LIME} \times (40 - 74) + \text{ACID} \times (96 - 98)] - (\text{FLOW} \times 0.086)]$$

where: TDSE = average effluent TDS, mg/l
TDSI = average influent TDS, mg/l
LIME = lime added, Kg/day
40 - 74 = mass ratio of Ca to $Ca(OH)_2$

ACID = acid added, Kg/day
96 - 98 = mass ratio of SO4 to H_2SO_4
FLOW = influent flow, l/s
0.086 = conversion factor

English

$$\text{TDSE} = \text{TDSI} + [[\text{LIME} \times (40 - 74) + \text{ACID} \times (96 - 98)] - (\text{FLOW} \times 8.34)]$$

where: LIME = lime added, lb/day
ACID = acid added, lb/day
FLOW = influent flow, mgd
8.34 = conversion factor

TABLE 6-20

Fixed O & M Cost Basis and Unit Cost Factors For Neutralization

Element	Cost Basis (Equivalent Unit Quantity)	Base Unit Cost
Labor (1,2)	0.20 Weeks (4.80 hrs/day)	$ 9.80/hr
Supervision (1)	10% Labor (.48 hrs/day)	$ 11.76/hr
Overhead (1)	75% Labor Cost	NA
Laboratory (3)	0.10 Shifts (0.57 hrs/day)	$ 10.70/hr
Maintenance	2.50% Capital	NA
Services	0.40% Capital	NA
Insurance & Taxes	2.50% Capital	NA
Service Water	0.00 L/s (0.00 Thou gpd)	$ 0.13/thou L ($ 0.50/thou gal)

NA - not applicable

(1) Labor may vary from 0.7 to 1.2 times the standard amount indicated depending on the overall scale of the plant. Labor, Supervision, and Overhead may be adjusted for the scale of the plant.

(2) One week = 7 days = 168 hours = 4.2 shifts

(3) One shift = 40 hours

Source: USEPA. 1982a. Treatability Manual, EPA - 600/2-82-001. Stock No. 055-000-00215-1 (4 volumes). Washington, D.C.: Unites States Government Printing Office.

(2) TSS Increase due to Neutralization

If calcium, sulfate, and carbonate are present in the wastewater, then additional suspended solids may be produced (USEPA 1982). The user should check first to determine if calcium sulfate may be generated (Step 1), and then check for calcium carbonate generation (Step 2) (USEPA 1982).

(a) Step 1. If calcium and sulfate are present in the influent in excess of the triggering values (1000 and 2000 mg/l respectively are used to trigger the need for this modification), the effluent TSS is calculated as follows:

Metric

TSSE = (CAL + SUL - 2500) + TSSI

where:
TSSE = effluent TSS, mg/l
CAL = total calcium dissolved solids, mg/l
= [LIME - (FLOW x 0.086)] x (40 - 74) + CALI
LIME = lime added, Kg/day
FLOW = influent flow, l/s
0.086 = conversion factor
CALI = influent calcium dissolved solids, mg/l
40 - 74 = mass ratio of Ca to $Ca(OH)_2$
SUL = total sulfate dissolved solids, mg/l
= [ACID - (FLOW x 0.086)] x (96 - 98) + SULI
ACID = acid requirement, Kg/day
SULI = influent sulfate dissolved solids, mg/l
96 - 98 = mass ratio of SO_4 to H_2SO_4
2500 = solubility limit of calcium sulfate, mg/l
= 800 mg/l calcium plus 1700 mg/L sulfate, (USEPA 1982)
TSSI = influent TSS, mg/l

English

TSSE = (CAL + SUL - 2500) + TSSI

where: CAL = total calcium dissolved solids, mg/l
= [LIME - (FLOW x 8.34)] x (40 - 74) + CALI
LIME = Lime added, lb/day
FLOW = influent flow, mgd
SUL = total sulfate dissolved solids, mg/l
= [ACID - (FLOW x 8.34)] x (96 - 98) + SULI

Note that the effluent values of calcium and sulfate may be set at their solubility limits after computing the TSS increase.

TSSI = influent TSS, mg/l
0.086 = conversion factor

English

TSSE = (CAL + SUL - 2500) + TSSI

where: CAL = total calcium dissolved solids, mg/l
= [LIME - (FLOW x 8.34)] x (40 - 74) + CALI
LIME = Lime added, lb/day
FLOW = influent flow, mgd
SUL = total sulfate dissolved solids, mg/l
= [ACID - (FLOW x 8.34)] x (96 - 98) + SULI

Note that the effluent values of calcium and sulfate may be set at their solubility limits after computing the TSS increase.

(b) Step 2. If calcium >200 mg/l and carbonate >200 mg/l and no sulfate:

Metric

TSSE = effluent TSS, mg/l
CARI = influent carbonate dissolved solids, mg/l
CAL = total calcium dissolved solids, mg/l
= [LIME - (FLOW x 0.086)] x (40 -74) + CALI
LIME = lime added, Kg/day
FLOW = influent flow, l/s
0.086 = conversion factor
CALI = influent calcium dissolved solids, mg/l
200 = solubility limit for calcium carbonate, mg/l
TSSI = influent TSS, mg/l

English

TSSE = (CARI + CAL - 200) + TSSI

where:
TSSE = effluent TSS, mg/l
CAL = total calcium dissolved solids, mg/l
= [LIME - (FLOW x 8.34)] x (40 - 74) + CALI
LIME = lime added, lb/day
FLOW = influent flow, mgd
8.34 = conversion factor

6.10 Solvent Extraction

6.10.1 Description

Solvent extraction, also referred to as liquid-liquid extraction, is the separation of the constituents of a liquid solution by contact with another immiscible liquid for which the impurities have a high affinity. The separation can be based either on physical differences that affect differential solubility between solvents or on a definite chemical reaction.

The solvent extraction process is shown schematically in Figure 6-24. The diagram shows a single solvent extraction unit operating on an aqueous stream; in practice this unit might consist of (1) a single-stage mixing and settling unit, (2) several mixers and settlers (single-stage unit) in series, or (3) extractor device (e.g., a column or differential centrifuge).

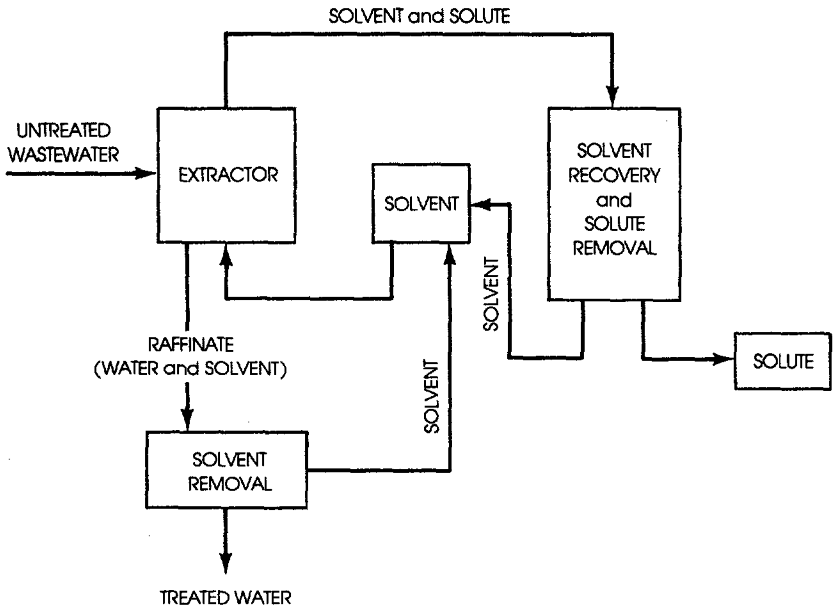
As the flow diagram indicates, reuse of the extracting solvent following solute removal) and recovery of that portion of the extracting solvent that dissolves in the extracted phase are usually necessary aspects of the solvent extraction process. Solvent reuse is necessary for economic reasons as the cost of the solvent is generally too high to consider disposal after use. Only in a very few cases may solvent reuse be eliminated. These cases arise where an industrial chemical feed stream can be used as the solvent and then sent on for normal processing, or where water is the solvent. Solvent recovery from extracted water may be eliminated in cases where the concentration in the water to be discharged is not harmful, and where the solvent loss does not represent a high cost.

The end result of solvent extraction is to separate the original solution into two streams: a treated stream (the raffinate), and a recovered solute stream (which may contain small amounts of water and solvent). Solvent extraction may thus be considered a recovery process since the solute chemicals are generally recovered for reuse, resale, or further treatment and disposal. A process for extracting a solute from solution will typically include three basic steps: (1) the actual extraction, (2) solvent recovery from the raffinate (treated stream), and (3) solute removal from the extracting solvent. The process may be operated continuously.

The first step, extraction, brings two liquid phases (feed and solvent) into intimate contact to allow transfer of solute from feed to solvent. An extractor unit can be a mixer-settler device in which feed and solvent are mixed by agitation, allowed to settle and separate into two liquid streams; or it can be a column in which two liquids are brought into contact by countercurrent flow caused by density difference. The process yields two streams, the cleaned stream or raffinate and the extract or solute-laden solvent stream. Both streams will contain extraction solvent and may require further processing to remove and/or to recover the solvent and solute. The treated stream or raffinate may require a solvent removal process if the solvent losses would add significantly to the cost of the process, or cause a problem with the discharge of the raffinate. Solvent removal may be accomplished by stripping, distillation or adsorption. The extract or solute laden stream may be processed to recover solvent and remove solute. The solute removal and solvent recovery can be via a second solvent extraction step, distillation, or some other process. For example, a second extraction, with caustic, is sometimes used to extract phenol from light oil, which is used as the primary solvent in dephenolizing coke plant wastewaters (USEPA 1982a). Distillation will usually be more common, except where problems with azeotropes are present. In certain cases, it may be possible to use the solute-laden solvent as a feed stream in an industrial process, thus eliminating solute

FIGURE 6-24

Schematic of Extraction Process



recovery. This is apparently the case at some refineries where crude or light oil can be used as a solvent (for phenol removal from water) and later processed with the solute in it. This application is particularly attractive since it eliminates one costly step.

Solvent extraction should be regarded as a process for treating concentrated, selected, and segregated wastewater streams primarily where material recovery is possible to offset process costs. Solvent extraction, when carried out on the more concentrated waste streams, will seldom produce a treat effluent (the raffinate) that can be directly discharged to surface waters; some form of final polishing will usually be needed. Solvent extraction cannot compete economically with biological oxidation or adsorption in the treatment of large quantities of very dilute wastes, and it will have trouble competing with steam stripping in the recovery of volatile solutes present in moderate to low concentrations.

6.10.2 Technology Status

Solvent extraction is a proven method for the recovery of organics from liquid solutions and may be the process of choice in some cases.

6.10.3 Representative Types and Modifications

There are two major categories of equipment for liquid extraction: single-stage and multi-stage equipment.

In single-stage equipment, the fluids are mixed, extraction occurs, and the insoluble liquids are settled and separated. A cascade of such stages may then be arranged. A single-stage unit must provide facilities for mixing the insoluble liquids and for settling and decanting the resulting emulsion or dispersion. In batch operation, mixing together with settling and decanting may take place in the same or in separate vessels. In continuous operation, different vessels are required.

In multi-stage equipment, the equivalent of many stages may be incorporated into a single device or apparatus. Countercurrent flow is produced by virtue of the difference in densities of the liquids, and with few exceptions, the equipment takes the form of a vertical tower which may or may not contain internal devices to influence the flow pattern. Other forms include centrifuges, rotating discs, and rotating buckets. Depending upon the nature of the internal structure, the equipment may be of the stagewise or continuous-contact type.

Commonly used solvents include crude oil, light oil, benzene, and toluene, Less common but more selective solvents include chloroform, ethylacetate, isopropyl ether, tricresyl phosphate, isobutyl ketone, methylene chloride, and butyl acetate. When crude oil or light oil is used, phenol is destroyed in downstream operations. Alternatively, extraction with light oil may be followed by phenol recovery via extraction of the oil with caustic. In this case, phenol is recovered as sodium phenolate.

6.10.4 Applications

Solvent extraction is presently applied in two main areas: (1) the recovery of phenol from aqueous wastes, and (2) the recovery of halogenated hydrocarbon solvents from organic solutions containing other water-soluble components. Solvent extraction is currently being used in the following industries, the major application is the extraction of phenolic materials from wastewater.

- Iron and Steel Manufacturing,
- Organic Chemicals Manufacturing, and
- Petroleum Refining.

The cokemaking sub-category of the Iron and Steel industry utilizes solvent extraction to dephenolize one waste stream. In the process, the benzene light oil, or other suitable solvent, extracts phenolic compounds from the wastewater. The phenolized solvent is then separated and extracted with caustic. Sodium phenolates separate out, and the dephenolized solvent is reused in the recovery system (USEPA 1982a).

Other applications of solvent extraction are briefly described below (USEPA 1982a).

- Extraction of thiazole-based chemicals from rubber processing effluent with benzene.

- Extraction of salicylic and other hydroxy-aromatic acids from wastewaters using methyl isobutyl ketone as the solvent.

- Deoiling of quench waters from petroleum operations via solvent extraction has been developed by Gulf Oil Corporation. Quench water containing about 6,000 mg/l of dissolved and emulsified oil is extracted with a light aromatic oil solvent, and the extract is recycled for refinery processing. Additional treatment of the water (e.g., via coalescence) is necessary for water reuse. It is not known if this process is in current use.

- Recovery of acetic acid from industrial wastewater is proposed to handle wastewaters that may contain acetic acid levels of 0.5% to over 5%. The extractant is a solution of trioctylophosphine oxide in a carrier solvent. This process is currently in the developmental stage, but has been demonstrated to be practical.

- Solvent recovery via solvent extraction is carried out in at least one hazardous waste management facility in Lowell, Massachusetts.

6.10.5 Advantages and Limitations

The main advantage of solvent extraction is its use as a recycle technique. Valuable solvents can be recovered for reuse in the process stream of an industry. There are relatively few insurmountable technical problems with solvent extraction. The most difficult problem is usually finding a solvent that best meets a long list of desired qualities including low cost, high extraction efficiency, low solubility in the raffinate, easy separation from the solute, adequate density difference with raffinate, no tendency for emulsion formation, non-reactive, and non-hazardous. No one solvent will meet all the desired criteria and, thus, compromise is necessary. There is a wide range of extraction equipment available today, and space requirements are not a problem.

Process cost is always a determining factor with solvent extraction, and has thus far limited actual application to situations where a valuable product is recovered in sufficient quantity to offset extraction costs. These costs will be relatively small when a single-stage extraction unit can be used (e.g., simple mixer-settler) and where solvent and solute recovery can be carried out efficiently. In certain cases, the process may yield a profit when credit for recovered material is taken. Any extraction requiring more than the equivalent of about ten theoretical stages may require custom-designed equipment and will be quite expensive.

Even if solvent recovery operations are utilized, the wastewater that remains after the solvents have been separated will still contain small amounts of these materials. Only one treatment technology has been demonstrated to be effective in complete solvent removal: steam stripping. In addition, solvent extraction systems seldom produce a raffinate that is suitable for direct discharge to surface waters and thus, a polishing treatment is generally required (e.g., biological treatment).

6.10.6 Reliability

Solvent extraction is highly reliable for proven applications, if properly operated.

6.10.7 Chemicals Required

Chemical solvents that are used include benzene (benzol), toluene, chloroform, ethylacetate, isopropyl ether, tricresyl phosphate, methyl isobutyl ketone, methylene chloride, and butyl acetate.

6.10.8 Residuals Generated

Solvent extraction generates no solid wastes. When mixed organic liquids are treated principally for the recovery of just one component, (e.g., the more valuable halogenated hydrocarbons), economics may make the purification of the other components (as required for resale or reuse) impractical resulting in a waste for disposal.

6.10.9 Design Criteria

Design is specific to the solute being recovered and the waste stream characteristics. The major design parameters are the choice of solvent, distribution coefficient, and wastewater flow rate.

6.10.10 Performance

Extraction reduces phenol concentrations from levels of several percent down to levels of a few parts per million. Removal efficiencies of 90 to 98% are possible in most applications, and with special equipment (e.g., centrifugal and rotating disc contactors) removal efficiencies of about 99% have been achieved. Subsequent data sheets provide performance data on the following industries:

- Petroleum Refining, and
- Organic Chemicals Manufacturing

(See Table 6-21).

6.11 Sludge Processing

Improvements in industrial wastewater treatment, unfortunately, are often accompanied by the production of increasing quantities of increasingly difficult-to-handle sludges. Sludges withdrawn from primary or simple sedimentation are as much as 97% water. Biological and many chemical sludges have higher water contents and are much more difficult to dewater than primary sludges. The cost of combined municipal and industrial sludge handling and disposal is often greater than the cost of treating the wastewater itself.

Although a large number of alternative combinations of equipment and processes are used for treating sludges, the basic alternatives are fairly limited. The ultimate depository of the materials contained in the sludge must either be land, air, or water. There are two basically different philosophical approaches in handling the sludges from wastewater treatment: reuse as opposed to disposal. The reuse approach is based upon recycling the sludges so that nutrients and organics contained in the sludges are beneficially reused. This is discussed in detail in the land disposal chapter, Section 7.2. The goal of sludge treatment in this case is to make the sludge compatible with the proposed reuse system (i.e., stabilize the sludge so that it will not cause nuisance conditions, eliminate pathogens to prevent disease problems, degrade toxic organics, etc.). The disposal philosophy considers the sludge a waste material. In some cases, such as ocean dumping, only limited pre-treatment of sludge is needed prior to disposal. However, most disposal systems incorporate treatment techniques to provide maximum reductions in volume and/or mass of sludge prior to disposal with little or no regard for the potentially beneficial components of sludge. Figure 6-25 summarizes general sludge handling alternatives.

Technologies discussed in this section include sludge conditioning, digestion, composting, thickening, and dewatering.

Note: The sludges produced by processing will require careful disposal as they may contain large concentrations of metals or precipitated toxic substances. See Chapter 7 for a discussion of disposal technologies.

TABLE 6-21

Control Technology Summary for Solvent Extraction

	Data Points		Effluent Concentration		Removal Efficiency, %	
Pollutant	Pilot Scale	Full Scale	Range		Range	Median
Classical pollutants, mg/L:						
COD	6		700 - 19,000	4,200	37 - 78	68
Chlorine, total	2		35 - 170	100	87 - 94	90
TOC	1			54		31
TOD	2		8,300 - 22,000	15,000		78
Toxic pollutants, ug/L:						
Xylene	3	1	<1,000 - 25,000	<1,000	96 - >97	>97
Phenol	12	1	77 - 9.6E6	66,000	3 - >99	72
Benzene	6	1	2,400 - 12,000	9,200	58 - 97	96
Ethylbenzene	1	1	4,000 - 4,400	4,200		97
Toluene	2	1	1,600 - 10,000	2,300	94 - 96	95
Styrene	1			<1,000		>93
Acetone	7		7,000 - 1.4E6	22,000	12 - 82	51
MEK	10		1,900 - 300,000	9,000	32 -99	64
1,2-Dichloroethane	2		<20,000 - 97,000	59,000	84 - >99	>92
1,1,2,2-Tetrachloroethane	1			4,200		91
1,1,2-Trichloroethane	1			16,000		90
Antimony		1		41		NM
Arsenic		1		140		48
Cyanide		1		16,000		27
Selenium		1		630		NM
Silver		1		<25		NM
Zinc		1		120		50
Total phthalates		1		1,900		5
1,2-Diphenylhydrazine		1		3,000		36
2,4,6-Trichlorophenol		1		ND		>99
2,4-Dimethylphenol		1		ND		>99
p-Chloro-m-cresol		1		ND		>99
4,6-Dinitro-o-cresol		1		ND		>99
o-cresol	9		2.3 - 330,000	4,000	83 ->99	93
m,p-cresol	1			25,000		91
Benzo (a) anthracene		1		ND		NM

TABLE 6-21 (continued)

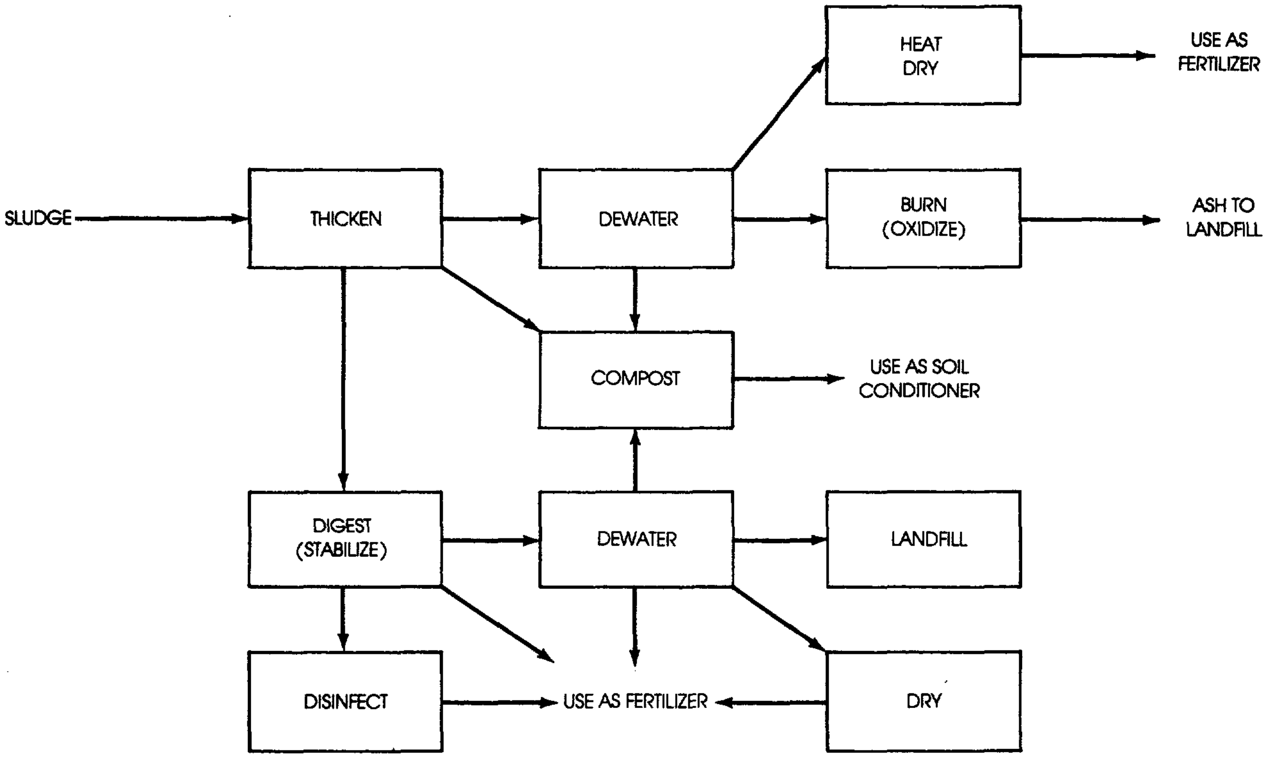
Control Technology Summary for Solvent Extraction

	Data Points		Effluent Concentration		Removal Efficiency, %	
Pollutant	Pilot Scale	Full Scale	Range		Range	Median
Benzo (a) pyrene		1		13		98
Acenaphthylene		1		1,600		50
Chrysene		1		95		67
Fluoranthene		1		500		49
Fluorene		1		190		75
Napthalene		1		5,900		NM
Phenanthrene		1		280		66
Chloroform		1		ND		NM
Methyl isobutyl ketone	1			4,000		97
Methanol	6		19,000 - 400,000	52,000	2 - 64	12
Ethanol	6		36,000 - 450,000	170,000	10 - 93	19
Isopropanol	1			3,000		99
n-Propanol	1			5,000		88
Isobutyraldehyde	1			16,000		88
2-Butanol	5		4,000 - 190,000	60,000	59- 99	75
n-Butanol	1			26,000		63
Monochloroacetaldehyde	2		290,000 - 4.3E6	2.3E6	10 - 96	53
Dichloroacetaldehyde, chloral, crotanaldehyde	1			3.0E6		45
Acentonitrile	2		560,000 - 650,000	600,000	10 - 25	18
1-Butanol	2		8,000 - 68,000	38,000	83 - 99	91
n-Propyl MEK	1			250,000		36
Benzene + other	1			ND		>99
Acetaldehyde	5		4,000 - 1.1E6	150,000	15 - 97	29
Acetic acid	7		110,000 - 1.4E7	2.7E6	8 - 91	77
Acetate	1			1.9E6		NM
n-Butyl acetate	1			7.3E6		NM
Formic acid	5		ND - 860,000	170,000	55 - >99	80
Propionic acid	1			23,000		71

Blanks indicate data not available.
ND = not detected.
NM = not meaningful.

FIGURE 6-25

Basic Sludge-Handling Alternatives



6.11.1 Conditioning

The usual purpose of sludge conditioning is to increase the rate and/or extent of dewatering achievable for a given sludge. A wide variety of physical and chemical techniques are used. The use of sludge conditioning prior to dewatering has become standard practice, and the combination of conditioning and dewatering reduces the moisture content of sludges from a range of 95 to 98% to one of 60-75 %. The most frequently encountered conditioning practices employ the addition of chemicals. In one method ferric chloride is added either alone or in combination with lime; while in another organic polyelectrolytes are added. The addition of lime by itself is a fairly popular conditioner for raw primary sludge, and ferric chloride alone has been used for conditioning biological sludges. Lime treatment to a high pH value of 12.0 has the added advantage of providing a significant degree of sludge disinfection.

Organic polymeric coagulants and coagulant aids are of three basic types:

(1) Anionic (negative charge) serve as coagulant aids complementing inorganic Al+++ and Fe+++ coagulants by increasing the rate of flocculation, size, and toughness of particles.

(2) Cationic (positive charge) serve as primary coagulants or in conjunction with inorganic coagulants.

(3) Non-ionic (equal amounts of positively and negatively charged groups in monomers) serve as coagulant aids in a manner similar to that of both anionic and cationic polyelectrolytes.

The popularity of polymers is primarily due to their ease in handling, small storage space requirements, and their effectiveness. All of the inorganic coagulants are difficult to handle and their corrosive nature can cause maintenance problems in the storing, handling, and feeding systems in addition to the safety hazards inherent in their handling. Many plants in the U.S. have abandoned the use of inorganic coagulants in favor of polymers.

The facilities for chemical conditioning are relatively simple and consist of equipment to store the chemical(s), feed the chemical(s) at controlled dosages, and mix the chemical(s) with the sludge. The cost of chemical conditioning is primarily a function of the quantity of chemical required which is affected by factors such as:

1. Solids concentration
2. Sludge particle size
3. Proportion of volatile matter in sludge
4. Reducing agents in the sludge (i.e., H_2S)
5. Alkalinity

Chemical requirements for any given sludge can be determined accurately only by tests on the specific sludge involved. Typical values appear in Table 6-22.

TABLE 6-22

Typical Dosages of Chemical Conditioners for Different Dewatering Processes

Process Chemical Added	Sludge Type gm Chemical Added per kg Sludge Solids		
	Raw Primary	Raw Primary & WAS	Anaerobically Digested Primary & WAS
Solid Bowl Centrifuge (Polymer)	1 - 2.5	2 - 5	3 - 5
Belt Filter Press (Polymer)	2 - 4	2 - 5	4 - 7.5
Vacuum Filter (Polymer[1])	2 - 5	3 - 6	---
(Lime[2])	80 - 100	90 - 160	150 - 210
(Ferric Chloride[2])	20 - 40	25 - 60	30 - 60
Filter Press (Lime[2])	110 - 140	110 - 160	110 - 300
(Ferric Chloride[2])	40 - 60	40 - 70	40 - 100

(1) Polymer can sometimes be substituted for lime and ferric chloride in conditioning raw sludges for vacuum filtration.

(2) Lime and ferric chloride are typically used together at these dosages.

Chemical costs also vary widely from one locale to another. Typical chemical conditioning costs in 1978 were $11-$28 per dry tonne of solids. The addition of inorganic chemicals can double the weight of sludge.

6.11.2 Digestion

The principal purposes of organic sludge digestion, a method of stabilization, are to render the sludge less putrescible, to reduce the pathogenic content, and to reduce the sludge quantity. Processes commonly used are: anaerobic digestion and aerobic digestion.

(i) Anaerobic Digestion

In this process, the organic matter in the sludge is stabilized in an anaerobic (oxygen devoid) environment. Most modern systems are "high-rate" systems utilizing one or two stages. A typical two-stage process is shown in Figure 6-26. The stabilization of the sludge occurs in the first stage, mixed and heated unit with the second stage digester providing settling and thickening. In a single stage system, the secondary digester may be replaced by some other clarification/thickening process. The digester is heated to 29-35'C and typically provides 15 days or less detention of the sludge.

Optimum conditions further include a pH of 6.8 to 7.2 and the absence of toxic materials. The process has been successful when primary sludge or combinations of primary sludge and limited amounts of waste activated sludge (WAS) constitute the system's feed. With the advent of wastewater treatment systems that are more efficient than simple sedimentation, large quantities of activated sludge are produced at the plants. This additional sludge, when placed in a two-stage anaerobic digestion process, can cause high operating costs and poor plant efficiencies. The basic cause of the problem is that the additional solids neither readily settle nor dewater after digestion.

The process converts about 50% of the organic solids to liquid and gaseous forms - providing a substantial reduction in the mass of sludge requiring disposal. A major component of the gaseous by-products (usually about two-thirds) is methane. The resulting gas has a typical heat value of 224 KJ/m^3 with about 0.94 m^3 of gas formed per kilogram of volatile solids destroyed.

The use of anaerobic digester gas has been practiced to some extent in wastewater treatment plants for many years. Typical design criteria are are presented in Table 6-23.

As noted the basic difference between conventional and high-rate systems is their loading rates. The high-rate units achieve their higher loadings because their design includes uniform temperature and greater mixing capacity.

The effect of temperature on time required for proper digestion is shown in Table 6-24.

FIGURE 6-26

Two-Stage Anaerobic Digestion

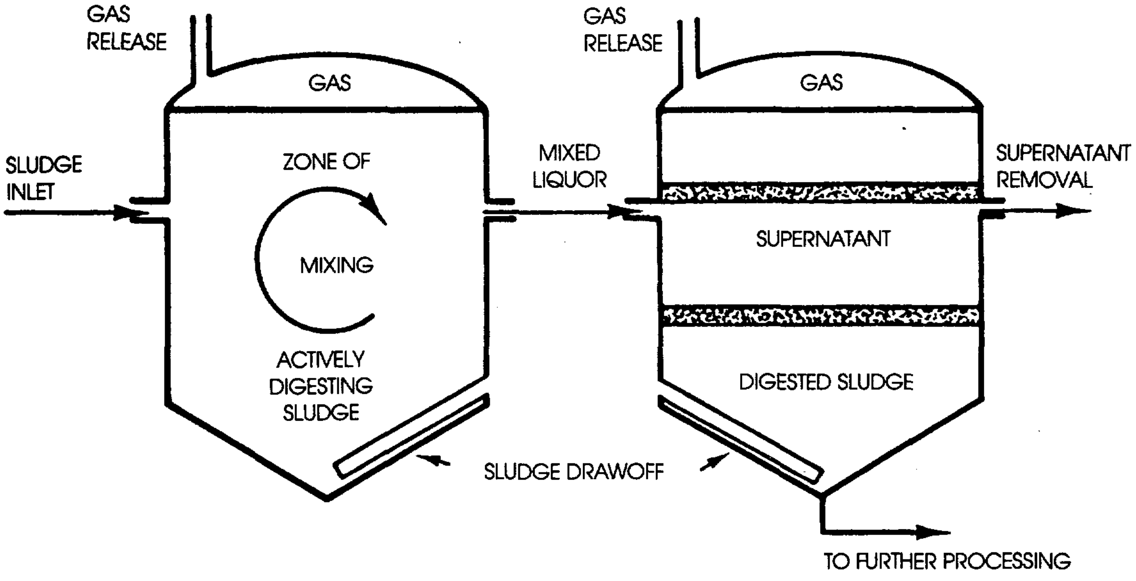


TABLE 6-23

Typical Design Criteria for Sizing Mesophilic Anaerobic Sludge Digesters

Parameter	Low-rate Digestion	High-rate Digestion
Volume criteria, (m^3/capita)		
Primary sludge	0.06 - 0.08	0.04
Primary sludge + Trickling filter humus	0.11 - 0.14	0.08 - 0.09
Primary sludge + Activated sludge	0.11 - 0.17	0.08 - 0.11
Solids loading rate (kg volatile suspended solid (VSS)/m^3/day)	0.64 - 1.60	2.40 - 6.40
Solids retention time, (days)	30 - 60	10 - 20

As noted, the basic difference between conventional and high-rate systems is their loading rates. The high-rate units achieve their higher loadings because their design includes uniform temperature and greater mixing capacity.

Source: Estrada, A. A. 1960. Design and cost considerations in high rate sludge digestion. Journal Sanitary Engineering Division. ASCE. 86(SA3):111.

TABLE 6-24

Effect of Temperature on Anaerobic Digestion Time

Temperature ('C)	Digestion Time (Days)
15	67.8
20	46.6
25	37.5
30	33.3
35	23.7
40	22.7
45	14.4
50	8.9
60	12.6

A typical conventional or low-rate anaerobic digester might be designed with a solids loading rate of 1.12 kg VSS/day/m3 and a detention time of 50 days, while a high-rate digester would have a solids loading rate of 4.4 kg VSS/day/m3 and detention time of 25 days.

Proper operation of a digester requires keeping the temperature variation to less than one degree per day, highest gas production, holding the volatile acids to alkalinity ratio between 0.1 and 0.25 and maintaining the pH between 6.8 and 7.2. As an example a volatile acids value of 200 mg/l and an alkalinity of 2,000 mg/l would give a volatile acids to alkalinity ratio of 0.1. Corrective measures may be needed if this value starts rising above 0.25 and approaching 0.35. Potential sources of problems include volatile acids, heavy metals, oxygen, sulfides and ammonia. Ammonia is inhibitory at levels of 1500 to 3000 mg/l and pH > 7.4, and it is toxic above 3000 mg/l. While it was originally thought that volatile acids concentrations above 2000 mg/l were toxic, as was already pointed out above, the important thing is the ratio of volatile acids to alkalinity. Soluble and total heavy metal concentrations are often greatly different because anions such as carbonate and sulfide can remove heavy metals from solution by precipitation and sequestering. Consequently, it is not possible to define precise total toxic concentrations for any heavy metal. Only the dissolved fraction of metals cause inhibition. Inhibition of anaerobic digestion occurs at soluble concentrations of approximately 3 mg/l for chromium, 2 mg/l for nickel, 1 mg/l for zinc and 0.5 mg/l for copper (Mosey 1976, DeWalle et al. 1979).

Digester gas is used at wastewater treatment plants to heat digesters and buildings and fuel engines that drive pumps, air blowers and electrical generators. The following criteria give estimates for gas and heat available from anaerobic digestion of a community's combined municipal and industrial wastewaters:

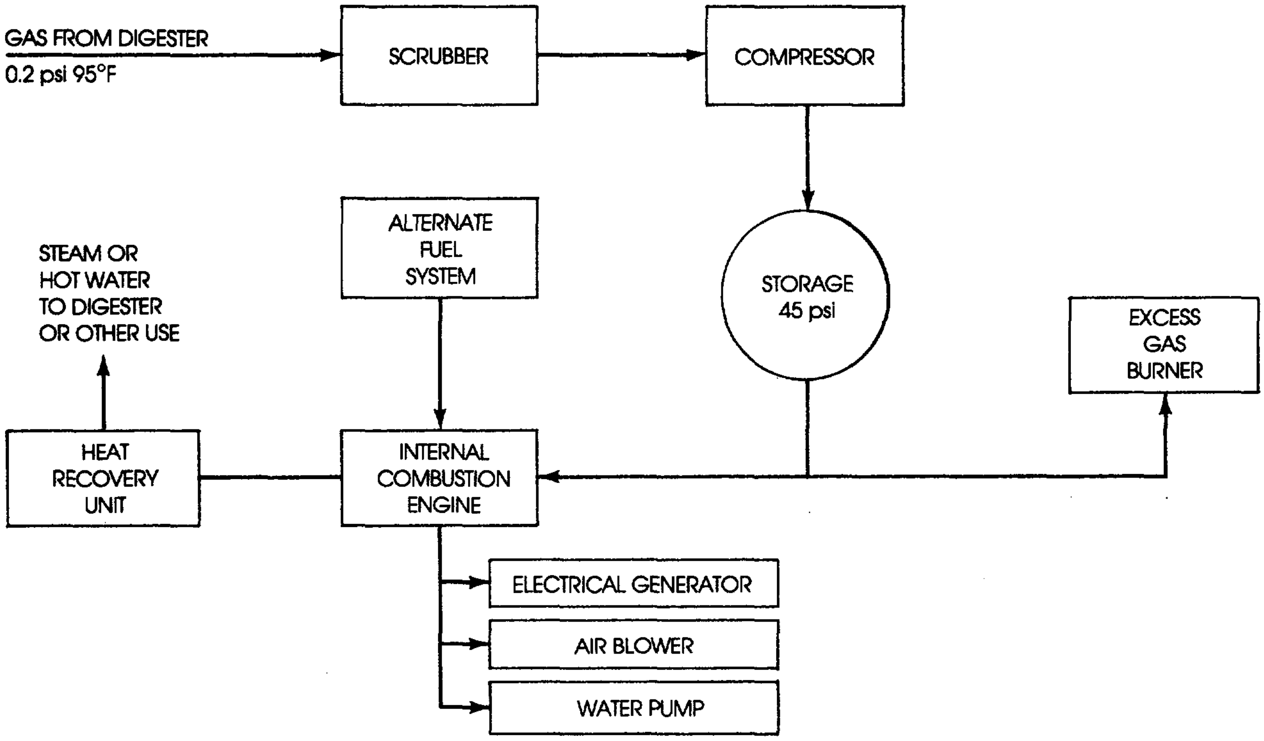
	Primary Sludge	Waste Activated Sludge	Total
Gas Produced, l/m3 of wastewater treated	38.72	42.42	81.14
Heat Available, kJ/m3 of wastewater treated	865.50	948.25	1813.70

A schematic of a typical system to utilize digester gas in an internal combustion (IC) engine is shown in Figure 6-27. As indicated in this figure the engine could be coupled to a generator, blower or pump. Typical IC engine efficiency is 36.4% (2750 kJ/MJ). An IC engine-generator's typical efficiency is 30 % (4479 kJ/MJ). The electrical energy which can be generated from anaerobic digestion of primary and WAS could supply about 85 % of the electrical energy required for an activated sludge plant while also providing over 50 % of the heat for the digestion process itself.

The process also provides substantial reductions of pathogenic bacteria (85-100 %).

FIGURE 6-27

Anaerobic Digester Gas Utilization System



The process disadvantages include: 1) process control requires considerable operator expertise and time to achieve optimum solids reduction and gas production; 2) the supernatants from the process are often high in BOD, solids and ammonia and impose an added load when recycled to the wastewater treatment system.

Costs on a per dry tonne basis in 1978 for two-stage anaerobic digestion follow:

Sludge Source	Tonnne Per Day Dry Solids to Digester			
	10	25	50	100
Primary	$19.10	$15.75	$14.50	$13.90
Primary + WAS	47.55	43.40	43.00	41.45

No credit for the fuel value of the digester gas is reflected in the above estimates. At a 1978 cost of $2.84/1000 MJ, the value of the fuel was $16/tonne dry solids for primary sludge digestion and $18.90/tonne for primary + WAS.

(ii) Aerobic Digestion

Aerobic digestion consists of the separate aeration of waste primary sludge, waste biological sludge, or a combination of waste primary and biological sludges in an open tank. It is usually used to stabilize excess activated sludges or the excess sludges from small plants which do not have separate primary clarification. Figure 6-28 is a schematic diagram of an aerobic digestion system. The advantages that the system offers over anaerobic digestion include: simpler operation, less capital cost, and better supernatant quality. Disadvantages are: higher operating costs, poor sludge dewatering characteristics, and net energy consumption rather than energy production.

Current practice is to provide >15 days of detention time for the stabilization of excess biological sludges. Additional time is required when primary sludge is included. Volatile solids reductions of 35-50% have been achieved. In addition about 2 gm of oxygen are required per gm of volatile solids destroyed when the liquid temperature is 45'C or less. A dissolved oxygen concentration of 1.0 mg/l should be maintained at all times.

Figure 6-29 well summarizes the results of several pilot and plant scale studies. This clearly shows the importance of both temperature and sludge age. Sludge age can be operationally defined as the total solids mass in the treatment system divided by the quantity of solids withdrawn daily. In a digester without recycle, the sludge age is equivalent to the hydraulic detention time. This figure is very useful for design. It shows, for example, with a liquid temperature of 20'C and a sludge age of 40 days that a 40 % volatile solids reduction can be expected.

FIGURE 6-28

Aerobic Digestion System

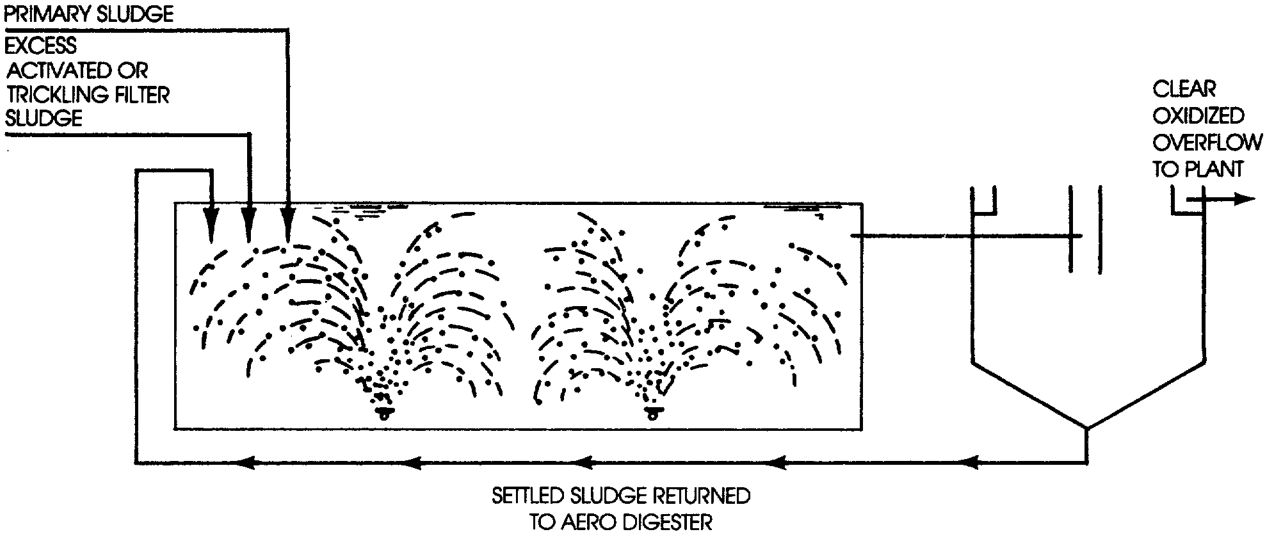


FIGURE 6-29

Volatile Solids Reduction as a Function of Digester Liquid Temperature and Digester Sludge Age

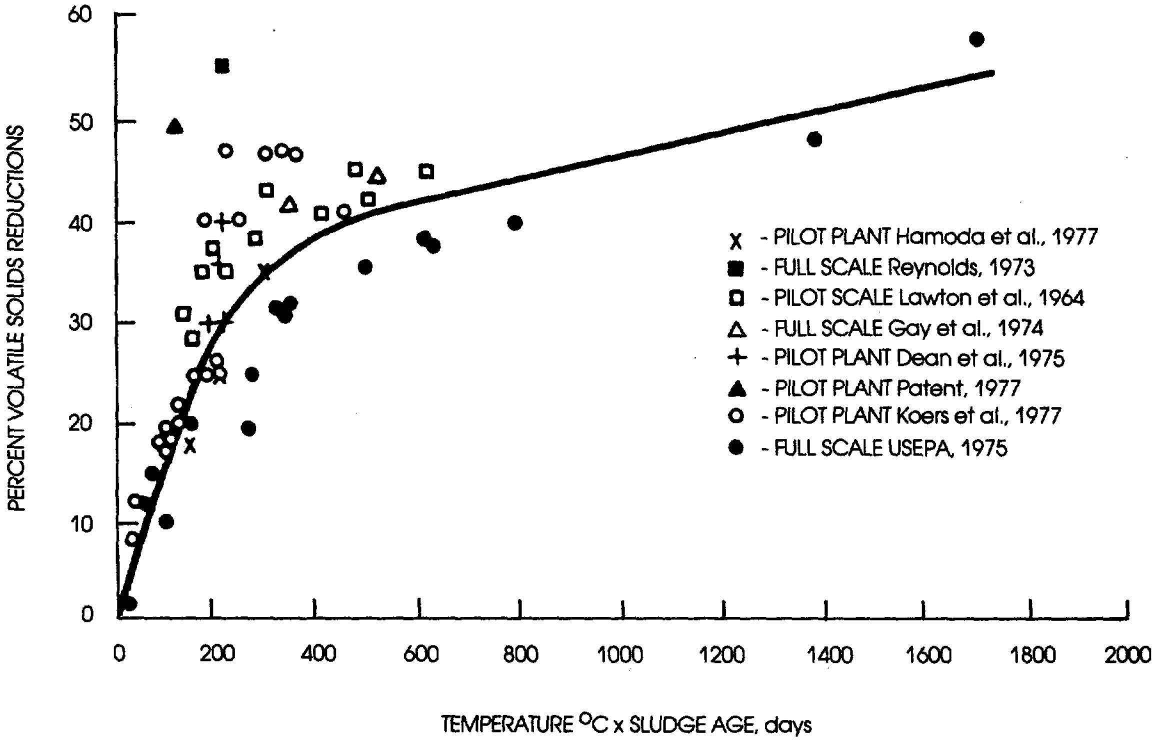


Table 6-25 presents 1978 costs calculated for the aerobic digestion of a combination of primary + waste activated sludges (exclusive of solids separation an/or thickening). Capital costs are significantly less than for two-stage anaerobic digestion. This is partly because total sludge detention time was assumed to be one-third that of two-stage anaerobic digestion. The required detention time to achieve the same level of solids destruction as with anaerobic digestion may actually be two times that of anaerobic digestions. O & M costs are significantly higher for the aerobic process due to power consumption. In order to properly compare these two systems, differences in thickening, dewatering and supernatant treatment costs must be added to each of the digestion system costs.

6.11.3 Composting

Composting is a method for the biological oxidation of organic matter in sludge under conditions which allow for the biological production of heat. Composting, properly carried out, will dewater sludge, destroy objectionable odor producing elements of sludge, destroy or reduce disease organisms, when it is operated at elevated temperatures (45-75'C) and produce an aesthetic and useful organic product. Compost, the end product, contains some available nitrogen, but its major value is its moisture holding capacity, which makes it an excellent soil additive. The soil/compost mixture is more workable and allows easier root penetration.

Composting of wastewater sludge differs from processing and composting of solid waste. Some of these differences are:

- Composting of solid waste is often preceded by a complex materials handling and separation process.
- Solid waste varies widely in composition which makes processing more difficult.
- Many past solid waste composting operations were operated and evaluated on the basis of profit making potential rather than as an alternative disposal means.
- For a given population, the volume of solid waste compost is several times the volume of wastewater sludge compost. Therefore, solid waste creates a much greater marketing or disposal task.

The general composting method is very similar for all processes. It is shown schematically in Figure 6-30. Important environmental considerations include moisture, temperature, pH, nutrient concentration and availability and oxygen concentration. Dewatered sludge (typically 20 % solids) is delivered to the site and mixed with a bulking agent. The purpose of the bulking agent is to increase the porosity of the sludge and thus help assure aerobic conditions during composting. If the composting material is too dense or wet, it may become anaerobic thus producing odors. The optimum moisture content of the composting mixture is in the range of 40-50 %. If it is too porous the temperature of the material

TABLE 6-25

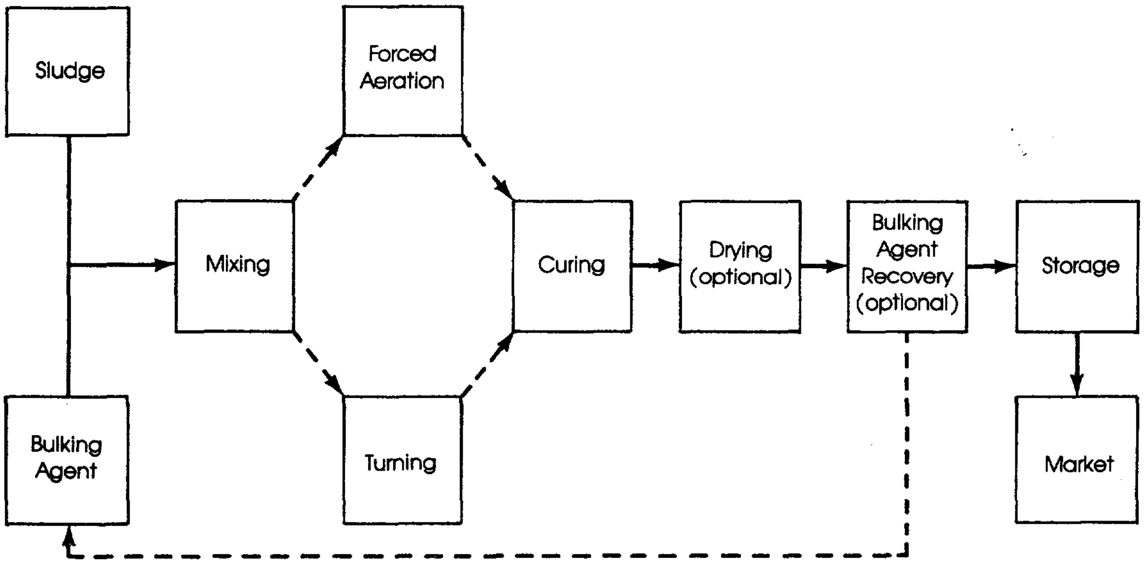
Aerobic Digestion Costs
Utilizing Conventional Air System

Item	Tons Per Day of Dry Solids (Primary + WAS) 10	25	50	100
Construction Cost	$800,000	$1,400,000	$2,200,000	$3,300,000
Engr., Legal, Adm., & During Construction (19%)	152,000	266,000	418,000	627,000
Total Capital Cost	$952,000	$1,666,000	$2,618,000	$3,927,000
O & M Labor @ $10/Hr.	32,000	50,000	80,000	105,000
Power @ $0.025/Kwh	37,500	75,000	150,000	300,000
Maintenance Materials	7,000	10,000	16,000	20,000
Annual O & M Costs	76,000	135,000	246,000	425,000
Annual Capital Cost (x.0944)	90,000	157,000	247,000	371,000
Total Annual Cost	$166,500	$292,000	$493,000	$796,000
Cost Per Ton of Dry Solids	45.61	$32.00	$27.00	$21.80

1 ton = 0.91 tonne

FIGURE 6-30

Composting Process Flow



will remain low. Low temperatures delay the completion of composting and reduce the kill of disease organisms.

Various bulking materials can be used, and suitable low cost materials include wood chips, bark chips, rice hulls, shredded tires and cubed solid waste. Unscreened finished compost has also been used. Generally, one part sludge having a 20 % solids concentration is mixed with three parts bulking agent, although this mixture can be varied depending on the moisture content of the sludge, type of bulking agent, and local conditions. The sludge-bulking agent mixture is then formed into the windrow or static pile as applicable, and the microorganisms cultivated there in a controlled environment proceed to reduce the biodegradeable organic material.

Following composting, the product is removed from the windrow or static pile and cured in storage piles for 30 days or longer. This curing provides for further stabilization and pathogen destruction. Prior to or following curing, the compost may be screened to remove a portion of the bulking agent for reuse or for applications requiring a finer product. The compost can also be used without screening. Removal of the bulking agent reduces the dilution of the nutrient value of the compost. The compost is then ready for distribution.

pH is a factor in all biological processes. Both high and low pH's reduce microbial activity. It is good to keep the process in a near neutral pH range. Another important environmental factor is the nutrient concentration required for the composting microorganisms. A basic measurement of this requirement is the carbon-nitrogen ratio. The optimum C/N ratio for composting is 20 to 25 parts of carbon to 1 part nitrogen. The final environmental factor of concern is the oxygen level in the composting mass. Oxygen is essential for the aerobic activity which takes place in a compost pile, and oxygen concentrations in the atmosphere within the compost on the order of 5 to 15 percent are generally necessary.

Composting systems generally fall into three categories: a) windrow, b) aerated static pile, and c) mechanized or enclosed systems. The windrow and aerated static pile systems have been used almost exclusively in composting sewage sludge because of their low cost and demonstrated performance. They are discussed in detail below.

Mechanized or enclosed systems have had limitedly usage in the U.S., but several systems are operating in Europe with sewage sludge. These systems offer certain advantages over natural processes. They are enclosed within a vessel or reactor, as a result environmental factors do not affect them. These processes require little land, can produce the highest solids content of any process and are most flexible with regard to odor control and operational control. However, they do have certain disadvantages. The facilities have the highest costs, since construction of the vessels and necessary equipment result in high capital expense. And, being mechanical in nature, the facilities have high maintenance costs and use more energy. These systems will not be discussed further.

(i) Windrow Composting

Windrow composting is the oldest process used exclusively for sludge. The windrow consists of long rows of sludge mixed with a bulking agent, often previously composted sludge. The sludge/bulking agent mixture must be thoroughly blended to assure proper composting. Oxygen is provided to the microorganisms through natural ventilation and periodic turning of the windrow.

The sludge-bulking agent mixture (normally 3 parts of bulking agent by volume to one part of sludge) is spread in windrows with a triangular cross section. The windrows are normally 3 to 4.9 meters wide at the base, 1 meter wide at the top and 0.9 to 1.5 meters high. An alternative method of mixing the bulking agent and sludge and forming the windrow consists of laying the bulking agent out as a base for the windrow. The sludge is then dumped on top of the bulking agent and spread. A composting machine (similar to a large rototiller) mixes the sludge and bulking agent and forms the mixture into a windrow. About 8 to 10 turns of the pile are necessary to adequately blend the two materials.

The windrow is normally turned daily using the composter; however, during rainy periods turning is suspended until the windrow surface layers dry out. Temperatures in the windrow interior under proper composting conditions range from 55 to 65'C. Turning moves the surface material to the center of the windrow for exposure to higher temperatures. The higher temperatures are needed for pasteurization and to kill off most pathogenic agents. Turning also aids in drying and increases the porosity for greater air movement and distribution.

The windrows are turned for a two week period or longer depending on the weather and efficiency of composting. The compost windrow is then flattened for further drying. The compost is moved to curing when the moisture content has decreased to approximately 30 to 40 percent. Proper windrow composting should produce a relatively stable product with a moisture content of 30 to 40 % and which has been exposed to temperatures of at least 50'C for a portion of time during the composting process.

The composting process requires longer detention times in cold or wet weather, therefore, climate is a significant factor with the windrow process in open spaces. Covering the composting area can significantly reduce the effects of cold weather and nearly eliminate the problems of wet weather. In any case, the curing area should be covered if operations are to be carried out during periods of precipitation.

(ii) Static Pile Composting

The static pile composting method (Epstein et al. 1976) as applied to raw sludge requires a forced ventilation system for control of the process. The pile then remains fixed, as opposed to the constant turning of the windrow, and the forced ventilation system maintains aerobic conditions by either forcing air into or drawing air through the pile by mechanical means.

A base is prepared for the pile consisting of a 0.3 m thick layer of bulking agent or previously composted unscreened product. A 0.1 m diameter perforated pipe is installed in the base as an aeration header. The piles are generally triangular in cross section, 4.5 to 7.5 meters at the base, 3 meters in height and 12 to 15 meters in length. The sludge-bulking agent mixture is piled on this base to a height of approximately 3 m to form a triangular cross section. Generally a ratio of 2:1 or 3:1 of bulking agent to dewatered sludge is used to absorb moisture and to provide porosity to the compost pile. The pile is capped with a 0.3 m layer of screened compost product. This top layer extends down the sides to help absorb odors and to act as a shield or roof against penetration of precipitation. A typical static pile is illustrated in Figure 6-31. An alternative configuration is the extended static pile method where subsequent piles are "added" to the initial static pile. This configuration saves space compared to a number of separate static piles.

The perforated underdrain pipe is attached to a blower by piping and fittings. The other side of the blower is piped to a smaller, adjacent pile of screened compost product, that is about 1.3 m high and 2.7 m across. Air and gases are drawn by the blower from the static compost pile and discharged through the small pile of product compost. The small pile effectively absorbs odors. The operating cycle of the blower is adjusted to maintain oxygen levels in the exhausted gases and compost pile within a range of 5 to 15 %. Temperatures within the compost pile will vary somewhat with monitoring location, but they should reach 60-65°C. Normally the blower is operated on an of-off cycle to maintain proper oxygen levels and temperatures within the pile.

After an average composting period of 3 weeks, the compost is moved to the curing area.

Outdoor temperatures as low as -7°C and rain totaling 0.2 m per week have not interfered with the successful outdoor operation of exposed static pile composting. Temperatures produced during static pile composting are generally above 55°C and often exceed 70 to 80°C.

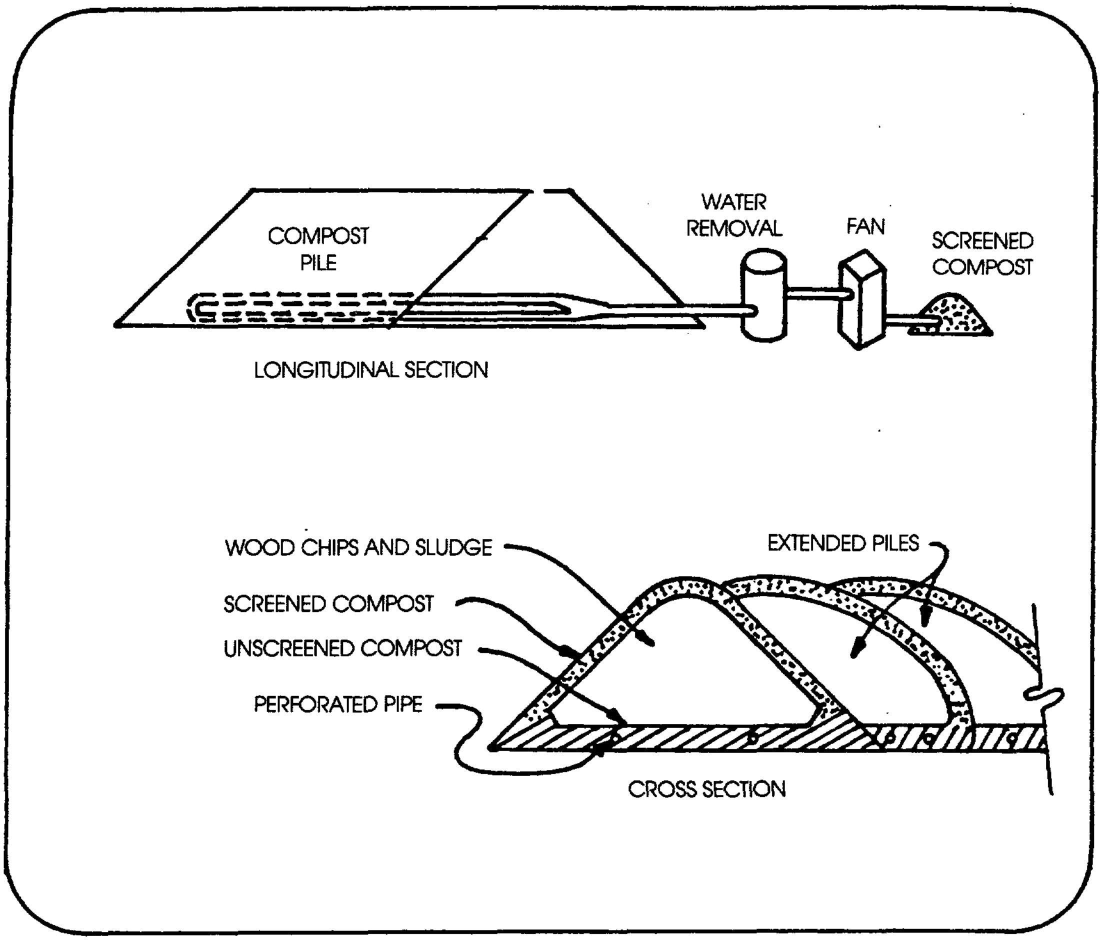
The compost product has a slight musty odor, is moist, dark in color, and can be bagged. The texture of the compost varies depending upon the degree of screening. Compost is valuable as a soil conditioner and low grade fertilizer and varies widely in content. Typical compost contains an average of 1.5 % nitrogen and 1.0 % phosphorous. Proper static pile composting should reduce total and fecal coliform and salmonella below detectable limits. Compost produced by the windrow process is likely to contain detectable pathogens because lower temperatures are produced. Composting has little effect on total heavy metal content of the sludge, but there is some dilution and also some indication of a lower uptake rate after composting.

Economics

The economics of composting are determined by two factors; the cost of producing the compost and the cost of (or income from) disposal of the

FIGURE 6-31

Static Pile Composting as Developed by
the Agricultural Research Service at Beltsville, Maryland



compost product. The marketing of the end product is a key to the success of a composting effort. One market study found several successful municipal sludge composting operations where all of the end product was sold or otherwise successfully used. The study concluded that the upper price limit for bulk sewage sludge compost in 1978 was $4-11/tonne and for packaged, bagged, sewage sludge compost, $66/tonne. Bagging costs could approach $33/tonne.

Those municipal sludge compost marketing operations that have been successful have generally:

- had favorable local publicity
- had the product available for pick-up (or make deliveries)
- offered guidelines for its use, or at least suggestions
- offered the product at no cost or inexpensively
- given the product a trade name

The cost of producing compost includes the following elements: 1) amortization of land, capital site improvements, and structures, 2) amortization of major mobile equipment costs and 3) operation and maintenance costs. Land requirements are affected by several factors but are typically 0.1-0.2 hectares/dry tonne for the static pile technique. Windrow techniques require 2-3 times more area.

The required site improvements and structures will vary depending on process used, availability of existing facilities, degree of mechanization of the process, and to a degree, the demands of the climatic region. Site improvements related to composting will generally include site access and improvements, bulking agent storage, bulking agent-sludge mixing area or mechanical fixed equipment, composting pads and appurtenances such as blowers, screening area, compost storage area, support facilities such as electrical, and fixed materials handling equipment.

Major mobile equipment includes screens, front end loaders, trucks, and testing equipment. The number and size of major equipment will depend on the capacity and type of operation. Operation and maintenance costs normally include:

- Labor for constructing the compost piles, handling materials, and screening the compost
- Labor for regular inspection of operations and performing tests on the piles
- Electrical energy for blowers, lighting, and other miscellaneous uses
- O & M costs for the equipment including front-end loader and screen
- Costs of transport of materials as required
- Cost of bulking agent, typically $3 to $5 per cubic meter in 1978.

This of course can be reduced if a material is employed that is being discarded as a waste product.

Tables 6-26 and 6-27 provide actual economic and operating data for small (< 0.9 dry mt/day) and medium (1.8 to 8.2 dry mt/day) static pile and

TABLE 6-26

Economic and Operating Characteristics of Small Static Pile Sludge Composting Facilities (<1 Dry Ton/Day)[a]

Facility	Average Sludge Composted[b] (Dry Tons/ Day	Solids Content of Sludge (%)	O&M Costs[c] ($/Dry Ton of Sludge)	Marketing Compensation ($/Dry of Sludge)	Capital Costs ($/Dry Ton of Sludge)	Person-hours Per Day Per Dry Ton Sludge	Dry Tons Per Hectare Per Day of Site Space
Swampscott, MA	0.3	25	423.08	21.45	146.666	5.8	2.23
Old Town, ME	0.3	11	408.40	43.00	916.666	16.3	0.49
Gardner, ME	0.9	20	-	-	-	8.3	1.11
Durham, NH	1.0	15	78.28	-	660.000	4.4	2.86

[a] Measurements are given in English tons. One English ton=0.907 mt.
[b] Based on a 5-day week.
[c] Based on values provided by the authority, may include dewatering and transportation.

TABLE 6-27

Economic and Operating Characteristics of Medium Sludge Composting Facilities (2-9 Dry Ton/Day)[a,b]

Facility	Average Sludge Composted[c] (Dry Tons/ Day	Solids Content of Sludge (%)	O&M Costs[d] ($/Dry Ton of Sludge)	Marketing Compensation ($/Dry of Sludge)	Capital Costs ($/Dry Ton of Sludge)	Person-hours Per Day Per Dry Ton Sludge	Dry Tons Per Hectare Per Day of Site Space
Bangor, ME	2.1	23	31.26	35.75	7,174	1.1	2.1
Morganton, NC	3.2	14	75.22	32.18	33,916	7.4	5.3
Occoquan, VA	3.6	19	44.70	0	412,500	2.8	4.4
S. Portland, ME	3.9	14	157.42	25.00	639,534	4.1	3.7
Stratford, CT	4.5	20	-	-	-	10.6	11.1
Middletown, NJ	4.5	21	169.22	0	946,000	5.3	5.9
W. Warwick, RI	5.0	14	37.77	0	-	2.2	3.1
Hampton Roads, VA	6.8	16	140.04	43.00	322,666	7.0	4.2
Marrimack, NH	8.0	21	101.73	7.15	22,988	3.00	9.8

[a] All facilities are static pile except Occoquan, VA, which is a windrow facility.

[b] Measurements are given in English tons. One English ton = 0.907 mt.

[c] Based on a 5-day week.

[d] Based on values provided by the authority; may include dewatering and transportation.

windrow facilities as of 1983. As these tables demonstrate, both capital and O&M costs vary substantially from one facility to another within each category. These variations are due to several factors such as method of cost accounting used, inclusion of dewatering costs, inclusion of transportation costs to offsite locations, amortization, and local considerations. Due to these variations, it is essential that each community perform its own cost estimate.

6.11.4 Thickening

The purpose of sludge thickening is to reduce the sludge volume to be stabilized, dewatered or disposed of Figure 6-32 illustrates the impact that thickening can have on sludge volume. Thickening a 1 % solids concentration sludge to one of 6 % solids reduces the volume of sludge to be handled by a factor of over 5. This reduction can provide significant savings in the cost of dewatering, digestion, or other downstream facilities. There are three commonly used methods for sludge thickening: gravity, flotation, and centrifugation.

(i) Gravity Thickening

Thickening by gravity is the most common concentration process in use at wastewater treatment plants. It is simple and inexpensive. Gravity thickening is essentially a sedimentation process similar to that which occurs in all settling tanks. But, in comparison with the initial waste clarification stage, the thickening action is relatively slow. The theoretical aspects of gravity thickening have been the subject of many studies and are well summarized in several papers (Dick and Ewing 1967; Edde and Eckenfelder 1968; and Fitch 1971).

Figure 6-33 illustrates a typical, circular gravity thickener. The units have a typical side water depth of 3 meters. Loading rates are expressed in terms of kilograms of dry solids in the sludge applied to the thickener surface area per day (kg/day/m^2) or per hour (kg/hr/m^2). Table 6-28 summarizes typical results achieved with gravity thickening and provides design criteria.

The degree to which waste sludges can be thickened depends on many factors; among the most important are the type of sludge being thickened and its volatile solids concentration. Bulky biological sludge, particularly that from the activated sludge process, will not concentrate to the same degree as raw primary sludge. Activated sludges, if thickened separately, are usually thickened by the flotation process. The degree of biological treatment and the ratio of primary to secondary (biological) sludge will affect the ultimate solids concentration obtained by gravity thickening. Hydraulic and surface loading rates are also of importance. Typical maximum hydraulic loading rates of 1,200 to 1,600 l/m^2/hr have been mainly used in the past for primary sludges. For sludges such as waste-activated or similar types, much lower hydraulic loading rates of 200 to 400 l/m^2/hr are applicable. Excessively low flow rates can lead to odor problems. If the sludge flow to the thickener is far below the design rate, pumping of secondary effluent to the thickener may be practiced to minimize odors.

FIGURE 6-32

Effect of Increasing Sludge Solids on the Final Sludge Volume

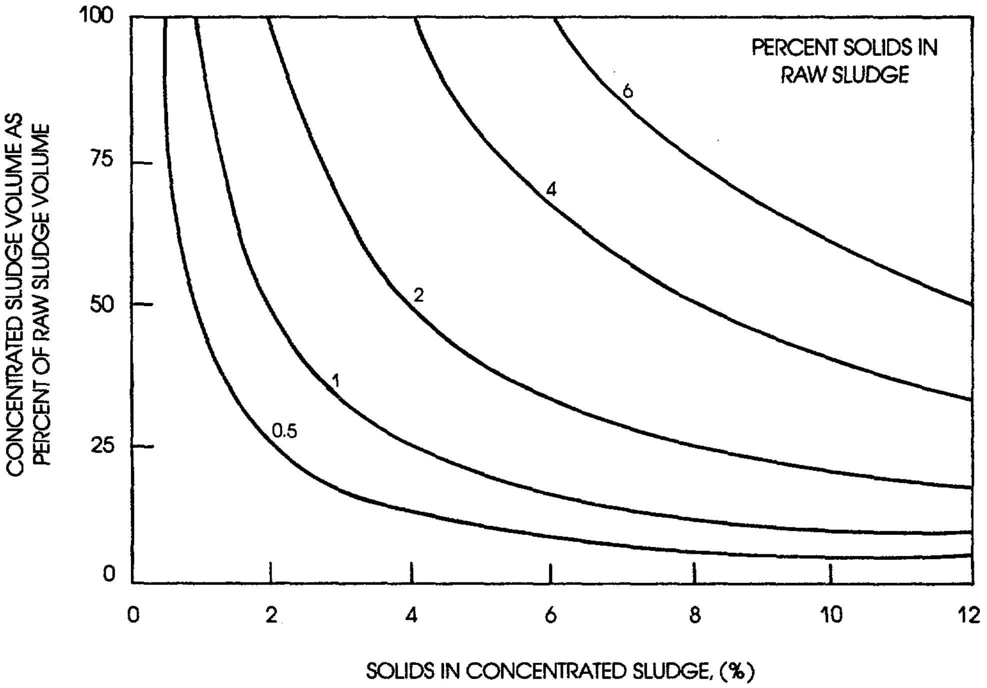


FIGURE 6-33

Gravity Thickener

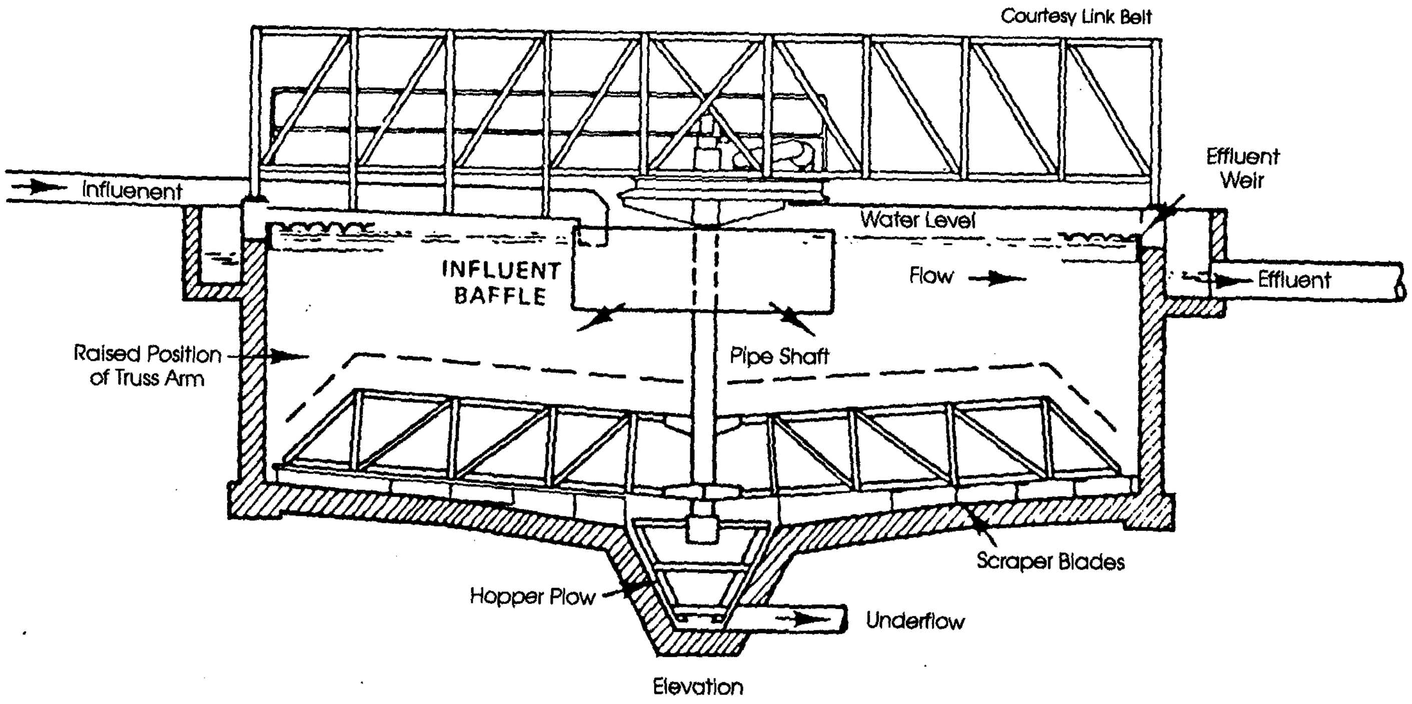


TABLE 6-28
Typical Gravity Thickener Surface Area Design Criteria[a]

Type of Sludge	Influent Solids Concentration, Percent Solids	Expected Underflow Concentration Percent solids	Mass Loading, lb/sq ft/hr[b]
Separate sludges:			
Primary (PRI)	2 - 7	5 - 10	0.8 - 1.2
Trickling filter (TF)	1 - 4	3 - 6	0.3 - 0.4
Rotating biological contactor (RBC)	1 - 3.5	2 - 5	0.3 - 0.4
Waste activated sludge (WAS)			
WAS - air	0.5 - 1.5	2 - 3	0.1 - 0.3
WAS - oxygen	0.5 - 1.5	2 - 3	0.1 - 0.3
WAS - (extended aeration)	0.2 - 1.0	2 - 3	0.2 - 0.3
Anaerobically digested sludge from primary digester	8	12	1.0
Thermally conditioned sludge:			
PRI only	3 - 6	12 - 15	1.6 - 2.1
PRI + WAS	3 - 6	8 - 15	1.2 - 1.8
WAS only	0.5 - 1.5	6 - 10	0.9 - 1.2
Tertiary sludge:			
High lime	3 - 4.5	12 - 15	1.0 - 2.5
Low lime	3 - 4.5	10 - 12	0.4 - 1.25
Alum	-	-	-
Iron	0.5 - 1.5	3 - 4	0.1 - 0.4
Other sludges:			
PRI + WAS	0.5 - 1.5	4 - 6	0.2 - 0.6
	2.5 - 4.0	4 - 7	0.3 - 0.7
PRI + TF	2 - 6	5 - 9	0.5 - 0.8
PRI + RBC	2 - 6	5 - 8	0.4 - 0.7
PRI + iron	2	4	0.25
PRI + low lime	5	7	0.8
PRI + high lime	7.5	12	1.0
PRI + (WAS + iron)	1.5	3	0.25
PRI + (WAS + alum)	0.2 - 0.4	4.5 - 6.5	0.5 - 0.7
(PRI + iron) + TF	0.4 - 0.6	6.5 - 8.5	0.6 - 0.8
(PRI + iron) + WAS	1.8	3.6	0.25
WAS + TF	0.5 - 2.5	2 - 4	0.1 - 0.3
Anaerobically digested PRI + WAS	4	8	0.6
Anaerobically digested PRI + (WAS + iron)	4	6	0.6

[a] Data on supernatant characteristics is covered later in this section.

[b] Typically, this term is given in lb/sq ft/day. Since wasting to the thickener is not always continuous over 24 hours, it is a more realistic approach to use lb/sq ft/hr.

1 lb/sq ft/hr = 4.9 kg/m^2/hr

The quality of the overhead liquid removed from the sludge solids is important in any thickening operation because this liquid is usually returned to the treatment processes. Generally, the supernatant is similar to that of raw sewage, 150 to 300 mg/l suspended solids and a BOD of about 200 mg/l. A well-operated thickener should have a minimum of anaerobic decomposition and a solids capture exceeding 90%. Thus, the overflow returned to the treatment process should not present an operational problem.

Table 6-29 summarizes the estimated 1978 costs of gravity thickening based on a loading rate of 4.1 kg/m2/hr. The electrical consumption of the process is low - on the order of 1-1.7 kwh/tonne of solids.

(ii) Dissolved Air Flotation Thickening

Flotation thickening units are especially popular in sewage treatment plants for handling waste activated sludges. There they have the advantage over gravity thickening tanks of offering higher solids concentrations and a lower initial cost for the equipment. In flotation thickening minute-sized air bubbles are attached to suspended solids causing the solids to separate from the water in an upward direction. This separation is due to the fact that the solid particulates have a specific gravity lower than water when the bubbles are attached.

Figure 6-34 illustrates the basic considerations involved in the process. A portion of the unit effluent, or plant effluent, is pumped to a retention tank (a pressurization tank) at 4.1-4.8 kg/cm2. Air is fed into the pump discharge line at a controlled rate and mixed by the action of an eductor driven by the reaeration pump. The flow through the recycle system is metered and controlled by a valve located immediately before the mixing with the sludge feed. The recycle flow and sludge feed are mixed in a chamber at the unit inlet. If flotation aids (such as polymers) are employed, introduction is normally in this mixing chamber. The sludge particles are floated to the sludge blanket and the clarified effluent is discharged under a baffle and over an adjustable weir. The thickened sludge is removed by a variable speed skimming mechanism. In practice bottom sludge collectors are also furnished for removal of any settled sludge or grit that may accumulate. Sludge thickening occurs in the sludge blanket, which is normally 20 to 61 cm thick. The buoyant sludge and air bubbles force the surface of the blanket above the water level, inducing drainage of water from the sludge particles.

Similar to gravity thickening, sludge type and quality affects unit performance. Flotation thickening is, as stated before, most applicable to activated sludges but higher float concentrations can be achieved by combining primary and activated sludges. However, equal or greater concentrations may be achieved by combining sludges in gravity thickening units. A high sludge volume index (SVI), representing a bulky sludge, results in poor thickener performance. Table 6-30 presents typical results of the flotation thickening process. As can be seen from the data polyelectrolyte chemicals are frequently used with the process. The overall effect is to increase the allowable solids loading, increase the

TABLE 6-29

Gravity Thickening Costs

Item	Tons Per Day of Dry Solids 10	25	50	100
Construction Cost	$120,000	$160,000	$320,000	$540,000
Engr., Legal, Adm., & Interest During Construction (19%)	22,800	30,400	60,800	102,600
Total Capital Cost	$142,800	$190,400	$380,800	$642,600
O & M Labor @ $10/Hr.	3,600	4,400	6,000	10,000
Power @ $0.025/Kwh	100	300	500	800
Maintenance Materials	400	700	1,300	2,200
Annual O & M Costs	4,100	5,400	7,800	13,000
Annual Capital Cost (x .0944)	13,500	18,000	36,000	60,700
Total Annual Cost	$17,600	$23,400	$43,800	$73,700
Cost per ton of dry solids	$4.80	$2.60	$2.40	$2.00

1 ton = 0.91 tonne
Assumption: 20 lbs/day/sf loading rate (98 kg/day/m^2)

FIGURE 6-34

Dissolved Air Flotation System

Courtesy Komline-Sanderson

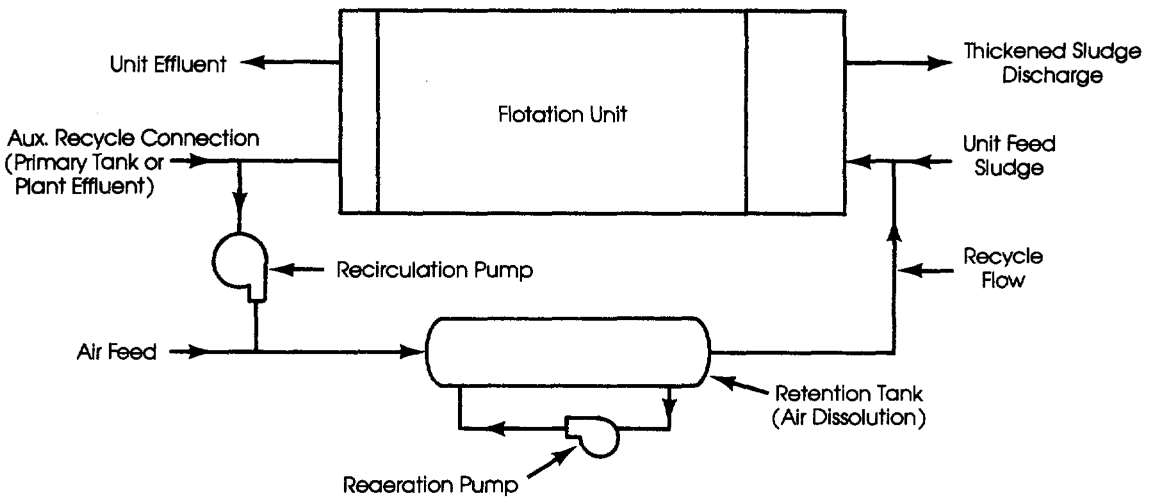


TABLE 6-30

Typical Results
Flotation Thickening

Sludge Type	Feed Solids Concentration (Percent)	Typical Loading Rate Without Polymer (lb/sqft/day)	Typical Loading Rate With Polymer (lb/sqft/day)	Float Solids Concentration (Percent)
Primary + WAS	2.0	20	60	5.5
Primary + (WAS + $FeCl_3$)	1.5	15	45	3.5
(Primary + $FeCl_3$) +WAS	1.8	15	45	4.0
WAS	1.0	10	30	3.0
WAS + $FeCl_3$	1.0	10	30	2.5
Digested Primary + WAS	4.0	20	60	10.0
Digested Primary +(WAS +$FeCl_3$)	4.0	15	45	8.0
Tertiary, Alum	1.0	8	24	2.0

1 $lb/ft^2/day = 4.9\ kg/m^2/d$

percentage of floated solids, and increase the clarity of the effluent. Cationic polyelectrolytes have been the most successful.

Typical design solids loading rates for flotation units are given in Table 6-31, which also shows the effect of sludge type on the solids loading rate (USEPA 1979). The maximum hydraulic loading rate must always be less than the minimum rise rate of the sludge/air particles to ensure that all the particles will reach the sludge float before the particle reaches the effluent end of the tank. Reported values have ranged from 0.54 to 2.7 l/min/m2, with a maximum value of 1.7 l/sec/m2 recommended for waste activated sludge.

Table 6-32 presents typical 1978 costs for flotation thickening at a loading rate of 8.2 kg/m2/hr not including any chemical feed costs. Polymer costs could add from $1-$17/tonne of solids. Electrical consumption is about 110 kwh/tonne of solids.

(iii) Centrifugal Thickening

Although centrifuges have been used widely for dewatering (See page 8), they have had less use for thickening. They have been used for thickening of WAS where space limitations or sludge characteristics make other methods unsuitable. WAS concentrations of 5-8 % were typically produced by a solid bowl conveyer type decanter centrifuge under the following conditions:

Operating method	Continuous
Bowl diameter	0.36 - 1.02 meters
Normal G range	1400 - 2100
WAS feed solids	0.5 - 1.5 %
Thickened WAS solids	5.0 - 8.0
Recovery	85 - 95
Polymer dosage range	0 - 3 kg/tonne

Other results are given in Section 6.11.5 (iii).

6.11.5 Dewatering

(i) Drying Beds

A widely used dewatering method that is especially popular in small wastewater treatment plants is drying of the sludge on open or covered sandbeds. Over 6,000 wastewater treatment plants in the United States use this method. Sandbeds possess the advantage of needing little operator skill. Air drying is normally restricted to well digested sludge, because raw sludge is odorous, attracts insects, and does not dry satisfactorily when applied at reasonable depths. Oil and grease discharged with raw sludge clog sandbed pores and thereby seriously retard drainage. The design and use of drying beds are affected by many parameters. They include weather conditions, sludge characteristics, land values and proximity of residences, and use of sludge conditioning aids. Climatic conditions are most important. Factors such as the amount and rate of

TABLE 6-31

Typical DAF Thickener Solids Loading Rates Necessary to Produce a Minimum 4 Percent Solids Concentration

	Solids loading rate, lb/sq ft/hr	
Type of Sludge	No chemical addition	Optimum chemical addition
Primary only	0.83 - 1.25	up to 2.5
Waste activated sludge (WAS)		
Air	0.42	up to 2.0
Oxygen	0.6 - 0.8	up to 2.2
Trickling filter	0.6 - 0.8	up to 2.0
Primary + WAS (air)	0.6 - 1.25	up to 2.0
Primary + trickling filter	0.83 - 1.25	up to 2.5

1 lb/sq ft/hr = 4.9 kg/m^2/hr

TABLE 6-32

Flotation Thickening Costs

Item	Tons Per Day of Dry Solids 10	25	50	100
Construction Cost	$200,000	$290,000	$380,000	$500,000
Engr., Legal, Adm., & Interest During Construction (19%)	38,000	55,100	72,200	95,000
Total Capital Cost	$238,000	$345,100	$452,200	$595,000
O & M Labor @ $8/Hr.	9,600	22,400	44,800	89,600
Power at $0.025/Kwh	10,000	20,000	40,000	75,000
Maintenance Materials	200	300	500	1,000
Annual O & M Costs	19,800	42,700	85,300	165,600
Annual Capital Costs (x. 0944)	22,500	32,600	42,700	56,200
Total Annual Cost	$42,300	$75,300	$128,000	$221,800
Cost per ton of dry solids	$11.60	$8.30	$7.00	$6.10

Assumption: 40 lbs/day/sf loading rate.

precipitation, percentage of sunshine, air temperature, relative humidity, and wind velocity determine the effectiveness of air drying. It is important that wastewater sludge be well digested for optimum drying. In well digested sludge, entrained gases tend to float the sludge solids and leave a layer of relatively clear liquid, which can readily drain through the sand. Typical design criteria are given below, and a summary of published guidance appears in Table 6-33.

Type of Digested Sludge	Area (m^2/capita)	Sludge Loading Dry Solids ($kg/m^2/a$)
Primary	0.09	134
Primary and standard trickling filter	0.15	107
Primary and activated	0.28	73.2
Chemically precipitated	0.19	107

Usually a deep layer of graded sand is laid over coarser gravel that is 20-45 cm deep. This sand-gravel filter is underlaid by pipes to drain off the clarified liquor from the sludge. Stable digested sludge is then applied to the bed at a depth of 20-30 cm, and the sludge is allowed to drain and dry for several weeks or months depending upon the weather. Especially adverse weather can result in drying times as long as 6 month.

Sandbeds can be enclosed by glass. Glass enclosures protect the drying sludge from rain, control odors and insects, reduce the drying periods during cold weather, and can improve the appearance of a waste treatment plant. Experience has shown that only 67 to 75 % of area required for an open bed is needed for an enclosed bed. Good ventilation is important to control humidity and optimize the evaporation rate. As expected, evaporation occurs rapidly in warm, dry weather. Adaptation of mechanical sludge removal equipment to enclosed beds is more difficult than to open drying beds.

Mechanical removal of sludge from drying beds has been practiced for many years at some large treatment plants, but now it is receiving more attention as the need to minimize problems with labor costs. Mechanical devices can remove sludges of 20 to 30 percent solids while cakes of 30 to 40 % generally require hand removal. Small utility tractors with front end loaders are often used for mechanical removal.

A major disadvantage in the larger plants likely to be involved in regional systems is the space required. For a 0.4 m3/s activated sludge plant, about 4.5 ha of drying beds would be required for the digested combination of primary and waste activated sludges at a loading rate of 73 kg/m2/a. The space requirements plus dependency on uncontrollable weather factors are severe restrictions on the use of drying beds in large, regional plants. Cost data available for large drying beds (adequate for 0.44-1.10 m3/s activated sludge plants) in 1978 projected a total costs of $77-99/ tonne of dry solids.

TABLE 6-33

Summary of Recognized Published Sand Bed Sizing Criteria for Anaerobically Digested Unconditioned Sludge

	Uncovered Beds		Covered Beds
Initial Sludge Source	Area m^2/cap	Solids Loading kg/m^2/a	Area m^2/cap
Primary			
	0.09	134	-
	0.09-0.14	-	0.07-0.09
N45° N latitude	0.12	-	0.09
Between 40°-45°N	0.10	-	-
S40° N latitude	0.07	-	0.05
Primary + Chemicals			
	0.2	110	-
	0.18-0.21	-	0.09-0.12
N45° N latitude	0.23	-	0.173
Between 40°-45°N	0.18	-	0.139
S40° N latitude	0.14	-	0.104
Primary + low-rate trickling filter			
	0.15	110	-
	0.12-0.16	-	0.09-0.12
N45° N latitude	0.173	-	0.145
Between 40°-45°N	0.139	-	0.116
S40° N latitude	0.104	-	0.086
Primary + waste activated sludge			
	0.28	73	-
	0.16-0.23	-	0.12-0.14
N45° N latitude	0.202	-	0.156
Between 40°-45°N	0.162	-	0.125
S40° N latitude	0.122	-	0.094

(ii) Vacuum Filtration

A vacuum filter basically consists of a cylindrical drum (See Figure 6-35 which rotates partially submerged in a vat of sludge. The filter drum is divided into compartments by partitions or seal strips. A vacuum is applied between the drum deck and filter medium causing filtrate to be extracted and filter cake to be retained on the medium during the pickup and cake drying cycle. In the drum filter shown in Figure 6-34 the cake of dewatered sludge is removed by a fixed scraper blade. There are alternative designs which use other methods for sludge removal.

The performance of vacuum filters may be measured by various criteria such as the yield, the efficiency of solids removal and the cake characteristics. Each of these criteria is of importance, but one or the other may be particularly significant in a given plant. Typical results are shown in Table 6-34.

Yield is the most common measure of filter performance. The yield expresses the filter output and is expressed in terms of kilograms of dry total solids in the cake discharged from the filter, per square meter of effective filter area, per hour.

The second measure of filter performance is the efficiency of solids removal. Basically, the vacuum filter is a device used for separating solid matter from liquid, and the actual efficiency of the process is the percentage of feed solids recovered in the filter cake. Solids removals on vacuum filters range from about 85 % for coarse mesh media to 99 % with close weave, long nap media. The recycled filtrate solids impose a load on the plant treatment units, and should normally be kept to a practical minimum. However, it may be necessary to reduce the filter efficiency in order to deliver more filter output and thus keep up with sludge production.

The filter cake quality is another measure of filter performance, depending upon cake moisture and heat value. Cake solids content varies from 20 to 40 % by weight, depending upon the type of sludge handled and the filter cycle time and submergence. Delivery of a very dry cake does not necessarily indicate good filter performance. Cake moisture should be adjusted to the method of final disposal, and it is inefficient to dry the cake more than is required. When incineration is practiced, a raw sludge cake having a fairly high moisture content can be burned without auxiliary fuel because of the higher volatile content, while a digested sludge cake will have to be dryer to burn successfully without make-up heat. One approach to improving the filtration and incineration characteristics of primary - WAS mixtures is to feed powdered coal as a conditioning agent prior to the dewatering step. It was found that a coal dose of about 0.3 kgs/kg dry sludge solids produced a sludge cake which permitted autogenous combustion with no effect on filter yield.

Illustrative costs based on an average loading rate of 20 kg/m2/h are shown in Table 6-35. Typical electrical consumption is 44-66 kwh/ tonne of solids.

FIGURE 6-35

Cutaway View of a Rotary Drum Vacuum Filter

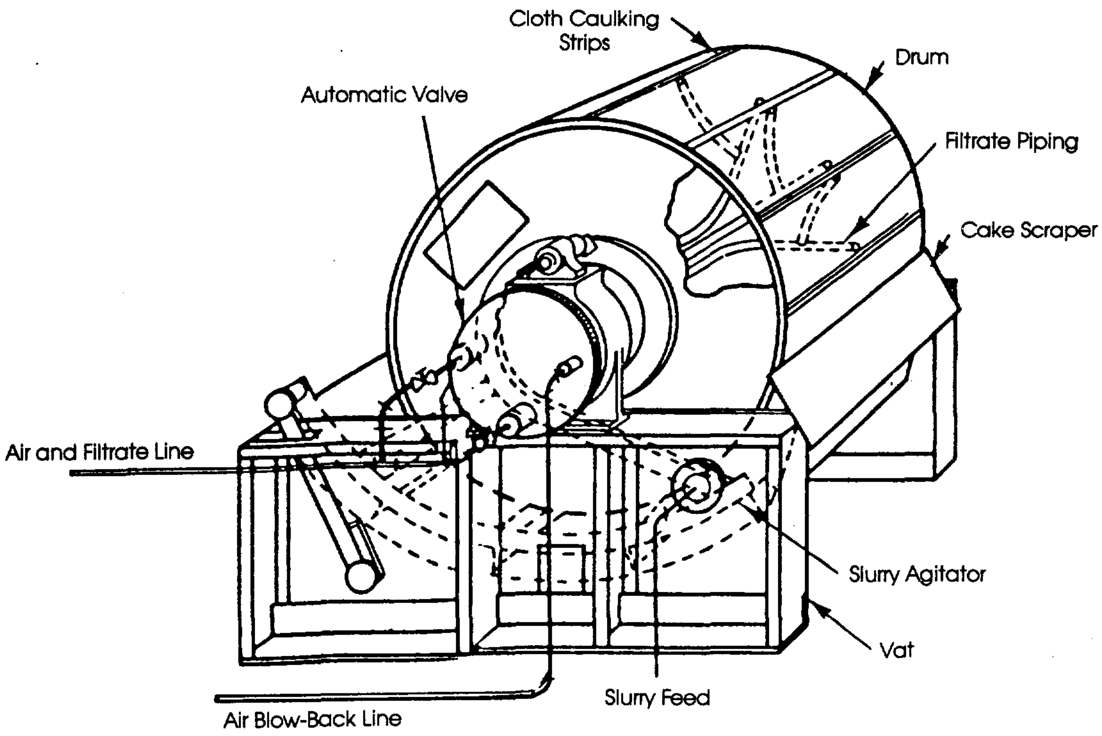


TABLE 6-34
Typical Results
Vacuum Filtration

Sludge Type	Design Assumptions	Percent Solids To VF	Typical Loading Rates (psf/hr)	Percent Solids VF Cake
Primary	Thickened to 10% solids Polymer conditioned	10	8-10	25-38
Primary + $FeCl_3$	85 mg/1 $FeCl_3$ dose	2.5	1.0-2.0	15.20
	Lime conditioning Thickening to 2.5% solids			
Primary + Low Lime	300 mg/1 lime dose Polymer conditioned Thickened to 15% solids	15	6	32-35
Primary + High Lime	600 mg/1 lime dose Polymer conditioned Thickened to 15% solids	15	10	28-32
Primary + WAS	Thickened to 8% solids Polymer conditioned	8	4-5	16-25
Primary + (WAS + $FeCl_3$)	Thickened to 8% solids $FeCl_3$ & lime conditioned	8	3	20
(Primary + $FeCl_3$) + WAS	Thickened primary sludge to 2.5% Flotation thickened WAS to 5% Dewater blended sludges	3.5	1.5	15-20
Waste Activated Sludge (WAS)	Thickened to 5% solids Polymer conditioned	5	2.5-3.5	15
WAS + $FeCl_3$	Thickened to 5% solids Lime + $FeCl_3$ conditioned	5	1.5-2.0	15
Digested Primary	Thickened to 8-10% solids Polymer conditioned	8-10	7-8	25-38
Digested Primary + WAS	Thickened to 6-8% solids Polymer conditioned	6-8	3.5-6	14-22
Digested Primary + (WAS + $FeCl_3$)	Thickened to 6-8% solids $FeCl_3$ + lime conditioned	6-8	2.5-3	16-18
Tertiary Alum	Diatomaceous earth precoat	0.6-0.8	0.4	15-20

1 psf/hr = 4.9 $kg/m^2/hr$

TABLE 6-35

Estimated Costs for Dewatering by Vacuum Filter

Item	Tons Per Day of Dry Solids			
	10	25	50	100
Construction Cost	$270,000	$470,000	$780,000	$1,400,000
Engr., Legal, Adm., & Interest During Construction (19%)	51,300	89,300	148,200	266,000
Total Capital Cost	$321,300	$559,300	$928,200	$1,666,000
Labor, Operations at $10/Hr.	47,000	90,000	160,000	270,000
Labor, Maintenance at $10/Hr.	8,000	17,000	28,000	51,000
Maintenance Materials, Chemical	46,000	100,000	180,000	310,000
Maintenance Materials, Other	24,000	45,000	72,000	120,000
Annual O & M Costs	$125,000	$252,000	$440,000	$751,000
Annual Capital Costs (x. 0944)	30,000	52,800	37,600	157,300
Total Annual Cost	$155,300	$304,800	$477,600	$908,300
Cost per ton of dry solids	$42.60	$33.40	$26.20	$24.90

1 ton = 0.907 tonne
Assumptions: Primary + WAS
4 lbs/hr/sf loading rate (20kg/m^2/hr)

(iii) Centrifugation

There are many types of centrifugal equipment available for a variety of specialized applications in industry. However, the solid bowl centrifuge is the most widely used type for dewatering of sewage sludge. The solid bowl-conveyer sludge dewatering centrifuge assembly shown in Figure 6-36 consists of a rotating unit comprising a bowl and conveyer. The solid cylindrical-conical bowl, or shell, is supported between two sets of bearings and includes a conical section at one end to form a dewatering beach or drainage deck over which the helical conveyer screw pushes the sludge solids to outlet ports and then to a sludge cake discharge hopper. Sludge slurry enters the rotating bowl through a stationary feed pipe extending into the hollow shaft of the rotating screw conveyer and is distributed through ports into a pool within the rotating bowl.

As the liquid sludge flows though the cylindrical section toward the overflow devices, progressively finer solids are settled centrifugally to the rotating bowl wall. The helical rotating conveyer pushes the solids to the conical section where the solids are forced out of the water, and free water drains from the solids back into the pool.

There are several variables which affect the performance of solid bowl centrifuges, and these are shown below. Several of them are preset by the manufacturer, and some can be controlled by the operator. Bowl speed is one of the prime variables since centrifugal force speeds up the separation of solids from liquids.

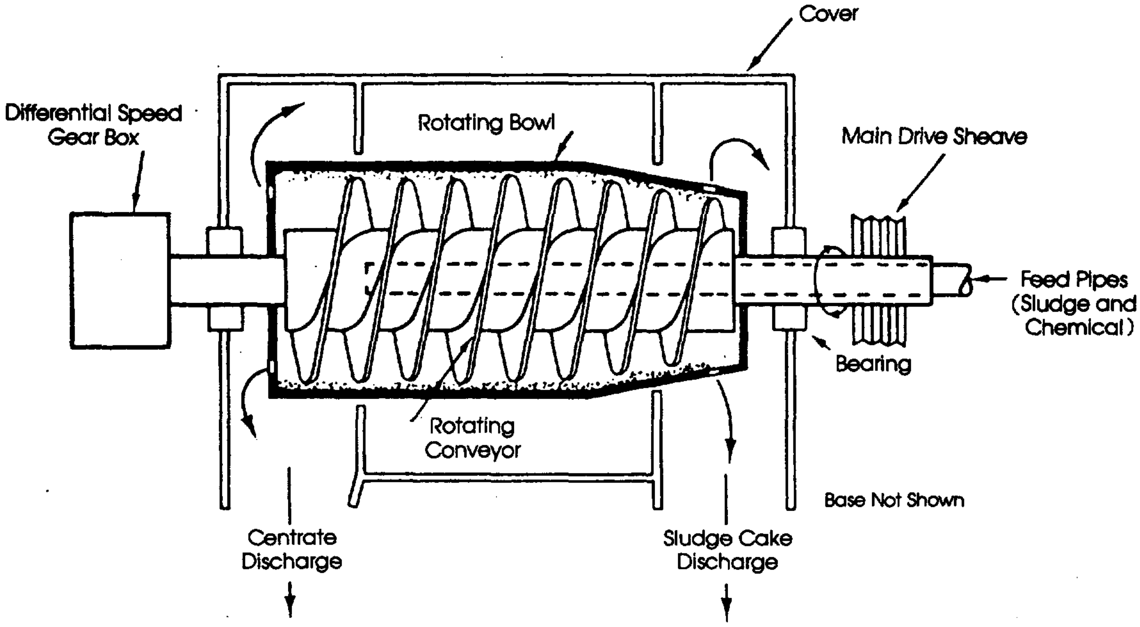
- bowl diameter
- bowl rotational speed
- beach length
- scroll rotational speed
- feed point of sludge
- condition of scroll blades
- bowl length
- beach angle
- pool depth
- scroll pitch
- feed point of chemicals

At any given pool depth, an increase in bowl speed provides more gravity-settling force, providing greater clarification. Typical G values for a solid bowl machine for many years were about 3,000. In recent years, units which operate at 700 G have been developed. These "low" speed units provide comparable results at lower power consumption. The introduction of polymers, has increased the range of materials that can be dewatered satisfactorily by centrifuges. The degree of solids recovery can be regulated over rather wide ranges depending on the amount of coagulating chemical used. Wetter sludge cake usually results from the use of flocculation aids because of the increased capture of fines. An increase in bowl speed will sometimes decrease polymer requirement, increase solids recovery and sometimes produce a drier cake. The bowl speed should not be operated at a higher speed than necessary due to the increased O&M cost.

In most centrifuge designs the differential speed between the bowl and the scroll can be varied without disassembly of the machine. The differential speed controls the solids residence time within the centrifuge and thus,

FIGURE 6-36

Continous Countercurrent Solid Bowl Conveyor Discharge Centrifuge



it can greatly influence cake concentration and machine capacity. Increasing the differential between the bowl speed and the scroll speed normally results in a wetter sludge cake and higher machine throughput. Conversely, a decrease in the differential speed produces a drier cake and decreased machine throughput and may result in poorer recovery or higher polymer dosage to maintain recovery.

Although the pool depth is variable on solid-bowl centrifuges, several hours of labor may be required to adjust the overflow weirs. As such, it is not a popular method of operational control. The pool depth regulates both the quality of clarification and the dryness of solids. Thus, while increasing the pool depth will normally result in better solids recovery at a specific feed rate, the cake produced will be wetter.

Extensive field trials were conducted at the Littleton/Englewood, Colorado Wastewater Treatment Plant with a high-speed and a low-speed centrifuge for both thickening waste activated sludge and dewatering a digested combination of primary plus waste activated sludges (USEPA 1987). The results of these tests are shown in Table 6-36. The high speed centrifuge (HSC) with a bowl diameter of 0.43 m and a length of 1.26 m was able to thicken the WAS to a 7 % solids concentration without any chemical conditioning. The low speed centrifuge (LSC) with a bowl diameter of 0.46 m and a length of 1.35 m, however, required $6.10 of chemicals per tonne of dry sludge solids to achieve a solids concentration of 7.0 % at a higher recovery. The results on the digested sludge were very similar. However, it was necessary to operate the LSC at higher than the customary G force of 1000-14000 G's to achieve equivalent performance.

Performance characteristics of solid bowl centrifuges with a variety of sludges are shown in Table 6-37 (USEPA 1987). The higher cake concentrations are generally achieved with the more favorable ratio of primary sludge.

Typical costs in 1978 for centrifugation are presented in Table 6-38 based upon 4 % influent solids concentration. Electrical requirements are a function of the bowl speed but are typically 22-66 kwh/tonne of solids.

Centrifuges have the advantage of being a totally enclosed process with limited space requirements.

(iv) Belt Press Filtration

Belt filter presses employ single or double moving belts to continuously dewater sludges through one or more stages of dewatering. Belt filter presses and solid bowl centrifuges have recently become the most frequently selected dewatering devices. All belt press filtration processes include three basic operational stages; chemical conditioning of the feed sludge, gravity drainage to a nonfluid consistency, shear and compression dewatering of the drained sludge. Figure 6-37 depicts a simple belt press and shows the location of the three stages. The dewatering process is made effective by the use of two endless belts of synthetic fiber. The belts pass around a system of rollers at constant speed and perform the function of conveying, draining and compressing.

TABLE 6-36

Centrifuge Tests at Littleton/Englewood STP, Colorado

	HSC		LSC	
	TWAS	D(P+WAS)	TWAS	D(P+WAS)
Feed - gpm	50	60	50	60
Cake - % TS	7	18	7	17
Recovery - % TSS	75	90	85	95
Polymer - $/TDS	0	16.00	5.50	16.00
Gravities	2500	2500	1185	1800
Power - kwh/TDS	400	52	300	40
Operating Costs - $/TDS	17.0	22.00	15.00	20.00

HSC - 16.75" O x 49.5" L

LSC - 18" O x 53" L

1 in = 2.54 cm

TABLE 6-37

Centrifuge Performance Characteristics

Sludge	Cake	Solids Recovery	Polymer*	
	% TS	%	kg/kT	$/kT
RP	28-34	90-95	1-2	5-10
DP	26-32	90-95	2-3	10-15
RWAS	14-18	90-95	6-10	30-50
DWAS	14-18	90-95	6-10	30-50
P(P + WAS)#	18-25	90-95	3-7	15-35
D(P +WAS)	17-24	90-95	3-8	15-40
$AL(OH)_3 + ALPO_4$	12-16	90-95	1-3	5-15
$Fe(OH)_3 + FEPO_4$	12-16	90-95	1-3	5-15
EAS	12-16	90-95	6-10	30-50
$Ca_x (PO_4)_x$	12-80	90-95	1.5-3	7-15
$CaCO_3$	40-50	90-95	0	0

R = Raw; D = Anaerobically Digested; P = Primary; WAS = Waste-activated Sludge; EAS = Extended Aeration or Aerobically Digested Sludge (SRT 25 days)
* Based pm $5/kg
\# 40:60 to 60:40 P:WAS mixture

TABLE 6-38

Estimated Costs for Dewatering By Centrifuge

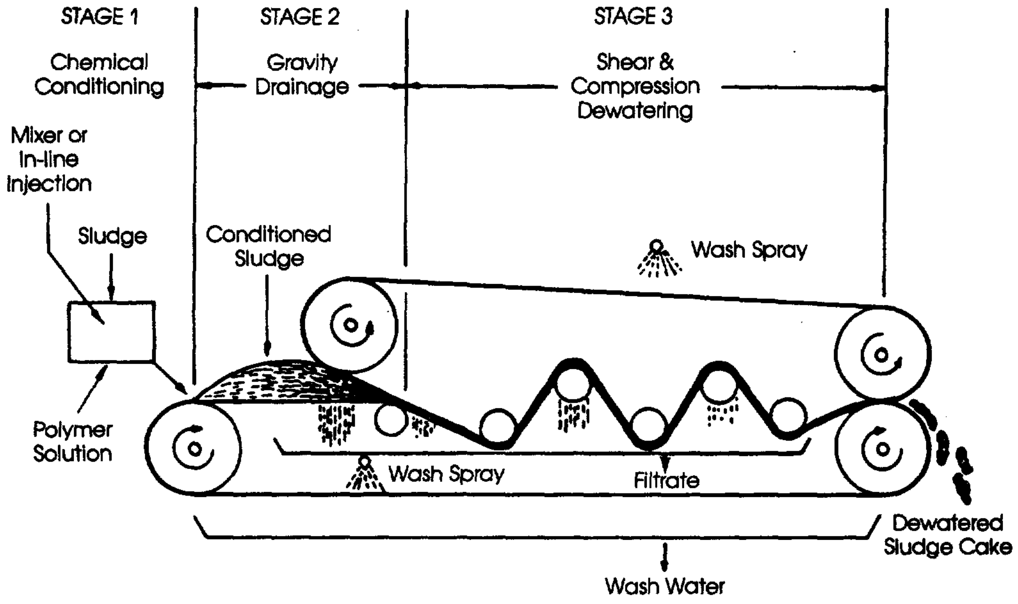
	Tons Per Day of Dry Solids			
Item	10	25	50	100
Construction Cost	$360,000	$580,000	$860,000	$1,400,000
Engr., Legal, Adm., & Interest During Construction (19%)	68,400	110,200	163,400	266,000
Total Capital Cost	$428,400	$690,200	$1,023,400	$1,666,000
Labor, Operation at $10/hr.	25,600	50,400	96,000	184,000
Labor, Maintenance at $10/hr.	6,200	13,600	22,400	45,600
Maintenance materials:				
Chemical	36,000	80,000	160,000	270,000
Other	27,000	52,000	85,000	140,000
Annual O & M Costs	94,800	196,000	363,400	639,600
Annual Capital Costs (x .0944)	40,400	65,200	96,600	157,300
Total Annual Cost	$135,200	$261,200	$460,000	$796,900
Cost per ton of dry solids	$37.00	$28.60	$25.20	$21.80

1 ton = 0.907 tonne

Assumed: 4% solids in influent

FIGURE 6-37

The Three Basic Stages of a Belt Filter Press



Good chemical conditioning is very important for successful and consistent performance of the belt filter press. A flocculant (usually an organic polymer) is added to the sludge prior to its being fed to the belt press. Failure of the chemical conditioning process to adjust to changing sludge characteristics can cause operational problems. If sludge is underconditioned, improper drainage occurs in the gravity drainage section, and either extrusion of inadequately drained solids from the compression section or uncontrolled overflow of sludge from the drainage section may occur.

Belt press performance is measured by the percent solids of the sludge cake, the percent solids capture, the solids and hydraulic loading rates, and the required polymer dosage. Several machine variables including belt speed, belt tension, and belt type influence belt press performance.

Belt speed is an important operational parameter which affects cake solids, polymer dosage, solids recovery, and hydraulic capacity. Low belt speeds result in drier sludge cakes. At a given belt speed, increased polymer dosages result in higher cake solids. With an adequate polymer dose, solids recoveries are improved by lowering belt speeds. Hydraulic capacity increases at higher belt speeds; however, the solids capture drops. Depending on desired performance, the belt speed setting can be used to produce a variety of different results.

Belt tension has an effect on cake solids, maximum solids loading, and solids capture. In general a higher belt tension produces a drier cake but causes a lower solids capture, at a fixed flow rate and polymer dose. A possible drawback of using higher tension is increased belt wear. For sludges with a large quantity of WAS, the belt tension must be reduced to contain the sludge between the belts. The maximum tension which will not cause sludge losses from the sides of the belts should be used. The high pressure zones on belt presses may cause problems with some WAS blends and may be unusable or require the lowest pressure setting possible.

Belt type is important in improving overall performance. Most belts are woven of polyester filaments. Belts are available with weaves of different coarseness and different strengths. A belt with a coarser and stronger weave may require higher polymer dosages to obtain adequate solids capture. The belt does require washing and flow rate required for belt washing is usually 50 to 100 % of the flow rate of sludge to the machine and pressure is typically 690 kPa or more.

Typical performance data for dewatering various types of sludges are shown in Table 6-39 (USEPA 1987). In general the sludge cake produced by dewatering on a belt filter press has a high enough solids concentration for any type of final disposal method including incineration. Belt filter presses have very low power requirements. The average power requirement is 8.2 kW/m belt width which is considerably lower than that of other types of mechanical dewatering equipment. Some units are as low as 0.8 kW/m belt width. Manpower requirements are also low.

Capital and O&M costs for the belt filter press tend to be similar to those for the solid bowl centrifuge. Where side by side tests have been

TABLE 6-39

Typical Performance Data for Dewatering Various Types of Sludges on Belt Filter

Type of	% Feed	Solids Loading Rate		Polymer	Dose	% Cake
Sludge	Solids	kg/hr/mbw	(lb/hr/mbw)	g/kg	(lb/dt)	Solids
Raw Sludges						
P	3-10	360-680	(800-1500)	1-5	(2-9)	28-44
WAS	0.5-4	40-230	(100-500)	1-10	(2-20)	20-35
P+WAS	3-6	180-590	(400-1300)	1-10	(2-20)	20-35
P+TF	3-6	180-590	(400-1300)	2-8	(3-15)	20-40
Anaerobically Digested Sludges						
P	3-10	36-590	(800-1300)	1-5	(2-10)	25-36
WAS	3-4	40-136	(100-300)	2-10	(4-20)	12-22
P+WAS	3-9	180-680	(400-1500)	2-8	(3-15)	18-44
Aerobically Digested Sludges						
P+WAS	1-3	90-230	(200-500)	2-8	(4-15)	12-20
P+WAS	4-8	135-230	(300-500)	2-8	(4-15)	12-30
Oxygen Activated Sludges						
WAS	1-3	90-180	(200-400)	4-10	(8-20)	15-23
Thermally Conditioned Sludges						
P+WAS	4-8	290-910	(600-2000)	0	(0)	25-50

performed and/or process feasibility studies made. It has been found that where in one instance the belt filter came out more expensive than the solid bowl centrifuge; in another situation the opposite held true.

(v) Recessed Plate Filter Press

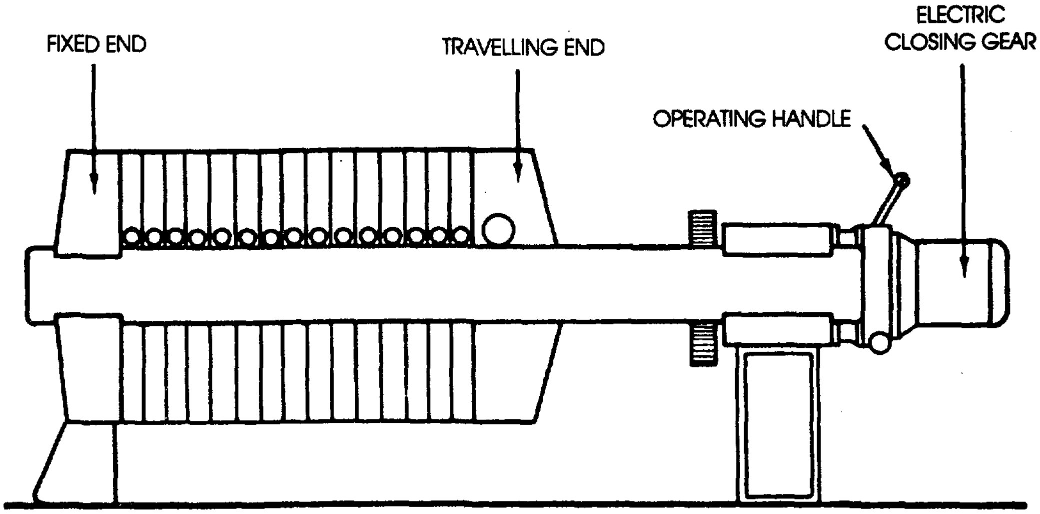
During the last several years there has been an increase in use of pressure filtration systems in treatment plants. Improvements in the equipment involved coupled with increasing quantities of difficult-to-dewater sludges account for the increase.

The filter press is a batch device, which has been used in industry and in European wastewater plants for many years. There are several variations in mechanical design and operating pressures. For purposes of illustrating the concept, a fixed volume vertical recessed plate filter press system will be described. Such a press consists of vertical plates which are held rigidly in a frame and which are pressed together between a fixed and moving end as illustrated in Figure 6-38. On the face of each individual plate is mounted a filter cloth. The sludge is fed into the press and water passes through the cloth, while the solids are retained and form a cake on the surface of the cloth. Sludge feeding occurs at pressures up to 1600 kPa and is stopped when the cavities or chambers between the trays are completely filled. Drainage ports are provided at the bottom of each press chamber. The filtrate is collected in these, taken to the end of the press, and discharged to a common drain. At the commencement of a processing cycle, the drainage from a large press can be in the order of 7.60 to 11.40 cubic meters per hour. This rate falls rapidly to about 1.9 cubic meters per hour, as the cake begins formation. When the cake completely fills the chamber, the rate is virtually nothing. The dewatering step is completed when the filtrate is near zero. At this point the pump feeding sludge to the press is stopped and any back pressure in the piping is released through a bypass valve. The electrical closing gear is then operated to open the press. The individual plates are next moved in turn over the gap between the plates and moving end. This allows the filter cakes to fall out. The plate moving step can be either manual or automatic. When all the plates have been moved and the cakes released, the complete pack of plates is then pushed back by the moving end and closed by the electrical closing gear. The valve to the press is then opened, the sludge feed pump started, and the next dewatering cycle commences.

There is also a variable volume recessed plate or diaphragm volume filter press, where sludge is pumped into the press at a low pressure (about 690 kPa) until the volume of the press has been filled with a loosely compacted cake. Then sludge pumping is stopped, and the diaphragm is inflated by pumping either air or water into the diaphragm at pressures up to between 1,480 kPa and 1,965 kPa for a preset time. For the diaphragm press, although most of the water removal occurs when sludge is being pumped into the press, a significant quantity of water is also removed after the diaphragm is inflated. When the diaphragm is deflated and the press opened, the cake drops out. Key advantages of the diaphragm include the fact that a dryer cake with a relatively uniform moisture content is produced and an overall shorter cycle time is needed and therefore a higher production throughput.

FIGURE 6-38

Side View of a Filter Press



Filter presses are normally installed well above floor level, so that the cakes can drop onto conveyers or trailers positioned underneath the press. The pressures which may be applied to a sludge for removal of water by the filter presses now available range from 5,000 to 20,000 times the force of gravity. In comparison, a solid bowl centrifuge provides forces of 700-3,500 g and a vacuum filter, 1,000 g. As a result of these greater pressures, filter presses may provide higher cake solid concentrations (30-50 % solids) at reduced chemical dosages. In some cases, ash from a downstream incinerator is recycled as a sludge conditioner.

Table 6-40 presents typical results from pressure filtration. As is readily apparent, the process produces a drier cake than either vacuum filtration or centrifugation. Table 6-41 presents illustrative costs for the assumed conditions as defined in the Table. Although the dewatering costs are higher than for vacuum filters or centrifuges under comparable conditions, the drier cake produced may result in savings in the downstream processes which are more than adequate to offset the higher costs. A diaphragm press has a higher initial cost than does a fixed volume press. Electrical consumption is primarily a function of influent solids concentration and ranges from 110-220 kwh/tonne of solids at 4 % influent solids to 55-110 kwh/tonne at 8 % influent solids.

(vi) Drying Lagoons

Lagoon drying is a low cost, simple system for sludge dewatering. Drying lagoons are similar to sandbeds in that the sludge is periodically removed and the lagoon refilled. Lagoons have seldom been used where the sludge is never removed, because such systems are limited in application to areas where large quantities of cheap land are available. Sludge is stabilized to reduce odor problems prior to dewatering in a drying lagoon.

Odor problems can be greater than with sandbeds, because sludge in a lagoon retains more water for a longer period than does sludge on a conventional sand drying bed.

Other factors affecting design include consideration of groundwater protection and access control. Major design factors include climate, subsoil permeability, lagoon depth, loading rates, and sludge characteristics.

Solids loading rates suggested for drying lagoons are 35 to 38 kg/m3/a of lagoon capacity. Other recommendations range from 0.1 m2/capita for primary digested sludges in an arid climate to as high as 0.3 to 0.4 m2/capita for activated sludge plants where the annual rainfall is 0.9 m. A dike height of about 0.6 m with the depth of sludge after decanting of 0.4 m has been used. Sludge depths of 0.8 to 1.2 m may be used in warmer climates where longer drying periods are possible.

Sludge will generally not dewater in any reasonable period of time to the point that it can be lifted by a fork except in an extremely hot, arid climate. If sludge is placed in depths of 0.4 m or less, it may be removed with a front-end loader in 3 to 5 months. When sludge is to be

TABLE 6-40

Typical Results
Pressure Filtration

Sludge Type	Conditioning	Percent Solids To Pressure Filter	Typical Cycle Length	Percent Solids Filter Cake
Primary	5% $FeCl_3$, 10% Lime	5	2 hours	45
	100% Ash		1.5	50
Primary + $FeCl_3$	10% Lime	4*	4	40
Primary + 2 stage high lime	None	7.5	1.5	50
Primary + WAS	5% $FeCl_3$, 10% Lime	8*	2.5	45
	150% Ash		2.0	50
Primary + (WAS + $FeCl_3$)	5% $FeCl_3$, 10% Lime	8*	3	45
(Primary + $FeCl_3$) + WAS	10% Lime	3.5*	4	40
WAS	7.5% $FeCl_3$, 15% Lime	5*	2.5	45
	250% Ash		2.0	50
WAS + $FeCl_3$	5% $FeCl_3$, 10% Lime	5*	3.5	45
Digested Primary	6% $FeCl_3$, 30% Lime	8	2	40
Digested Primary + WAS	5% $FeCl_3$, 10% Lime 100% Ash	6-8*	2	45
Digested Primary + (WAS + $FeCl_3$)	5% $FeCl_3$, 10% Lime	6-8*	3	40
Tertiary Alum	10% Lime	4*	6	35
Tertiary Low Lime	None	8*	1.5	55

*Thickening used to achieve this solids concentration

TABLE 6-41

Estimated Costs for Dewatering by Filter Press

Tons Per Day of Dry Solids	10	25	50	100
Construction Cost	$820,900	1,085,000	2,170,000	3,498,600
Engr., Legal, Adm., and Interest During Construction (19%)	156,000	206,200	412,300	664,700
Total Capital Cost	$976,900	1,291,200	2,582,300	4,163,300
Labor, O & M at $10/hr.	76,000	120,000	180,000	310,000
Chemicals	46,000	100,000	180,000	310,000
Electric Power @ $0.025/Kwh	3,500	6,500	10,500	16,800
Sidestream Treatment, $8.80/T	32,000	80,000	160,000	320,000
Maintenance Materials as 3% of Capital Costs/yr.	29,300	38,700	77,500	124,900
Annual O & M Costs	186,800	345,200	608,000	1,081,700
Annual Capital Costs (x .0944)	92,200	121,900	243,800	393,000
Total Annual Cost	$279,000	$467,100	$851,800	$1,474,700
Cost per Ton of Dry Solids	$76.40	$51.20	$46.70	$40.40

1 ton = 0.907 tonne

Assumed: 5% $FeCl_3$, 10% Lime for Conditioning
2.4 hour cycle time
Cake Solids = 40%
4% influent solids

used for soil conditioning, it may be desirable to stockpile it for added drying before use. One proposed approach utilizes a 3-year cycle in which the lagoon is loaded for 1 year, dries for 18 months, is cleaned, and allowed to rest for 6 months. Definitive data on lagoon drying are scarce. Sludge may be dewatered from 5 % solids to 40 to 45 % solids in 2 to 3 years using sludge depths of 0.6 to 1.2 m.

Limited cost data on large lagoon systems indicate 1978 costs in the range of $11-22/tonne exclusive of land costs.

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CHAPTER 7 - Technical Requirements for the Safe Disposal of Hazardous Wastes

Introduction

Hazardous waste management involves prevention, treatment and disposal. Prevention includes both waste and volume reduction. One aim of any hazardous waste management strategy should be to reduce to a minimum the quantity of hazardous waste that must be managed and that requires disposal. Treatment and disposal techniques render the wastes less hazardous and dispose of them in a manner such that environmental and human health problems do not occur. Land disposal of hazardous waste can be an economically and technically sound management approach but it requires an engineered design to control pollutants. A good understanding of the stringent requirements of proper landfilling is of prime importance for the management of hazardous wastes. Treatment prior to the land disposal of hazardous wastes can be accomplished by: (a) detoxification of the wastes, (b) separation and concentration of the hazardous constituents in a reduced volume, and (c) stabilization and solidification of the waste to inhibit leaching.

Land disposal currently comprises the following processes which are discussed in this section.

Landfills (Section 7.1)

In a number of countries where controls over hazardous wastes are either recent or have not yet even been introduced, the general practice is still characterized by the uncontrolled disposal of hazardous wastes on municipal or other landfill sites.

As a result of serious problems with old and abandoned dump sites many countries today favor an "engineering concept(s) landfill," where hazardous wastes are segregated in a site which is selected for its natural containment properties and then engineered so as to isolate the wastes as far as possible from the environment. Any leachate generated by the waste is collected and treated, and both ground and surface water in the vicinity of the site are monitored.

In some countries, notably the USA, technical standards for site construction (e.g., in terms of providing a double liner and an impermeable cover, are stringent). There are also standards for monitoring, which is required for at least 30 years after closure. Insurance coverage to pay for any necessary corrective action is a current issue in a number of countries, and is already mandatory in the USA.

Controlled Co-disposal (Section 7.1.6)

Carefully controlled codisposal is advocated in the United Kingdom as an alternative concept of landfill. It is argued that properly controlled codisposal of selected hazardous wastes up to a maximum loading rate can result in the degradation of certain organic contaminants or the attenuation of inorganic contaminants to background levels. In either case, the ideal result is that the waste is no longer hazardous. The proponents of codisposal thus claim that it is less likely to lead to future problems of contaminated sites than the alternative policy of deliberately segregating and entombing hazardous wastes, which is, in effect, a policy of controlled long-term storage.

Surface Impoundments (Section 7.3.1)

Aqueous wastes may be treated in surface impoundments such as pits, ponds and lagoons. This practice may be regarded either as storage or as a form of waste treatment, allowing the settling of solids and perhaps some biological degradation prior to discharge of effluents to surface waters. Surface impoundments are used primarily in the United States, where they are now subject to control as are other forms of land disposal.

After treatment in surface impoundments, wastewater treatment or physical-chemical treatment facilities, the aqueous effluents are generally discharged to sewers or directly to surface waters. In most countries such effluents are not regarded as hazardous wastes.

Land Treatment (Land farms) (Section 7.2)

Land-farming comprises the mixing of sludges into the top surface of the soil, usually by ploughing. The method was developed for refinery sludges in the United States, but has also found some application in Europe. Land-farming of oily sludges is a treatment method aimed at the biological breakdown of the sludge. The land is not used for agriculture. This is in contrast to the application of sewage and other biological treatment sludges to farmland, which is a common practice in a number of countries. Control is necessary when the sludges are contaminated with heavy metals. Under carefully controlled conditions, wastes, in addition to oily sludges that have been successfully land treated, include: pesticides/herbicides like aldrin, dieldrin, parathion, malathion, 2, 4-D, DDT, kepone, and piperonylic acid; and other chemicals like ethylbenzene, pentachlorophenol, and pulp mill lignins.

Sub-Surface Disposal (Section 7.3.2)

Three forms of sub-surface disposal may be distinguished: deep-well injections; disposal of liquid wastes via mine shafts; and deep burial. Deep well injection of hazardous wastes is mainly confined to the United States. Some 30 million tons of aqueous wastes--11% of all hazardous wastes--were disposed of by this method in 1981. However, under new regulations deep well injection is expected to be phased out.

Disposal of liquid waste to mine shafts is practiced to a limited extent in the United Kingdom. The technique includes the re-injection of saturated brine and certain other liquid wastes into salt cavities. A major commercially available facility disposes of aqueous wastes contaminated with organic materials in a mine shaft which is claimed to be totally sealed. In Spain a gypsum mine is used for the disposal of dewatered residues from chemical treatment.

Deep burial of wastes is exemplified by the long-term storage of containerized wastes in salt mines. The major commercial facility in West Germany receives about 50,000 tons per annum of particularly difficult and toxic wastes from most European countries. The placement of each individual shipment within the mine is recorded, so that wastes can be recovered later for recycling if appropriate.

Land disposal of hazardous wastes represents the permanent placement of solid, liquid, semi-solid, or contained gases in or on the land. It is expected that a portion or all of the wastes will be present at the site at closure. Unless the waste is totally and permanently contained, a portion of the wastes can be expected to migrate from the location at which they are placed. It is this migration of (a) components of the original waste or (b) decomposition or reaction by-products as runoff, leachate or gaseous emissions that must be controlled at a land disposal site.

Careful evaluation of the relative risk associated with the land disposal of hazardous wastes is needed. Table 7-1 indicates determinations that are important when land disposal alternatives are being considered and evaluated.

Figure 7-1 illustrates some of the environmental concerns that can occur at a land disposal site. As precipitation and surface water percolate through the disposal area, contaminants can be solubilized and carried to the water table where they are transported through the groundwater. The contaminated leachate will exist as a plume in the groundwater because of incomplete mixing and diffusion. Transportation to and through the saturated zone can be slow. Clay soils retard such transport to a greater extent than sands or gravel. If drinking water or irrigation wells intercept the contaminated leachate or if the leachate enters the surface waters, adverse environmental and public health impacts can occur. Surface waters can also be contaminated by runoff from the disposal site.

Other adverse environmental impacts can occur. Air may be contaminated through volatilization of waste components or emission of gases from the site and by windborne contaminated particles. In addition, vegetation growing on the site may be contaminated by waste that adheres to leaves and by uptake of constituents such as metals and other chemicals.

A hazardous waste land disposal facility must be designed and operated to avoid human health exposure and to minimize migration of contaminants from the site. Emphasis is placed on approaches that reduce the possibility of contaminating surface or ground waters, that control gaseous emissions and wind erosion, and that eliminate adverse food chain impacts. These approaches involve one or more of the following: (a) a natural impermeable containment possibly reinforced by a man-made impervious liner, (b) diversion of off-site surface run-on, and control of any on-site runoff, (c) incorporation of the wastes in the soil, (d) an impermeable cover for landfills and (e) avoidance of food chain vegetation on the surface of the site.

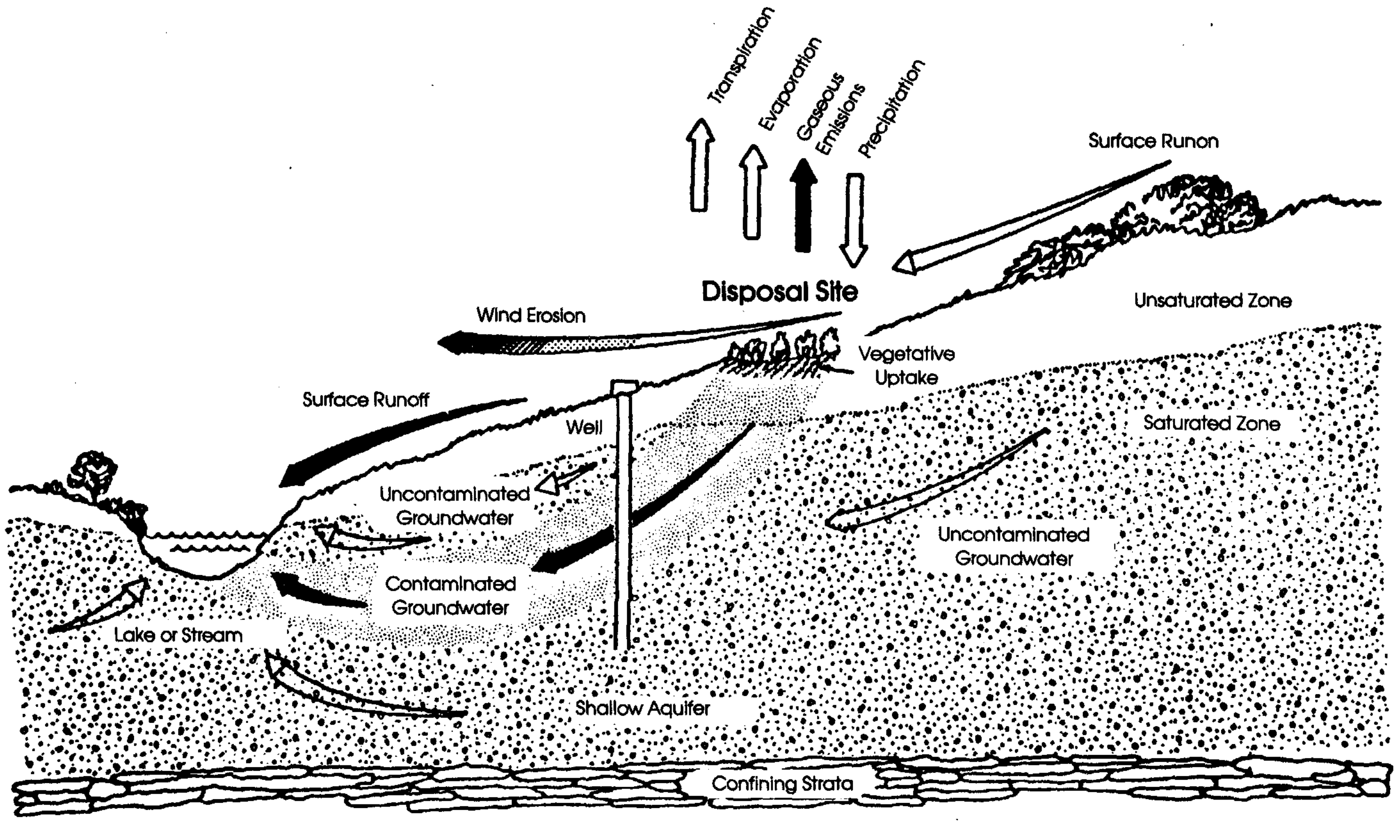
TABLE 7-1

Determinations Important to the Evaluation of a Potential Hazardous Waste Land Disposal Site

- Analysis of the Wastes to be Applied
- Identification of Reaction or Decomposition By-Products Expected to Occur
- Determination of Soil, Hydrogeological and Surface Characteristics
- Estimation of the Transport and Fate of Mobile Waste Constituents and By-Products
- Estimation of the Environmental and Health Impact of the Mobile Components if Such Components Reach Critical Receptors (Humans, Animals, Plants) in the Ecosystem
- Estimation of the Type and Amount of Food Chain, Human Health, and Environmental Exposure that May Occur

FIGURE 7-1

Contaminant Transport from a Land Disposal Site



7.1 Landfills

7.1.1 Description

A landfill is a disposal facility where hazardous wastes are placed into and stored in the soil. Landfills for hazardous wastes frequently are considered a technology of last resort to be used after every effort has been made to reduce or eliminate the hazards posed by the wastes. The intent is to bury or alter the wastes so that they are not environmental or public health hazards. Landfills are not homogeneous and are usually made up of cells in which a discrete volume of a hazardous waste is kept isolated from adjacent waste cells by a suitable barrier. Barriers between cells commonly consist of a layer of natural soil (clays) which restrict downward or lateral escape of the hazardous waste constituents or leachate.

Figure 7.1-1 presents a cross-section of a hazardous waste landfill. The daily, intermediate and final cover that represent proper operating conditions, the discrete cells of the landfilled material, and the use of liners and a leachate collecting system are portrayed. Discussion of liners, covers, operating conditions and closure and post-closure of landfills are discussed in subsequent sections.

Landfilling relies on containment rather than treatment or detoxification for control of hazardous wastes and technologically is an unsophisticated disposal method. Landfills are a common method of hazardous waste management for both untreated wastes and the residues from treatment technologies and require careful construction as well as continuous maintenance and monitoring.

The cross-section of a completed and closed secure landfill is shown in Figure 7.1-2. Appropriate liners to protect the groundwater from contaminated leachate, run-off control, leachate collection and treatment, monitoring wells and appropriate final cover design are integral components of an environmentally sound hazardous waste landfill.

Although there have been instances of groundwater contamination resulting from landfills, they remain a key hazardous waste strategy except in unsuitable hydrogeologic locations. The majority (about 68%) of hazardous wastes handled at North American facilities and about 47% of such wastes handled at European facilities are landfilled (Hrudey 1983).

In the United States, a hazardous waste landfill is designed and operated to be a secure permanent burial site for the hazardous waste. The design and construction of such a landfill attempt to prohibit contact between the landfill contents and the surrounding environment.

The primary concern at landfills is to prevent groundwater contamination. Design and management emphasize prevention of leachate formation and migration. Prevention methods include: (a) elimination of free liquids (liquid waste should be dewatered or solidified before placement), (b) diversion of surface waters (runon), (c) use of relatively impermeable

FIGURE 7.1-1

Schematic Cross-Section of a Landfill that Contains Hazardous Wastes

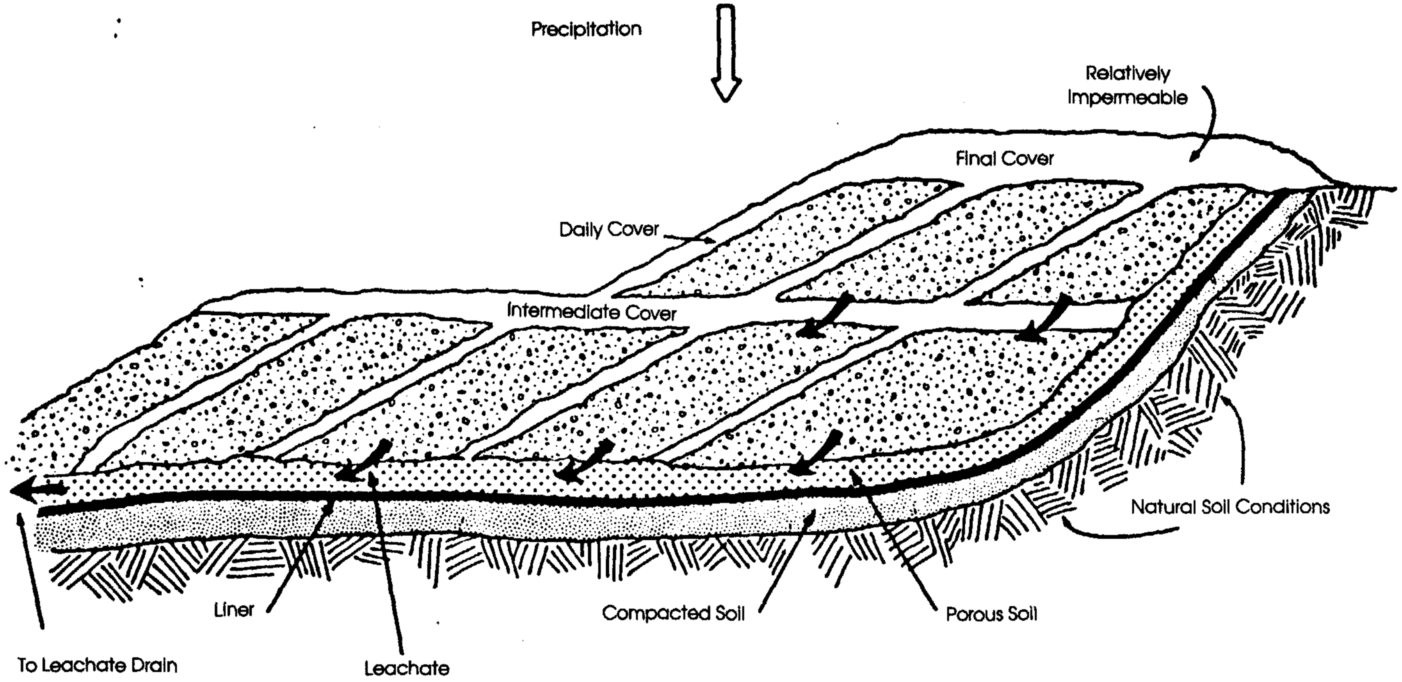
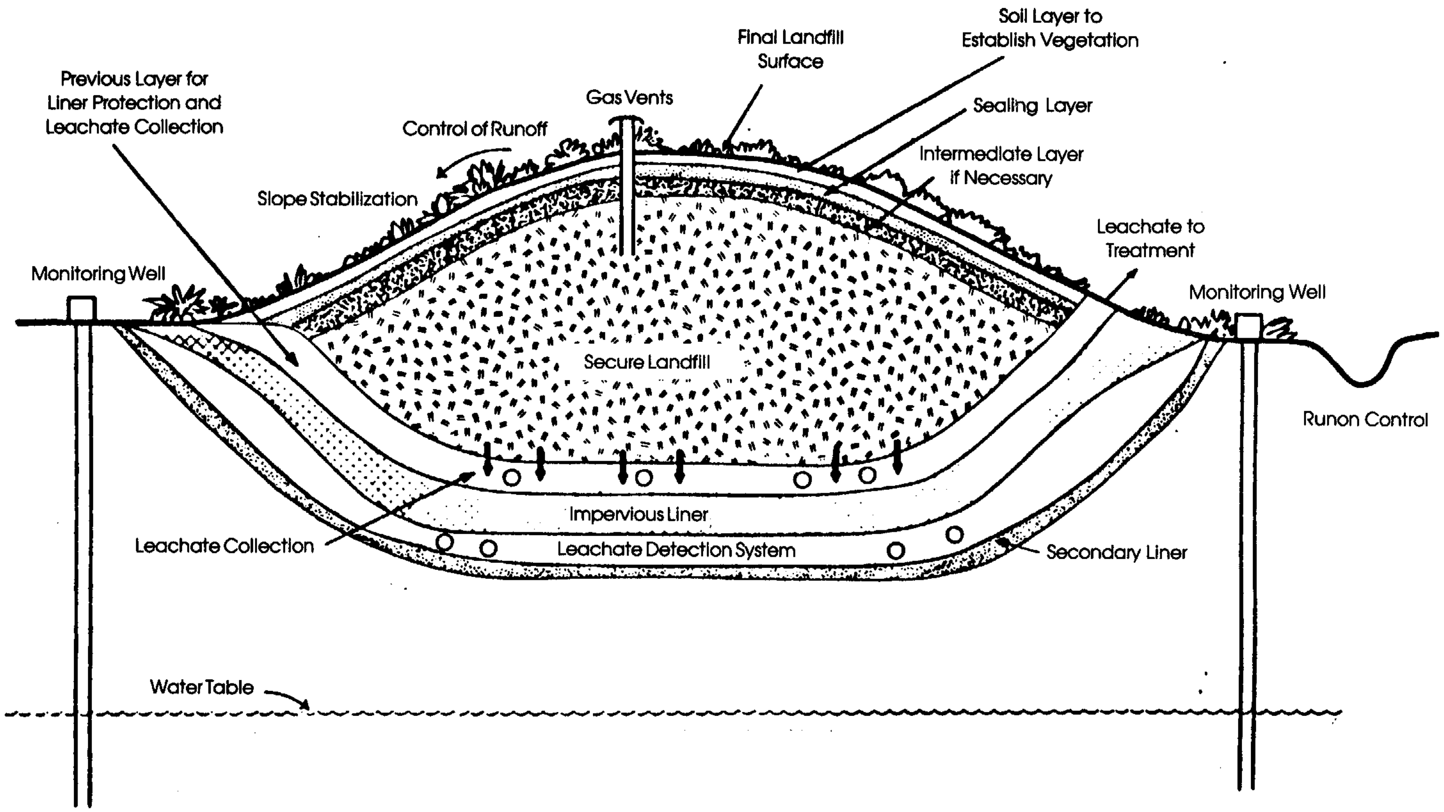


FIGURE 7.1-2

Schematic Cross-Section of a Secure Landfill



daily and final cover to minimize infiltration of precipitation, (d) compaction of wastes, (e) use of cells throughout the landfill, (f) collection and treatment of leachate, and (g) groundwater monitoring. Approaches to keep water out of landfills are noted in Table 7.1-1.

The ideal hazardous waste landfill is one which is underlain by many meters of impermeable clay in a non-seismic zone. Hazardous waste landfills should not be placed above a draining water aquifer.

Plans for adequate record-keeping should be made. The location and dimensions of each cell in the landfill should be recorded, as should the contents of each cell. For future needs, the location and dates of each type of waste material placed in the cells should be recorded.

An initial step in developing a secure landfill for hazardous wastes is to determine the type and characteristics of the material that will be placed in it. Information on the estimated waste volume, and the physical and chemical properties of the materials expected to be landfilled is important to the actual design and management, as well as to the determination of area and volume requirements and equipment needs.

7.1.2 Site Selection

(i) General Evaluation

The feasibility of land disposal for hazardous wastes depends on factors such as the type, amount and characteristics of the wastes, laws and regulations, public perception and acceptance, and the soil and site characteristics. This section focuses on those characteristics that are important when selecting a site for land disposal options. The design and management of hazardous waste land disposal options are site specific. The technical and economic feasibility of these options will depend on the topography, soils, climate and hydrology of the site, on transport distances from the waste source to the site and on current and projected land uses.

The site selection approach can be considered to have at least three major components: (a) general evaluation -- problem definition and initial solution evaluation, (b) detailed analysis of the feasible options; and (c) final site selection and design. The general evaluation screens out site alternatives that clearly are not feasible. The detailed analysis closely evaluates the site, soil, and groundwater characteristics, includes economic and political considerations, and narrows site options. Final site selection and design involves detailed economic, technical, and political evaluations of specific sites.

The general site evaluation is concerned with defining technical feasibility (e.g., is it worth considering these methods in greater detail or are other methods more appropriate). The general area requirements are identified after consideration of waste and site characteristics, and possibilities to modify waste characteristics to reduce the needed land area. With the general land area needs and locations identified,

TABLE 7.1-1

Approaches to Keep Water out of Hazardous Waste Landfills

- Proper siting to avoid wetlands, flood plains and areas of high groundwater
- Diversion of surface run-on
- Minimize exposed waste surfaces
- Avoid ponding of precipitation on the site
- Proper use of intermediate cover material
- Prompt covering and closing of inactive areas
- Proper closure and post-closure management

approximate costs can be estimated. Such costs and pertinent factors such as site ownership and obvious public reaction can be used to judge whether a site continues to be a possible hazardous waste disposal option. If a positive conclusion is reached, detailed technical, economic, and political analyses of the feasible sites can be made.

Because of the need to minimize pollutant migration and public and environmental risk, there may be only a few potential sites for evaluation. For many hazardous waste generating industries, such sites may be on their own property.

No site should be used for the landfill disposal of hazardous waste unless the geological and hydrogeological properties of the site have been carefully investigated and found to be satisfactory. In addition, the general evaluation should consider all of the general site selection criteria in Table 7.1-2.

A hazardous waste landfill should not be located in a floodplain or be in contact with groundwater. The seasonal height and flow of the groundwater should be established so that the potential for water contamination can be assessed and the location of monitoring wells be established.

The site should not be in an area of seismic activity or an area that includes cavities, faults or sinkholes. The integrity of the landfill may be compromised in such areas. The soils at the bottom of the hazardous waste landfill site should be clays and should be several feet thick and relatively impermeable, preferably with a permeability of no greater than 10^{-7} cm/sec. If such clay soils are not available, appropriate soils may have to be brought to the site to serve as liner material and/or a synthetic flexible membrane liner will have to be installed. Suitable soils should be available as landfill cover material or else appropriate soils will have to be brought to the site.

The site should not be located where normal runoff will inundate the site. Drainage from the surrounding land should be diverted around the site.

A detailed analysis is initiated after a land disposal option has been estimated to be potentially feasible. This analysis must: (a) evaluate critical site characteristics of each option, (b) identify the land disposal method to be investigated for each candidate site, (c) define the detailed design requirements for each land disposal method and candidate site, (d) develop a cost-effectiveness ratio for each alternative, (e) encourage public participation throughout the planning stages, (f) determine whether land disposal continues to be feasible and (g) identify a site or sites for intensive field investigation . Although this analysis primarily uses available published information, field evaluations are desirable.

The actual or projected use of the land at or adjacent to a potential site will impact the economic, political, and social aspects of site selection. These factors can be important constraints on the use of a possible site. Land use restrictions should be evaluated as part of the site selection process.

TABLE 7.1-2

Landfill Site Selection Criteria

Engineering

Physical site. Should be large enough to accommodate waste for life of production facility.

Proximity. Locate as close as possible to production facility to minimize handling and reduce transport cost. Locate away from water supply (suggested minimum 500 feet) and property line (suggested minimum 200 feet).

Access. Should be all-weather, have adequate width and loan capacity, with minimum traffic congestion.

Topography. Should minimize earth-moving, take advantage of natural conditions. Avoid natural depression and valleys where water contamination is likely.

Geology. Avoid areas with earthquakes, slides, faults, underlying mines, sinkholes, and solution cavities.

Soils. Should have natural clay liner or clay available for liner, and final cover material available.

Environmental

Surface water. Locate outside 100-year floodplain. No direct contact with navigable water. Avoid wetlands.

Groundwater. No contact with groundwater. Base of fill must be above high groundwater table. Avoid sole-source aquifer. Avoid areas of groundwater recharge.

Air. Locate to minimize fugitive emissions and odor impacts.

Terrestrial and aquatic ecology. Avoid unique habitat area (important to propagation of rare and endangered species) and wetlands.

Noise. Minimize truck traffic and equipment operation noise.

Land use. Avoid populated areas and areas of conflicting land use such as parks and scenic areas.

Cultural resources. Avoid areas of unique archaeological, historical and paleontological interest.

Legal/regulatory. Consider national, regional and local requirements for permits.

Public/political. Gain local acceptance from elected officials and local interest groups.

TABLE 7.1-2 (continued)

Economic

Property acquisition. Actual land cost plus related costs.

Site development. Excavation, grading, liner, new roads, and other development costs.

Annual costs. Fuel costs, operating labor, maintenance, land preparation, utilities, and overhead.

Salvage value. Do not consider; site probably will not be an asset.

Preferably, a land disposal operation should be located at the site of or immediately adjacent to the waste generator. However, for the majority of waste generators this is not feasible. If hazardous wastes must be transported to an off-site facility via public roads, rail systems or other means, the transporter must comply with applicable regulations concerning transport of hazardous materials.

(ii) Site Characteristics

There rarely is an ideal land disposal site and some characteristics that must be considered when selecting a site are summarized in Table 7.1-3. Unacceptable site characteristics for hazardous waste land treatment and disposal landfills are noted in Table 7.1-4.

When evaluating site suitability, the site characteristics can be identified from a soil survey. However, modern soil surveys are not available for many areas of the world and airphotos and information from remote sensing studies may provide pertinent information on site hydrogeology and soil features.

Landforms that generally are unsuitable for a hazardous waste disposal site are noted in Table 7.1-5. Other geologic features which should be identified during the site selection process include: (a) depth of subsurface soils: (b) bedrock outcrops: (c) type of bedrock; and (d) fissures, faults, fractures, crevices, joints, caves, springs, sinkholes, seeps, limestone cavities, or other irregularities in the bedrock.

Unsuitable Hydrogeologic Conditions

The human health and environmental impacts that may occur when there is a failure at a hazardous waste land disposal facility differ dramatically depending on the physical location of the facility. Some site hydrogeologic characteristics favor rapid release and transport of waste constituents and therefore have a high potential for adverse impacts. These hydrogeologic conditions should be avoided when siting such facilities.

Unsuitable sites are those that contain hydrogeologic conditions that are unstable or have the potential to damage containment structures. Unsuitable sites are those prone to: (a) floods, (b) seismic impacts, (c) volcanic impacts, (d) landslides, or (e) subsidence. Other unsuitable areas are those: (a) characterized by weak and unstable soils and (b) above aquifers used for drinking water.

Criteria for Suitable Sites

General criteria for identifying suitable sites for hazardous waste land disposal facilities exist (Table 7.1-6). Use of these criteria and available site-specific data will allow these facilities to be sited at locations where subsequent human health and environment problems will be minimal.

TABLE 7.1-3

Site Characteristics Important When Selecting Land Disposal Sites

- LAND USE

- HYDROLOGY
  - Surface
  - Subsurface

- GEOLOGY

- SOIL PROPERTIES
  - Physical
  - Chemical
  - Hydraulic

- TOPOGRAPHY

- CLIMATE
  - Wind
  - Temperature
  - Moisture

TABLE 7.1-4

Site Characteristics that are Unacceptable for Hazardous Waste Land Disposal Sites

- Geology
  - Bedrock Outcrops
  - Irregularities such as fissures or faults overlying groundwater
- Hydrology
  - Aquifer recharge zone
  - Flood prone areas
  - Wetlands
  - Seasonally high water tables
  - Near private or community water supply wells or reservoirs
- Climate
  - Extremely wet or cold conditions
- Topography
  - Overly steep slopes
  - Broken terrain
- Soils
  - Thin soil above groundwater
  - Highly permeable soils above shallow groundwater
  - Soils with extreme erosion potential
- Land Use
  - Areas formerly used for
  - landfills
  - Areas contaminated with persistent residues from past chemical spills or waste treatment processing.

Source: Ryan, J. R., and R. C. Loehr. 1981. Site selection methodology for the land treatment of wastewater. Special Report 81-28. Hanover, New Hampsire: U.S. Army Corps of Engineers, Cold Regions Research and Engineering Laboratory.

TABLE 7.1-5

The Suitability of Specific Land Forms as Land Disposal Sites*

Land Form	Comment	Suitability for Land Disposal Site
Moraine	Highly heterogeneous material	Detailed soil analysis required
Floodplain	Frequent flooding	Exclude
Drumlins	Limited areas and steep slopes	Exclude
Filled Valleys	Heterogeneous material, dependent on parent material and climate	Detailed soil analysis required
Delta	Frequent flooding	Exclude
Beach ridges	Limited area, high ground water	Exclude
Coastal plains	Highly heterogeneous material	Detailed soil analysis required
Tidal flats	Frequent flooding	Exclude
Sand dunes	Wind erosion, Unstable landform	Exclude

Source: Ryan, J. R., and R. C. Loehr. 1981. Site selection methodology for the land treatment of wastewater. Special Report 81-28. Hanover, New Hampshire: U.S. Army Corps of Engineers, Cold Regions Research and Engineering Laboratory.

TABLE 7.1-6

Criteria for Determining Suitable Sites for Hazardous Waste Land Disposal Facilities

Site Characterization

The inherent geologic, hydrogeologic and soil characteristics of the site can be fully characterized.

Location

The site provides a stable location of the land disposal facility and is not likely to be damaged by natural or man-made events. The site does not contain unsuitable hydrogeologic conditions.

Monitoring

All potential groundwater and other contaminant transport flow paths can be monitored.

Protected Land

The site is not in designated historic areas or parks or on significant agricultural land, significant wildlife refuges or habitats of endangered species.

Source: U.S. Environmental Protection Agency. 1985. Permit Writers Guidance Manual for the Location of Hazardous Waste Land Storage and Disposal Facilities: Phase 1, Criteria for Location Acceptability and Existing Regulations for Evaluating Locations. Washington, D.C.: Office of Solid Waste.

(iii) Landscape

Landfilling inevitably has a visual impact. Sites in scenically attractive areas and in designated areas such as National Parks or Areas of outstanding natural beauty or sites that contain features of special interest or are of special scientific interest must be avoided. A landfill site should intrude as little as possible on the landscape during its operational life and should not result in the unnecessary loss of features of interest and significance. Before commencing landfill operations, plans should be approved to avoid adverse visual impacts and to ensure that restoration progresses as quickly as possible. A landfill site may be screened from areas of public access by either a tree or shrub planting scheme or by using natural ground contours to assist in visual screening. A restored landfill should quickly become an integral part of the landscape.

However, a sensible balance has to be struck. For example if a site contains a geological outcrop of significant interest, it may be possible to preserve it only if the exposure is near the surface of the site or alternatively in an area of the site not to be landfilled. A decision therefore must be made on whether waste disposal or site preservation is to prevail. Such a decision must take account of the location of the site in terms of the waste disposal strategy, the importance of the features it is wished to preserve, and the economic and environmental consequences of alternative options.

(iv) Topography

The configuration of the landscape determines the pathway and rate of surface water movement and often subsurface water movement. The mode of waste transportation to the site also requires an evaluation of the topography. Both the topography of a potential site and that of the adjacent area should be evaluated. A level site adjacent to higher sloping land can receive considerable surface runoff and subsurface flow. Surface and subsurface hydrologic considerations can help evaluate such conditions and the need for and type of surface water diversion systems.

(v) Ecology

Past practices have located landfill operations in quarry or mineral excavations, on derelict land or less frequently on undisturbed ground or agricultural land. However, quarry or mineral excavations are usually poor locations for landfills due to the lack of suitable clay soils, hydrogeologic considerations, and difficulty of installing liners. All sites will differ in landscape and ecological values and in other site characteristics but for all of them a landfill operation is likely to have a major, and possibly irreversible, effect on the ecology of the site and its immediate area.

(vi) Local Community

One of the more obvious effects of landfill on a local community and one which is a continuing source of complaint is the additional traffic generated by its presence. Heavy lorry traffic gives rise to nuisance from noise, vibration, exhaust emissions, dust, dirt and visual intrusion. Whenever possible heavy vehicles should be routed to reach their destination by traveling on major roads. This can be achieved either by voluntary agreement, or specified in a waste disposal contract where one exists.

In endeavoring to achieve a good relationship with the local community and demonstrate his competence, the operator should give prompt and sympathetic attention to complaints. By dealing with them quickly the operator will demonstrate that he is determined to operate his site efficiently and is taking due care to safeguard the environment and the community.

7.1.3 Soil Properties and Contaminant Control Mechanisms

(i) Soil Properties

The soil properties requiring evaluation depend upon the waste constituents and the type of land disposal method. Measurement of soil properties should occur as part of the detailed analysis and the field investigations. General soil properties of interest include: permeability, pH, organic matter content, silt, sand and clay content, and cation exchange capacity. These and other relevant parameters are discussed in Section 7.1.2.

Data collected from a soil profile can be used to prepare a detailed soil map of the site. The steps that should be followed in acquiring specific field soils data, especially for landfill and land treatment system sites, are noted in Table 7.1-7.

(ii) Soil Contaminant Control Mechanisms

Knowledge of contaminant control and removal mechanisms in soil is important to decisions about the type of waste that should be disposed of on or in the land and to the proper design and operation of land disposal sites. The soil characteristics that exist at a site and the biological and chemical reactions that take place influence the mobility of constituents or by-products. For instance, in the certain soils wastes that have a high or low pH can be neutralized, inorganic constituents can be converted to less mobile toxic forms, and organic compounds can be degraded.

Soil consists of solids separated by pore spaces. The solid matter consists of mineral particles from rock fragments and organic matter from plant and animal decay. Microorganisms are also a component of the organic matter. In the top layer of soil, organic matter may comprise 1-10% of the solid matter. The pore spaces between the solids are filled with air or water and dissolved organics and inorganics.

TABLE 7.1-7

Acquisition of Field Site Information

Step 1

- Prepare soil maps from air photos, soil survey and topographic map overlays or other available information.
- Determine approximate soil boundaries and location of monitoring points.

Step 2

- Lay out grid at a site and determine elevation of individual grid points.
- Conduct soil borings and determine soil characteristics at designated grid points.
- Install temporary groundwater monitoring wells.

Step 3

- Develop topographic maps of soil surface elevations, impermeable horizon elevations, groundwater elevations and depth to bedrock.
- Analyze soil descriptions and correlate to known soil series.
- Develop soil map and indicate location of sampling units for each soil type.
- Determine the chemical and hydraulic parameters to be measured.

Step 4

- Collect samples for chemical analysis.
- Conduct hydraulic tests at selected grid points.
- Conduct deep borings to determine subsurface geology.

Source: Ryan, J. R., and R. C. Loehr. 1981. Site selection methodology for the land treatment of wastewater. Special Report 81-28. Hanover, New Hampshire: U.S. Army Corps of Engineers, Cold Regions Research and Engineering Laboratory.

The movement of water is a function of the nature and amount of pore space while the mobility of waste constituents is related to both the movement of water and the physical and chemical characteristics of the constituents and the soil. Soil characteristics are variable in both space and time. The percentage of inorganic matter, for instance, increases with depth.

Soil texture refers to the size of individual mineral particles. Table 7.1-8 lists the basic textural classes. Other terms commonly used to describe soils also are noted. By definition, sand must make up 70% or more of the weight of a sandy soil and a clay soil must contain at least 35% clay.

The size of particles in soils cannot be changed unless drastic physical or chemical processes are involved. As a result, the proportion of a particle size group in a soil (i.e., the texture) is a basic property of a soil. The size ranges of gravel, sand, silt and clay are presented in Table 7.1-8. To determine size of particles in a soil, the fine sand and larger fractions are separated into groups by the use of sieves. The silt and clay percentages are then determined by methods that depend on the rate of settling of the particles from a water suspension.

Permeability refers to the ease with which air and water move through the soil and is determined by soil pore size. Individual pores range in size from a few microns to several millimeters in diameter. Generally the finer the texture of the soil, the lower the permeability. Clays have a very low permeability (Table 7.1-9). Permeability is expressed in units of distance per unit time (inches or centimeters/hour). The common permeability classes are noted in Table 7.1-9.

Another important characteristic of soil is its capability to retain dissolved ions and compounds. The net charge on clay and humus particles is negative.

Positively charged ions (cations) such as ammonium, calcium, magnesium and sodium are attracted to and held by the clay and humus. The negative charge on soil organic matter, and to a minor extent, on clay particles is pH dependent with the net negative charge increasing with increasing pH.

As a result of this phenomenon, positively charged ions (cations) will be retained by soils with a high clay and humus content while negatively charged ions (anions), such as nitrate and chloride ions, will not be retained and will be mobile, moving with the water in the soil. The greater the cation exchange capability (CEC) of a soil, the greater the potential of the soil to retain charged waste constituents and the more effective the soil is for waste treatment and disposal. The CEC of a soil is expressed in terms of milliequivalents (meq) per 100 grams of soil (meq/100g) and is a function of the organic matter content and the type and amount of clay in the soil. The CEC of pure humus is about 200 meq/100g while the CEC of montmorillonite and kaolinite clays are about 90 and 80 meq/100g, respectively. The CEC of most soils ranges between 10 and 30 meq/100g.

TABLE 7.1-8

Soil Textural Classes and Common Terms Used to Described Soils

Texture	Common Terminology	Permeability Range (cm/hr)
Coarse	Sandy, sandy, loamy sandy	> 5
Moderately coarse	Sandy loam, fine sandy loam	1.6 TO 5
Medium	Very fine sandy loam, loam, silt loam, silt	1.6 TO 5
Moderately	Clay loam, sandy clay loam, silty clay loam	0.5 TO 1.6
Fine	Sandy clay, silty clay, clay	<0.10 TO 0.5

Particle	Size Range (mm)*
Gravel	
o coarse......................	.12.7 to 76.2
o fine........................	2.0 to 12.7
Sand	
o very coarse.................	1.0 to 2.0
o coarse.....................	0.5 to 1.0
o medium.....................	0.25 to 0.5
o fine.......................	0.1 to 0.25
o very fine..................	0.05 to 0.1
Silt.........................	0.002 to
Clay.........................	<0.002

* U.S. Department of Agriculture (USDA) Classification System

TABLE 7.1-9

Common Permeability Classes*

Class	**Permeability** (cm/hr)	(cm/sec)
Very Fast	>25.0	>7 x 10-3
Fast	12.5 to 25.0	3.5 to 7 x 10-3
Moderately Fast	6.25 to 12.5	1.7 to 3.5 x 10-3
Moderate	2.0 to 6.25	0.6 to 1.7 x 10-3
Moderately slow	0.5 to 2.0	1.4 to 6 x 10-4
Slow	0.125 to 0.5	3.5 to 14 x 10-5
Very Slow	<0.125	<3.5 x 10-5

*Classes used by the U.S. Soil Conservation Service (USDA)

Soil texture, permeability and soil CEC are the major factors affecting the movement of waste constituents through soils. The general relationship between soil types, permeability, and sorption capacity is indicated in Figure 7.1-3. There are, however, other factors that affect the transport and fate of constituents applied on or into the land. Basic soil science texts (Brady 1984) and reference documents (Page et al. 1983, and Loehr et al. 1979) should be consulted for details. Chemical reactions in the soil affect the mobility of dissolved ions or compounds with the result that some constituents are retained in the site indefinitely while others are more mobile. Adsorption and chemical precipitation are two important chemical reactions governing the movement of constituents at a land disposal site with the cation exchange capacity (CEC) being the most important absorption phenomena. Many of the potentially toxic metals such as cadmium (Cd), nickel (Ni), zinc (Zn) and copper (Cu) are positively charged and their immobilization and retention by soil is related to the CEC of the soil.

Because the CEC of a soil is generally correlated with the particle size distribution and type of soil (soil texture), the loading rate of metals in a waste applied to land has been related with this property. This has been done in establishing cumulative limits for metals in sludge applied to agricultural cropland (USEPA 1983, Table 7.1-8). The amount that may be applied is greater for soils with larger CEC values. The allowable amounts of the metals generally are in the following order: Pb> Zn> Cu> Ni> Cd. The maximum amounts of a waste constituent that should be applied is a function of the kind and nature of the waste and of the soil characteristics.

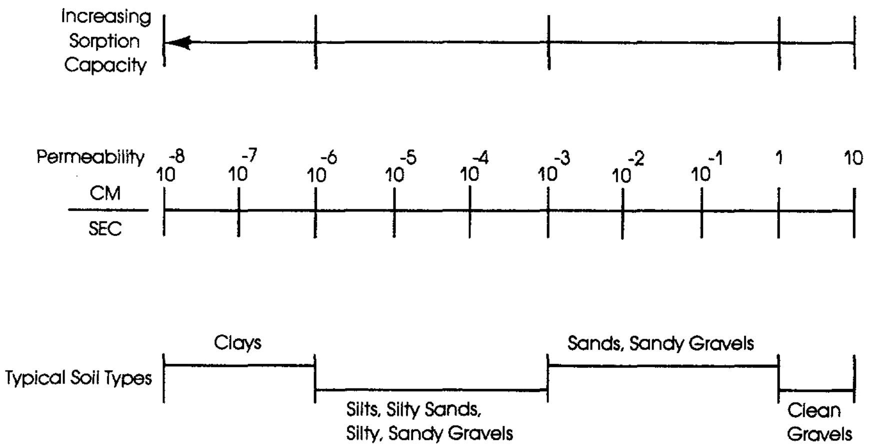
(iii) Microbial Decomposition

A paramount role in the reduction of organic soil contaminants is attributed to microbial decomposition. This is mostly utilized in land application of wastes, where the soil and wastes are most intimately mixed and optimum conditions are created for decomposition and destruction. Soil microorganisms can decompose organic matter in many wastes. Such degradation occurs by facultative or anaerobic organisms if the wastes are not toxic or inhibitory to the organisms. In general, soils have a high capacity for organic carbon decomposition. Degradable organic loading rates as high as 1100 kg/ha/day have been applied (Loehr et al. 1979) without adverse environmental conditions. Hazardous wastes, however, contain organics that are not readily degradable. Organics such as the chlorinated hydrocarbons are relatively resistant to decomposition in soils, unless soil organisms are preconditioned to the specific waste under carefully controlled conditions.

In a conventional sanitary landfill aerobic conditions will prevail initially. However, anaerobic conditions are rapidly established with methane and carbon dioxide being the final products of carbon metabolism. Carboxylic acids are major intermediates and can remain as the end products of carbon metabolism. The metabolism of landfilled domestic refuse is dominated by carbohydrates; these constitute about 50% of the dry weight while proteins and lipids account for only 5 to 10% of the dry weight of refuse in the United Kingdom.

FIGURE 7.1-3

Soil Permeabilities and Sorptive Properties of Soils



Most studies of landfill microbes are dominated by public health considerations. Attention has been focused on the presence of organisms involved in the decomposition of domestic refuse in a landfill. Animal or human pathogens in the applied wastes should not survive at a controlled hazardous waste land disposal site. Some pathogens could be in surface runoff, if such runoff occurs immediately after the waste is applied, but there should be little concern about pathogens at hazardous wastes disposal sites since such sites are designed to contain the applied wastes.

In-place density affects the decomposition processes. If wastes are deposited without compaction (0.35 ton/m^3), air ingress into the mass can readily occur. Decomposition is then essentially aerobic, the principal products being carbon dioxide and water with temperatures greater than 80^{o}C being observed. At refuse densities in excess of 0.5 ton/m^3 the dominant decomposition process becomes anaerobic with the appearance of carboxylic acids in leachate and methane in the gas phase. As the in-place density of waste is increased, water penetration will be more difficult and the overall moisture content will not rise significantly. Since the rate of decomposition increases with increasing moisture content, high compaction will tend to lower the rate of decomposition. However, in some situations compaction to a high density may mean that saturation occurs with lower liquid inputs. High levels of gas generation have been observed at in-place densities of 0.8t/m^3 or more.

7.1.4 Water Quality Protection

(i) Landfill and Water Quality

The infiltration of rainfall and surface water into a landfill coupled with biochemical and chemical breakdown of wastes produces a liquor or "leachate" high in suspended solids and with a high organic and inorganic content. If this leachate enters surface or groundwaters before sufficient dilution has occurred, it will cause pollution.

Water authorities, as well as waste disposal authorities, have a duty to ensure that landfill operations do not adversely affect water quality.

While water authorities should seek to provide maximum protection to waters used for potable supplies, they also have to safeguard other water interests. Water in undeveloped aquifers must be protected as if they were developed, unless the waters are permanently unusable. Water authorities, too, must be concerned with other aspects of landfill development, if these bear on their land drainage responsibilities.

Landfill Site Characteristics

Few sites are naturally ideal for landfill, but many that are not can with careful planning be engineered to be suitable, though at a cost. Landfill sites fall into a spectrum in terms of the extent to which leachate is contained with the landfill. It is convenient to distinguish two broad categories, that is 'attenuate and disperse' sites and 'containment sites' which correspond to two different approaches to the protection of water resources.

Attenuate and Disperse Sites

Such sites allow the slow release of leachate from the landfill and rely on various attenuation mechanisms operating within the body of the waste and in the unsaturated and saturated zones of the underlying strata to ameliorate the polluting characteristics of the leachate. The attention mechanisms include those of dilution and dispersion which help to decrease the effect of leachate on water resources. Geological formations best suited to attenuation and dispersal are those with significantly high amounts of clay minerals present and where leachate movement will be through pores or micro fissures. Attenuation may be improved at some sites by the use of permeable liners. Some landfill sites allow leachate migration at rates too fast for much attenuation to take place through chemical or biological processes, though dilution and dispersion in the underlying saturated zone will still be important. These sites in their natural state are usually suitable only for relatively inert wastes, but they can be engineered to accept other wastes.

Containment Sites

Containment sites are intended to isolate wastes and leachate from the environment for a considerable time (perhaps for decades, or even hundreds of years). This aim may be achieved by taking advantage of strata with a permeability low enough to prevent significant seepage of leachate through the base or engineered by use of a liner. At containment sites, attenuation processes will take place almost entirely within the body of the waste. In time these processes will normally reduce the organic strength of the leachate, although in some circumstances, particularly where difficult wastes are involved, leachate quality may deteriorate.

Containment sites are not generally suitable for the disposal of large volumes of liquid wastes. Ultimately such sites will become saturated and overflow unless the ingress of liquid is sufficiently reduced by either a suitable cover or low permeability cap; it should be noted that low permeability can be as important as a liner in safeguarding water quality. Leachate control systems will need to be employed at containment sites, enabling the extraction of leachate for treatment and disposal when necessary. The restoration and aftercare of containment sites call for particular care in planning and management.

(ii) Protection of Water Quality

Landfill sites must be selected and engineered to ensure the protection of water resources. Fundamental to the protection of water quality is the control of leachate. The basis of this control is an understanding of the water balance of the landfill taking account of liquid inputs, storage capacity and run-off, so that estimates of the amount and timing of leachate generation can be made and the control measures needed to deal with the leachate can be determined. Such estimates, and the effectiveness of the control measures taken, will need to be continually checked by monitoring (See Figure 7.1-4). Once leachate leaves the boundary of the landfill it has the potential to pollute both surface and groundwaters. Liquids of all types should be kept out of a hazardous waste landfill. Liquids build up the hydraulic head on the bottom liner, increase the amount of leachate that must be collected and treated, and may eventually cause leakage.

Surface Waters

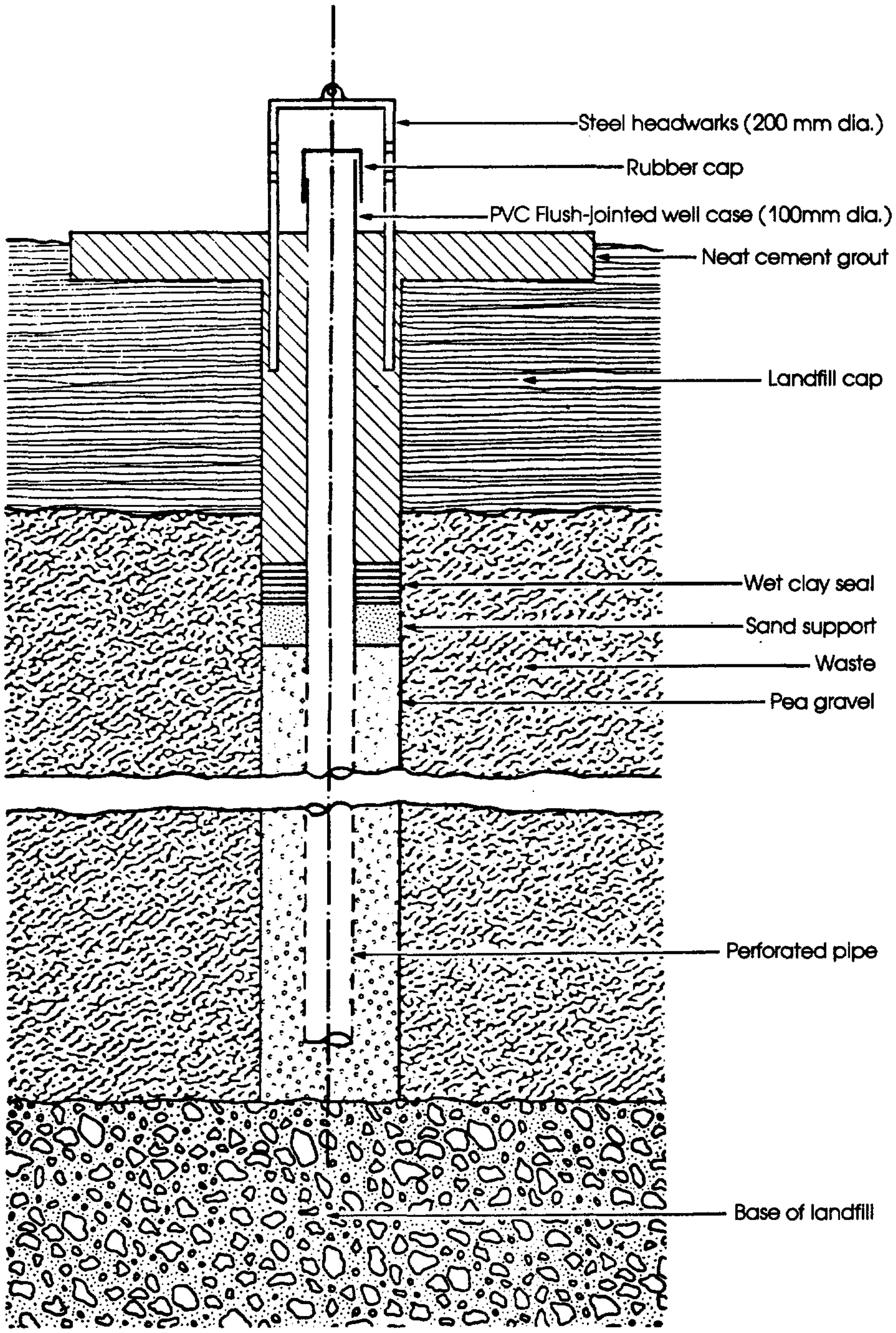
Unless water authority consent has been given, the discharge of leachate to surface waters must be prevented. Surface waters can usually be protected by the design and engineering of drainage systems; however, leachate may escape by various routes and hydrogeological investigations are needed to ensure that all significant routes are detected and dealt with in the design. Leachate may pass through the base or sides of a landfill to issue at the surface at a lower level and flow into surface waters. Alternatively sites having a relatively impermeable base and sides will retain leachate so that, if liquid inputs are not controlled, it will accumulate and ultimately overflow or break through. Egress of leachate is a risk particularly with some older mineral excavations where drainage adits may have been constructed.

Small V-sectioned valleys may be used for landfilling by constructing a culvert over the natural channel and landfilling on top. However, there is the risk that leachate will gravitate to the culvert track and possibly discharge into it. Valley sites should be surveyed to identify seasonal and permanent springs, small streams and water courses and the nature and extent of their catchment must be ascertained to determine the maximum capacity of the culvert or diversion channel. Landfilling over culverts is not regarded as good practice unless a well designed and properly engineered culvert is provided. It is better to divert upstream water around such sites if all practicable. Old railway cuttings may also be considered as artificial valleys, though it should always be kept in mind that railway engineers would have made provisions to drain their cuttings and these drains are likely to remain. It must be determined whether drains are present and, if so, whether they should be retained, removed, grouted up or diverted. It is important to remember that the culverts should not be put on permeable material otherwise they will provide a flow of water. The culverts must be surrounded by a suitable thickness of impervious material (about 0.5 metre of clay may be sufficient).

Subsidence basins can be formed as a consequence of differential settlements of the land surface resulting from certain methods of underground extraction of minerals such as coal and salt. In wet weather water may accumulate in such basins. This may be aggravated by subsidence affecting the natural drainage characteristics of the area and result in permanently submerged hollows. Alternatively, basins may be dry owing to fractures affording ready egress of water into underlying strata. Such sites would not normally be candidates for a landfill unless they could be adequately engineered to prevent pollution, including any that might occur if further subsidence were to take place.

FIGURE 7.1-4

Leachate Monitoring Borehole



Some landfill sites are raised above the level of the surrounding land. In such cases leachate will flow either through the base of the fill material into the underlying deposits or else emerge onto lower-lying surrounding land. It needs to be established whether local drainage systems exist; if so, their extent must be determined and decisions made on whether they should be utilized, protected or removed.

Groundwaters

The protection of groundwaters is more difficult than that of surface waters and calls for a thorough investigation of the hydrogeological characteristics of the site to establish the fate of any leachate. At sites where relatively rapid migration occurs, leachate will move away through fissures and coarse sediments, or more slowly in an unsaturated zone consisting of diverse intergranular pathways, before entering the groundwater. At other locations, movement of leachate from the site will be severely restricted, but nevertheless, some migration must be expected to occur in time. For all sites it is necessary to assess, in the light of hydrogeological information and other relevant factors, the likely effectiveness of attenuating mechanisms, both within the landfill and in underlying strata, in reducing pollution risks to an acceptable level. From such investigations the type and extent of microbes reacting in the waste mass can be determined.

The main components in the leachate from landfill sites may be conveniently grouped into four classes, as follows:

(a) major elements and ions such as calcium, magnesium, iron, sodium, ammonia, carbonate, sulphate and chloride.

(b) Trace metals such as manganese, chronium, nickel, lead and cadmium;

(c) a wide variety of organic compounds which are usually measured as Total Organic Carbon (TOC) or Chemical Oxygen Demand (COD); individual organic species such as phenol can also be of concern; and

(d) microbiological components.

All household waste and most industrial waste will give rise to leachate.

Household waste is reasonably consistent in composition over all landfill sites, as is the resulting leachate; typical figures for leachate composition for recent and aged household waste are shown in Table 7.1-10. Leachate composition at sites taking industrial waste is much more variable; some data providing a comparison between leachates from household waste and those from sites taking industrial waste as well as from household/industrial waste co-disposal sites are shown in Table 7.1-11.

TABLE 7.1-10

Typical Composition of Leachates from Recent and Aged Domestic Wastes at Various Stages of Decomposition (all results in mg/l except pH-value)

Determined	Leachate from Recent Wastes	Leachate from Aged Wastes
pH- value	6.2	7.5
COD (Chemical Oxygen Demand)	238000	1160
BOD (Biochemical Oxygen Demand)	11900	260
TOC (Total Organic Carbon)	8000	465
Fatty acids (as C)	5688	5
Ammoniacal-N	790	370
Oxidized-N	3	1
o-Phosphate	0.73	1.4
Chloride	1315	2080
Sodium (Na)	960	1300
Magnesium (Mg)	252	185
Potassium (K)	780	590
Calcium (Ca)	1820	250
Manganese (Mn)	27	2.1
Iron (Fe)	540	23
Nickel (Ni)	0.6	0.1
Copper (Cu)	0.12	0.3
Zinc (Zn)	21.5	0.4
Lead (Pb)	8.4	0.14

TABLE 7.1-11

Composition of Leachates from Solid Wastes in Landfills

(all results in mg/l except pH-value)

	Household Waste	Pitsea (UK) (43%) Industrial)	Rainham(UK)* (Industrial/ Household)	Granmo (Norway) (66% Industrial	Cedar Hills (USA) (Industrial Household)
pH-value	5.8-7.5	8.0-8.5	6.9-8.0	6.8	5.4
COD	100-62400	850-1350		470	38800
BOD	2-38000	80-250		320	24500
TOC	20-19000	200-650	77-10000	100	
Volatile acids (C_1-C_6)	ND-3700	20	600-10000	10	7100
Ammoniacal-N	5-1000	200-600	90-1700	120	
Organic-N	ND-770	5-20		62	
Nitrate-N	0.5-5			0.04	
Nitrate-N	0.2-2	0.10-10	8.0		
o-Phosphate	0.02-3	0.20		0.6(Total)	11.3(Total)
Chloride	100-3000	3400	400-1300	680	
Sulphate	60-460	340	150-1100	30	
Sodium (Na)	40-2800	2185	2000	462	
Potassium (K)	20-2050	888	50-125	200	
Magnesium (Mg)	10-480	214		66	
Calcium (Ca)	1.0-165	88		188	
Chromium (Cr)	0.05-1.0	0.05	0.5	0.02	1.05
Manganese (Mn)	0.3-250	0.5			
Iron (Fe)	0.1-2050	10	0.6-1000	70	810
Nickel (Ni)	0.05-1.70	0.04	0.5	0.1	1.20
Copper (Co)	0.01-1.15	0.09	0.5	0.09	1.30
Zinc (Zn)	0.05-130	0.16	1.0-10	0.06	155
Cadmium (Cd)	0.005-0.01	0.02		0.0005	0.03
Lead (Pb)	0.05-0.60	0.10	0.5	0.004	1.40
Monohydric phenols		0.01	ND-2.0		
Total cyanide		0.01	0.09-0.52		
Organochlorine pesticides		0.01			
Organophosphorous pesticides		0.05			
PCBs		0.05			

* Samples obtained from boreholes within the fill.

Effects of Water Pollution by Leachate

Good design and management, in particular the full use of attenuation processes, can ensure that water resources are not put at risk. Nevertheless, the potential to severely affect water resources exists and should be understood. The effects of substantial pollution of surface waters and groundwater by leachate are briefly described below.

Surface Waters

The input of high strength organic leachate and inorganic solutions of metals in a reduced state of oxidation into a water course will deplete the oxygen content of the water and ultimately will result in the extinction of all oxygen-dependent life. If the leachate contains non-biodegradable organic compounds these will persist for some time. When such compounds are assimilated into food chains they may adversely affect aquatic species. Although the effect on aquatic fauna of individual compounds can be assessed or predicted with some confidence the combined effects of several compounds cannot. In addition temperature, pH and dissolved oxygen concentration all have an influence on the degree of toxic effects on a particular aquatic species. For many pollutants, no single concentration can be given which is universally applicable for all aquatic environments.

Groundwater

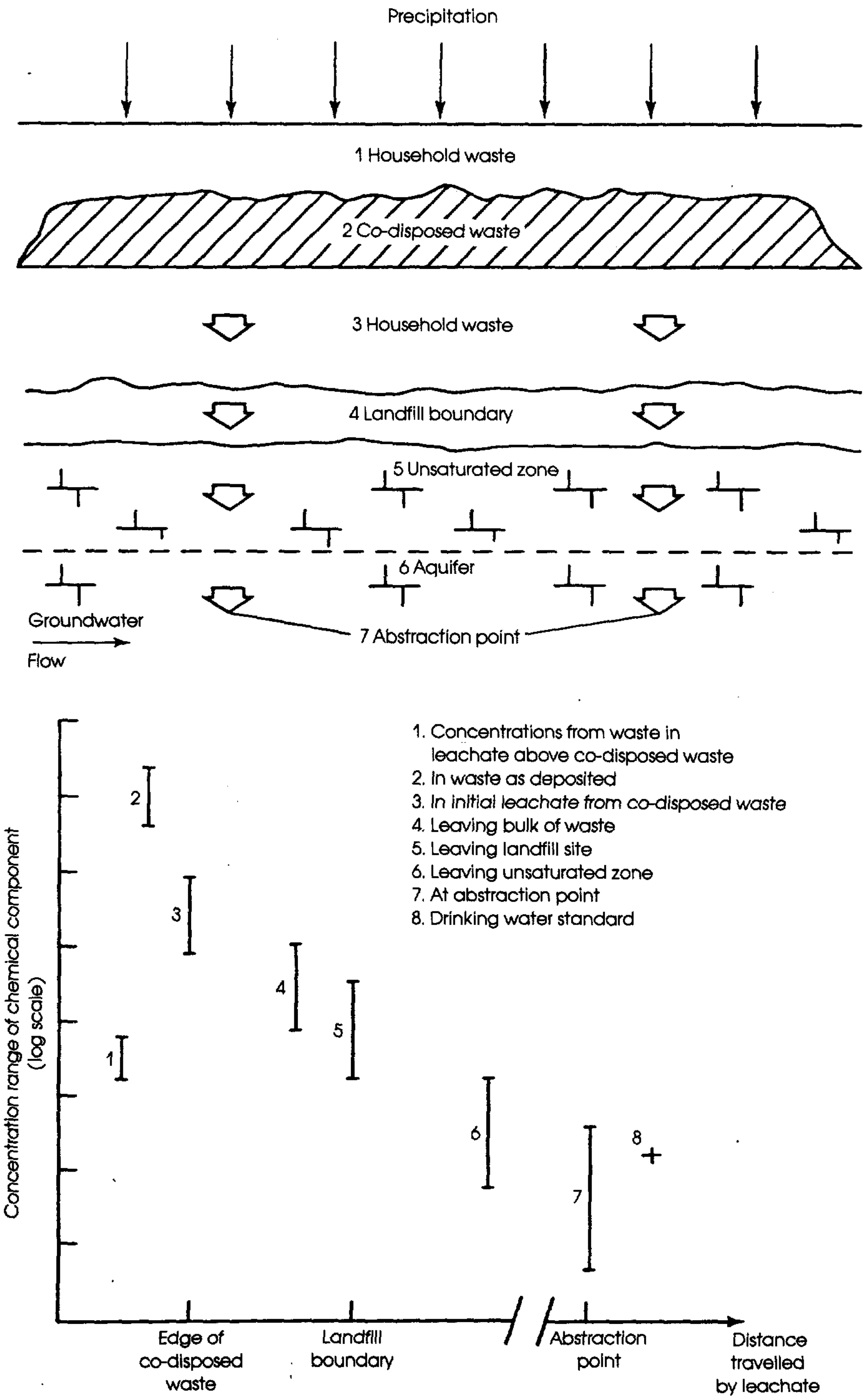
The effects of high strength organic leachate on groundwater will persist for a long time due to the limited amount of dissolved oxygen available and the low rates of dispersion. Once the groundwater has become polluted it may be unsuitable as a source of potable water supply for the foreseeable future. The nature of the strata and groundwater flow rates will control the extent of the pollution plume and its movement down the groundwater gradient away from the waste disposal site. Some constituents in leachate are chemically stable while others are not amenable to attenuation by physical processes in the groundwater environment. The identification and quantification of a pollution plume is difficult to achieve unless a large number of boreholes are constructed and groundwater samples abstracted and analyzed at regular intervals.

Attentuation

When wastes are landfilled, various processes including dilution and dispersion act within the landfill to reduce the polluting potential of the leachate. The action of these processes as a whole, which may be physical, chemical or biological in character, is termed 'attenuation' (see Figure 7.1-5). Attenuation processes, which have already been discussed, will only briefly be mentioned here. They vary according to the nature and quantities of the wastes deposited and depend on both the geochemistry of the strata and prevailing hydrogeological condition.

FIGURE 7.1-5

The Various Factors which can Contribute to the Attenuation of Chemical Species by Landfills and Surrounding Geological Strata



Attenuation may result in either the effective removal of a particular polluting component in the leachate, or its delayed release into the environment. The principal attenuation processes are summarized in Table 7.1-12. However, for many components of hazardous wastes, attenuation does not have any impact.

(iii) Water Balance Calculations

The importance of understanding the water balance of a landfill has already been noted. Leachate generation should be minimized. The effects of the infiltration of rainfall make it imperative that both the ground and surface water ingress into the landfill are controlled (see Figure 7.1-6). A useful aid in calculating and assessing the likely rate and extent of leachate production is to apply water balance equations. The main factors contributing to the water balance of a landfill site are:

(a) water input including effective rainfall (precipitation minus runoff and evapotranspiration), surface and groundwater infiltration and liquid waste disposal,

(b) surface area,

(c) nature of wastes,

(d) site geology, and

(e) surface liquid storage (e.g., leachate balancing lagoons).

In estimating the amount of leachate likely to be generated each year from operational areas the following water balance equation may be used:

$L_o = 1 - E - aW$ where

L_o is the free leachate retained at the site (equivalent leachate production minus leachate leaving the site) in m^3/annum (U.S. Congressional Budget Office 1985). (1) is the total liquid input (precipitation plus liquid waste plus any surface or ground water inflow) in m^3/annum. E is the evapotranspiration losses (evaporation plus minimal transpiration) in (m^3/annum). a is the absorptive capacity of the waste (m^3/tonne of waste as received) and W is the weight of waste deposited in tons/annum.

Good landfill practice normally requires that the site is operated so that L_o is always negative or zero. A positive value for L_o implies leachate build-up in the site.

The equation requires modification to cover the situation after final restoration has taken place. There may then still be some further

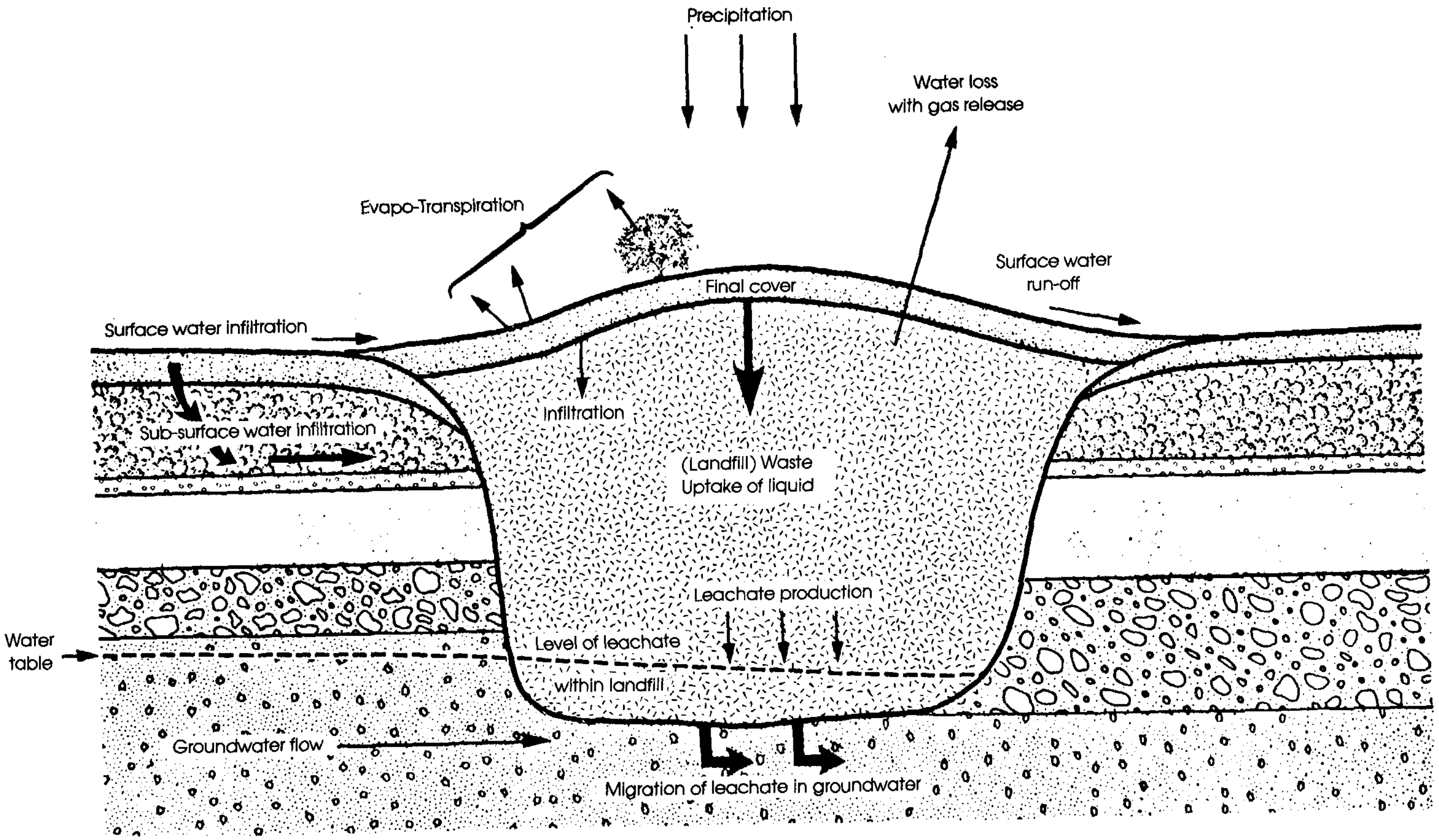
TABLE 7.1-12

Attenuation Process

Physical	Absorption, adsorption, filtration, dilution dispersion.
Chemical	Acid-base interactions, oxidation, reduction, precipitation, co-precipitation, ion-exchange, complex ion formation.
Biological	Aerobic and anaerobic microbial degradation.

FIGURE 7.1-6

Landfill Water Balance



absorptive capacity for liquid within the waste, (U) and an additional quantity (R) must be introduced to allow for surface water runoff. The revised water balance equation is then:

> $L_r = 1 - E - R - U$ where lr is the leachate retained in the site after restoration (Brady 1984).

For sites having a low permeability cap, R will be very high. The term U may be calculated by summing the values of L_o using the first equation during the operation of the site. The major factors controlling leachate production are described below, with reference to the terms in the water balance equations.

Total Liquid Input (1)

Rainfall, which in countries like the United Kingdom ranges between 0.55 and 2.0 meters per year is normally the largest contributor of water to a landfill site. As far as is practicable, surface and groundwater infiltration into a site should be eliminated. Any residual input, however, should be estimated and included in the equation. Some sites are licensed to accept liquid wastes and for the purposes of water balance calculation these should be included as part of the total liquid input. Spraying leachate generated on site back over fresh waste or the completed site is a relatively common practice. This leachate will already have been accounted for in water balance calculation for the operational landfill. However, if it is sprayed over new areas of the site or over completed parts it should be included as part of the input term in the equation as appropriate.

Evapotranspiration (E)

Wind, temperature, humidity and atmospheric pressure influence the rates of evaporation/evapotranspiration, and are reflected in the value of term E in the above equations. However, while climatic conditions vary from one area to another, the resulting range in the value of E is likely to be much less than the differences that occur between values at operational and completed landfills. Healthy vegetation on completed landfills increases evapotranspiration and also binds the soil, thus minimizing erosion. However, the ability of the final cover to limit infiltration and encourage surface water run-off may be impaired by root penetration.

The effective rainfall at a site is given by the actual rainfall minus E. The amount of rainfall that infiltrates a landfill will be the effective rainfall less any run-off. Experimental data suggest that infiltration rates through typical daily cover soils range from 20% in summer months to 100% during the winter. Such high infiltration rates, which are apparently independent of the quality of typical daily cover used, indicate the importance of limiting operational areas and excluding other sources of liquid ingress. Similar conditions apply to intermediate cover; evidence suggests that the infiltration rate of precipitation is not significantly altered by the type or quantity of intermediate cover used. If, during infilling, surfaces are formed with slopes greater than 10 percent (1 in 10) then some run-off can be expected but this will not substantially reduce overall infiltration, unless collection and removal of any run-off water is provided. For these reasons it is most unlikely that run-off will be effective at an operational site and it has therefore not been included in the first equation.

The position is very different for completed landfills where, with the use of contouring and the incorporation of drainage systems (and, at containment sites, the use of a low permeability cap) a substantial proportion of precipitation can be diverted away from the site as run-off. (Hence, the introduction of the factor (R) in the second equation above.) Typical infiltration values through completed sites are 25-30% of annual effective rainfall. In addition to quantity, depth and compactness of the cover, run-off will also be affected by the frequency, intensity, and duration of rainfall events.

Absorptive Capacity of Solid Waste (a)

There are two main mechanisms for moisture retention within waste. First, liquid can be absorbed into the waste and held by capillary attraction within the micro-structure of waste particles. Second, free water can be stored in voids within the waste. Typical values for overall voidage are between 20 and 35%. Intermediate cover materials or highly compacted wastes can also give rise to localized areas of saturation known as perched water tables. Thus water retention values are strongly dependent on both the density of the waste and the presence or absence of voids or of impermeable barriers which will inhibit the continuous downward percolation of liquid. At many landfill sites waste densities of 0.7 to 0.8 tons/m^3 of waste as received are achieved and at such densities it is likely that about 0.1 to 0.2 m^3 of added liquid per cubic meter of waste as received can be absorbed before substantial leachate generation commences. However, at higher compaction densities absorptive values will fall. For example, at placement densities in excess of 1.0t/m^3 the absorptive capacity may fall to as low as 0.02-0.03 m^3 of as received waste.

Waste (W)

Finally, a value for the annual quantity of waste to be deposited (W) is required to calculate an overall water balance for the site. Generally speaking, the higher the rate of waste input the less likely it is that leachate will be generated during the operational phase of a site. This can mean that the absorptive capacity of the waste will not be utilized and landfill gas generation as a consequence of biodegradation cannot be promoted in the absence of sufficient water input.

7.1.5 Landfill Site Development

The preparations needed at a site before waste can be deposited may be considered under two headings: infrastructure, that is, the position of the building, roads and facilities that are necessary to the efficient running of the site; and the basic engineering works needed to shape the site for the reception of wastes and, generally, to meet the technical requirements of the working plan.

(i) Site Infrastructure

The infrastructure provided at a landfill sets the general framework within which the site will be operated. Careful planning is essential if the landfill is to operate efficiently and safely throughout its operational phase.

Site Entrance

At many sites, and particularly at operational or recently abandoned mineral workings, access arrangements will already be established. At other sites access will have to be provided. In either case an adequately splayed bellmouth, properly surfaced and capable of taking traffic in both directions will be required. Road safety considerations may necessitate the provision of an entrance set back from the highway with good sight lines and possibly of an acceleration and deceleration lane. Vehicles using the landfill should not be required to queue on the highway. The design of the site entrance will depend on a number of factors including the number and types of vehicles using the site and the classification of the highway from which access is to be gained. The appearance of the entrance has a major influence on how a landfill site is perceived by the public. An untidy entrance with mud on the roads and highway does little to instill confidence in the operation. Judicious landscaping and tree planting at the site entrance will provide a screen for the operation and be a pleasing feature.

Site Accommodation

The size, type and number of buildings required at a landfill depend on factors such as levels of waste input, the expected life of the site, environmental factors and the availability of other facilities and depots. Buildings will, therefore, range from single portable cabins/messroom units to purpose-built complexes incorporating offices, messrooms, stores, garages and workshops. However, regardless of the number of units to be provided, certain features are common, should be borne in mind and are listed below.

- The need to comply with planning, building, fire, and health and safety regulations and controls.
- Security and resistance to vandalism.
- Durability in service and the possible need to relocate accommodations during the lifetime of the site.
- Ease of cleaning and maintenance.
- Appearance.
- The availability of services such as electricity, water, drainage and telephone.

Accommodation provided at a landfill should meet the needs of site control and record keeping, welfare and possibly stores, garages and workshops. At large landfills and particularly those taking difficult industrial wastes, provision may also be needed to accommodate management, technical support staff and a small laboratory for check analysis of waste on its reception.

Site Control Office

All landfill operators need to control and keep records of vehicles entering and leaving a landfill site. Generally site control office is needed to achieve this. The type, size and location of the office will depend on factors such as those listed below.

- The use made of the site by vehicles not operated by the landfill operator.
- Whether the site is within a factory curtilage.
- Whether a weighbridge is installed.
- Whether other uses are to be made of the accommodation.

In most cases the control office should be located away from the entrance to allow traffic to queue off the highway. At busy sites, and particularly those with a weighbridge it is beneficial to position the office on a kerbed island, ideally to the offside of approaching vehicles. In designing the control office, consideration should be given to providing the control point with a barrier or traffic light system to regulate entry into the landfill. At small sites a combined site control office and accommodation unit will usually suffice.

Welfare/Messroom Unit

Except when welfare and messing facilities can be provided close to the landfill such as at a works canteen, special provisions for the welfare of site employees should be made. Facilities may be incorporated as a separate room or areas within the site control office, but at large sites they are best provided in a separate unit. The facilities listed below should be provided.

- Lockers for clean and dirty clothing and other belongings together with drying arrangements and separate storage for safety equipment.
- Adequately heated and lighted accommodation containing a table and chairs or benches for use during meal breaks.
- Facilities for heating food and providing hot water.
- Adequate first aid equipment.
- A wash basin with hot and cold water.

- Shower facilities.
- Sufficient lavatories both for employees and visitors.
- Accommodation for the crew of household waste collection/delivery vehicles at sites where site rules allow only the driver to proceed to the disposal area.

Stores

Space should be provided for the storage of materials used on-site. Potentially harmful substances such as insecticides and weedkillers, flammable substances and liquified gas containers will require special facilities. Other materials such as diesel fuel, oils and greases should be stored in correctly marked tanks or containers. The Fire Prevention Officer should be consulted over the provision of storage facilities at landfill sites.

Garages

At some landfills garage and workshop facilities for the equipment may be provided. If a garage is to be used for maintenance purposes it will require adequate lighting, heating, ventilation and insulation. Consideration should be given to providing a low voltage electricity supply for hand tool operation.

Plant and Vehicle Cleaning

Arrangements should be made for the plant to be cleaned regularly. Areas set aside for cleaning require a good water supply, adequate drainage and specially protected electrical equipment.

Additional Accommodations

At large landfill sites, accommodation should be provided for management meetings and for a display of site development, operation and restoration plans.

Recording Waste Input

Reliable site monitoring is dependent on an accurate record of inputs. A weighbridge at the site entrance is the best way of providing such data. Alternatively, void-usage maybe surveyed on a regular basis.

Weighbridges

In selecting and siting a weighbridge there are a number of considerations to be borne in mind. Unless the landfill is to be restricted to regular users, vehicles will have to be weighed both going into and out of the site, and this might cause congestion. The weighbridge must also be located far enough away from the site entrance to prevent queuing on the public highway.

The type of vehicles using the site will determine platform size and capacity. Two types of platform weighbridges are in general use. One type is flush with the road surface; the installation of which calls for fairly extensive civil engineering work. The other type is mounted on the road surface and requires much less engineering. However, because the bridge can be some 350 mm above the level of the road, ramps must be provided. This type of unit is moveable and can be installed at other sites at reasonable cost.

Any weighbridge installed at a landfill site is subject to inspection by a local trading standards authority and will be required to meet the prescribed accurate standard. Where accuracy is not of paramount concern, the installation of axle weighers may provide a relatively low cost method of weighing. The weighers are usually installed in a shallow pit. Vehicles can be weighed without stopping as the weighers can be linked to a remotely sited load indicator.

Wheel-cleaning

One means of ensuring that mud is not taken onto the highway is to provide site roads constructed to both a high standard and of sufficient length that any mud and debris trapped on vehicles is likely to be shaken onto these roads where it can be regularly swept up. However, in many cases site limitations will require the provision of mechanical wheel cleaning equipment. Nevertheless, it is still essential that any wheel cleaning equipment is positioned far enough into the site to leave an adequate length of access road between it and the site entrance to enable any retained mud or slurry to be removed before vehicles drive onto the highway.

Sunken Bath

This relatively low cost system of wheel cleaning involves the provision of a shallow trough, formed in concrete or a similar material, filled with water and through which all traffic leaving the site must be driven. A series of humps or bars set into the road surface within the bath provides vibration which assists in shaking off loose material. Care must be taken to ensure that vehicles do not enter the bath at too high a speed.

Elevated Shaker Bars

These are essentially an oversized cattle grid set level with the road, or alternatively elevated with a ramp at either end. The vibration from vehicles passing over the bars causes the mud and trapped material to drop through the grid bars into a trough or sump. The sump requires cleaning at regular intervals.

Site Notice Boards

It is important that adequate information on the operation and regulation of a landfill is displayed at the site. At the site entrance a notice board of adequate size should be located in a position where it can be read. It is essential that users and visitors to the site are fully informed of site health and safety procedures. In general, agreement on these procedures should be reached by discussions between the site operator and contractors using the site. The key points of the agreed procedures should be clearly displayed on a notice board in a position where it will be seen by all people entering the site. Suitable speed restrictions, directions and other signs such as "No Smoking" should be positioned as necessary throughout the site.

Disposal of Bulky Household Waste Delivered by the Public

Though not strictly falling within the subject of this review, it is recognized that it may sometimes be desirable, particularly in rural areas, to provide a disposal facility for direct delivery by the public of bulky items of household waste near the entrance to the landfill. Such a facility should be in a separately fenced area situated near the entrance of the site.

Roads

Good quality roads are an essential requirement for efficient landfill disposal operations. Access roads to both the site and disposal area must be maintained in good repair at all times. The access road from the highway to the site control office is likely to remain in use throughout the life of the site and accordingly should be designed and constructed to meet such requirements. It should be of sufficient length and width to allow vehicles to both pass and queue on it and be surfaced with either tarmacadam or concrete. Suitable road markings and signposts should be used. The secondary site roads between the site control office and landfill area will probably need to be relocated during operations as new areas are brought into use.

Security Fencing

Except where a landfill site has a distinctive and effective natural barrier, adequate fencing should be provided to prevent unauthorized access. Unrestricted access presents significant health and safety problems not only for trespassers but also to authorized site users and occupants of adjacent properties.

Quarry Working

Many landfill sites are in active or abandoned quarries. It is essential that quarry sides remain safe during landfilling activities. Strict controls over the conduct of mineral excavation have been introduced in recent years to prevent quarry sides being left in an unsafe condition, but such controls may not apply to quarries which have been abandoned for some time. Operators planning to infill quarry workings should seek specialist advice on the stability of the sides, overburden and spoil heaps, adits and shafts and on the appropriate measures to be taken. Periodic inspection of quarry sides during the operation of a landfill is advised. It must be noted that any movement of spoil or work on the faces of a quarry is a quarrying operation and a qualified quarry manager will therefore need to be appointed.

(ii) Site Engineering

Earthworks

The working plan may require extensive earthworks to be carried out before deposition of waste can take place; this is particularly so if artificial liners are to be used. Such works may involve grading the base or sides of the site or the formation of embankments and the like, and may require large tonnages of overburden or similar material to be moved. Material may also have to be placed in stockpiles for later use at the site. All such operations will require the use of earth moving machinery. The number and type of machines needed will depend on the nature of the earthworks to be carried out.

Various features of landfill design may require substantial earthworks. The cell method of operation, which is being increasingly adopted at modern landfill sites, requires the construction of cell walls. This method of operation has the advantage that infiltration of rainfall is reduced and leachate generation is thus more readily controlled. The size of cells required will be influenced by the rate of waste deposition. It has been found that a 30 m wide cell can handle an input of some 1000 tons/day of household waste. It should be noted that when clay bunds are formed to a height greater than 4 m, specialist advice should be obtained on their design and construction.

At some sites earthworks may be needed to construct earth banks around part or all of the site perimeter to screen the landfill operations from the public. Trees or shrubs may then be planted on the banks to enhance the screening effect. Major earthworks may also be involved in the provision or improvement of access roads as well as the construction of haul roads within the site itself. Numerous minor earthworks may be required. For example, the construction of drainage ditches and sumps in unfilled areas of the site for the separate collection of landfill leachate and uncontaminated rainwater.

(iii) Lining Landfll Sites

General

The principal aim of lining and capping a site is to contain the leachate, thus preventing pollution of surrounding land and waters. Lining may also assist in leachate control by reducing groundwater infiltration into the landfill. Lined landfill sites can usually accept a wider range of wastes than would otherwise be possible. Artificial liners are constructed of materials which are to all intents and purposes impermeable. Natural lining materials, such as heavy clay soils, exhibit low permeability (for example 10^{-8} cm/sec). As well as impending the flow of liquid, natural liners may also attenuate the leachate.

Site Preparation for Liners

Where the use of a liner is envisaged, the suitability of a site for lining will have been evaluated in the site investigation stage, and if found to be suitable, plans for its lining will have formed part of the working plan for the landfill operation. Table 7.1-13 lists some of the factors that should be taken into account when assessing the suitability of a potential landfill site for lining. Liners should not be installed until the site has been properly prepared. It should be stressed that soils must be excavated and recompacted for form liners. In-situ clays almost always contain areas of variable permeability. The engineering of landfill sites for lining is a technically complex measure which should be entrusted only to competent operatives. Some of the work that may be necessary is described below.

Earthworks

The area to be lined should be free of objects likely to cause physical damage to the liner, such as vegetation including stumps and roots, hard objects, sharp rocks and the like. The area to be lined should be allowed to dry sufficiently to ensure that the surface can support men and equipment during installation of the liner. U.S. EPA has determined that most synthetic liner failure results from damage occurring during installation of the membrane. Therefore, special precautions are being developed. Particular care should be taken to deal with the consequences of rainfall during the operation. Any accumulation of water should be removed from the site without delay and installation allowed to proceed only when ground conditions are suitable. Where a smooth base to the site cannot be obtained or where synthetic lining materials are being used, a binding layer of suitable fine-grained material should be laid to support the liner. A layer of similar material should also be laid above the liner to protect it from subsequent mechanical and environmental damage. Where synthetic liners are used, an 'anchor trench' around the periphery of the site to which the liner can be secured is usually required.

Underdrainage

Whenever possible, sites in which a high groundwater table exists should be rejected as being unsuitable for lined landfills. However, it is recognized that, occasionally, such sites may have to be lined, in which case an adequate underdrain system will be required. If failure of a liner would, regardless of other precautions, result in unacceptable deterioration of water quality, such a site should be used only for inert wastes.

Liners

A hazardous waste landfill should have a liner that is designed, constructed, and installed to prevent migration of wastes or by-products out of the landfill into adjacent subsurface soil, groundwater or surface water at anytime during the active life of the landfill. The active life includes the closure and post-closure period of the landfill. The liner

TABLE 7.1-13

Factors in the Assessment of the Suitability of a Site for Lining

1.	Grading of sides:	Side slopes of less than 1:3 are required to accommodate the plant and equipment used in lining: it must be possible to grade the sides of sites to meet this requirement.
2.	Hardness of base:	A base of hard rock will usually preclude the use of sealants such as bentonite which require renovation of the site prior to installation.
3.	Stability of bedrock:	Lining materials are unable to cope with sudden or differential settlement.
4.	Underlying impermeable:	At some sites it may be possible to excavate the materials: overburden to reach naturally impermeable materials.
5.	Groundwater inflow:	The emergence of groundwater beneath either a natural
6.	Base below water table:	In such cases the base will have to be raised with inert materials before the liner can be installed. Temporary lowering of groundwater by pumping is not
7.	Stability of subgrade:	If bund and cell walls cannot be engineered without fear of slippage there will be problems when laying the liner on slopes.
8.	Treatment or disposal	Consider availability of facilities to deal with of leachate: leachate in excess of site capacity.

should be constructed of materials that have chemical properties and sufficient strength and thickness to prevent failure due to (a) pressure gradients, (b) physical contact with the waste or leachate to which they are exposed, (c) climatic conditions, (d) the stress of installation, and (e) the stress of daily operation.

The liner should be placed on a foundation or base capable of providing support and resistance to pressure gradients above and below the liner to prevent failure of the liner due to settlement, compression, or uplift. The liner also should be installed to cover all surrounding earth likely to be in contact with the waste or leachate.

In some countries, such as the United States, a hazardous waste landfill is designed with two liners, a leak detection system, and a leachate collection system. The leak detection system detects any migration of liquid into the space between the liners, and the leachate collection system allows removal, treatment and proper control of any leachate that does occur.

Figure 7.1-7 illustrates the types of liners that can be used to protect the groundwater. Combinations of compacted layers of low permeability soils, such as clays, and flexible membrane liners (FML), such as man-made plastics, can be used to contain any leachate that may be produced. Figure 7.1-8 provides additional details about liners used at hazardous waste landfills.

Liners function to: (a) impede the flow of the pollutant and pollutant carrier and (b) absorb or attenuate suspended or dissolved constituents. A liner with low permeability is needed to impede the pollutants. The absorptive or attenuative capability of a liner depends on the chemical composition of the liner and its mass.

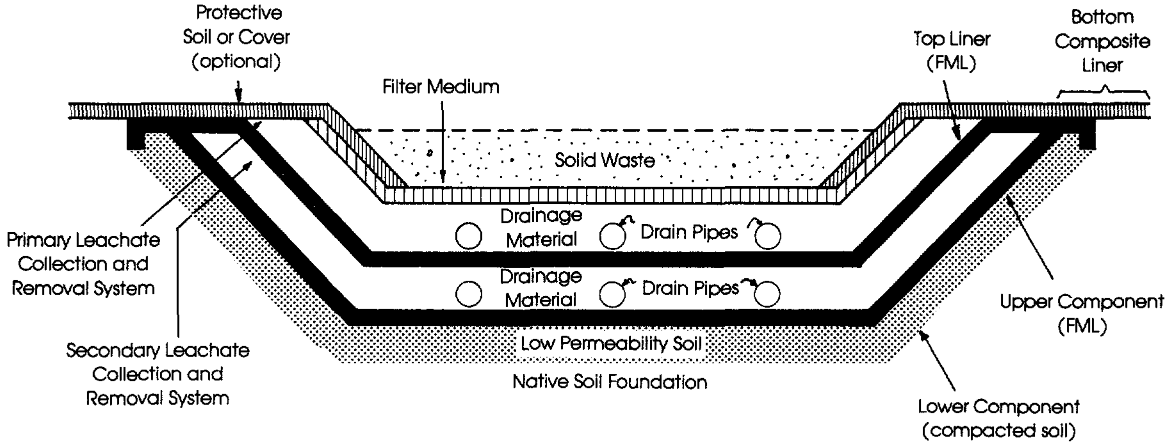
Most liners incorporate both functions (i.e., impede and absorb, but to different degrees). Man-made membrane liners, such as plastic sheeting, are the most impermeable but have little sorptive capacity. Soils, such as silts and clays, have larger sorptive capacity but can be more permeable.

In general, the thicker the soil liner, the slower the movement of pollutants through it. Due to their general availability, clay soils normally are considered as the first alternative for hazardous waste landfill liners. There is no such thing as an "impermeable" liner. Liquid is transmitted through all liners to some degree. Hazardous waste landfills require liners with a very low permeability.

Permeability is a numerical measure of the ability of a soil or other liner material to transmit a fluid. When water percolates through a soil, the permeability often is called hydraulic conductivity. The units of permeability normally are in terms of centimeters per second (cm/sec). Such units can be put into practical perspective by identifying the relative rate of movement associated with different permeabilities (Table 7.1-14). Thus a liner with a permeability of about 1.0×10^{-7} cm/sec would permit liquids to move only about 0.03 meters/year (0.1 feet/year).

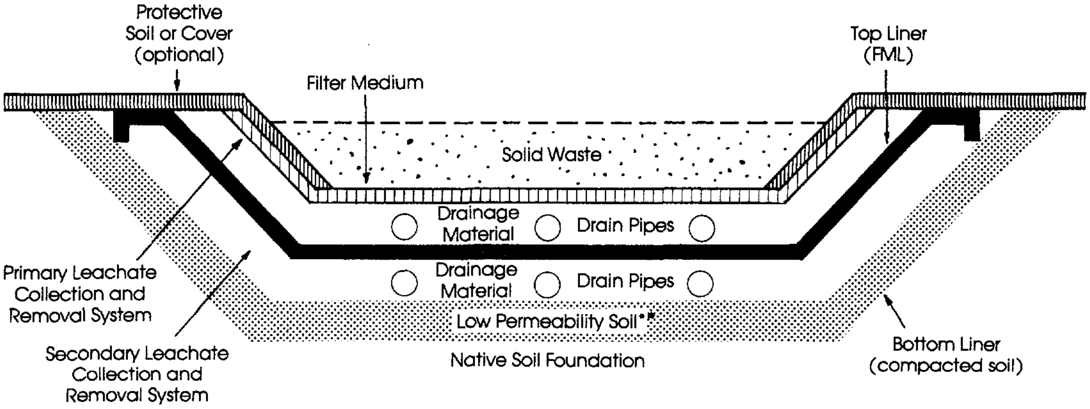
FIGURE 7.1-7

Schematic Diagram of a HSWA* Double Liner System and an EPA Synthetic/Clay Liner for a Landfill



Source: EPA.

SCHEMATIC DIAGRAM OF A COMPOSITE DOUBLE LINER SYSTEM FOR A LANDFILL



Source: EPA.

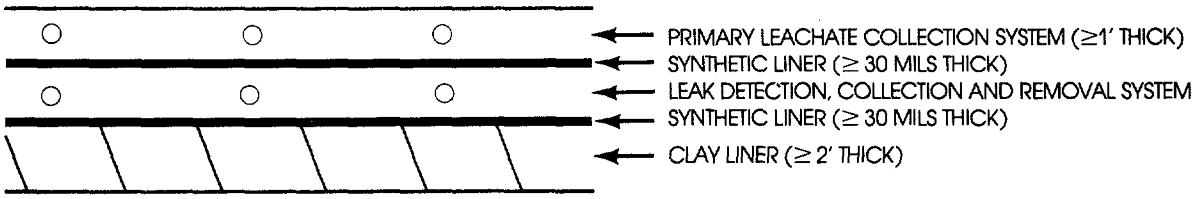
SCHEMATIC DIAGRAM OF A HSWA* DOUBLE LINER
SYSTEM AND AN EPA SYNTHERIC/CLAY LINER FOR A LANDFILL

*Thickness to be determined by
breakthrough time for EPA synthetic/clay liner
Thickness ≥feet for HSWA* double liner

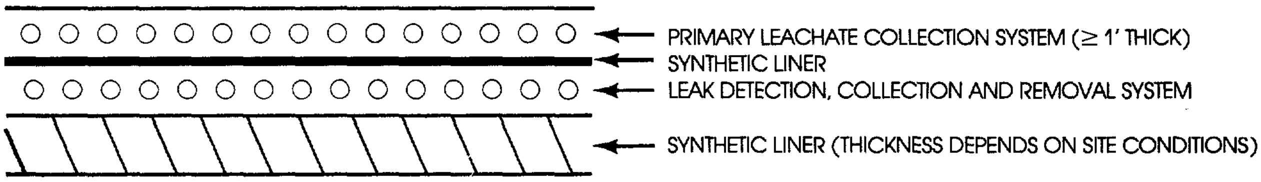
*Hazardous & Solid Waste Amendments of 1984 (USA).

FIGURE 7.1-8

Schematic Diagram of Two Double Liner Designs for Landfills



DESIGN 1



DESIGN 2

Source: U.S. Environmental Protection Agency. 1985. Handbook - Remedial Action at Waste Disposal Sites. EPA - 625/6-85-006. Cincinnati, Ohio: Office of Research and Development.

TABLE 7.1-14

Significance of Soil Permeability Values

Permeability Value (cm/sec)	Equivalent Movement of Liquid and Associated Contaminants	
	meters/year	(feet/year)
1.0 x 10-3	315	(1000)
1.0 x 10-4	32	(100)
1.0 x 10-5	3.2	(10)
1.0 x 10-6	0.3	(1.0)
1.0 x 10-7	0.03	(0.1)
1.0 x 10-8	0.003	(0.01)

Liners for hazardous waste landfills are recommended to have a permeability of about 1 x 10^{-7} cm/sec or less. Fine sands have permeabilities in the range of 10^{-1} to 10^{-3} cm/sec, clayey soils in the range of 10^{-3} m to 10^{-5}, silt in the range of 10^{-5} to 10^{-7} and clays in the range of 10^{-6} cm/sec and less. Most undisturbed soils have permeabilities in the range of 10^{-3} to 10^{-7} cm/sec.

All other factors being constant, a reduction in the void ratio of a soil results in a lower permeability. With compaction, several changes occur in a soil: (a) decrease of the effective area available for flow, (b) a decrease in the media pore size, and (c) a reduction in the void ratio. Thus, compaction reduces the permeability of a soil. Soil liners are compacted by suitable equipment as a hazardous waste landfill is constructed.

If existing soils are not adequate to serve as landfill liners, the soil permeability can be decreased by the addition of imported clays or polymetric materials. Clays such as monmorillonite or bentonite are available commercially and can be incorporated into existing soils to provide an adequate liner. The amount of additive used should be such that the amended soil has a permeability of about 10^{-7} cm/sec or less.

Choosing the liner materials compatible with the wastes to be put in the landfill is important. Information on the compatibility of synthetic liners is provided in Table 7.1-15. Research studies have shown that concentrated organic chemicals such as acetic acid, ethylene glycol, acetone, xylene and heptane can increase the permeability of clay soils. However, dilute organic chemicals, up to about 1400 mg/l, do not appear to affect the permeability of clays. Dilute chemicals are likely to be in landfill leachate. Field experience with liner integrity for hazardous waste landfills is limited and greater information on liner longevity and effectiveness is needed.

The estimated service life of a liner in a particular exposure condition is an important factor in selecting a liner material. For temporary holding situations such as impoundments and waste piles, a short life may be satisfactory. For hazardous waste landfills, a very long service life is required.

Physical, chemical and biological failure of liners can occur. Table 7.1-16 identifies adverse conditions that affect liner performance. Physical failures are commonly related to inadequate subgrade preparation, subsoil movement, inadequate operating conditions at the landfill, and changing hydrostatic pressures. Chemical failures normally are related to the incompatibility of the waste and the liner.

During construction or installation, liners should be inspected for uniformity, damage and imperfections such as holes or cracks. Immediately after construction or installation, synthetic liners and covers must be inspected to ensure tight seams and joints and the absence of tears or punctures. Soil-based and admixed liners and covers must be inspected for imperfections, including cracks, channels, root holes, or other

TABLE 7.1-15

Characteristics, Advantages, and Disadvantages of Selected Synthetic Liners

Liner Material	Characteristics	Range of Costs	Advantages	Disadvantages
Butyl rubber	Copolymer of isobutylene with small amounts of isoprene	M	Low gas and water vapor permeability; thermal stability; only slightly affected by oxygenated solvents and other polar liquids	Highly swollen by hydrocarbon solvents and petroleum oils; difficult to seam and repair
Chlorinated polyethylene	Produced by chemical reaction between chlorine and density polyethylene	M	Good tensile strength and elongation strength; resistant to many inorganics	Will swell in presence of aromatic hydrocarbons and oils; high elongation, poor memory
Chlorosulfonated polyethylene	Family of polymers prepared by reacting polyethylene with chlorine and sulfur dioxide	M	Good resistance to ozone, heat, acids and alkalis, easy to seam	Tensile strength increases on aging; good tensile when supported; poor resistance to oil
Epichlorohydrin rubbers	Saturated high molecular weight, aliphatic polyethers with chloromethyl side chains	H	Good tensile and tear strength; thermal stability; low rate of gas and vapor permeability; weathering; resistance to hydrocarbons, solvents, fuels, and oils	Difficult to field seam or repair
Ethylene propylene rubber	Family of terpolymers ethylene, propylene and non-conjugated hydrocarbon	M	Resistant to dilute concentrations of acids, alkalis, silicates, phosphates and brine; tolerates extreme temperatures, excellent resistance to weather and ultraviolet exposure	Not recommended for petroleum solvents or halogenated solvents; difficult to seam or repair; low seam strength

TABLE 7.1-15 (continued)

Liner Material	Characteristics	Range of Costs	Advantages	Disadvantages
Neoprene	Synthetic rubber based on chloroprene	H	Resistant to oils, weathering, ozone and ultraviolet radiation, resistant to puncture, abrasion, and mechanical damage	Difficult to seam or repair
Polyvinyl chloride	Produced in roll form in various widths and thicknesses; polymenzation of vinyl chloride monomer	L	Good resistance to inorganics, good tensile, elongation, puncture, and abrasion resistant properties, wide ranges of physical properties, easy to seam	Attacked by many organics, including hydrocarbons, solvents and oils; not recommended for exposure to weathering and ultra violet light conditions
Thermoplastic elastomers	Relatively new class of polymeric materials ranging from highly polar to nonpolar	M	Excellent oil, fuel, and water resistance with high tensile strength and excellent resistance to weathering and ozone	None reported
High Density Polyethylene	Blow or sheet extended P.E.	M to H (based) on thickness)	Good resistance to oils and chemicals, resistant to weathering; available in 20 to 150 mils thicknesses, resistance to high temperature	Thicker sheets require more field seams; subject to stress cracking; subject to puncture at lower thicknesses. Poor tear propagation

Notes: 1. Cost ranges: L =\$1.4/yd^2, M = \$4.8/yd^2, H = \$8.12/yd^2 (installed costs.

2. All ratings are based on property compounded materials designed for that specific application.

Source: U.S. Environmental Protection Agency. 1985. Handbook - Remedial Action at Waste Disposal Sites. EPA - 625/6-85-006. Cincinnati, Ohio. Office of Research and Development.

TABLE 7.1-16

Summary of Adverse Site Conditions Affecting Liner Performance

Unfavorable Condition	Potential Liner Problem
o Geotechnical/Hydrogeologic	
Moderate to active seismic area	Instability; liner failure
Settlement or subsidence	Cracks in clay or tears in synthetic liners
High groundwater table	Lifting or rupturing of liner
Voids	Cracking of liner
Sinkholes	Liner failure
Subsurface gas	Lifting of liner prior to backfilling
High permeability soils	Piping of subgrade
o Climatic	
Frozen ground/ice	Cracking, tearing
Wind	Lifting and tearing liner
Sunlight	Dehydration of clay liner (permitting cracks to develop)
	Destruction of some synthetic liners (caused by ultraviolet radiation)
High humidity	Poor seam adhesion caused by absorption of moisture by the solvents.

Source: U.S. Environmental Protection Agency. 1985. Handbook - Remedial Action at Waste Disposal Sites. EPA - 625/6-85-006. Cincinnati, Ohio. Office of Research and Development.

structural non-uniformities that may cause an increase in the permeability of the liner.

A flexible membrane liner should be protected during installation and operation by using sand or homogeneous soil layers above and below the membrane liner. The material on which the membrane liner is placed can exert pressure if rocks, roots or similar materials remain in contact with the liner.

Flexible membrane seaming techniques are important so that all membrane liner joints are solid and continuous. Earthmoving equipment spreading a protective soil layer above the liner can also stress and damage a liner. The liner should have sufficient strength to resist failure from any settlement or movement of the foundation material. All of these possibilities should be considered in the design and operation of a hazardous waste surface impoundment.

Climatic conditions also can affect liner integrity. Of particular concern is the stress caused by seasonal freeze-thaw cycles and that caused by runoff-related erosion of above ground support structures.

Operational Considerations for Artificially Lined Sites

Special considerations that apply to the operation of lined landfill sites are highlighted here. During the early phase of operation, particular care must be taken to ensure that the traffic does not damage the liner. Traffic movements over cover material should be minimized and hardcore roads underlain by heavy-duty fabric should be provided to permit vehicles to unload and turn. Particular care should be taken in placing the first lift of refuse, and buildup of water and leachate should also be controlled. Where a drainage system has been installed above the liner, pipes should be cleaned out after the first lift of refuse. Subsequently, annual cleaning is recommended.

One problem unique to landfills lined with polymeric materials is that should leachate penetrate the liner, it may then undergo further degradation and produce landfill gas. Gas buildup has been known to result in billowing up of the liner. Some provision for gas venting may therefore be desirable by using a permeable substrate on which the liner is placed. The performance of a liner should be closely followed by monitoring the quality of groundwater close to the site. However, it must always be borne in mind that in the event of a leak occurring, pinpointing its origin and repairing the damage is almost impossible. One option to minimize further groundwater pollution is to ensure that the level of leachate within the site is kept to a minimum and to cap the site with an impermeable material as soon as possible. Even then it may be necessary to continue to pump out leachate at regular intervals.

7.1.6 The Disposal of Hazardous Wastes

(i) General

This section deals with the landfilling of 'hazardous wastes.' The term 'hazardous wastes' covers a range of industrial wastes, the disposal of which calls for special procedures because of either their hazardous nature or physical characteristics. The principles and practices set out in previous sections relating to the assessment and design of landfill sites, and to their preparation, operation and restoration, are generally applicable to sites taking hazardous wastes. Table 7.1-17 is a good checklist. However, as would be expected, such wastes necessitate special consideration at all stages of landfill development. The characteristics of difficult wastes are wide ranging, therefore each site needs to be judged on its merits as to the types of waste it can receive. Appropriate control procedures must be adopted; generally, more stringent measures are called for compared to those employed for the disposal of household or similar wastes. Experience indicates, however, that with good management and operating practice, the landfilling of difficult wastes need pose no greater risks to human health or the environment than that presented by, say, household waste.

(ii) Selection of Disposal Technique

Types of Landfill Sites

Two broad types of landfill are considered: 'attenuate and disperse' sites in which reliance is placed on leachate attentuation processes acting within the landfill, followed by dispersion in surrounding geological strata as any leachate migrates away from the landfill, and 'containment' sites where the aim is to isolate any leachate from the environment, thus relying on attenuation within the landfill and on leachate management practices to prevent water pollution. Because of the nature of some hazardous wastes, doubts may exist about the attenuating mechanisms that are likely to operate and hence about the characteristics of the leachate. These doubts will dictate additional care when considering the use of attenuate and disperse sites for the disposal of difficult wastes. If components of particular wastes attenuate only very slowly or in ways which are not understood, a containment approach may need to be adopted, particularly if water resources are potentially at risk. It must be understood that not all wastes will undergo attenuation, no matter how long the retention time. Particular aspects of attenuation applicable to the disposal of difficult wastes are discussed later.

Disposal Options

In addition to having some choice in the type of disposal site to be operated, an operator has the choice of four main techniques. These are: codisposal, monodisposal, multidisposal and pre-treatment followed by disposal via the other options. These techniques are now described.

Codisposal

Codisposal is the conscious deposition of difficult wastes with household or other similar wastes to achieve specified objectives. A central objective is the maintenance of a balanced input of wastes to ensure that the attenuation processes are not overwhelmed. Controlling the rate of

TABLE 7.1-17
Solid Waste Landfill Design Checklist

Step	Task
1	Determine solid waste quantities and characteristics
	a. Existing
	b. Projected
2	Compile existing and generate new site information.
	a. Perform boundary and topographic survey
	b. Prepare base map of existing condition on-site and near-site
	(1) Property boundaries
	(2) Topography and slopes
	(3) Surface water
	(4) Utilities
	(5) Roads
	(6) Structure
	(7) Land use
	c. Compile hydrogeological information and prepare location map
	(1) Soils (depth, texture, structure, bulk density, porosity permeability, moisture, ease of excavation, stability, pH, and cation exchange capacity)
	(2) Bedrock (depth, type, presence of fractures, location of surface outcrops)
	(3) Groundwater (average depth, seasonal fluctuations, hydraulic gradient and direction of flow, rate of flow, quality, uses)
	d. Compile climatological data
	(1) Precipitation
	(2) Evaporation
	(3) Temperature
	(4) No. of freezing days
	(5) Wind direction
	e. Identify regulations (Federal, State, and local) and design standards
	(1) Loading rates
	(2) Frequency of cover
	(3) Distances to residences, roads, and surface water
	(4) Monitoring
	(5) Roads
	(6) Building codes
	(7) Contents of application for permit

TABLE 7.1-17 (continued)

Step	Task
3	Design filling area
	a. Select landfilling method based on:
	(1) Site topography and slopes
	(2) Site soils
	(3) Site bedrock
	(4) Site groundwater
	b. Specify design dimensions
	(1) Trench width, depth, length
	(2) Cell size
	(3) Cell configuration
	(4) Trench spacing
	(5) Fill depth
	(6) Interim cover soil thickness
	(7) Final cover soil thickness
	c. Specify operational features
	(1) Use of cover soil
	(2) Method of cover application
	(3) Need for imported soil
	(4) Equipment requirements
	(5) Personnel requirements
4	Design facilities
	a. Leachate controls
	b. Gas controls
	c. Surface water controls
	d. Access roads
	e. Special working areas
	f. Structures
	g. Utilities
	h. Fencing
	i. Lighting
	j. Washracks
	k. Monitoring wells
	l. Landscaping
5	Prepare design package
	a. Develop preliminary site plan of fill areas
	b. Develop landfill contour plans

TABLE 7.1-17 (continued)

Step	Task
	(1) Excavation plans - including benches
	(2) Sequential fill plans
	(3) Completed fill plans
	(4) Fire, litter, vector, odor and noise controls
	c. Compute solid waste storage volume, soil requirement volumes, and site life
	d. Develop final site plan showing:
	(1) Normal fill areas
	(2) Special working areas
	(3) Leachate controls
	(4) Gas controls
	(5) Surface water controls
	(6) Access roads
	(7) Structures
	(8) Utilities
	(9) Fencing
	(10) Lighting
	(11) Washracks
	(12) Monitoring wells
	(13) Landscaping
	e. Prepare elevation plans with cross-sections of:
	(1) Excavated fill
	(2) Completed fill
	(3) Phased development of fill at interim points
	f. Prepare construction details
	(1) Leachate controls
	(2) Gas controls
	(3) Surface water controls
	(4) Access roads
	(5) Structures
	(6) Monitoring wells
	g. Prepare ultimate land use plan
	h. Prepare cost estimate
	i. Prepare design report
	j. Submit application and obtain required permits
	k. Prepare operator's manual

input of difficult wastes is therefore always necessary. Much of the recent research effort on landfill processes has been concentrated on the effects of co-disposing difficult waste with household refuse. The results obtained have indicated that, when properly managed, codisposal can be regarded as a safe and efficient disposal option for many difficult wastes. (See Figure 7.1-9)

Codisposal normally utilizes properties in household-type waste to attenuate those constituents in difficult wastes which are polluting and potentially hazardous and thereby make their impact on the environment acceptable. The practice requires special precautions and management of all operations to ensure that it is both safe and environmentally acceptable. Difficult wastes destined for codisposal must be critically assessed and only those wastes which are compatible with household waste should be accepted for codisposal.

Monodisposal

Monodisposal is the disposal of wastes having the same general physical or chemical form by landfilling or lagooning. Following disposal the waste need not, necessarily, remain in the same physical form as it was produced. For example, Pulverized Fuel Ash (PFA) from power stations is almost always landfilled at monodisposal sites and frequently it is pumped there as a slurry and allowed to dewater. Producers of bulk inorganic chemicals often dispose of large quantities of waste at monodisposal landfills (e.g., gypsum from phosphoric acid production).

Multidisposal

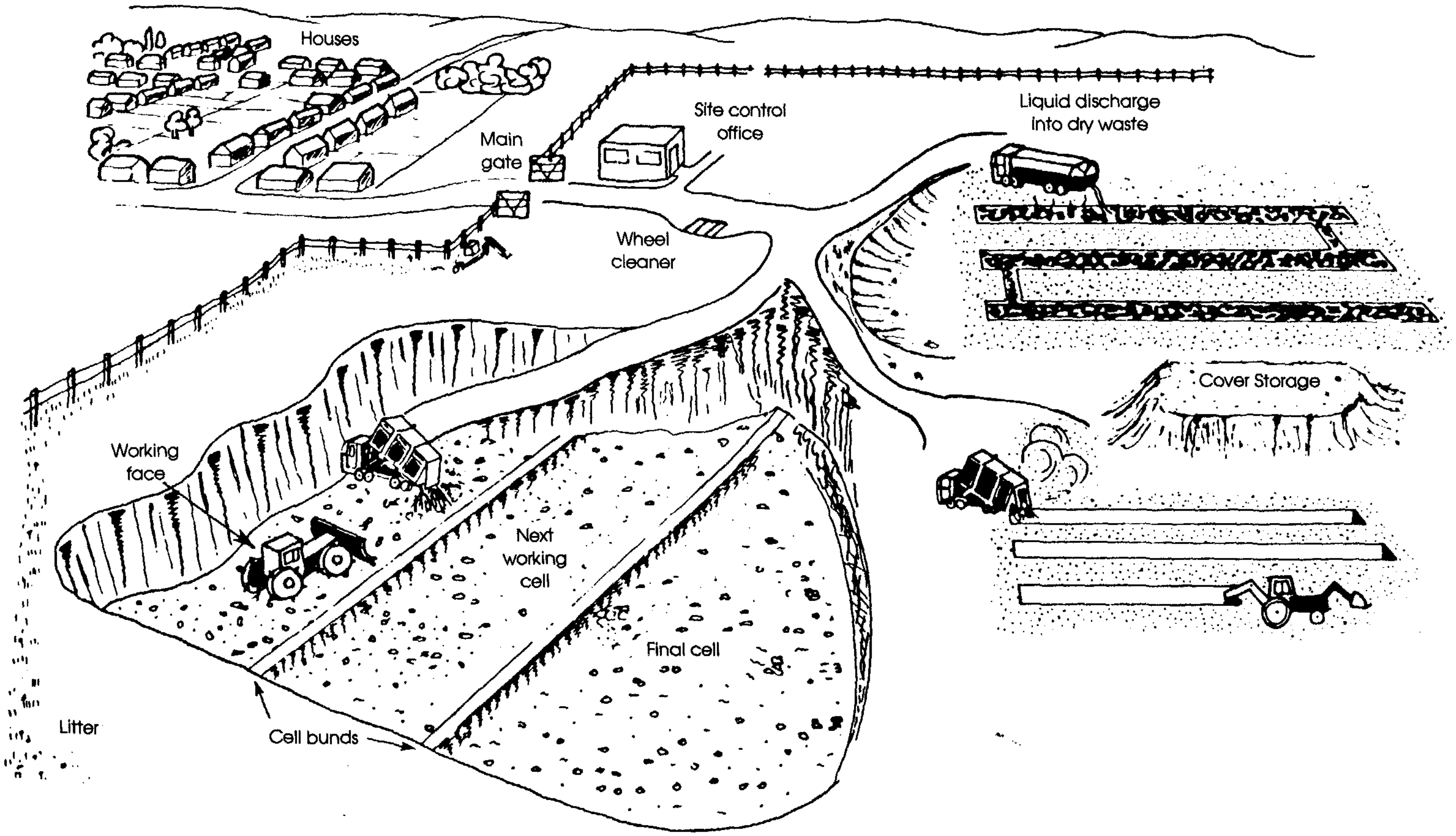
The aim of multidisposal is to turn deposited mixed heterogeneous waste into a no more hazardous and preferably more environmentally acceptable form than the individual component wastes. Multidisposal is generally used to describe the disposal of chemically different wastes which, like liquids or sludges, have similar physical forms. The deposit of mixed wastes either as liquids into lagoons or at sites accepting both inert and degradable industrial and commercial solid wastes may also be regarded as examples of multidisposal operations.

Where mixed wastes are to be deposited, it is vital that they not react to produce either worse polluting or more hazardous compounds, or significant concentrations of dangerous or noxious gases and vapors. Detailed chemical knowledge of the wastes and an evaluation of the products of reactions should be an integral part of any multidisposal operation.

A further type of multidisposal operation is the disposal of difficult wastes into inert industrial solids. While this is similar to codisposal, in this instance attenuation processes within the landfill do not occur. Finally, multidisposal has occasionally been used to describe disposal sites which can accept a range of different wastes which are landfilled in separate discrete areas of the site. Mixing of the wastes therefore does not take place. Such a practice is not significantly different from monodisposal other than possibly in scale. Accordingly this type of operation should be regarded as monodisposal. The term co-deposit is sometimes used.

FIGURE 7.1-9

Typical Co-disposal Landfill Site



Pre-treatment and Disposal

Some wastes are not suitable for landfill disposal in the form in which they arise by virtue of their physical or chemical properties and thereby present an unacceptable risk to personnel or the environment. Pre-treatment, often undertaken before the waste arrives at the landfill, can include simple bulk reduction such as: sludge dewatering to reduce the total quantity of waste, precautionary measures such as the secure bagging of loose asbestos to preclude fibre release during deposit at the landfill, various chemical treatment processes or the use of solidification processes (see Chapter 6).

(iii) Potential Impact on Water Quality

An appreciation of how landfill sites may affect water quality has already been given. Particular interest is attached to the potential impact of codisposal landfills since many of the components in an industrial waste will undergo some form of attenuation within a landfill when codisposal is practiced. Freshly deposited household wastes have been found to possess very different attenuating properties to those exhibited by mature wastes. For many difficult wastes, codisposal with mature (that is 1 to 5 years old) household waste has been found to be preferable if optimum attenuation is sought. Experience and research over the past decade has demonstrated that with proper management attenuation processes can act such that the codisposal of difficult wastes does not result in unacceptable environmental pollution; the polluting potential of the leachate should be no greater than that from household or similar waste. However, it must also be recognized that the detailed behavior of individual wastes, let alone that of complex mixtures within the physico-chemical and biological environment of a landfill, is not as yet fully understood. Some attenuation processes such as those involving pH-dependent reactions, ion exchange and adsorption are to some extent reversible. In order to prevent the sudden release of pollutants into the surrounding environment, a codisposal landfill operation must ensure that conditions which could give rise to undesirable reactions are not allowed to develop. For example, the disposal of chelating agents or acidic wastes into landfills which have received significant amounts of metallic sludges should be avoided.

(iv) Treatment of Hazardous Wastes Prior to Landfilling

Pre-treatment of waste should be considered prior to landfilling to achieve: (a) bulk reduction (e.g., dewatering of sludges); (b) reduction of hazard potential when handling or transporting to a landfill; or (c) conversion of the waste to an acceptable form for landfill disposal. Pre-treatment usually involves the use of physical or chemical processes such as phase separation, bulk reduction and solvent or solute separation. Chemical treatment may include neutralization, oxidation and reduction, precipitation, or may aim to change the physical form of the waste, for example, through solidification or encapsulation (see Chapter 6).

(v) Loading Rates and Compatibility

Chemical Interactions Between Wastes

When two or more wastes are to be deposited at the same location on a landfill site it is the responsibility of the site operator to ensure that the concentrations of reactive species in the wastes are sufficiently low that they may be safely deposited together. To achieve this it is essential that the composition of all wastes delivered to a landfill site are known before they are accepted at the site. Steps which must be taken to prevent the co-deposition of incompatible wastes include: (a) prior knowledge of the constituents in a waste, (b) disposing of non-compatible wastes in separate parts of the site, (c) the clear labeling of trenches for liquid waste reception, and (d) competent supervision of the discharge of wastes. Undesirable reactions which can occur when mixing incompatible wastes are given below.

- The generation of heat by chemical reaction which in extreme cases may result in fires or even explosions (e.g., alkali metals, metal powders).
- The generation of toxic gases (e.g., arsine, hydrogen cyanide, hydrogen sulphide).
- The generation of flammable gases (e.g., hydrogen acetylene).
- The generation of gases such as nitrogen oxides, carbon dioxide, sulphur dioxide, chlorine, and
- Dissolution of toxic compounds including heavy metals (e.g., complexing agents like chelates).

A simplified version of an extensive study comparing the reactions between the 12 most common categories of hazardous wastes is presented in Figure 7.1-10. Very little information is available on adverse reactions involving three or more wastes although codisposal involving several waste streams has been successfully undertaken on the basis of accumulated experience. It is important that where such experience is lacking and where adverse reactions are possible, small scale testing should be undertaken by competent staff to establish the compatibility of wastes before they are landfilled.

Waste-liner Compatibility

The integrity of the lining material is vital. It has been recognized recently that certain wastes may adversely affect the performance of a liner. It is important, therefore, to ensure compatibility between the liner material used and the wastes to be deposited. In view of the uncertainties that exist, especially on the long-term behavior of liners, certain precautions described below appear desirable in controlling the deposition of difficult wastes.

FIGURE 7.1-10

Compatibility of Selected Hazardous Wastes

1	Oxidising Mineral Acids	1											
2	Caustics	H	2										
3	Aromatic Hydrocarbons	H F		3									
4	Halogenated Organics	H F GT	H GF		4								
5	Metals	GF H F			H F	5							
6	Toxic Metals	S	S				6						
7	Sat Aliphatic Hydrocarbons	H F						7					
8	Phenols and Cresols	H F							8				
9	Strong Oxidising Agents		H	H F		H F		H		9			
10	Strong Reducing Agents	H F GT			H GT				GF H	H F E	10		
11	Water and Mixtures containing Water	H			H E		S				GF GT	11	
12	Water Reactive Substances	← Extremely reactive, do not mix with any chemical or waste material →											12

E	Explosive
F	Fire
GF	Flammable Gas
GT	Toxic Gas
H	Heat Generation
S	Solubilisation of Toxins

Clay and Bentonite Liners

In the light of very limited current knowledge it is suggested that incompatible difficult wastes should not be deposited adjacent to in-situ clay which is less than 10 meters thick. Liner materials based on bentonite exhibit properties which make them compatible with a wide range of chemicals. Unlike many synthetic liners, bentonite is not adversely affected by oils and organic solvents except at high concentrations (>50%) although these would not be deposited at bentonite lined sites. There is no known experience of a codisposal site having been lined with bentonite.

Polymeric Membranes

Various polymeric membranes are available which, though compatible with inorganic wastes, may to varying degrees be susceptible to attack by organic chemicals. Such membranes should not be used therefore to line sites that are expected to receive organic chemical wastes until their compatibility with the wastes has been established. Difficult wastes should not be deposited within 5 meters of an artificial liner.

Loading Rates

There is a need to limit and control the quantities of wastes deposited at a landfill site in order to maintain appropriate proportions between various types of waste. Codisposal sites in particular depend on careful regulation of waste types and quantities deposited to ensure that attenuating processes do not become overwhelmed. Table 7.1-18 summarizes some design recommendations for municipal sludge landfills, some of which are refuse landfills that also receive sludge. For difficult wastes to be co-disposed with household or commercial wastes their rate of deposition will be limited by the attenuation capacity of the latter.

Many codisposal landfills are licensed to take a wide range of wastes containing a large number of potentially polluting components, and it is necessary therefore to identify those constituents in wastes which will require individual monitoring. It is considered unnecessary to set loading rates for every constituent since the analytical requirements would be too onerous and could not be technically justified. Similarly it is impracticable to measure precise inputs of particular components in all waste consignments received. Loading rates should take such imprecision into account. Nevertheless, at all sites there will be certain chemical species in wastes received that must be carefully estimated, regulated and monitored. Conditions vary from country to country depending on the composition of the waste and climatic factors. Hence, any loading rates suggested below are only indicative and must be confirmed by careful monitoring at the particular site.

Guidance on the Loading Rates for Specific Wastes

Liquid Wastes

In general, it is the volume of liquid wastes, rather than the capacity of the site to attenuate constituents in the waste, which limits the addition

TABLE 7.1-18

Design Criteria for Mixed Hazardous Wastes

Method	Slude solids content	Trench width	Bulking required	Bulking agent	Bulking ratio[a]	Cover Thickness Interim	Cover Thickness Final	Imported soil required	Sludge application rate (in actual fill areas)	Equipment
Narrow trench%	15-20%	2-3 ft.	No[c]	---	---	---	2-3 ft.	No	1,200-5,600 yd^3/acre	Backhoe with loader excavator, trenching machine
	20-28%	3-10 ft.	No[c]	---	---	---	3-4 ft.			
Wide trench	20-28%	10 ft.	No[c]	---	---	---	3-4 ft.	No	3,200-14,500 yd^3/acre	Track loader, dragline, scraper, track dozer
	228%	10 ft.	No[d]	---	----	---	4-5 ft.			
Area fill mound	220%	---	Yes[d]	Soil	0.5-1 soil 1 sludge	3 ft.	3-5 ft.	Yes	3,000-14,000 yd^3/acre	Track loader, backhoe with loader, track dozer
Area fill layer	215%	---	Yes[d]	Soil	0.25-1 soil: sludge	0.5-1 ft.	2-4 ft.	Yes	2,000-9,000 yd^3/acre	Track dozer, grader, track loader
Diked containment	20-28%	---	No[b,c]	Soil	0.25-0.5 soil 1 sludge	1-2 ft.	3-4 ft.	Yes	4,800-15,000 yd^3/acre	Dragline, track dozer, scraper
	228%	---	No[b,c]							
Sludge/ refuse mixture	23%	---	Yes[d]	Refuse	4-7 tons refuse: 1 wet ton sludge	0.5-1 ft.	2 ft.	No	500-4,200 yd^3/acre	Dragline, track loader
Sludge/ soil mixture	220%	---	Yes	Soil	1 soil: 1 sludge	0.5-1 ft.	2 ft.	No	1,600 yd^3/acre	Tractor with disc, grader, track loader

---

[a] Volume basis unless other wise noted.

[b] But sometimes used.

[c] Land-based equipment

[d] Sludge-based equipment

1 ft = 0.305 m
1 yd^3 = 0.765 m^3
1 acre = 0.405 ha

of more liquids to landfills. For management purposes water balance calculations must be used throughout the lifetime of any landfill site accepting liquid wastes. The quantity of liquid waste deposited will be sensitive to, and may be temporarily controlled by, other events which contribute to water balance, such as above average rainfall or the unavailability of other wastes having sufficient absorptive capacity. The effects will be most noticeable at containment landfills. At such sites it is therefore desirable to control the addition of liquid wastes according to the results obtained from monitoring the liquid levels in the site.

Acid Wastes

Concentrated acids should not be directly landfilled since they are extremely corrosive and if landfilled have the potential to cause fires and produce toxic gases by chemical reaction. When for technical reasons landfill is the best practicable option, the acid should at least be diluted prior to deposit. Household and similar wastes do have a significant capacity to neutralize acids, and this capacity is normally maintained until degradation of the waste is almost completed.

Codisposal of acidic wastes should only be at a rate which will maintain a near neutral pH value in any leachate produced. In addition four other effects must be considered. First, acid wastes can dissolve metals present in other waste which will subsequently need to be attenuated to acceptable levels. Second, the quantity of liquid represented by any acids deposited must be within the sorptive capacity of the landfill. Third, acid inevitably will have a toxic effect on the local microbiological population although evidence suggests that recovery is relatively rapid. Fourth, acids can react with other materials in a landfill to produce toxic gases. Such effects should be monitored in areas adjacent to those receiving acids and input quantities regulated accordingly.

The codisposal of acidic wastes into trenches excavated into previously deposited wastes depends upon the availability of a sufficient quantity of absorptive waste for neutralization to take place as the acid waste moves away from its immediate point of disposal. Mature household waste (1 to 5 years old) usually affords good buffering capacity. The lifetime of the trenches will vary. Where vertical flow can occur, at sites located on permeable strata, a lifetime of only several months can be expected; while at sites having low-permeability base material, where only lateral leachate migration can occur, there is evidence that the neutralizing capacity may not be exceeded for several years.

For direct disposal to landfill the concentration of hydrochloric and sulphuric acids should not exceed 20%, nitric acids 5% and chromic acids 5% (as chromium VI). Chromic acids must be neutralized or diluted to pH 4 to attenuate uncontrolled reactions with organic matter. Mixed acids require special consideration since their reactivity is generally greater than that of the individual acids alone.

The loading rate for sulphuric acid should be limited to a maximum of 20 kg of acid (by weight of pure acid) per ton of mature household waste. When calculating initial loading rates at sites located on permeable strata it will be necessary to ensure that there is sufficient waste directly below and immediately adjacent to the disposal trench. Where lateral spread of waste acid is expected, larger amounts of household waste will be available for neutralization. Under these conditions acceptable quantities may be much larger but this can only be established by careful monitoring within the site in relation to the permeability of wastes previously deposited, the available absorptive capacity remaining around the trench and the binding effect of precipitation reactions occurring at the base of the trench.

In the absence of lateral liquid movement, a loading rate of no more than 5 kg of hydrochloric acid per ton of mature household waste is suggested, since hydrochloric acid is particularly effective at mobilizing metals. The mobilization of heavy metals is a good indicator of the effect of acid codisposal when monitoring sites. Zinc and nickel are typically the first to appear in leachate. The degree of mobilization is likely to depend as much on the competition by other ions in the waste acid as on the effect of the acid itself. Dilute solutions of hydrochloric or sulphuric acid (less than or equal to 1%) are less aggressive in a landfill and loading rates will probably be controlled by the quantity of liquid rather than any chemical reaction.

Heavy Metal Wastes

In terms of environmental impact the principal heavy metals of concern are cadmium, chromium, copper, lead, nickel and zinc. They are often found together in a single waste stream. Wastes which may contain heavy metals include hydroxide sludges, ashes, slags and some paint pigment wastes. Such wastes are almost always landfilled in a water insoluble form, and care is needed to ensure that they remain so in the landfill environment. Excluding cadmium, heavy metals are frequently considered together when setting discharge limits to sewers for leachate. The concentration of zinc in leachate from fresh refuse (generally about 10 mg/liter) may require separate handling. Cadmium should also be considered separately. The loading rates for heavy metal wastes will depend on their leachability potential and on subsequential absorption/desorption and precipitation reactions.

The concentration of heavy metals found in household and similar wastes can vary widely. Codisposal of heavy metal wastes should not be allowed if it is likely to increase their concentrations beyond those observed in leachate derived from fresh refuse; codisposal is best carried out using household waste more than 6 months old.

Limited evidence suggests that an initial loading rate of up to 100mg of soluble chromium, copper, lead or zinc per ton of mature household waste is unlikely to produce a significant change in leachate concentrations 3 meters distance from the heavy metal waste. The likelihood of insoluble metal compounds becoming soluble also needs consideration. In the absence of field data demonstrating the contrary it should be assumed that nickel will not undergo significant attenuation other than by dilution and dispersion. The codisposal of heavy metal wastes may inhibit microbial organic degradation processes within the waste mass. Resistant populations can, given time, be re-established. It is desirable, therefore, to monitor the micro-biological activity in such instances.

Arsenic, Selenium and Antimony

Provided it can be demonstrated that the concentration of arsenic does not exceed 10mg/liter in leachate from a codisposal site, industrial wastes containing up to 1% total arsenic may be acceptable for codisposal. In wastes containing soluble arsenic compounds pre-treatment will usually be required to ensure that the concentration of arsenic in leachate from the site does not exceed 10mg/liter.

Isolated small quantities of concentrated arsenical wastes containing less than 0.5 kg of total arsenic may usually be co-disposed without pre-treatment. However, such wastes should be covered immediately by at least 2m of refuse, and, except for arsenic sulphides, should where practicable be mixed with lime. The production of nascent hydrogen which can lead to the generation of arsine gas must be obviated in all landfill operations involving arsenical wastes.

Wastes containing selenium and antimony should be treated in a similar manner to wastes containing arsenic since their chemical behavior is generally analogous. However, they do not form stable hydrides and there is therefore less likelihood that they will present a problem. Nevertheless, loading rates for selenium and antimony should be based on similar criteria to those used for arsenic.

Mercury

Before establishing loading rates for wastes containing mercury in a landfill, it is essential to know the chemical form in which the mercury is present. Although all mercury compounds have the potential to accumulate in tissue, organic and particularly alkyl mercury compounds are the most toxic. Loading rates should therefore be set to prevent elevated concentrations of mercury occurring in leachate and consideration given to the possibility of mercury vapor being released.

Where significant quantities of mercury-bearing wastes are involved, their disposal to landfill should be restricted to sites where little or no horizontal leachate flow occurs. A special disposal area should be designated for wastes containing more than 20 mg/kg inorganic or elemental mercury or more than 2 mg/kg organic mercury. Such an area will require special restoration treatment and its after-use may be restricted. For non-specific mercury-bearing wastes the loading rate should not exceed twice the average concentration of 2g/ton found in household and similar wastes. Individual deposits of solid wastes containing more than 1 kg of mercury are not acceptable, unless distributed through large areas of uncontaminated refuse.

Such initial loading rates may not take full advantage of the efficiency with which soluble mercury is precipitated as mercuric sulphide in the anaerobic zone of a landfill. However, mercury-bearing wastes containing viable concentrations of non-organic mercury should be recovered, and those containing more than 100 mg/kg organic mercury should not be accepted for direct landfill.

Phenolic Wastes

Phenolic compounds such as phenol, cresol and xylenol are moderately soluble in water. Their presence at extremely low concentrations in drinking water will, on chlorination, impart an unpleasant taste and odor to the water. The codisposal of phenolic wastes therefore requires particular care to prevent contamination of water used as a source of potable water. Phenolic wastes in landfill sites are attenuated by both aerobic and anaerobic degradation following sorption. Long residence time in the landfill and good contact between the waste and leachate is necessary to achieve significant attenuation.

Phenols have a greater affinity for organic materials than for water. Low permeability immediate cover material should be employed to delay leaching and enhance the contact between the phenolic and the organic fractions of codisposal waste. Landfill sites on low permeability strata and the use of discrete cells offer distinct advantages for the codisposal of phenols.

Leachates from fresh refuse have been found to contain total phenol concentrations of 1 to 10 mg/liter whereas leachates from older refuse contain about 1 mg/liter. Codisposal using refuse more than 6 months old therefore offers distinct advantages. To maintain concentrations within the 1 to 10 mg/liter range water infiltration should be minimized. Deposition of solid wastes containing phenols may be covered with a minimum of 300 mm of clay or similar low permeability material to minimize water infiltration into the waste. Phenolic wastes should not be deposited where perched water tables are likely to be a problem. Liquid phenolic wastes are more likely to be acceptable at containment landfill sites but in any case only in very small quantities or at low concentrations.

The loading rate of phenolic wastes to landfills should not normally exceed 2 kg of total phenols per ton of refuse, unless site-specific data confirm that biodegradation of phenols is particularly effective. Acceptable attenuation cannot be inferred by an observed low initial rate of leaching. There is evidence that concentrations of 1 mg/liter phenol are reduced quite rapidly in the unsaturated zone outside a landfill site in strata possessing slow intragranular flow. Attenuation is thought to be by anaerobic degradation which may occur within many landfills containing only household wastes, but this attenuation mechanism cannot be relied upon to occur in all cases all the time.

Oily Wastes

Oily wastes typically occur as three different types: free oils, industrial oil-water emulsions and oily mousses, which are usually a solid material such as that derived from marine oil spills. Attenuation of oils in landfills is largely through sorption by solid wastes. Degradation rates of the sorbed material are extremely slow. Aerobic conditions using land farming techniques have been shown to be more efficient in degrading oils. For these reasons wastes containing oils should go to landfill only when other means of disposal are not practicable, but it is recognized that landfilling may be necessary for oil removed from beaches after an oil spill at sea. Recovery of oil from waste, for reuse or use as a fuel, should be undertaken whenever possible. Oily wastes can be incinerated as a source of heat when reuse is not possible. Oils which cannot be recovered or incinerated are likely to be so because they have become contaminated with unacceptable materials. Loading rates must take into account the exact nature of such contaminants in addition to the oils.

The concentration of mineral oils found in leachate from household wastes is typically 10 mg/liter. Where oils are deposited at loading rates of not more than 2.5 kg of oil per ton of well compacted refuse, the concentration of oil in the leachate is unlikely to exceed 10 mg/liter after passage through a few meters of household wastes.

Oil-water emulsions, such as cutting oils which also normally contain bactericides and additives to stabilize the emulsion, have been found to infiltrate landfills and to be absorbed by household waste in a similar manner to aqueous wastes. Subsequent breakdown of the emulsion allows absorption of the free oil by solid waste in the same way as with mineral oils, the aqueous phase becoming leachate. In order to keep oil concentrations in leachate within the range of those from household waste, loading rates should not exceed 40 kg of oil emulsion or 0.4 kg of oil per ton of refuse. If disposal trenches are used, it should be noted that some emulsions, on breakdown, tend to bind the base and sides of trenches.

There is some evidence to suggest that containment sites are more permeable to oils than to water. Site loadings must therefore be limited to such amounts that leachate concentrations typical of those from refuse alone are not exceeded. In addition, due to their stability, oils are unsuitable for deposition within the top layer of refuse at codisposal sites.

Pesticide Wastes

Concentrated pesticide wastes are unsuitable for landfill disposal and should be destroyed by high temperature incineration. Wastes containing low concentrations of pesticides (e.g., tank washings, may be acceptable for codisposal although such practices should be closely monitored).

Maximum concentrations of 10 g/m3 or 20 mg/kg of pesticide active ingredient in the waste after deposition are recommended. Leaching of pesticides appears to be largely controlled by their water solubility so that it is possible that these concentrations could be increased for some pesticides having low solubility. However, evidence will be required by the licensing authority to show that such materials can be deposited in an environmentally safe manner. Deposition should take place in trenches excavated in mature household waste. Immediate backfilling is recommended for solid waste, although trenches used for liquid wastes may be left open until an acceptable loading has been achieved.

Polychlorinated Biphenyl (PCB) Wastes

PCB's are persistent in the environment and there is no evidence that they degrade significantly even under the anaerobic conditions within a landfill. They are relatively immobile and their water solubility is low, typically 0.01 to 0.001 mg/liter. Sorption of PCBs is likely to occur on solids with high organic content. The disposal of anything but minimal amounts of PCBs dispersed through a landfill is unacceptable. Recent guidance from the UK Department of Environment has indicated that 20 mg/kg PCB in waste after deposition, in a mixture with bioactive waste within the body of a landfill, should be regarded as a maximum concentration.

Solvent Wastes

Solvent wastes which burn unsupported at 40 degrees or below should not be landfilled. Other solvents of lower flammability and particularly those mixed with other wastes can be landfilled only in small quantities.

Acid Tars

Acid tars arise mainly from three industries: the production of metallurgical coke, the manufacture of highly purified mineral oils such as the white oils used by the pharmaceutical industry and the re-refining of used oils. They all contain a mixture of often highly viscous thixotropic organic material and sulphuric acid of variable but usually high concentration and in some cases even oleum. Acid tars do not degrade to any significant extent in a landfill environment and when buried have a tendency to migrate upwards, leading to uncontrolled releases at the surface.

The landfilling of acid tars raises very difficult problems and it is doubtful whether such wastes should be landfilled at all. Very few landfills are capable of safely accepting acid tars and none should do so without pre-treatment; as a minimum the free acid should be neutralized and any free oil separated before the residue is landfilled by codisposal.

Cyanide Wastes

Cyanide can be present in waste in various chemical forms such as: 'simple cyanide', 'complex cyanide', 'cynate', 'thiocyanate' and 'nitriles'. In all instances it is the ability of the waste to produce free cyanide which dictates the loading rate in the landfill. In general, landfilling cyanide wastes are recommended only for small quantities.

Cyanide wastes should be deposited in segregated areas of a landfill site where the pH will not fall below 5, thereby minimizing production of hydrogen cyanide. Complex cyanide wastes, such as soils contaminated with gas works waste containing 'spent oxide,' should be deposited at loading rates which take account of the concentration of free cyanide and other contaminants which are likely to be present such as phenols. Spent oxide typically contains 10 to 50 mg/kg free cyanide.

In general, cyanide waste should be deposited at a loading rate of no more than 1 g of free cyanide per ton of refuse. An exception may be wastes from the redevelopment of old gas works sites where significant quantities may require disposal during a limited period of time. In such cases, it may be acceptable to exceed the recommended maximum loading rate provided such a deposit does not overwhelm the attenuation processes operating within a landfill which could lead to long-term environmental problems. Containment landfills may offer considerable advantages for the disposal of these wastes.

Tannery Sludges

The loading rate for the codisposal of tannery sludges depends on both the waste content and chromium content. For a typical sludge solids content of 2% the loading rate should not exceed 1 ton of sludge per 15 tons of refuse. However, dewatering of the sludge prior to landfill is preferable, the loading rate of the sludge being adjusted accordingly. Tannery sludges typically contain up to 3.5% chromium (III) in the dry solids. In practice the leaching rates of chromium from these wastes are very slow and elevated chromium concentrations are rarely observed, nevertheless, monitoring for the presence of chromium in leachate is advisable.

Monitoring

Monitoring is essential at all sites where codisposal is practiced. The objective of monitoring is to build up a picture of the general behavior of a site. The distribution of key species in leachate within a site, together with physical properties such as temperatures and water levels, indicate the general state of a landfill and show the effect of wastes deposited. Initially, wastes deposited should be limited to household and other bioactive general waste to allow the monitoring network to become established and to ensure that a sufficient depth of sorptive waste is available before codisposal commences. The impact of the initial deposits of co-disposed industrial waste can thus be monitored and provide information on appropriate loading rates for the site.

It is unlikely that the loading rates initially set will remain relevant for the lifetime of a site. The monitoring data, which should be interpreted and discussed among the site operator, water authority and disposal authority, will enable a clear picture of site behaviour and its attenuating properties to be established which, in turn, will indicate whether or not loading rates require adjustment.

(vi) Site Operations

Monodisposal

Monodisposal sites normally occupy several hectares. Consequently, large areas of land may be committed to this use for some considerable time. With proper management it is possible to minimize the impact on the environment to an acceptable level. In many instances such sites can be operated on the principle of progressive restoration, using a cell method which allows completed areas of the landfill to be made available for other uses as each cell is restored. Even during operations, sympathetic landscaping can reduce the level of intrusion and help reduce the environmental impact.

Monodisposal normally makes little or no use of attenuation processes within the landfill. The restricted concentrations and chemical composition of the waste and its relatively uniform nature usually limit the physical, chemical and biological attenuation mechanisms that can operate.

Codisposal

The way in which codisposal is carried out depends on the nature of the waste and the requirement to achieve satisfactory loading rates. The following paragraphs give general guidance, but it should be recognized that codisposal operations are site specific.

Codisposal of Liquid Wastes

Before any liquid wastes are accepted for codisposal, the capacity of the site to receive them should be assessed by making a water balance calculation. Specific areas of the site filled with medium density household or similar wastes (0.6-0.7 tons/m3) should be designated for liquid disposal to facilitate its absorption. Highly compacted wastes may have a lower capacity for absorbing liquids. Deposition of liquids into baled wastes is inadvisable, since high density bales limit the quantity of liquid which can be absorbed as well as encouraging fissure flow. It is preferable to deposit liquids into wastes which are at least 6-12 months old.

Liquid Reception Facilities

Liquid wastes are usually taken directly to their point of disposal on a landfill or sometimes stored temporarily in tanks or sumps at the site entrance from which they can later be pumped into trenches or lagoons on the landfill. Precautions to ensure that only compatible liquids are mixed in temporary storage tanks is essential.

Methods of Codisposal of Liquid Wastes

Four methods are available for the applications of liquid wastes to landfills: trenching or lagooning, injection, spraying, and surface irrigation.

Trenching and Lagooning

Trenches or lagoons should not be excavated in areas of the site previously used for the deposition of wastes which will be incompatible with the incoming wastes. Other possible hazards must also be considered; for example, no excavations should be allowed where asbestos has been previously deposited. All excavated material should be dealt with in such a way that no nuisance is created, such as windblown material or odors. Trenches and lagoons should always be clearly marked and fenced. Trenches should always be used for the codisposal of odorous wastes, nuisance from odors can be reduced by covering the trenches with such materials as railway sleepers, plastic, corrugated iron or purpose built covers.

The distance between trenches should be at least 5 meters and they should be at least 10 m away form the site boundary. At lined containment sites it is necessary to leave sufficient underlying waste and take other measures needed to protect the integrity of the liner. Precautions should be taken to ensure that the sides of the trenches are stable and fencing provided to prevent unauthorized or inadvertent access. Edge protection may be necessary where access to trenches is required.

The aim of trenching is to provide sufficient internal surface area to allow seepage into the waste mass already deposited. The number and disposition of trenches required will depend both on the quantity and type of incoming wastes. Separate trenches should be allocated to different types of waste. The liquid levels should not be allowed to reach the top of the trench at any time, and surface waters should be diverted away from them. Fine material can accumulate on the base of a trench and retard seepage. This may be alleviated by providing a number of interconnected trenches using an overflow 'weir' system. Liquid codisposal operations should always be monitored.

Lagooning is the disposal of pumpable fluids into a bounded area on the surface of a landfill, or more usually into shallow pits excavated in waste already deposited. Where codisposal lagooning is practiced, care must be taken to ensure that the sorptive capacity of the underlying solid waste is not exceeded and disposal must be carefully controlled and managed to avoid chemical incompatibility. Liquids are dissipated by seepage into the waste, and during summer months by evaporation. Completed lagoons may have poor load-bearing characteristics, and should remain fenced until the land becomes stabilized.

Injection

Liquid wastes can be injected below the surface of a completed landfill using facilities which have been installed during the lifetime of a site. Such facilities might include perforated pipes or columns of pipes installed while the landfill was operational. They can extend from several meters above the base of the landfill up to the final level for waste deposition. The number of 'wells' provided and the outlets at different depths to utilize fully the capacity of all deposited solid waste will depend on prospective waste arisings, and the size of the site. Specific 'wells' should be designated for specific wastes.

The main advantage of injection into completed sites is that a more accurate assessment of the water balance can be made before liquid wastes are accepted. The quantity of incoming liquid wastes can therefore be regulated to ensure that the absorptive capacity of the landfill is not exceeded. Injection can also offer certain advantages for the disposal of odoriferous wastes.

Spraying and Surface Irrigation

The spraying of liquid wastes over the surface of deposited wastes on landfill sites makes use of evaporation to reduce the volume and possibility of real oxidation to replace the organic loading of liquid wastes, and can also possibly improve leachate quality by allowing further reaction with the solid waste mass to occur. Care must be taken to avoid pollution of surface water and excessive ponding. Spray drift and potential health hazards to operators also need to be considered.

Codisposal of Sludges

Sludges and slurries can be deposited in trenches in a similar manner to liquid wastes but rapid or blinding may occur. Alternatively trenches can be excavated in waste already deposited in advance of a working face, the sludge deposited and immediately covered. Sludges may also be deposited with incoming solid waste provided that due regard is taken of water balance. Sludges should be deposited at the base of the working face and covered immediately with solid waste. Care should be taken to avoid running vehicles over the sludge.

Codisposal of Solid Wastes

Generally, solid difficult wastes should be dispersed along the base of the working surface and covered immediately. Such wastes should not be deposited within 5 m of a site boundary or within the first lift of waste deposited on a site. If difficult solid wastes are to be deposited at sites where thin layering techniques are employed, special areas must be designated away from the working face.

Impermeable wastes are not suitable for depositing in layers over large areas since they are likely to give rise to perched water tables and channelling of the leachate. Attenuation processes within the landfill are consequently likely to be inhibited under such circumstances.

Codisposal of Awkward Wastes

Dusty Wastes

Occasionally wastes arise which are extremely fine and light. Unless they are handled and landfilled with due care and attention, such wastes can give rise to significant dust problems, both on the landfill and possibly outside its boundaries.

Whenever practicable, such wastes should either be packaged, for example in biodegradable material, or dampened sufficiently to suppress dust and then deposited in trenches. The area around the trenches should be kept damp to prevent dusty material drying out, and the trenches should be backfilled as soon as practicable. Where it is not practicable to moisten the waste beforehand, it should be fed by gravity using water sprays into the trench through a large diameter pipe. It will often be necessary for site workers to wear appropriate respiratory protective equipment during disposal of dusty wastes. Dusty wastes which present an additional acute hazard due to the presence of toxic materials should not be landfilled directly but should be processed to remove the hazard before disposal.

Asbestos

The landfilling of asbestos requires particular care. All fibrous and dusty asbestos wastes should be accepted at a landfill site only in robust plastic sacks or similar wrapping, as required by regulation. These should be sufficiently strong that they do not burst during loading, transport and unloading. Intact articles made from asbestos cement or bonded asbestos need not necessarily be separate consignments. On arrival, the delivery vehicle should be directed to the designated area. The current practice is for fibrous or dusty asbestos to be deposited at the base of a working face or preferably into previously excavated trenches. The asbestos waste should be carefully deposited and the bags should not be dragged around. At least 0.5 m of other suitable wastes must be immediately spread over the bagged loose asbestos, but 0.2-0.25 m is sufficient for hard bonded asbestos, such as asbestos cement. In addition, the top, sides and face of the deposited asbestos should not be less than 0.5 m from the surface or flanks of the current working face. Asbestos wastes should not be deposited within the top 2 m of a site.

Procedures should be prepared for the eventuality that bags containing fibrous or dusty asbestos waste burst during handling or disposal at a site. The procedures should ensure that exposure of all personnel to airborne asbestos fibres is kept as low as possible and in any case does not exceed the relevant control limits. Such procedures should include:

- Immediate damping down of any asbestos spillage;

- immediate covering of any spillage with a substantial layer of other waste in a manner which avoids the generation of airborne dust;

- where personnel are likely to be exposed to asbestos they should wear adequate respiratory protection of a type approved by the Health and Safety Executive;

- if personnel become contaminated they should change outer clothing, shower and dispose of contaminated clothing after suitable bagging; and

- if a vehicle or other equipment becomes contaminated it should be washed thoroughly and the washings disposed of in an acceptable manner.

Odorous Wastes

A number of industrial wastes, including those from the intensive rearing of animals and by-product rendering and some chemicals manufacture, can give rise to odors. Odors emanating from landfill sites must be prevented generally and particularly when the site is located near housing. Adequate waste reception and handling facilities are essential to controlling obnoxious industrial wastes. Prior notification of delivery of such wastes is normally necessary and its reception and disposal should be undertaken only in suitable weather conditions. Various techniques are available to minimize odors from wastes being landfilled. These include below surface injection of liquids and the immediate covering of solids with materials possessing odor-suppressing properties.

Drummed Wastes

As a general principle, drummed waste is unacceptable at landfill sites. Large quantities of drums pose stability problems when landfilled since it is impossible to eliminate all void space between the drums and there is invariably ullage space in the drums. Drums eventually corrode away and release their contents. Landfilling drums which contain unacceptable wastes therefore pose a very high risk. However, when in exceptional circumstances and for sound technical reasons, the landfilling of waste packaged in drums is the best practicable environmental option, the total volume of such wastes landfilled should be carefully regulated. Practices used should prevent localized concentrations of drums occurring. Only full drums of particular types of drummed solid wastes (e.g., certain distillation residues with softening points well above temperatures likely to be encountered in landfill sites, should be allowed).

Wastes in drums are frequently heterogenous and are often difficult to identify. Careful and detailed labeling by the waste producer of the contents of each drum should be required. To ensure that the wastes he is accepting are as described, the landfill operator should inspect all drums on arrival at a landfill site and confirm that the contents conform to their description and labelling. To facilitate this inspection, only lidded open-top drums should be used to contain the waste.

Even with regular, consistent labeling on the part of the waste generator, the landfill operator cannot be absolutely certain that the composition of each consignment of waste he receives has not been subject to significant changes. Landfill operators should not therefore accept any drummed waste unless they are satisfied that they know by inspection or analysis the contents in each drum and that the acceptance of these materials is permitted by their disposal license.

Inevitably, the contents of drums for disposal will not always be adequately identified; this is particularly the case when factories are being cleared out. A waste disposal contractor will therefore need to be provided with the results of chemical analysis of each drum of waste before he can decide how best to deal with it. Good landfill practice demands that only those wastes whose composition is known to be compatible with the landfill environment should be landfilled. Chemical treatment or incineration will be the appropriate method of disposal for some wastes. In any case the landfill operator will need to be satisfied that the contents of the drums are appropriate for landfill disposal based on the results obtained from the sampling and analysis. Sampling should be carried out at the point of arising.

Before any drum is sampled, attempts should be made to establish at least the range of substances which might be in each drum. During the opening and subsequent sampling, precautions taken and the protective equipment worn should be appropriate to the risk posed by the most hazardous substances likely to be found. Once the contents have been identified each drum should be labelled accordingly.

At landfill sites accepting any drummed waste a designated reception and storage area is required. This should be of a suitable size to allow lorries to maneuver and park safely. The storage area should afford adequate security and be of a size sufficient to hold at least one days quantity of drums suitable for land disposal. As far as practicable, inspection and sampling of all drums should take place with the drums still on the vehicle. Wastes found to be unsuitable for landfilling should be returned to the waste producer for more appropriate disposal elsewhere (e.g., at a waste treatment facility). Only when the inspection has proven that the wastes are acceptable should the drums be unloaded. Unloading should be supervised and appropriate handling equipment employed to avoid accidental damage to the drums and spillage of their contents.

The disposal of all drummed wastes found to be acceptable should be undertaken without delay. Drums should be spaced out across the base of the working face in intervals of not less than 0.5 m and immediately completely surrounded and covered with compatible waste or other cover materials. Some drummed waste (e.g., solid pesticide formulations, should be emptied out and the contents spread as widely as possible in a mixture with biodegradable waste within the landfill).

The disposal of drums containing liquid wastes presents many problems. Such drums should be emptied and their contents disposed of in accordance with good landfill practices such as trenching and lagooning. Normally, decanting and bulking up of drummed liquids should always be undertaken at their point of arising. However, it is recognized that in isolated instances it will not be practicable or safe to do so. In such cases, the drums should not be landfilled but their contents decanted at the landfill site into trenches or lagoons designated for the particular type of waste. Again, suitable precautions must be taken at all stages to ensure the safety of operatives. It must also be ensured that only compatible wastes are deposited and that liquid loading rates are not exceeded. Drums that have contained liquid wastes should be reused as containers for other material only if they can be safely decontaminated. Reuse of drums that have contained dangerous waste is not a practice that can be recommended; generally drums should be disposed of separately after crushing or sold as scrap metal after having first been thoroughly cleaned and crushed.

Disposal of Tires

The landfilling of tires in large quantities can lead to considerable difficulties especially from fires, instability and settlement, and deteriorating leachate quality. Consequently, tires should not be landfilled unless they have previously been finely shredded or are deposited in extremely small numbers. It is recommended that the quantities of shredded tires landfilled should be strictly regulated and should not exceed 5% by volume of the total input of solid waste. Intact tires may be used in limited quantities for the construction of injection wells.

Within a landfill conditions are such that a wide range of organic compounds can be leached from tires. Leachate migration in such circumstances can cause severe pollution problems. Unfortunately, there have also been a number of fires involving tires at landfill sites. These can be difficult to contain and extinguish. Fires involving large quantities of tires can result in the pyrolytic breakdown of rubber, with the production of complex mixtures of highly polluting products.

At landfills where significant quantities of tire fragments (up to 5%) are deposited they should be placed in specially constructed cells or trenches not more than 2 m deep. On completion, the cells should be covered with at least 1 m of incombustible material. Special fire fighting procedures should be laid down and incorporated into a contingency plan agreed upon with the local fire brigade.

Equipment

Mobile pumps which are capable of handling a range of liquids and sludges will be required. Pumps will also be required for spraying and irrigating liquid wastes. For the disposal of solid difficult wastes special equipment is not normally required except for handling drums. It is important to ensure that any equipment that becomes contaminated does not lead to waste being spread throughout the site. Equipment, therefore, should be cleaned regularly. An efficient wheel cleaning facility for vehicles is essential.

The provision of washing and first aid facilities, and the provision of protective clothing has been discussed. Additionally, mess and changeroom facilities must include a shower and preferably an emergency water drench in addition to any drench showers provided in the working area.

Monitoring of Wastes

At sites accepting difficult wastes it is particularly important to analyze leachate as well as incoming wastes.

Analysis of Leachate

Leachate at codisposal sites should be monitored regularly to ensure that attenuation mechanisms are not being overloaded and that attenuation is taking place as predicted. For sites where leachate treatment is undertaken or where leachate is being discharged to a sewer, monitoring is required to ensure that effluent quality is maintained within the design parameters of the plant and meets the consent conditions for its discharge.

Analysis of Incoming Waste

Inspection and analysis of incoming waste is necessary to verify the waste producer's description of the waste, to ensure compliance with disposal license requirements as well as the health and safety of personnel handling the waste, and to confirm that the waste is suitable for the method of disposal chosen. Detailed analysis on-site is not normally necessary. Nevertheless, regular monitoring and sampling of wastes received should be carried out and if necessary samples sent for more detailed analysis to a central laboratory.

The analytical facilities required at a landfill site accepting difficult wastes will depend on the nature and quantity of incoming wastes. Small sites which can accept only a limited range of difficult waste may need to provide only basic analytical equipment and rely on external consulting analysts for any detailed information required. Common tests include:

(a) appearance;

(b) odour;

(c) pH;

(d) flammability; and

(e) specific gravity.

Equipment for a laboratory serving a landfill site accepting difficult wastes is suggested in Table 7.1-19.

Manning Levels

Manning levels and the required qualifications of staff will be determined by the quantity and type of waste handled. Monodisposal lagoons and some multidisposal in-house landfills may not need to be attended, except when disposal is taking place from vehicles rather than pipelines, and for regular inspections and security checks. However, codisposal landfills will inevitably require additional manpower in comparison with sites accepting only household or similar wastes. No precise advice can be given, but the following may be regarded as key personnel in the satisfactory operation of a landfill taking difficult wastes.

Site Manager

A landfill site accepting significant quantities of difficult wastes must have a permanent site manager responsible for ensuring that all waste

TABLE 7.1-19

Site Laboratory Equipment

Equipment	Parameter
1. Sampling devices for liquid/sludge wastes	
2. pH papers, or pH meter	pH
3. Titration equipment	acid/alkali strength
4. Absorbent paper and matches	flammability
5. Glassware	appearance
6. Dropper bottles of selected reagents	reactivity
7. Bellows and a selection of gas detection tubes	volatility and gas type
8. Filtration equipment	
9. Hydrometer	specific gravity

disposal operations are carried out in accordance with the operational plan and that all regulations are complied with. He will also have to make decisions on whether to accept wastes, suitable loading rates, appropriate handling procedures for differing types of waste and the action to be taken if unplanned events occur. Site managers, therefore, should have considerable experience in all aspects of landfill operations.

Site Foreman or Chargehand

He will be responsible for the day-to-day running of the site. His primary duty must be to ensure that the site is run in accordance with the procedures laid down in the Operations Manual.

Face-marshall

At sites where the volume of incoming traffic is high and where a wide range of waste are delivered, a face marshall will normally be required. He will be responsible for directing vehicles to the appropriate operational area and ensuring that their loads are discharged in a satisfactory manner.

Gate-marshall or Weighbridge Operator

A record of all incoming loads is essential. Where wastes are consigned under the "Special Waste" regulations the gate marshall will be responsible for checking that the consignment note which accompanies such wastes agrees with the load and is expected.

Machine Operators

The number of operatives required will depend on the quantity of equipment used on the site at any one time.

Training

All personnel should be trained to a standard which satisfies Health and Safety requirements particularly for the safe operation of equipment. They should be aware of the working plan for the site and the standards of operation to be carried out. The nature and types of material being handled, and the difficulties involved in ensuring continuous supervision of all employees makes it essential that the staff receive a particularly high standard of training.

Health and Safety

A major responsibility when running a landfill site which accepts difficult wastes is the protection of site operatives and delivery personnel from risks to their health and safety. An additional consideration is the safety of visitors and the general public who may live near the site.

The effective and continuous protection of landfill site operatives from risks may be difficult to achieve. Such difficulties, however, cannot be used as an excuse to ignore the issue. The standards of safety, training and supervision set should be commensurate with the risks involved and should be monitored regularly to ensure that they are effective.

Many hazards which exist at landfill sites are common to other areas of employment such as the extensive use of mobile plant vehicles. It is essential that those responsible for safety at a landfill site recognize and deal with both the normal hazards and those which are unique to landfill operations. The latter include:

- the hazards which may result from consignors failing to declare the true and precise nature of a consignment of waste;
- the hazards which may result from the mixing of incompatible wastes -- this hazard may be aggravated by a lack of knowledge of the precise composition of the waste (e.g., gypsum in a mixture with household waste can generate hydrogen sulphide and under similar conditions arsine can be produced by arsenic compounds).

Employees at landfill sites work in all weather conditions and will need to be provided with suitable windproof wet weather clothing. Weather conditions can also make the handling of some wastes more difficult, for example the handling of dusty wastes will require special care in dry, windy conditions, while heavy rain may increase the difficulty of spreading sludge. The disposal of even small quantities of solvent-bearing wastes may be hazardous in hot, still weather.

Most hazards can be minimized by adopting working practices to lessen the hazard. In most instances the provision of appropriate handling equipment and suitable protective clothing will be necessary. The type of protective clothing used should be carefully selected to ensure not only that it is adequate for the task to be performed but that its design ensures an employee's safety and is comfortable and practicable when worn under the arduous conditions which can prevail on landfill sites.

Operational Procedures

All wastes should be checked on delivery to the site and if found to be acceptable should be directed to the appropriate disposal area. here a reception sump for liquid wastes is not available, a site operative should accompany the tanker to or receive it at the designated disposal area. Vehicles carrying difficult solid wastes or sludges should be directed to either the base of the working face, or to some other designated area of the working site. In both cases it will be necessary to ensure that the site roads are adequate for the traffic flow. Under bad site conditions (particularly in winter) it may be necessary to transfer loads to on-site vehicles for carriage to the disposal area.

Machine drivers and face marshalls should be made aware of the importance of inspecting loads as they are being loaded and unloaded to ensure that unacceptable wastes are segregated for disposal elsewhere.

'Totting' should not be permitted on landfill sites taking difficult wastes. As far as practicable working alone should not be allowed on such sites, and if workers are alone there should be some means of regularly checking the well-being of all personnel on the site.

Fires at landfill sites accepting difficult wastes can present additional special problems due to the noxious and possibly toxic fumes and particulates which may be generated. Operators manning equipment to bring fires under control should wear self-contained breathing apparatus. If dense fumes are being produced, such apparatus should be worn even if the cab of a site vehicle is air conditioned.

First Aid Arrangements

First aid facilities have been discussed. There is always a possibility that employees could be splashed by irritating or corrosive materials. Equipment such as eye wash bottles or other means of irrigating the eyes should be provided. Contamination of large areas of the body is also possible, and emergency showering or drenching facilities with a continuous supply of clean water should be provided near the working area as well as centrally. Good communications are essential on large landfill sites accepting difficult wastes. The usefulness of radio telephones for rapid site communications has been proven.

7.1.7 Leachate Collection and Treatment

(i) Leachate Collection

A leachate collection and removal system at a hazardous waste landfill must be installed and managed in a manner that will allow it to be used for all of its projected life. A leachate collection and removal system must be:

(a) compatible with the characteristics of the leachate to be collected,

(b) of sufficient strength to resist collapse by pressures exerted by equipment used at the site and by the accumulated waste and cover materials, and

(c) capable of withstanding the hydrostatic pressure.

Precipitation and run-off are important factors affecting the volume of leachate. External run-off should be diverted from the landfill site and intermediate and final covers should help divert the precipitation that falls on the site. Assuming that there is no surface water infiltration or groundwater moving through the landfill, the only leachate production will come from the landfilled wastes.

(ii) Leachate Treatment

Once leachate has been collected, numerous alternatives exist for treatment and disposal. If the facility is not yet in operation, the quantity and characteristics of the leachate to be treated must be estimated. The leachate characteristics depend on the nature of the landfilled wastes and on any fermentation that occurs in landfill. If the characteristics of the collected leachate indicate it is a hazardous waste, the leachate must be managed as such in accordance with the applicable permits and requirements. Typical compositions of leachate are given in Tables 7.1-20 and 7.1-11.

Biological treatment processes may be adversely affected by toxins occurring in both household and industrial waste but various factors may act to moderate such effects. The micro-organisms which form the basis of biological treatment may prove resistant to transitory shock discharges of toxic substances. Even if the microbial population is severely depleted, the situation can usually be retrieved since the regeneration of some micro-organisms is rapid. Finally, many biological populations have the capacity to acclimatize to the point of actually utilizing many toxins. Biological treatment of leachate can be accomplished by both aerobic and anaerobic processes.

In addition to their ability to oxidize certain toxic organic substances, aerobic processes can also acclimatize to the presence of heavy metal ions. In such cases the heavy metals are not oxidized by biological processes but are absorbed and adsorbed by the biological floc. Even though biological treatment processes can be effective in removing many toxic substances, pre-treatment may be necessary if they are present at abnormally high concentrations. The compounds which may adversely affect biological treatment include:

(a) Metals - If inhibition by metals is a problem simple pre-treatment with lime to precipitate the metals as hydroxides may be necessary.

(b) Carbon compounds - Very high concentrations of degradable organic compounds can be oxidized biologically by acclimatized systems. Low concentrations of chlorinated solvents will rapidly volatilize during aerobic treatment.

(c) Ammonia - High concentrations of ammonia (up to 2,500 mg/l) appear to be tolerated both by aerobic and anaerobic systems but the biological treatment rate will be reduced at high ammonia concentration.

(d) Chloride - Chloride can be tolerated at relatively high concentrations (up to 20,000 mg/l) by aerobic processes. Anaerobic processes appear slightly more sensitive with 10,000 mg/l inhibiting gas production.

TABLE 7.1-20

Typical Composition of Leachates from Recent and Aged Domestic Wastes at Various Stages of Decomposition (all results in mg/l except pH-value)

Determinant	Leachate from Recent Wastes	Leachate from Aged Wastes
pH-value	6.2	7.5
COD (Chemical Oxygen Demand)	23800	1160
BOD (Biochemical Oxygen Demand)	11900	260
TOC (Total Organic Carbon)	8000	465
Fatty acids (as C)	5688	5
Ammoniacal-N	790	370
Oxidised-N	3	1
o-Phosphate	0.73	1.4
Chloride	1315	2080
Sodium (Na)	960	1300
Magnesium (Mg)	252	185
Potassium (K)	780	590
Calcium (Ca)	1820	250
Manganese (Mn)	27	2.1
Iron (Fe)	540	23
Nickel (Ni)	0.6	0.1
Copper (Cu)	0.12	0.3
Zinc (Zn)	21.5	0.4
Lead (Pb)	8.4	0.14

(e) Sulphide - Anaerobic digestion processes usually tolerate up to 200 mg/l as soluble sulphide and little effect has been observed at concentrations up to 400 mg/l. In contrast aerobic processes can satisfactorily treat 1,000 mg/l without impairing performance.

Biological treatment process plants used include anaerobic lagoons or reactors and as aerobic systems, aerated lagoons and activated sludge systems. Physiochemical processes considered to be of most use on landfill leachates include air stripping, pH adjustment, chemical precipitation, oxidation and reduction. Advanced techniques such as carbon adsorption, ion exchange and solvent extraction may be necessary for certain leachates. These treatments are further described below.

Aerobic treatment systems generally use aerated lagoons or the activated sludge process. In both cases the leachate is aerated in the presence of an active biological floc. For aerated lagoons retention periods in excess of five days may be required to achieve greater than 95 per cent removal of the chemical oxygen demand (COD) from the leachate. To maintain biological activity within an aerobic system, a supply of essential nutrients is necessary. Leachate is normally deficient in phosphorus and occasionally nitrogen, and therefore addition of these nutrients may be required.

An activated sludge system differs from aerated lagoons by using sludge recirculation and a separate clarifier. The overall oxygen demand is thereby reduced and a more active biological community can be maintained within the reactor allowing the hydraulic retention time to be reduced and treatment rate increased.

Aerobic systems can be simple to design, construct and operate using lined basins fitted with a surface aerator or coarse bubble aeration plant. Biological sludges produced in aeration basins can be landfilled but are difficult to dewater.

Anaerobic Reactors

Anaerobic biological processes offer several different advantages over aerobic ones when used for the treatment of leachate. The advantages include the generation of methane gas and a much lower rate of production of sludges. Also, an anaerobic system does not need aeration equipment. The main disadvantage is that ammonia is not effectively treated.

In an anaerobic treatment system, complex organic molecules present in the leachate are fermented by bacteria to carboxylic acids, which are then partially converted to methane and carbon dioxide by methanogenic bacteria. Most of the development work carried out using anaerobic biological processes has been on a small scale using mixed digesters or anaerobic filters. Although filters have been found to be more efficient than mixed digesters both systems are generally considered to be unsuitable for use with leachates for on-site treatment, because of the high outlet BOD and ammonia requiring further treatment.

Physical and chemical processes have been investigated as an addition to, or replacement for, aerobic or anaerobic biological treatment of leachate. Chemical methods used to treat leachate include the addition of selected chemicals to precipitate, oxidize or reduce both inorganic or organic fractions. Precipitation and coagulation treatment using lime, ferric chloride or alum has been shown to have little effect on the removal of organic matter. Ion exchange processes have been found to be effective in the removing of color, suspended solids and heavy metals, but ion exchange is an expensive process.

Attempts to remove organic material from leachate using chemical oxidants have been made. Hydrogen peroxide, ozone, and less commonly, calcium hypochlorite and potassium permanganate have been evaluated. Excellent removal of color and iron has been achieved. To obtain any significant reductions in COD high dose rates were required. The amelioration of certain odors, such as that due to hydrogen sulphide, has been successfully carried out by treatment with hydrogen peroxide. Oxidizing sulphide in leachate to sulphate before discharge to the sewer has also been carried out successfully.

Leachate often contains high concentrations of nitrogen compounds, principally as ammoniacal nitrogen. Removal of ammoniacal nitrogen has been evaluated at both full-scale and pilot-plant scale. For air stripping of ammoniacal nitrogen it is converted to a gaseous form (ammonia) at pH-values 10.5-11.5 and removed by contact with air either in a stripping tower or pond system. Tower stripping is impracticable at very low temperatures (below freezing) because of icing problems. The use of shallow tanks and coarse bubble aerators has been successful and is recommended.

Other On-Site Treatment Options

Two further approaches to the treatment and disposal of leachates are described below. The first of them, land irrigation, can be carried out onsite only if sufficient land is available.

Spraying of leachate onto land (as opposed to landfill sites) can result in significant reduction in its volume due to evapotranspiration. Additionally, as the leachate percolates through vegetated soils opportunities are provided for the microbial degradation of organic components, the removal of inorganic ions by precipitation or ion exchange, and possibly for the rapid uptake of constituents such as ammonia by plants.

Evaporative losses may be maximized by using standard agricultural mist sprays. Intermittent spraying throughout each day will provide more effective evaporation than a single daily application. Transpiration by vegetation will account for a substantial proportion of total loss. The possibility of spreading harmful pathogens by spraying leachate needs to be considered but evidence to date suggests that this is not a problem provided the operation is properly managed.

Little information is available on the long term effects of continual spraying of leachate onto land. The spraying of leachates containing metals or persistent organic compounds is not to be recommended because of their accumulation in soil and plant material. In this respect reference should be made to publications on the application of sewage sludge to land and contaminated land trigger concentrations.

This form of leachate treatment requires large areas of land. If treatment gives rise to run-off away from the landfill a consent to discharge will be required from the water authority. Land irrigation as a treatment process may require planning permission and a disposal license separate from those granted for the landfill site operation.

Recirculation Through the Landfill

Recirculation of leachate through solid wastes, particularly where the biochemical activity of the waste has not been exhausted, potentially offers advantages both in reducing the volume of the liquid by evaporation, and reducing its strength. In addition it has been reported that increasing the moisture content of the landfill gives rise to a more rapid stabilization of wastes, and enhances gas production.

Although it is being extensively utilized, most of the available data on leachate recirculation has come from pilot-scale studies. The major benefit shown has been the production of a low strength leachate in a relatively short period of time. The leachates were similar in composition to those produced by wastes at least five years old. The solid wastes also appear to have degraded and stabilized more rapidly. Like land irrigation, recirculation will be most effective in the summer months when ambient temperatures and consequent evaporative losses are high and leachate production is at a minimum.

(iii) Leachate Monitoring

A theme running through this chapter is that the landfill is a complex reactor where physical, chemical and biological processes act to transform potentially polluting wastes into environmentally acceptable deposits. Because of the complexity of these processes and their potential environmental effects, monitoring is needed to confirm that the landfill is behaving in the ways predicted when the site was first planned and licensed and to provide information needed for management decisions. In this respect landfill is no different from most industrial processes that rely on complex chemical or biological reactions. There is, however, one difference; while closure of an industrial plant usually means that the need to monitor it ceases, this is rarely the case for a landfill.

Processes acting within or close to the landfill can continue for many years after the last deposit of waste. Therefore, monitoring should also continue beyond cessation of operations. Proper planning of the total landfill operation, including aftercare, should overcome this problem.

Leachate monitoring plays a central role in the management of all landfills. In the first place, data on leachate volume and composition are essential to the proper control of leachate so as to ensure the protection of water quality. While the formation of leachate is almost inevitable in a UK-like climate, by proper management its uncontrolled generation within a landfill is not. Modern landfilling therefore usually requires some form of leachate treatment. Such treatment can be successful only if the leachate conforms to predicted quality and treatment plan design parameters. Even where leachate can be simply pumped to sewers, its volume and quality needs to be controlled to satisfy water authority consent conditions. Monitoring of leachate volume is therefore linked to monitoring of its chemical composition. A knowledge of the chemical make-up of a leachate is necessary for the effective working of most leachate treatment schemes. In addition, monitoring data also provide the means of assessing the general behavior of a landfill. For example, knowledge of the chemical composition of a leachate is required to confirm that attenuation processes within the landfill are proceeding as expected. Changes in chemical composition of leachate can act as a warning system and help identify problems caused, for example, by overloading one part of the site with a particular waste. Remedial action can thus be taken. Bearing in mind the long life of a modern landfill, during which time significant changes could take place in landfill practices and in the character of the wastes deposited, monitoring is necessary to ensure that the measures taken for environmental protection remain effective.

Leachate monitoring should not be confined to the landfill itself, but should also take place outside the landfill boundary. Two types of landfill have been identified. One relies on attenuation mechanisms acting reasonably rapidly within and near the landfill to reduce the polluting components of leachate to acceptable levels. The other relies on containing leachate within the landfill for some appreciable time so that attenuation processes have time to act. In both cases, monitoring is imperative; in the first, to confirm that attenuation processes are proceeding in the manner predicted, and in the second, to confirm the integrity of the site, particularly if artificial liners are employed.

Any monitoring scheme must provide detailed information on the development of leachate within, and beyond, a landfill. The scheme will be site specific; it should be drawn up at the site investigation stage and implemented, as far as practicable, as the first stage in the site preparation. It cannot be stressed too strongly that it is vital to obtain pre-operational background readings against which the results of later monitoring may be judged. Without such background readings it is very difficult to determine what, if any, effect a landfill has on the environment.

Monitoring of a landfill and its environs is not solely the responsibility of the landfill operator though he must bear the major share. The disposal authority will also wish to undertake some monitoring to ensure that the conditions specified in the disposal license are being adhered to and remain appropriate. Similarly, in exercising its water pollution prevention duties, the water authority will wish to undertake some monitoring. With three parties involved, there is a need to ensure that duplication of effort does not take place and that an appropriate monitoring scheme is undertaken. All parties should therefore get together early in the site preparation stage to agree on the monitoring scheme and particularly who is to be responsible for what. Analyses undertaken by the three parties should be on a consistent basis to facilitate the comparison and pooling of results.

7.1.8 Treatment of Landfill Gas

Introduction

During the decomposition of the organic materials by biological activity in the waste, various gases and vapors known collectively as landfill gas are involved. The gas poses hazards, in particular through the risk of fires or explosions, and appropriate precautions must be taken in the design and operation of landfill sites to minimize these risks. In some circumstances gas may be produced in such quantities that its extraction as a fuel becomes worthwhile. This section provides a brief account of the generation and composition of landfill gas and the environmental problems it creates, and of various aspects of site design that relate to the control of gas migration through the site. The considerations that apply to the use of landfill gas as a fuel are also outlined. However, the control and use of landfill gas raises many issues that require special consideration including: legal aspects, the health and safety of personnel, the selection and design of plant and equipment, and monitoring requirements. A detailed discussion of these issues is presented in 'Landfilled Wastes' (U.K. D.O.E. 1986).

Several factors influence the rate at which landfill gas is generated. The moisture content of the waste is important as increased moisture enhances decomposition and hence gas generation. The decomposition process is complex and generally anaerobic. Optimum conditions for anaerobic digestion normally occur in unsaturated wastes with greater than 40 per cent moisture content. Methane forming bacteria can survive only within a narrow pH range, the optimum range for methane production being from pH 6.4 to 7.4. The optimum temperature for anaerobic decomposition lies between 29°C and 37°C. After the waste is deposited the oxygen content of trapped air is quickly utilized. Any further ingress of air, which would inhibit methane generation, is likely to be minimal unless excessive extraction of landfill gas results in air being drawn in. Recirculation of leachate is also unlikely to introduce sufficient oxygen to inhibit gas production. Finally the co-disposal of wastes with household wastes may affect the rate of gas production. Some industrial wastes can inhibit gas production by reducing the efficiency of the microbial processes, while, on the other hand, codisposal of wastes with a high putrescible content may enhance gas production.

Composition of Landfill Gas

Biological and biochemical decomposition of wastes takes place over a number of years, and during this time considerable changes occur in the nature and quantity of the gas evolved. These changes are summarized in Figure 7.1-11. Initially, air entrapped in waste when deposited is rapidly replaced by carbon dioxide and hydrogen. The production of hydrogen, which may reach concentrations of approximately 20 per cent by volume of total gas, normally occurs during the first few months. Typically, some 6 to 12 months deposition substantial amounts of methane will start to be produced. The concentration of methane will gradually increase until it reaches about 65 per cent of the landfill gas, with a corresponding decrease in the carbon dioxide content. When the methane has reached its maximum concentration, the methane-carbon dioxide mixture will continue to evolve over a period of some years. It is difficult to give a timescale for the evolution of these gases or for the period during (over) which they will be produced. However, at large, deep landfills the production of methane at maximum concentration for periods in excess of 10 years can be expected. At more shallow sites the period will be less and possibly much less if aerobic conditions persist for an extended period of time.

A large number of minor constituents at low concentrations have been identified in domestic waste landfill gas. Some of these compounds are often responsible for odors associated with the gas, but are unlikely to represent a health hazard. Hydrogen sulphide is unlikely to make any significant contribution to landfill gas odor unless high concentrations of sulphate-bearing wastes are present. In the presence of domestic refuse, sulphate sludges or plasterboard have been shown to produce large quantities of hydrogen sulphide, which is both odorous and toxic. Odors from landfill gas differ from those from leachate since the smell of the latter is predominantly due to carboxylic acids which are only present at low concentration in the gas.

Environmental Problems Arising From Landfill Gas

Problems due to landfill gas fall into the following categories:

(a) explosions or fires due to gas collecting in confined spaces, such as buildings, culverts, manholes, or ducts on or near landfill sites;

(b) asphyxiation of people entering culverts, trenches or manholes on landfill sites;

(c) when released through fissures at the surface, landfill gas may be ignited with a risk of setting fire to the waste;

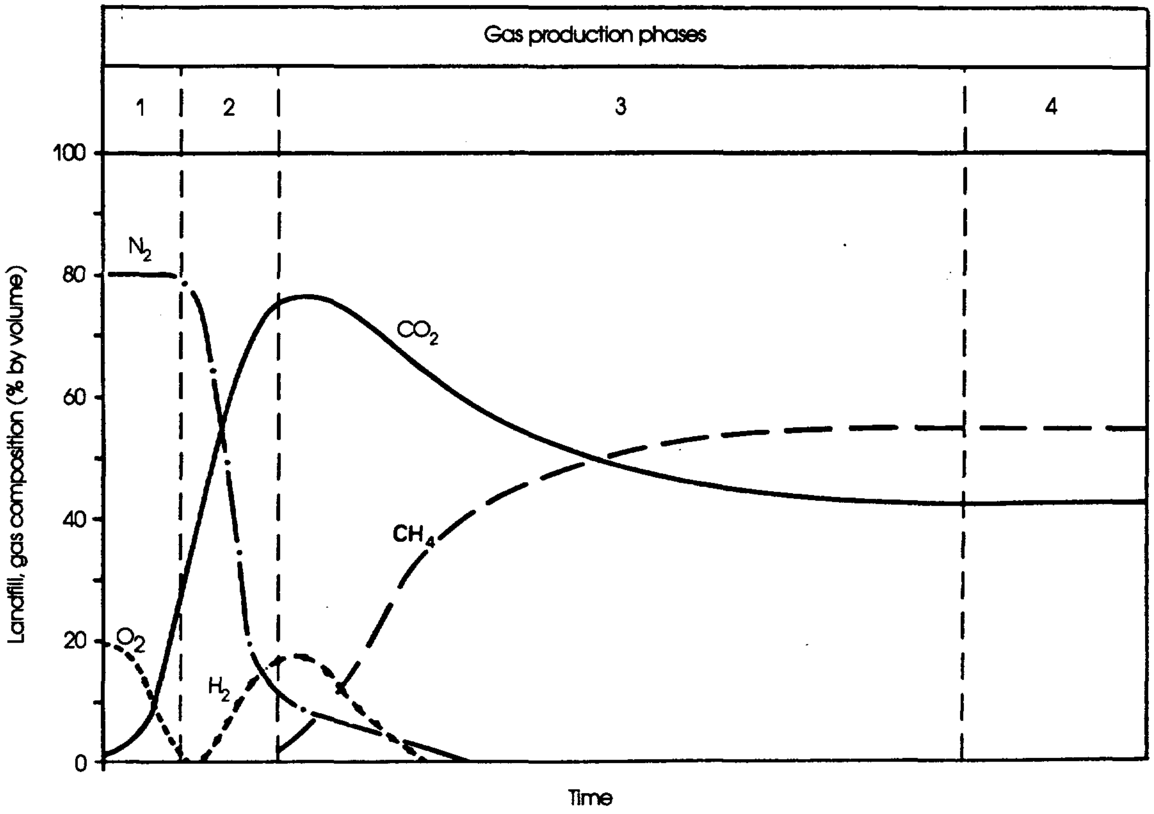
(d) detrimental effects on crops or vegetation on or adjacent to landfill sites;

(e) risks to human health from gas emissions; and

(f) nuisance problems, especially odor.

FIGURE 7.1-11

Typical Production Pattern for Landfill Gas



It is apparent from recent surveys that the number of incidents involving landfill gas is increasing. However, to date, serious incidents have been very few.

Landfill gas has also been responsible for damage to vegetation particularly on restored landfills. Methane displaces the normal soil atmosphere and prevents diffusion of oxygen from the air into the soil resulting in the soil being rendered anoxic. Phytotoxic compounds may also inhibit plant growth.

Landfill gas is diluted above a site typically by between a thousand and over a million fold with the air. Although several organic compounds may be present within the landfill gas at significant concentrations these are also diluted to several orders of magnitude below any toxicity threshold but nevertheless may still be odorous as they are released into the atmosphere. Measurements of metals as volatile derivative or in particulate matter have shown that concentrations are very low at both domestic and industrial waste sites. It should be noted, however, that at sites taking difficult wastes care must be taken to ensure that toxic gases are not generated by the mixing of incompatible wastes.

Landfill odors can cause considerable nuisance, as the dilution required to eliminate the smell may not be achieved under some weather conditions. Potential sources of odor include wet refuse which has been allowed to decompose partially before deposition, or large gas vents releasing gas directly to atmosphere. Minimum disturbance and finely graded cover material will do much to alleviate the smell nuisance.

Lateral Diffusion of Landfill Gas

On sites situated in strata of medium or high permeability such as sands, gravel, chalk or fissured rock, gas can migrate laterally for considerable distances beyond the landfill site boundary. While in most circumstances the gas will eventually vent to the atmosphere, at some landfills it may collect in buildings situated around the site and create a potential hazard. Gas migration should therefore be considered when planning a proposed landfill operation or any development adjacent to a landfill site.

Three types of systems may be used, either singly or in combination, to prevent lateral migration of landfill gas:

(a) impermeable barriers;

(b) passive venting; and

(c) gas pumping.

Materials which have been used to prevent gas migration include puddled clay, bentonite, plastic sheeting and PFA cement grouts. For sites up to 5 meters in depth, a trench excavated to about the depth of the base of the refuse around the perimeter of the site and lined with a barrier material should be sufficient to prevent gas migration. The width of the trench does not appear to be critical. For deeper landfills impermeable barriers should be installed as they are to prevent leachate migration.

Alternatively lateral migration of gas is effectively prevented by surrounding the site with a zone of high permeability material through which the gas can readily vent to the atmosphere. For sites up to 5 meters deep a perimeter trench filled with coarse aggregate or builders' rubble has proven satisfactory. Care must be taken, however, to prevent sealing of the top surface with mud, snow, ice, and the like. Such blockages can be overcome by installing a horizontal perforated pipe in the aggregate filled trench, the pipe being connected to vertical vent pipes at intervals of some 20 to 30m. With this system the upper surface of the trench may be sealed with clay or similar material. At deeper landfills a passive venting system could be created by depositing coarse aggregate between the side walls and the waste. However, gas release rates may then be high and could cause a nuisance. Alternatively, boreholes with slotted casing extending to the base of the refuse may be drilled either around the perimeter of the site or within the waste itself. However, unless a sufficient number of such boreholes are installed it is likely that pumping will be required to ensure an adequate rate of ventilation.

The quality and quantity of intermediate and top cover used in operations and restoration also influences the extent of lateral gas migration. For example the specification of a low permeability cap as part of the restoration may well encourage lateral migration by effectively preventing gas escape.

Landfill Gas Monitoring

As indicated above, landfill gas is one of the major routes through which landfilling may impact the environment and the local community. It is also in some cases a potential source of energy supply. Landfill gas therefore needs to be monitored to ensure that its production does not place a landfill site or its surroundings at risk as a result of gas migration or dangerous concentrations being allowed to accumulate. There is a temptation to combine landfill gas and leachate monitoring at the same borehole. Care must be taken to ensure that any equipment used to monitor leachate is compatible with a landfill gas environment. The practice of lowering pumps into monitoring boreholes to pump out leachate is not to be encouraged at sites where gas is evolving.

Collection and Utilization of Landfill Gas

At some landfills it may be worthwhile extracting gas for use as a fuel but to be successful a number of requirements have to be met. These are:

(a) A suitable use for the gas must be identified.

(b) The landfill must have a minimum depth of at least 10m of biodegradable material.

(c) There must be a large quantity of waste already deposited. Experience suggests that at least 0.5 million tons is required.

(d) The waste should not be too old. Wastes deposited for between 5 and 10 years seem generally to produce the highest gas yields.

(e) The water level should be at least 5m below the landfill surface. Saturated conditions are not conducive to landfill gas collection.

By these criteria, venting gas to the atmosphere or flaring will remain the only control option for most landfills. However, at some landfills it may be worthwhile utilizing the gas as a process fuel, for electricity generation or, conceivably, as a chemical feedstock.

A preliminary assessment of quality and potential quantities of landfill gas is essential and can best be achieved by drilling wells at appropriate locations and determining the quantity of gas that can be extracted at various suction pressures. It is important to determine the sphere of influence for each extraction well and this is usually measured by placing peizometers at various distances from the wells. The sphere of influence will vary but usually has been found to be between some 30-50 m radius. The number of wells required can vary, around 2 wells per hectare being an average figure. The rate at which gas can be withdrawn is also variable, a typical figure being about $3m^3$ per minute. A typical calorific value for landfill gas is 20 MJ per m^3, corresponding to about 55 per cent methane.

Usually a landfill gas extraction well has a diameter of between 0.3 to 0.6 meters and is drilled either to the base of the refuse or until the water table is reached. The well casing is usually some 0.1m in diameter and perforated for the bottom half, the annulus around the lower part being backfilled with permeable material, followed by clay or other sealing material.

Gas production wells are joined together by plastic pipe, usually of at least 100 mm diameter, which in turn is connected to a suction pump. The pipes, which are often laid on the surface of a landfill, are normally sloped to facilitate the collection and removal of condensate.

7.1.9 Closure and Restoration of Site

(i) Covers

Ultimately hazardous waste landfills reach the end of their useful life. When the landfill is closed, it should be sealed with an impermeable layer of suitable material and graded to prevent ponding of surface water and to minimize erosion. Natural vegetation (grasses and weeds) are permitted to grow on the cover to reduce wind and surface erosion. Deep rooted vegetation should be avoided since it can penetrate any impermeable barrier.

The prime function of a final cover is to minimize infiltration of precipitation. Other functions include preventing: (a) contamination of surface run-off, (b) wind dispersion of hazardous waste and (c) direct contact of the hazardous waste with humans and animals. For long term performance with minimum maintenance, the final cover is designed to promote surface drainage, minimize erosion, avoid build-up of gas pressures in the landfill and accommodate settling and subsidence.

Generally a layered cover will be needed to meet the multiple requirements of a proper cover (i.e., leveling of irregular waste surfaces, gas venting if needed, an impermeable layer, and soil for vegetative growth). Figure 7.1-12 illustrates the components of a layered cover. The buffer soil can be sand or a good grade of soil which is free of clods and rocks that may interfere with the leveling action. Both the buffer and the topsoil protect the impermeable layer of clay or membrane from punctures, tears and cracks. The buffer soil smooths irregularities and provides protection from surface traffic. Typical thicknesses of these cover materials are: topsoil -- at least two feet; sand buffer soil -- one foot; and clay -- two feet. The topsoil should be capable of supporting a desirable vegetative cover and should not crack under drying or freezing conditions. The impermeable layer can be natural off-site clays or a synthetic membrane, but should be at least as impermeable as the bottom liner to prevent the build-up of liquids in the landfill. Table 7.1-21 summarizes the general characteristics and advantages of soils used as landfill cover material.

When gaseous emissions require control, a porous layer (Figure 7.1-12) and vents can be designed into the cover. The gas transmission layer can be a perforated pipe or a gravel layer above the buffer layer or can be a permeable buffer layer.

(ii) Post-closure Plan

At the end of its useful life, a hazardous waste landfill should be closed and associated treatment and storage facilities decontaminated in a safe and environmentally acceptable manner. If final closure takes place properly further maintenance and control will be limited to the extent necessary to protect human health and the environment and to prevent the post-closure escape of hazardous waste or contaminated emissions. Such emissions include leachate, contaminated rainfall, and waste decomposition products. Components of a hazardous waste landfill closure plan are presented in Table 7.1-22.

After closure the landfill owner should:

(a) maintain the integrity and effectiveness of the final cover, repairing the cap as necessary to correct the effects of settling, subsidence, erosion, or other events;

(b) maintain and monitor the leak detection system;

FIGURE 7.1-12

Typical Layered Cover System

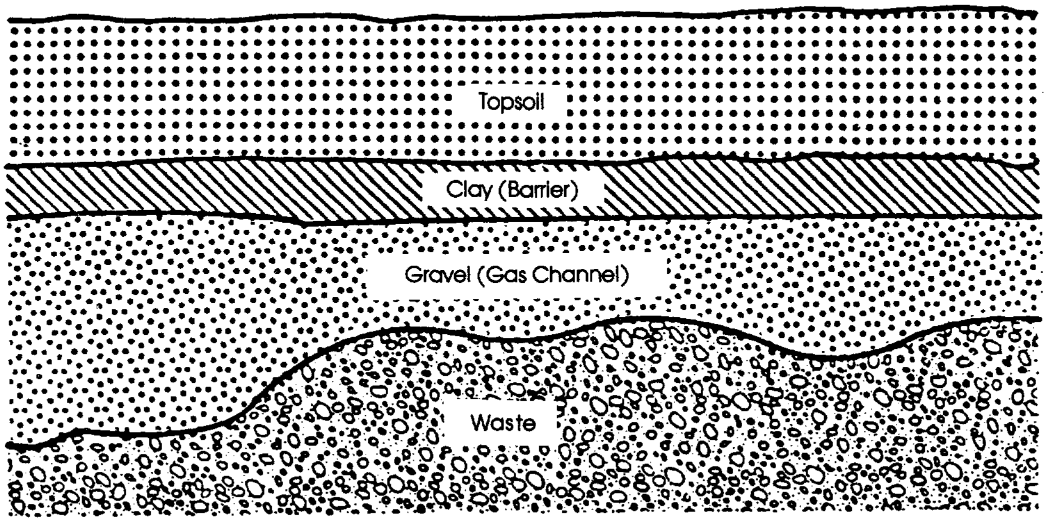


TABLE 7.1-21

General Characteristics of Landfill Cover Soils

Cover	Clean Sand	Clayey/Silty Gravel	Clayey/Silty Sand	Silt	Clay
FUNCTION					
Wind erosion dust, and litter control	Excellent, few fines	Fair to poor control until water and wind erosion remove fines from top inches		Poor	Poor
Odor control	Poor-Fair	Fair-Good	Good-Excellent	Excellent	Excellent
Trafficability and handling	Good	Poor			
*Water Content *Strength	Bearing strength excellent in all moisture ranges; however dry coarse grains can roll and not offer friction resistance (traction).	Bearing strength drops with addition of moisture, in relation to the fines content. Good friction resistance when dry.			
Infiltration, percolation and gas control	Freely draining. Poor control of infiltration and percolation.	Fair to poorly draining. Good control of infiltration and percolation if compacted.		Very poor draining. Excellent control of infiltration and percolation.	
*permeability (k) must be $<10^{-5}$ cm/s	K = 1.0 to 10^{-3}	K $=10^{-4}$	K $=10^{-4}$ to 10^{-6}	K $=10^{-4}$ to 10^{-6}	K $=10^{-7}$ to 10^{-9}
Water erosion control	Poor control, Easily eroded	Excellent control, Poorly draining and massive	Fair-Poor control; without vegetation is poorly draining and easily eroded	Poor control, without vegetation, easily eroded	
		With compaction and cement stabilization these soil groups can become excellent for erosion control.			

TABLE 7.1-21 (continued)

Cover	Clean Sand	Clayey/Silty Gravel	Clayey/Silty Sand	Silt	Clay
Slope Stability	Generally stable at angle of repose up to 1:1. If unaffected by seepage, steep slopes (greater than 1:1) are vulnerable to shallow surface sloughing and to seepage effects.			Steep cuts (1:1 and up) are temporarily stable in clays and silts which exhibit good bearing strength and are at water contents close to the plastic limit.	
Animal and control	Good	Fair-Good	Poor	Poor	Poor
Fire control	Fair	Good-Excellent	Good-Excellent	Excellent	Excellent
Crack resistance	Excellent	Fair-Excellent	Good	Poor	Poor
Vegetation support	Unacceptable	Good Salt content is very important to most type of vegetation.	Excellent	Good	Fair- to Excellent
CLIMATE EFFECTS					
*A supply of unfrozen cover *Salt additives will lower the freezing temperature of soils.	Freezes early and thaws early. Freezes to greater depths than fine-grained soils. Can be placed frozen with minimal settlement upon thawing. Well-drained borrow areas prevent freezing effects from becoming severe.	Freezes late and thaws late. Excavation and placement may be difficult when frozen. When thaw takes place and ice melts, wet voids will settle quickly.			

TABLE 7.1-22

Components of a Hazardous Waste Landfill Closure Plan

Upon closure the following should occur:

- Decontaminate and decommission any hazardous waste treatment and storage facilities.
- Provide a final appropriate cover for the landfill.
- Control pollutant migration from the landfill via surface water, groundwater, and air.
- Maintain the existing groundwater monitoring network for the required period of post-closure maintenance.
- Continue to divert run-on from the landfill.
- Prevent soil and wind erosion.
- Control surface waste infiltration and ponding at the closed site.
- Maintain any gas leachate collection, removal and treatment system.
- Maintain the integrity of the final cover and any liners.
- Note, in a document examined in a title search, such as a deed, that the land has been used to manage hazardous waste and that its use is restricted.
- Restrict access to the landfill as appropriate for the post-closure area.

(c) continue to operate the leachate collection and removal system until leachate is no longer detected;

(d) maintain and check the groundwater monitoring system;

(e) prevent run-on and run-off from eroding and otherwise damaging the final cover; and

(f) protect and maintain any surveyed benchmarks.

Each permitting agency that regulates hazardous waste landfills normally has guidelines or regulations that identify the steps to be followed in preparing a closure plan, closing a landfill, and providing for post-closure care.

The post-closure plan should provide care for an extended period of time after closure, such as for at least 30 years. This is an arbitrary time period which can be lengthened or reduced depending upon technical information related to post-closure pollutant migration. Properly closed hazardous waste landfills can be used for general use purposes such as parking areas and open spaces. For such uses, a satisfactory cover and post-closure monitoring of surface emissions is critical.

7.1.10 Issues Regarding Codisposal of Municipal and Hazardous Waste

(i) Introduction and Purpose

A major portion of this chapter is devoted to the practice of codisposal of municipal solid waste and hazardous waste as a sound and practical method for disposal of hazardous wastes. The purpose of this section is to alert the user of this manual to some of the long-term adverse consequences of this practice that may occur if adequate resources are not committed to selecting, designing, operating and monitoring a codisposal landfill. This aim is addressed by describing some of the experience in the State of California, U.S.A., where codisposal was widely practiced over the past 15-20 years.

(ii) Past Practices in California

Twenty years ago, a survey of the Status of Solid Waste in California found major deficiencies in the way in which solid wastes were managed (California State Department of Public Health 1986). These deficiencies ranged from inadequate planning, financing and standards to poorly developed technology and fragmented authority. Compounding the problem, it was found that very little was known about the volumes, types of wastes, and ultimate disposition of hazardous wastes until this survey was undertaken. It was found that hazardous wastes were being generated by a broad spectrum of industry represented in the State. The estimated volume in 1968 was about 12 million metric tons/year or about 1.8 kilograms/person/day. These wastes were being disposed of in private dumps on the premises where they were generated, placed in open dumps off-site, and dumped indiscriminately. In general, there was very little control of where and how these hazardous materials were disposed.

By 1974, the State published Guidelines for Hazardous Waste Land Disposal Facilities (California Department of Public Health 1974) which recommended codisposal as one of the methods for disposal of both solid and liquid hazardous wastes. The report cited the "large sponge-like absorption capacity of a sanitary landfill containing community-type refuse," and the isolation of the hazardous waste under tons of refuse as the primary advantages of codisposal. Another advantage cited was the low cost of codisposal compared to other methods including ponding and land spreading which required dedicated land and equipment for hazardous waste disposal. These guidelines also included siting criteria, operations and maintenance procedures, recommendations for subsequent use of disposal areas and other precautions similar to those mentioned in this guidance document. The 1974 guidelines also mentioned the need for property use controls to limit or eliminate the potential of exposure to landfill gases, design features in buildings that provide for venting of gases, and the control of liquid input so as not to exceed the absorption capacity of the fill. Excess liquids would cause leachate generation and possible groundwater contamination.

The methods of codisposal included incorporation of liquid and solid hazardous wastes at the working face of the landfill, disposal into trenches cut into the fill, and injection of liquids into large diameter wells drilled through the fill. At the working face, a berm was created using refuse and then hazardous wastes were discharged into the pit formed by the berm. More refuse was then dumped into the pit to absorb the hazardous wastes. The working face was then covered at the end of the day with 30 cm. of clean soil. Using this method, the hazardous wastes were essentially mixed in with municipal refuse as part of the normal operations at a sanitary landfill. This was the most widely practiced mode of controlled codisposal.

Injection of liquids into wells was practiced at a few large landfills which accepted liquid waste in bulk and where enough fill was already in place to absorb these liquids (Tinnan 1978). Typically, these wells were 1 to 2 meters in diameter and 20 to 50 meters deep. A concrete and metal cap with a hose hook-up was placed at the opening of the well to facilitate the safe and efficient discharge of the liquids. The capacity of these wells were typically determined empirically by the ability of the well to accept more liquids.

These methods were employed at one time in the late 1970s by at least ten authorized landfills in the State. Six of these, which happened to be the larger sites, were located in the drier Southern portion of the State. Codisposal was practiced through the mid-1980s until regulatory constraints and incidents of adverse environmental and public health impacts put an end to this practice.

(iii) Public Health and Environmental Problems Encountered

Many of yesterday's codisposal facilities are now being investigated for leakage of leachate and landfill gases contaminated with hazardous substances which were disposed in the landfill (California Department of Health Services). In its 1986 version of the list of hazardous substance release sites which are candidates for investigation and possible remediation using public funds or "Superfund" money, the State identified twelve former landfills where codisposal occurred. The sites on this list are in general the highest priority sites in the State which require some kind of response to a potential public health or environmental problem due to uncontrolled release of hazardous substances into the environment.

Of these twelve sites, eight were authorized to accept large quantities of hazardous wastes for codisposal with municipal refuse. Four of these were over 200 hectares in size and accepted volumes of hazardous wastes which ranged from 200,000 metric tons to 700,000 metric tons. The types of wastes disposed included: acid wastes, alkaline wastes, pesticides, solvents, lead sludges, oily wastes, a variety of tank bottom sludges, paint sludges and various other industrial process wastes.

In addition to these sites which are being investigated as "Superfund" sites, many other refuse disposal sites are being investigated for leaks through other State and Federal statutes and programs. As a result of the findings of releases of hazardous substances from the above mentioned sites, the State legislature mandated a program that involves systematically checking for hazardous substance releases into the groundwater or via landfill gas in all known landfills in the State, whether or not they were authorized to accept hazardous wastes.

One incident at a facility in Southern California in 1984 brought into focus the potential problems which have been encountered as a result of codisposal of hazardous wastes and municipal refuse. This was one of the largest codisposal facilities in the State and had been in operation for over ten years. In 1984, the finding of combustible levels of landfill gas containing significant levels of vinyl chloride and other volatile organics in homes surrounding the site prompted the evacuation of 20 families from their homes for eight months. Investigations promptly lead to redoubled efforts at the landfill to construct gas extraction wells and other control measures to mitigate the hazardous situation. Groundwater investigations after this incident also showed off-site contamination with volatile organics. Another similar incident at another Southern California codisposal facility where residences were constructed immediately adjacent to the landfill further emphasized the need to critically evaluate the long-term consequence of codisposal.

In both instances, the problem was intensified because of the proximity of residences to the landfill. Both were very large and accepted large quantities of hazardous wastes. Had the landfill not expanded over time and had the encroachment of residential housing near the landfills not occurred, the public health problem would not have been as immediate.

Another serious result of the codisposal practices is the cost of investigations and remediation of the problem. Investigations involve at a minimum the installation of groundwater monitoring wells at various depths around the landfill where the cost of each well can range from 10,000 to 100,000 dollars U.S. The sampling and analysis of water from these wells may cost from 50-1,000 dollars U.S. per sample depending on the type of constituents being sought. Monitoring of volatile organics in ambient air and in the unsaturated zone of soil at various locations around the site is also usually needed at a cost of 200 to 1,000 dollars U.S. per sample. Other costs related to performing health surveys and conducting a public participation and information program are also typically incurred. These costs altogether can easily run into the millions of dollars for a typical landfill site.

Remediation programs for this type of site typically include both gas migration control and leachate migration control through a series of wells designed to capture hazardous substances which may be migrating off-site via these routes. These extraction programs are usually coupled with a treatment scheme that involves either extraction of the hazardous constituents from the gas or water or destruction of the constituents by incineration or some other appropriate techniques. The costs of these programs can easily run over 10 million dollars U.S., especially when it is likely that these systems may have to be operated for periods in excess of 15 to 20 years.

(iv) Summary and Recommendations

Although this chapter seems to promote the concept of codisposal of municipal wastes with liquid hazardous wastes because of its relative ease, safety and low cost, the users of this manual must be aware of potential long-term disadvantages of this practice. As recently as 10 years ago in the State of California, this was accepted as the method of choice for disposal of a wide variety of liquid hazardous wastes including acids, bases, solvents, various sludges and others. It was thought that the relatively dry climate of California and high absorption capacity of municipal wastes would prevent leaching of these toxic compounds to the groundwater. It was also thought that these sites were optimally located geologically and geographically as to minimize the potential for impact of contaminated leachate and fugitive gas emissions on nearby residents.

Unfortunately, many of these facilities that were once authorized to accept liquid hazardous wastes for codisposal with municipal refuse are now considered "problem" sites or sites where releases of hazardous substances have occurred to adversely affect public health. Groundwater has been contaminated with toxic volatile organics at these sites and in several locations residents were temporarily evacuated due to combustible levels of landfill gas and significant accumulation of volatile organics in homes. The encroachment of residential development has compounded the public health risks posed by releases of toxic substances from these sites.

Experience in the State of California has shown that unless enough resources are committed to properly select, design, operate and monitor a codisposal landfill, the consequences can be very serious and very costly. If these resources cannot be applied prior to and during operation of such a facility, the cost of defining the extent of contamination caused by releases from it and retrofitting it for the control of leachate and gas/vapor migration can be far greater. Furthermore, it is critical that land use restrictions be imposed not only on the facility but also on a reasonable area surrounding it so as to provide a buffer zone between the site and any potential human receptors. It is clear that the incidents mentioned earlier would not have been so severe if these buffer zones were established and enforced.

Codisposal upon initial consideration appears to be a reasonable and cost effective means of disposing of hazardous wastes, however, it may become just the opposite in the long-term. Although each country must make its own decisions on how much of its scarce resources to commit to hazardous waste management, it should consider the potential long-term costs of codisposal and closely evaluate the appropriateness of more permanent solutions like incineration or other treatment technologies.

7.1.11 References 7.1 - 7.1.10

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Telex: WUI 64145 WORLDBANK
RCA 248423 WORLDBK
Cable Address: INTBAFRAD
WASHINGTONDC

European Office
66, avenue d'Iéna
75116 Paris, France
Telephone: (1) 40.69.30.00
Facsimile: (1) 47.20.19.66
Telex: 842-620628

Tokyo Office
Kokusai Building
1-1, Marunouchi 3-chome
Chiyoda-ku, Tokyo 100, Japan
Telephone: (3) 214-5001
Facsimile: (3) 214-3657
Telex: 781-26838

THE WORLD HEALTH ORGANIZATION

Headquarters
CH - 1211 Geneva 27, Switzerland
Cable Address: UNISANTE GENEVA
Telex: 27821
Telephone: (022) 91 21 11
Facsimile: (022) 91 07 46

Regional Office for Africa
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Headquarters and Regional Office for Africa
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Regional Office for Asia and the Pacific
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Industry & Environment Office (IEO)
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