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ABSTRACT

EVALUATION OF ENVIRONMENTAL CHARACTERISTICS OF WTP RESIDUALS FOR DISPOSAL IN MONOFILLS

by
Ping Tian

Disposal of water treatment plant (WTP) have raised great concerns due to the stringent water quality standards and environmental regulations, and limited availability of land for ultimate disposal. The ultimate disposal of water treatment plant residuals by using monofill may be one of the most economical, and feasible options. Design and operation of such a monofill will be simple because of the uniformity of the characteristics of WTP residuals.

The objectives of this project were to determine the environmental characteristics of WTP residuals and to apply these characteristics for the design of monofills and development of the metal leaching model to predict the primary metals movement from WTP residuals in long term.

Samples of WTP residuals were collected from ten treatment facilities. Average solids content of raw residuals was 1.5 % and solids content of dewatered residuals varied from 15% to 82% in this study. It was noted that if the solids content of the residuals were below 15%, the residuals usually failed the paint filter tests. The pH of WTP residuals mainly depends on the coagulant added. The pH of alum and ferric residuals were generally neutral and the pH of lime residuals ranged from 9 to 12. The pH value and buffer capacity of lime residual were high. This is beneficial since it prevents metal from

leaching. A high cation exchange capacity (CEC) of WTP residuals (20 to 35 meq/100g) generally connotes potentially a high capacity to maintain contaminants in the residuals.

There is a direct relationship between what were inside the residual and what leached out. Metal contents, pH, and CEC in dewatered residuals are found to be related to types of water sources, impurities of water sources, and chemicals added during water purification and dewatering processes.

The toxicity characteristics leaching procedure (TCLP) analysis for WTP dewatered residuals from ten water treatment facilities were conducted. Analytical results show that the concentrations of the TCLP regulated compounds in all samples were below the regulation limits. This indicates that the WTP residual tested were non hazardous.

There was no significant change in solids content, volatile solids content, and primary metals contents among the fresh, six-month, and one year-old samples taken from minimonofill. The results of field study indicated that most metals and organics remained in WTP residuals and no significant biodegradation noticed.

A mathematical predictive leaching model for major metal ions in WTP residuals are developed using diffusion theory, metals solubility, and adsorption mechanisms. This leach model can be used to predict long term primary metals leaching and to quantify immobilization of these elements for WTP residuals. WTP monofill disposal is better than MSW landfill disposal based on the leachbility studies and leaching model prediction.

EVALUATION OF ENVIRONMENTAL CHARACTERISTICS OF WTP RESIDUALS FOR DISPOSAL IN MONOFILLS

by
Ping Tian

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APPROVAL PAGE

EVALUATION OF ENVIRONMENTAL CHARACTERISTICS OF WTP RESIDUALS FOR DISPOSAL IN MONOFILLS

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This thesis is dedicated to
my dear wife Ying Li and my family

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CONTENTS

Chapter Page

4 ENVIRONMENTAL CHARACTERIZATION OF WTP RESIDUALS

Chapter **Page**

LIST OF TABLES

Table **Page**

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

1.1 General

Water treatment plant (WTP) residual materials are the solids removed from coagulation, softening, sedimentation, and filter backwashing processes at water treatment facilities. The major types of WTP residuals are alum, ferric, and lime based on the types of coagulants added. Their composition depends on impurities such as clay minerals, organics, inorganics, etc. and the chemicals added WTP residuals are relatively homogeneous in composition as compared with wastewater sludges or other industrial solid wastes. The disposal of these residuals has grown into a formidable challenge for the water industry, especially in light of the current emphasis on a clean environment.

Each year several million tons of residuals are produced by water treatment facilities in the United States. Handling and disposal of these residuals have always been important considerations in water treatment. Due to the stringent regulations, the problem of how and where to dispose of these residuals is continuing to receive great attention.

In past years, the disposal methods of water treatment plant residuals has included direct discharge of these into sanitary sewers, waterways, land disposal, and ocean dumping with or without prior dewatering, and incineration. However, some of these alternatives are no longer feasible due to the current regulations that limit the direct discharge of wastes into water courses. These regulations include the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500), the Clean Water Act of 1977, the Solid Waste Disposal Act (PL 91-512), and the Resource Conservation and Recovery Act of 1976 (PL 94-580).

Disposal of WTP residuals to surface waters or sewers falls under the jurisdiction of the Federal Water Pollution Control Act Amendments (FWPCA) and the Clean Water Act (CWA). The FWPCA called for the establishment of national pollutant discharge elimination system (NPDES) permits. Under NPDES, a permit must be obtained for any industry discharge, including WTP residuals. The Marine Protection Research and Sanctuaries Act (MPRSA) were enacted at almost the same time as FWPCA. As a result of numerous amendments to these regulations, no ocean dumping of industrial wastes is permitted.

On the other hand, disposal of WTP residuals on land is governed by the Resource Conservation and Recovery Act (RCRA), which was amended in 1986 as the Hazardous and Solid Waste Amendments (HSWA). The amended RCRA constitutes the framework for federal regulation of waste treatment, storage, and disposal facilities in the U.S. Subtitle D of RCRA deals with municipal solid waste. If a WTP residual is disposed of in a landfill, then the activity must compile with the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980. If a water treatment utility disposed of its residuals at landfill that also accepted other wastes that contaminated the groundwater, the water utility could be liable for cleanup based on its use of the landfill, even if its residuals did not directly cause the problem.

At present, landfilling and land application of water treatment residuals are the logical alternatives because of their environmental and economical advantages over other methods. Yet finding an appropriate landfill for final disposal is another problem that a water utility may face. Monofill, a landfill that is exclusively set aside for accepting only WTP residuals, thus becomes the simple, economical option for ultimate disposal. Due to the relative homogeneity of the WTP residuals, some of the considerations associated with the design, construction, and operation of the Municipal Solid Waste (MSW) landfills can be eliminated, modified, or relaxed for the WTP residual monofills.

1.2 Scope and Objective of Research

The objective of this research work was to evaluate the environmental characteristics of WTP residuals for disposal in monofills. As a first step in monofilling WTP residuals, WTP residuals will be analyzed by use of the TCLP test as defined by RCRA requirements to prove this residuals being non-hazardous. having been shown to be non-hazardous, WTP residuals would be allowed to the landfilling under municipal solid waste (MSW) regulations. For landfilling WTP residuals in an MSW landfill, requirements will generally be that the residuals have be no free water as determined by the paint filter test. States may also require a minimum solids concentration, often in the 20 percent range. For creating a WTP monofill, the strict MSW requirements are not necessary if the landfilling of WTP residuals will not impact groundwater or surface water based on the study of physical, chemical, biological, and geotechnical properties of various types of water treatment plant residuals. In order to assess the potential for leaching contaminants from a WTP residuals and thereby impact environment, multiple batch leaching test and a one year field studies were conducted.

This investigation was divided into three major phases as follows:

1. *Laboratory study of physical and chemical properties of WTP residuals*
    - To set up protocol for environmental experimentation including analytical instrumentation, anaerobic reactors, and methods of analysis.
    - To analyze water treatment plant (WTP) residuals, including sample collection, preservation, preparation, handling, testing, and QA/QC.
    - To characterize the WTP residuals based on the results of these studies and EPA regulations.
2. *Field study of WTP residuals in monofills*
    - To set up minimonofill in New Castle Water Treatment Plant Monofill in New Castle, Pennsylvania
    - To evaluate the environmental characteristics of aged WTP dewatered residuals and leachates during one year field experiment.
3. *Metal leaching model development for WTP Residuals*

The major concern in water facilities is whether the primary metals, such as aluminum, iron, lime, will leach out, since the leaching of such metals will impact the subsurface environmental and affect the methods of disposal. It is essential to develop leaching model for predicting the long term metal leaching rate of WTP residuals.

Predictive Leaching model is developed using material balance and diffusion theory. Experimental leaching tests are performed by multiple batch extractions. Thermodynamics and kinetics of leaching and effects of leaching characteristics are discussed. Finally, a mathematical method suitable for computer implementation for solving time-dependent primary metals leached from three types of WTP residuals is present.

1.3 Organization of this Study

The second chapter of this study is devoted to introduction and review of prior literature regarding WTP residuals. The sources of the samples collected for this study, the treatment processes and dewatering methods employed in these WTP facilities are presented in Chapter 3. In this Chapter, the testing methods for the characterization of the residuals are discussed. Chapter 4 discusses the environmental characterization of the residuals based on laboratory tests. The field testing was conducted on a model monofill over the period of a year. The findings of these tests are reviewed in Chapter 5. Chapter 6 presents the development of primary metals leaching modeling for WTP residuals and prediction of long term metals leaching in the monofill or MSW landfill. All the results were based on the multiple batch extraction leaching test and short term testing results. The summary of the evaluation of environmental characteristics of WTP residuals was presented in Chapter 7.

CHAPTER 2

BACKGROUND

2.1 Disposal of Water Treatment Plant Residuals

Handling and disposal of WTP residuals have always been important considerations in the water industry. During the past ten years, increasingly stringent legislation has made the disposal of WTP residuals quite difficult.

Many studies have been conducted on methods of dewatering and disposal of WTP residuals. (Barlow 1973, Cornwell et al. 1992, Dempsey 1992, George et al. 1991, Harry and Michael 1973, Malcolm Pirnie 1978, Raghu and Hsieh 1987a) The most effective process involved gravity thickening of the residual followed by mechanical dewatering and truck hauling to a landfill site. Barlow (1973) investigated various methods of processing and disposal of residuals and pointed out that determination of the characteristics of the residuals was essential to develop criteria for disposal and handling of the residual. Until 1980, analyses for the testing of water treatment plant residuals was performed using general physical and chemical tests. Toxicity testing was not considered.

2.1.1 Water Treatment Plant Residuals

In water treatment plants, residuals are most commonly produced during the following treatment processes: presedimentation, sedimentation, softening, and filtration (filter backwash) (Cornwell and Westerhoff 1981). When surface waters are withdrawn from water courses containing a large quantity of suspended materials, presedimentation prior to coagulation may be practiced to reduce the accumulation of solids in subsequent units. The settled material generally consists of fine sands, silts, clays, and decomposed organic products.

Aluminum and iron salts are generally used to accomplish coagulation. The chemical reactions of both these salts are similar. After chemical reaction, alum and iron flocculate with water impurities and form insoluble aluminum and iron hydroxides. Approximately 0.44 pounds (0.10 kg) of chemical residuals are produced for each pound (0.23 kg) of alum added (Cornwell and Westerhoff 1981). Alum residual leaving the sedimentation basin generally has a suspended solids content of 1 percent. Out of this, 20-40 percent are volatile and the remainder is inorganic clays or silts. The BOD_5 of alum residual is usually around 100 mg/L. However, the chemical oxygen demand (COD) of the residual is considerably higher than BOD_5. The pH of alum residual is generally in the range of 5.5 to 7.5. Alum residuals from sedimentation basins may include large numbers of microorganisms, but generally do not exhibit an unpleasant odor. The residual flowrate is often in the range of 0.3 to 1% of the treatment plant flow (Cornwell and Westerhoff 1981).

The residuals from softening by precipitation with lime ($Ca(OH)_2$) and soda ash (Na_2CO_3) varies from a nearly pure chemical to a highly variable mixture. Theoretically, each mg/L of calcium hardness removed produces 2 mg/L of $CaCO_3$ residual, and each mg/L of magnesium hardness removed produces 2.6 mg/L of residual. A survey conducted by the state of Ohio found actual residual production at lime softening plants to be 1.785 times the theoretical values (Ohio Department of Health 1969). However, many of plants surveyed were softening surface supplies, which produce a highly variable material. In this case, softening was often carried out in conjunction with chemical coagulation and, therefore, may contain large quantities of silts, clays, and precipitated metal coagulants.

All water treatment plants that employ rapid filtration produce large volumes of washwater containing a low concentration of suspended solids. The volume of backwash water is usually 2 to 3 percent of the treatment plant flow. The characteristics of solids in the backwash water resemble those found in the sedimentation units. Since filters can support biological growth, the filter backwash may contain a larger fraction of volatile solids than solids from sedimentation basins. Any suspended solids present in the water will produce an equal amount of residual

Generally speaking, the water treatment plant residual consists of fine sands, silt, clays, precipitated metal coagulants, conditioning agents, and organic matter.

2.1.2 Handling and Disposal of WTP Residuals

Many methods exist for handing and disposal of water treatment plant residuals, but only a few are widely used. These generally can be divided into seven categories

1 Direct discharge to receiving stream,
2. Co-disposal with sewage sludge at a waste treatment plant,
3. Nonmechanical dewatering methods,
4. Mechanical dewatering methods,
5. Incineration,

6. Land application, for soil stabilization and for agriculture, and
7. Landfilling

A method of disposal is to discharge the WTP residual to a sanitary sewerage system. Co-disposal of alum residual has been practiced with success in four large U.S. cities -- Detroit, Michigan; Wilmington, Delaware; Washington, DC; and Philadelphia, Pennsylvania (Malcolm Pirnie 1978).

Nonmechanical dewatering can be accomplished either in open lagoons or in sand-drying beds. With the use of either of these alternatives, freezing-thawing of the residual might be considered to increase the solids content of the residual. Dewatering of WTP residuals using nonmechanical methods has been found to occur via two basic mechanisms: (1) residual drainage — draining of the free water from the residual; and (2) residual drying -- evaporation of water remaining after draining. Cornwell et al. (1981) have described nonmechanical dewatering techniques of alum residuals in which alum is extracted by treatment with acid. Since 1980, the lagoons and drying beds have been widely used at water treatment plants because they can lead to an economical way of treating water treatment residuals.

Various dewatering systems have been used to produce a residual cake suitable for land disposal and a liquid stream suitable for recycle or discharge (Westerhoff and Daly 1974). Centrifugation, vacuum filtration, belt filtration, and pressure filtration have been widely tested and successfully used.

Land application of WTP residual is achieved by spreading the residual over the ground surface or blending it within the topsoil. WTP residual is applied to agricultural and forest lands for its nutrient value and disturbed areas for soil amendment and reclamation purposes (Bugbee and Frink 1985).

Incineration of WTP residuals is best viewed as two operations: first, evaporating the moisture or drying the solids, and second, burning the dried solids. There are several types of incinerators on the market. They generally perform well if properly operated, however, it should be noted that alum residual has been found to cause an unacceptable build-up of aluminum slag on incinerator walls in some cases.

The landfill (Weiss, 1974) is a method of disposing of WTP residual on land, without creating nuisances or hazards to public health or safety. If properly designed, constructed, and operated, it provides an economical method of ultimate disposal of WTP residuals.

2.2 Characteristics Of Water Treatment Plant Residuals

Characteristics of WTP residuals vary from one treatment plant to another. They depend on the raw water quality, treatment processes, chemicals added, and methods of dewatering. A detailed analysis of residual characteristics should be conducted for each residual. However, there are some common characteristics among particular types of residuals.

2.2.1 The Composition, Structure and Related Properties of WTP Residuals

Residuals from water treatment processes are composed mainly of soil particles, coagulants, organic materials, and water. Sources of the soil particles and organic materials are the colloidal and suspended materials in raw water sources.

Surface waters generally contain a variety of colloidal impurities that cause turbidity and impart color. These impurities include inorganic and organic solids, ions, and microorganisms. Colloids are substance whose particle sizes range from about 1nm (1nm = 10^{-6} mm) to 1μm (1μm = 10^{-3} mm) when they are mixed with water or other substance (Bohn et al. 1985).

Suspended inorganic solids are mainly clay-sized fraction particles with a size range from 1nm to 1μm . They are fine crystal mineral sheets resulting from the weathering products of rocks. Minerals of the clay-sized fraction commonly include the carbonate and sulfur minerals, the layer silicates, and various oxides. These solids usually are not involved in any chemical reactions during water treatment processes (Bohn et al. 1985).

The most abundant carbonate mineral is calcite ($CaCO_3$). The most important secondary mineral in clay is silicate. When the particle size of this mineral is reduced to clay or colloidal size fractions, the influence of these minerals on the properties of the residuals is greatly enhanced. They will not only affect the plasticity and cohesion, swelling and shrinkage, but cation exchange capacity (CEC) as well. Other important constituents are so-called free oxide minerals such as allophane. Allophane is a general name for amorphous aluminosilicate gels. The composition of allophane varies widely but includes mostly hydrated Al_2O_3, SiO_2, and Fe_2O_3. Aluminum oxide (Al_2O_3) is called alumina. Clay contains a large proportion of it. Silicon dioxide (SiO_2) is called silica. Pure form of silica (usually called quartz), for example, quartz sand, is extremely stable. But other modifications which consist of extremely small siliceous skeletons are far more reactive than ordinary quartz (Czernin 1962). Allophane may also have a high CEC and a large surface area (Bohn et al. 1985). Layer silicates, allophane and the chemical additives from water treatment process can play important roles in affecting the properties of WTP residuals.

Organic materials are mainly colloidal polymers called humus produced by the degradation of nonhumus materials undergoing enzymatic and chemical reactions. Humus is a complex mixture of brown to black colored amorphous and highly colloidal substance. It can hold up to 20 times its weight in water which prevents drying and shrinking. Humus helps to stabilize the structure by forming structural units called aggregates, which are similar to flocs. It also increases the cation exchange capacity of the residual. In general, 20 to 70% of the total CEC of many soils is due to the organic matter present in them (Bohn et al. 1985).

Raw water usually contains some metal ions like calcium (Ca^{2+}), magnesium (Mg^{2+}), ferric (Fe^{3+}), etc. which cause hardness and impart color to water. During the water treatment process, hardness and color are usually removed by addition of lime and/or soda ash. Colloidal particles are removed from water by the addition of coagulants which induce flocculation. Some ions in the residual are retained in soils by cation and anion exchange, precipitation, weak electrostatic attraction and as a complex with organic matter.

Alum, ferric chloride, ferric sulfate, and lime are mainly used as coagulants in water treatment process. Some water treatment plants use lime as conditioning agent in dewatering processes. These additives are used to promote colloidal aggregation by destroying the forces that stabilize colloidal particles. When these destabilized colloidal particles move toward each other and become attached during the sedimentation or dewatering process, flocculation occurs.

Clay colloidal impurities in receiving water usually carry negative charges. Since like charges repel, these similarly charged colloids are held apart from each other by their electric charges and thus are prevented from aggregating into larger particles. When coagulant is added (like alum or ferric sulfate, etc.), it will dissociate to yield Al^{3+} or Fe^{3+} ions, which then hydrate to form a variety of aquametal complexes such as $Al(OH)^{2+}$, $Al(OH)_2^+$ and other polynuclear species such as aluminum hydroxide polymer ($Al_8(OH)_{20}^{4+}$). These species are capable of being adsorbed at the surface of colloidal particles, reducing the surface potential. The negative charges of colloids are neutralized by a swarm of positive ions in the solution called diffused double layer (DDL), resulting in the destabilization of the colloidal particles. The DDL imparts a net positive charge at the edges of the colloidal particles, so when these particles are very close together (for example during flocculation), this edge charge can participate in an edge-to-face linkage between particles to form a so called floc-structure (Figure 2.1). The structure formed is sometimes called salt type floc-structure (Lambe and Whitman 1969).

2.2.2 Water in WTP Residuals

Water in WTP residuals can be classified into four categories as discussed below (Knocke and Wakeland 1983, and Huang 1979):

1. Free water: This kind of water is capable of moving freely by gravity. It is very easy to remove free water by using dewatering equipment such as presses. Free

water can also be removed by processes such as evaporation and drying due to weather, which can take a very long time. Typically, WTP residual samples collected from lagoons contain large quantities of free water, and dewatered cake samples have very little free water.

2. Floc water: This kind of water is free water trapped within the voids of the floc structure. Dewatered residual cake contains mostly floc water. Unless the floc surfaces are moved, the floc water trapped in them can not be removed. Significant quantities of energy may be required to achieve floc water removal. This can be accomplished by processes such as heating, freeze and thaw, air drying, and mechanical squeezing. Discussion of some of these processes will be presented elsewhere in this study.
3. Capillary water is held tightly to the particle surface by surface tension. These forces are believed to be short-ranged but very close to the particle surface resulting in the appearance of highly structured water molecules. In WTP residuals, this water can be within the flocs as well, as long as it is associated with a solid surface. The major difference between capillary water and floc water is that the latter is free to move and can be removed by mechanical dewatering. The capillary water is not free to move but adheres to solid surfaces. It can be removed by mechanical force if the flocs are broken up and the capillary water become free water.
4. Adsorbed (Bound) water: This kind of water is bound (adsorbed) within the molecular structure of colloidal solids. Water molecules are held together by

hydrogen bonding. Due to the elliptical shape of these molecules, the centers of gravity of the positive and negative charges are not the same. So, the water molecules tend to assume a configuration such that their positively charged portions are close to the negatively charged surfaces of the colloids present in the WTP residuals. These water molecules are adsorbed and so they can not be removed by mechanical effects such as dewatering, evaporation, and normal drying processes. A great deal of energy must be expended in breaking the bond between the water molecules and the negatively charged surfaces. This can be accomplished by heating the residual to a very high temperature or by applying a very large amount of electrical energy.

Change in the water content (amount and distribution) of residual materials is the greatest single cause of variation in their geotechnical properties. It will not only alter the floc structure and the particle sizes of the solids but also will change the ion concentration and complex formation within the residuals.

Generally, geotechnical behavior of WTP residual material is not only a function of the physical and chemical composition of its solid contents but also a function of the type, amount, and chemical nature of the pore fluids. As the structure and the solids content of the WTP residuals change, interactions between the solid and the liquid phases such as cementation takes place. This will affect the geotechnical properties such as compaction (handling), shear strength and permeability.

The major constituent of any of the WTP residuals is water. In the case of a lime residual, the dewatering characteristics of the residual are related to the calcium and magnesium molar ratio. A residual with a calcium and magnesium ratio less than two will be difficult to dewater (AWWA 1981). Studies have also shown that the size of the calcium carbonate formed during the chemical reaction also affects the residual thickening and dewatering (Judkin and Wynne 1983). Softening and coagulation residuals tend to be thixotropic and the coagulation residual is generally gelatinous (AWWA 1978). In the case of alum residual, the chemically bound water is about 40 percent of the total water content and therefore, it will be difficult to dewater a residual in the form of chemical hydroxide to greater than a 60 percent solids concentration with a mechanical device. In practice, dewatering is limited to 45 to 50 percent solids concentration. With the passage of time, because of the alternate freezing and thawing and wet and dry environment the solids concentration increases up to 90 percent.

2.2.3 Prior Studies Related to WTP Residual Characteristics

Elliott et al. (1990) conducted a study on the land application of WTP residual. For the 20 WTP residuals tested, they found that the typical WTP residuals were predominantly inorganic. On an average, the residuals contain 3 percent by weight of organic carbon which is stable and resistant to degradation. The nitrogen content is similar to that present in soil, which is 0.6 percent by weight (Elliott et al. 1990). Trace metal concentrations in WTP residuals are between those of soils and sewage sludges. The total concentrations of the six metals studied, which were cadmium, copper, chromium, nickel, lead, and zinc, varied considerably. Only two WTP residuals had total Ni levels near the maximum recommended level (200 mg/kg) allowed in Pennsylvania for waste materials to be land applied. WTP residuals contain very low concentrations of phosphorus and large amounts of aluminum and iron hydroxide solids. The microbial toxicity of the residuals was very low except for two samples which had high concentrations of extractable Ni and Cd.

In order to assess the potential for contaminants to leach from coagulant residuals and thereby, impact water sources, alum residual toxicity studies were conducted by George et al. (1991), the results showed that toxicity occurred with acidic extracts and basic extracts and not with circumneutral extracts, acidic extracts were most toxic. He also reported that no chronic toxicity was measured with the alum residual extracts from the four water treatment plants tested. The data indicate that water utilities may adversely affect aquatic primary production by discharging alum residuals in acidic receiving waters and soft surface waters with a hardness less than 40 mg $CaCO_3$/L. A pilot evaluation study for toxicity of WTP residuals was conducted by Cornwell et al. (1992). For the three residuals studied, they found that all three residuals were nonhazardous (USEPA. 1986a, 1991) based on the results of Toxicity Characteristics Leaching Procedure (TCLP) tests. Some degree of leaching of arsenic, copper, iron, manganese, and zinc was observed from all residuals; however, the percentage of the total contaminant that leached was under 3 percent and often below 1 percent.

Historically, physical characteristics of WTP residuals have not been a major concern. This is because ultimate disposal practices did not have stringent strength requirements. Disposal of dewatered residual was typically governed by solids concentration (Cornwell et al. 1992). To establish design guidelines and criteria for residual handling, determination and understanding of the fundamental properties of these residuals are essential.

In one of the early studies on this subject, Raghu and Hsieh (1987a, 1987b) performed modified proctor compaction tests on a lime/alum type residual and a lime type residual. Compaction curves exhibited typical one-hump shape. For the lime/alum residual, the optimum water content was about 65 percent and the maximum dry unit weight was about 51 lb/pcf. The corresponding numbers for the lime residual were 28 percent and 84.5 percent respectively. Cornwell et al. (1992) reported the results of standard Proctor compaction test on a coagulant residual that had been stored at a water plant for an extended time. The test results showed the typical one-hump compaction curve. Optimum moisture content and maximum dry unit weight were approximately 17 percent and 105 pcf, respectively. Wang et al. (1992) reported the engineering behavior of one iron and two alum coagulant WTP residuals. Test results indicate that the residuals were extremely plastic and highly compressible. The compaction curve for the iron type residual exhibited typical one-hump shape, whereas that for the alum residuals showed no peak. Wang et al. (1993) found that the compaction curve for alum type WTP residual was not that of the typical one-hump. Instead, it exhibited a monotonically decreasing pattern, the dry density decreased with increasing water content from a maximum near zero water content (Vesilind, 1974).

Raghu and Hsieh (1987a, 1987b) conducted conventional consolidation tests to determine the coefficient of permeability of the two WTP residuals mentioned above. The coefficients of permeability were found to be on the order from 10^{-6} to 10^{-7} cm/s. Using the method of dewatering under different pressure differentials, Knocke and Wakeland (1983) investigated the compressibility of four residuals,--alum (low density), alum (high density), conditioned alum, and lime residuals. The average coefficient of compressibility for the first three alum residuals ranged between 0.94 and 0.97, whereas that of the lime residual was 0.79. These results indicate that the lime residual was considerably less compressible than the alum residual. Wang and Tseng (1993) reported the permeability characteristics of an alum water-treatment residual treated separately with a slaked lime and fly ash showed that, for both the treated and the untreated residuals, the permeability-versus-void ratio relationship was expressed as a power function as opposed to an exponential function. Under the same void ratio, the treated sample was noted to have a greater permeability than the untreated one. It was found that within the void ratio of approximately between 7 and 17, the coefficient of permeability varies from about 3×10^{-8} to 8×10^{-6} cm/s for untreated residual samples. All these residuals indicate that the WTP residuals to be impervious.

Available information on the shear strength characteristics of WTP residuals concentrates mainly on the undrained strength with regard to sludge handleability (or workability). Raghu and Hsieh (1987a) conducted unconfined compression and direct shear tests on a lime type residual at a dry density of 84.5 pcf and a moisture content of 28 percent. A value of 9.6 psi was obtained for the unconfined compression strength. An angle of internal friction of 28 degrees and a cohesion of 2.5 psi were determined from an undrained direct shear test. Most other studies used the vane shear test method to determine the strength (Cornwell and Koppers 1990).

Novak and Calkins (1975) studied five sludges using a torvane and reported 0.04 tsf to be the minimum undrained shear strength for easy handling of residuals. This minimum shear strength value is about 2.5 times lower than 10 kPa (0.104 tsf) currently used in Germany and Netherlands for determining the acceptability of WTP residuals for disposal in a landfill (Cornwell and Koppers 1990). Using both a torvane and a viscometer, Huang (1979) presented, a relationship between the undrained shear strength and solids concentration for two residuals.

Reports about the study of the durability of WTP residuals are scarce. Most of the previous work in this area focused on dewatering by freezing-thawing. Ferrell et al. (1970) reported on the design of facilities for dewatering alum residuals by natural freezing. Logsdon et al. (1971) studied the effective dewatering freezing rate. It was suggested that high freezing rate caused effective dewatering and residuals could be effectively frozen in thin layers on a flat surface. Vesilind et al. (1990) presented a conceptual model for freezing of WTP residuals that explained the improvement of dewaterability of WTP residuals by freezing.

2.2.4 Leaching Consideration of WTP Residual

Because the high concentration of primary metals presents in WTP residuals, such as aluminum, iron, and lime, the key characteristeic of the residual is the degree to which ions will leach from the solid form. A variety of tests have been developed to measure leaching under a variety conditions (Bishop, 1982, 1988). The Toxicity Characteristic Leaching

Test (TCLP) and the Extraction Procedure (EP) are the most important leach tests (Ham et al, 1979).

In order to help assess the potential for leaching contaminants from coagulant sludges and thereby impact water sources, a six-month continuous column leaching test was conducted by cornwell et al. (1992). Simulated acid rain was used as a leaching solution. Leachate was analyzed for six months with the equivalent of ten years of rain having been applied. The results were concluded that none of the metal concentrations in the leachate exceeded drinking water MCLs and therefore no groundwater impact would be expected.

In the literature review, there is no leaching models for WTP residuals developed to predict performance on the TCLP test, to predict long-term leaching behavior in the environment, to correlate experimental data, and to improve understanding of WTP residuals.

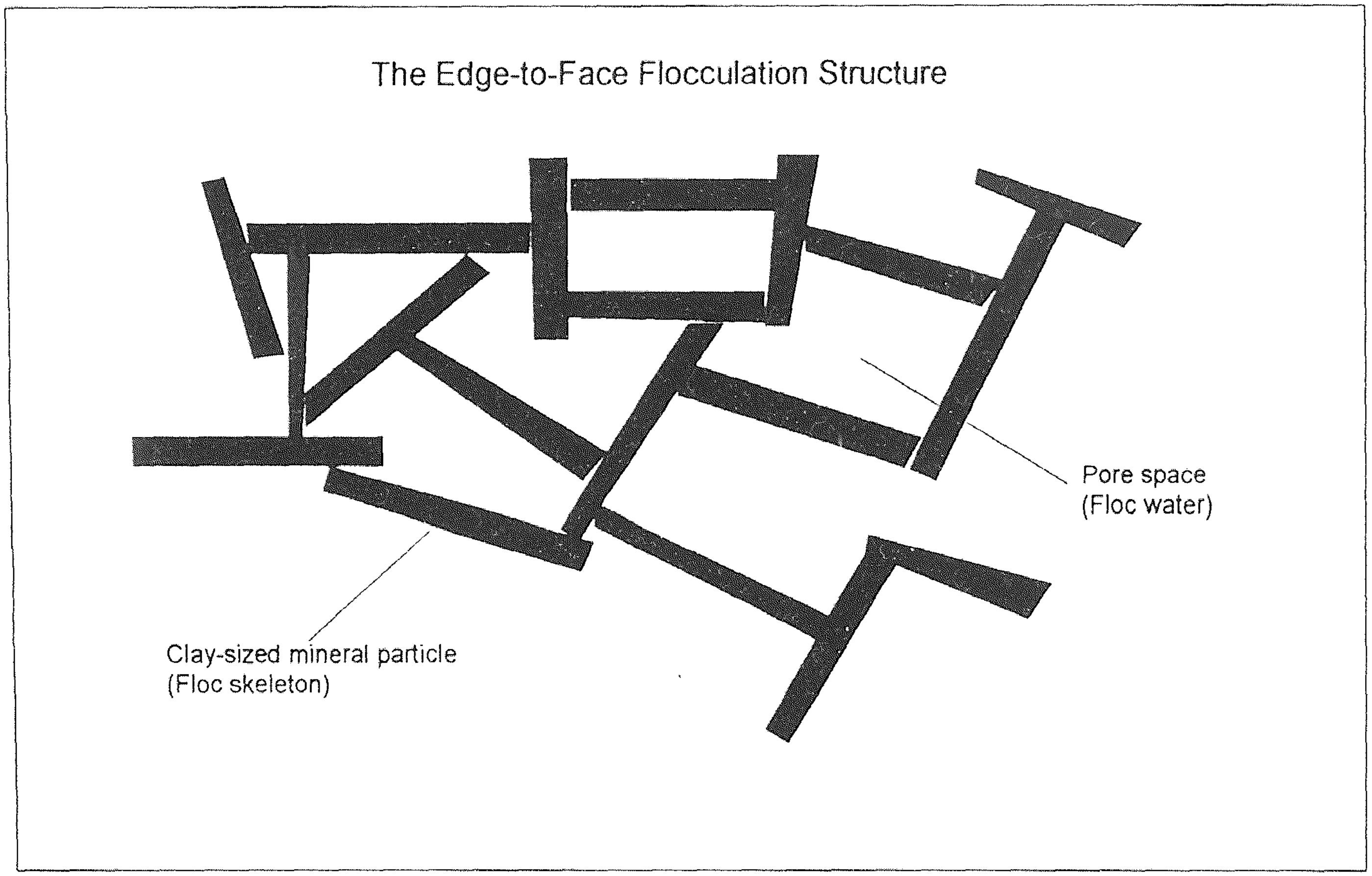


Figure 2.1 Illustration of flocculated structure of WTP residual

CHAPTER 3

MATERIALS AND METHODS

3.1 Introduction

Characterization of WTP residuals must be undertaken prior to the development of design, construction and operation of WTP residual monofills. Origin, generation and the parameters affecting the characteristics of WTP residuals were presented in Chapter 2. In general, waste characterization is done to address the following issues: (Bagchi 1994)

- Whether the waste is hazardous.
- Whether the waste can be monofilled.
- Probable leachate constituents (necessary for judging liner compatibility, treatment plant design and groundwater monitoring program design).
- Volume rate of waste generation.
- Physical properties of waste necessary for the design, construction and operation of monofill and
- Identification of waste reduction alternatives.

The first three issues can be addressed by conducting environmental tests such as TCLP tests and paint filter tests. The fourth and the sixth issues are treatment plant specific and are beyond the scope of this study. The fifth issue can be investigated by performing selected environmental and physical tests on residuals.

Samples collected for testing for this study were analyzed according to the procedures and requirements specified in relevant EPA procedures. The testing parameters and methods utilized are listed in Table 3.1. These were selected to fully investigate the environmental and the geotechnical properties of the residual. However, since these tests may not be exactly suited for testing residuals, it became necessary to modify these procedures to suit our requirements. Such modifications and/or changes in procedures are also described in this Chapter. (Table 3.1)The quality assurance and quality control procedures as specified in each method are presented in Appendix A.

3.2 Origin and Production of Residual Samples Tested for this Study

Samples of WTP residuals for this study were collected from ten treatment facilities located in the U.S. The basis for selecting the types of WTP samples and the plants from which the samples were obtained involved several factors,

- These include major types of WTP residuals (based on coagulants added);
- different raw water sources;
- treatment processes;
- types of dewatering methods;
- geographical location; and
- diversity of monofills.

Major types of WTP dewatered residuals produced in the United State are alum, ferric, and lime. Most WTP residuals are generated from coagulation, water softening, sedimentation and filtration processes. The raw water sources of the samples were reservoirs and rivers. The treatment processes are discussed in the following sections in this study. Dewatering processes utilized for the samples tested were mechanical (belt press, filter press, and centrifuge), and non-mechanical (lagoon and drying bed).

For convenience of discussion, symbols were assigned to each of the samples collected. These symbols are listed in Table 3.2 below and will be used throughout this study.

Locations of the treatment plants from which the samples were obtained are shown in Table 3.3. Also presented in this table is information describing the types of WTP residuals, water sources, properties of water sources, water treatment processes, dewatering processes, and sampling processes.

3.2.1 Sample Collection and Handling

Trip blanks and field blanks were prepared for residual samples collected from treatment plants located in New Jersey. Samples were collected directly from the drying beds or dewatering machines. They were properly labeled and placed in the coolers with freeze packs. Chain of custody forms were prepared and maintained. Appropriate procedures were followed during transportation, storage, and preparation for testing so that samples would not be disturbed or cross contaminated (USEPA 1986). For the samples collected from outside of New Jersey, the dewatered cake samples were requested to be placed in a 4-mil-thick polyethylene bag. The bag was then enclosed in a plastic container and sent to NJIT, in Newark, New Jersey.

Sample labels included the following information:

Name of collector,

Date and time of collection,

Location and place of collection,

Type of residual, and

Dewatering method.

The physical, chemical biological, and geotechnical tests were conducted within the time limit specified by EPA and ASTM methods.

For testing volatile organic compounds (VOCs) in WTP residuals, the liquid samples were collected in 40 ml-wide-mouth amber glass jars, sealed with Teflon cap, and then put in the cooler with freeze packs to keep the samples under 4°C during the transportation. The solid samples were collected in wide-mouth amber glass jars, sealed with Teflon cap, and then put in the cooler with freeze packs to keep the samples under 4°C. EPA regulations stipulate that the samples have to be tested within two weeks from the date of collection.

As part of field testing, samples of water treatment plant residuals and water were collected by the NJIT research team from the minimonofill at New Castle, Pennsylvania. Sample collection, transportation, and storage were the same as these residual samples collected from New Jersey WTP. These samples were obtained over a period of one year. The three phases of sample collection for testing were immediately, six month, and one year after the installation of the monofill.

3.2.2 Treatment Plants from Which Samples Were Collected

1. Jersey City Water Treatment Plant at Boonton, NJ

Water treated in Jersey City Water Treatment Plant (Figure 3.1) is collected from the Rockaway River and Boonton Reservoir. This plant produces 47 to 80 mgd (1.7 to 3.0 x 10^5 m^3/day) of water. Impurities in raw water include turbidity, color, iron, and hardness (Table 3.3). Treatment processes consist of rapid mix coagulation, flocculation, filtration and chlorination. Lime, alum, and polymers are used as coagulants. WTP residuals, produced from coagulation and filtration, are conditioned by adding 59 percent of lime and dewatered with plate frame filter press. The amount of residual cake generated in this plant is approximately 8,000 lb/day (3,616 kg/day). The dewatered residual has a solids content of about 25-35 percent.

2. Passaic Valley Water Treatment Plant at Clifton, NJ

The Passaic Valley Water Treatment Plant (Figure 3.2) has a capacity of 39 - 50 mgd (1.5 to 1.9 x 10^5 M^3/day). The impurities in raw water include color, turbidity, and iron. Treatment processes utilized are flocculation, multiple media filtration, and disinfection. Alum is the coagulant used. WTP residuals are withdrawn from four sedimentation basins and pumped to two holding tanks. Lime is used as conditioning reagent and the WTP residuals are dewatered with two pressure frame filter presses to produce a dewatered cake with solids content of 30 percent.

3. Wanaque Water Treatment Plant at Wanaque, NJ

The flow in the Wanaque water treatment plant (Figure 3.3) varies from 40 to 140 mgd (1.5 to 5.3 x 10^5 M^3/day), and the average capacity is 105 mgd (4.0 x 10^5 M^3/day). Water is stored in Wanaque Reservoir before treatment. Impurities in raw water mainly consist of turbidity, color, iron and manganese (Table 3.3). The water treatment process involves pretreatment (premix basins and reaction basins), coagulation, sedimentation, and filtration. Alum and polymer are used as coagulants. WTP residuals, produced from settling basins and filters, flow to lagoons and have an average solids content of 0.25 to 1.0 percent. The residuals were dewatered with a belt press dewatering machine to a concentration of 14 percent solids content and then air dried in a drying bed.

4. *West River Water Treatment Plant in Woodbridge, CT*

The water sources for West River Water Treatment Plant (Figure 3.4) are Lakes Dawson, Glen and Watrous . The plant has a capacity of 10 mgd (3.8×10^4 M^3/day) and an average annual production of 7 mgd (2.6×10^4 M^3/day). Impurities in the raw water are turbidity, color, iron, and manganese (Table 3.3). The water treatment facility is a direct filtration process including a rapid mix chamber where the coagulant is added. Ferric chloride and cationic polymer are used as coagulants. Backwash water from gravity filtration is directly sent to lagoons. Residuals from the lagoons are pumped to a drying bed twice a year. Four types of samples were collected from this plant as shown as in Figure 3.4. Sample 1 was collected from backwash water in dual media filters. From the lagoons, sample 2 was collected from the lagoons. The remaining samples, 3 and 4, were collected from the drying bed. Sample 3 (RWA) was obtained from a region in the drying bed below the zone of frost penetration. This sample was not subjected to freezing and thawing and was in a paste form. Sample 4 (RWAF) was a freeze-thawed residual because it was collected from a region close to the surface and it had frozen. This sample was in a dry (granular) form. Samples 1 and 2 were not utilized in this study.

5. Manasquan Water Treatment Plant in Monmouth County, NJ

The Manasquan Water Treatment Plant (Figure 3.5) has a capacity of 4 mgd (1.5 x 10^4 M^3/day). The water sources are Manasquan River and Manasquan Reservoir. The impurities include turbidity, iron and manganese, and color (Table 3.3). Water treatment processes utilized are ozonation, mixing, clarification, mixed media filtration, granular activated carbon adsorption, and chlorination. The coagulant used is alum. Coagulant aid (LT20 Percol) is used year round. Clarifier residual and filter backwash water are directed to the lagoons. The residual is placed in the lagoon for about six months and then air dried in field. Sample for testing was collected from the air drying field with a shovel.

6. Treatment Plant of Minneapolis Water Works, Minneapolis, MN

The Minneapolis Water Works Plant (Figure 3.6) produces an average flow of 72 mgd (60 mgd (2.7 x 10^5 M^3/day) in the winter and 180 mgd (6.8 x 10^5 M^3/day) max. in the summer). The sole water source is the Mississippi River. The impurities include hardness, color, turbidity, and taste and odor (Table 3.3). Water treatment processes include softening, carbon adsorption, occasionally potassium permanganate, carbonation, flocculation, sedimentation, and filtration. Alum and iron are used as coagulants and lime is added for softening. Powder activated carbon is used for taste and odor control. The residuals from softening and backwash processes are thickened to a solids content of 7 to

10 percent and fed into a centrifuge. The sample collected and tested was centrifuged cake.

7. Haworth Water Treatment Plant at Harrington Park, NJ

Haworth Water Treatment Plant (Figure 3.7) has a peak capacity of 160 mgd (6.1×10^5 M^3/day) at two treatment plants and an average capacity of 103 mgd (3.9 x 105 M^3/day). The surface water from Hackensack River and Pascack Brook is stored in the reservoir, and is then pumped into an ozone contactor. Water is then subjected to flotation/skimmer, dual media filtration, and finally it flows into the distribution system after disinfection. The impurities in raw water include hardness and color (Table 3.3). Alum is used as coagulant in the coagulation process. Activated carbon is used occasionally. Ozone is used as the primary disinfectant. The backwash water from dual media filters is pumped into the lagoons and is transported to the drying bed after thickening. Two samples were collected. One sample (HWR) was collected from outlet of backwash water tank, and another sample (HWD) was collected from the drying bed.

8. Ellwood City Water Treatment Plant, Ellwood City, PA

The water source for the Ellwood City Water Treatment Plant (Figure 3.8) is Slippery Rock Creek. Average water production is 8 mgd (3.0×10^4 M^3/day). Impurities in water are turbidity, taste and odor, iron, and hardness (Table 3.3). Water treatment processes include coagulation with lime, sometimes oxidation with $KMnO_4$, presettling, sedimentation, filtration, and chlorination. Ferric chloride and cationic polymer are used as coagulants. Backwash water from dual media gravity filtration and sedimentation residuals are directly sent to lagoons. Then the residuals from the lagoons are moved to a drying bed and transported to monofill after drying. Two types of samples were collected from this plant. Sample 1 (ELR) was collected from lagoon and sample 2 (ELD) was from the drying bed. The environmental test results of sample 2 (ELD) are presented in this study.

9. Bradenton Water Treatment Plant, City of Bradenton, FL

Two types of residual samples were sent to the laboratory at the Bradenton Water Treatment Plant (Figure 3.9) on May 18, 1993, and were received on May 27, 1993. One sample was labeled "from drying bed" and the other "prior to monofill". Based on the information from the plant, this facility has a capacity of 32 - 40 mgd (1.2 - 1.5 x 10^5 M^3/day). The average capacity is 36 mgd (1.4 x 10^5 M^3/day). The water source is Lake Manatee. The impurities include color, taste and odor, organics, and hardness (Table 3.3). Water treatment processes utilized are powdered activated carbon addition, flocculation with alum, lime, and polymer, sedimentation followed by postmixing with chlorine, ammonia, and lime, mixed media filtration, and chlorination. The main coagulant used is alum. Coagulant aid (polymer) is used during summer. Flocculation and sedimentation residual and filter backwash water are directed to the residual lagoons. The residual is placed in the drying bed and discharged into the landfill. Three samples were collected from this plant. Sample 1 (FLR) was collected prior to the residual entering the drying bed, sample 2 (FLD) was collected from the drying bed, and sample 3 (FLDM) was collected prior to the residual disposed in the landfill.

10 St. Louis County Water Company, St. Louis, MO

South County Plant of St. Louis County Water Company (Figure 3.10) has a capacity of 40 mgd (1.5×10^5 M^3/day) with an average flow of 15 mgd (5.7×10^4 M^3/day). The raw water from Meramec river is subjected to softening, settling, filtration, and released to the distribution system after disinfection. Ferric sulphate is used as coagulant and lime is used in the softening process. The residual from water treatment process is pumped into the lagoons and is transported to the landfill after thickening. A residual sample was collected from the landfill, which was one year old.

3.3 Environmental Testing Methods

As shown in Table 3.1, environmental characteristics tested include general physical and chemical test, biodegradability, and the toxicity characteristic leaching procedure (TCLP) test.

3.3.1 General Physical and Chemical Tests

General physical and chemical tests for determination of physical and chemical properties of WTP residuals include paint filter test, solids content, volatile solids, pH, metals composition, and cation exchange capacity.

3.3.1.1 ***Paint Filter Test:*** Based on federal regulations [RCRA Subtitle D], solid wastes to be disposed of in a sanitary landfill should be adequately dewatered and should not contain freely flowing liquids. A test called paint filter test was performed in the laboratory to determine the amount of the free liquid. EPA method 9095 was used for this test. 100 grams of dewatered WTP residual sample was put in a funnel with a 60 mesh filter. If any liquid passed through the paint filter during a 5-min. test period, the sample was deemed to contain freely flowing liquids.

3.3.1.2 Composition Tests: The composition tests include determination of solids concentration, volatile solids, fixed solids, and primary metals extracted from WTP residual.

Total solids were determined gravimetrically as outlined in Standard Methods (APHA, AWWA, and WEP 1992). A well-mixed WTP sample was placed in a weighing dish and dried to constant weight in an oven (Fisher, BLUE M) at temperature of 103 to 105 °C. Volatile solids were determined by placing the sample in a furnace and heating to a temperature of 550 ± 50 °C. The remaining solids represented the fixed solids, while the weight lost was the volatile solids.

The alkalinity of WTP residual is its capacity to neutralize acids. Because of the high alkalinity of lime residual, determination of alkalinity was conducted by measuring the volume of 5.0 N sulfuric acid required to titrate 4 grams of sample dissolved in 100 mL of distilled water to the designated pHs to obtain hydroxide (pH=10), carbonate (pH=8.4), and bicarbonate alkalinity's (pH=4.3) with a calibrated pH meter (Orion Model 420A).

Inductively Coupled Plasma - Mass Spectrometry (ICP-MS) was used to analyze primary TCLP extracted metals and total metals in extracted acid digestion procedure. The primary metals and ion tested included iron, aluminum, calcium, sodium, manganese, zinc, magnesium, and copper.

3.3.1.3 pH: EPA method 9045 is a procedure for measuring pH in calcareous and non calcareous soils. The above EPA method was used to determine pH of WTP residuals for this study. The samples were mixed either with Type II water or with a calcium chloride solution, depending on whether the WTP residual was considered calcareous or non-calcareous. The pH of the solution was then measured with a pH meter (Orion Model 420A). The buffers of pH 4, pH 7, and pH 10 were used to calibrate the pH meter.

3.3.1.4: Cation Exchange Capacity: Cation exchange capacity (CEC) can be defined as the capacity of the soil to adsorb cations including heavy metal ions during the leaching process. The procedure for determining the cation exchange capacity of WTP residual included two steps. The first was to exchange the cation in the residual with sodium ion. The second step was to measure the sodium with atomic adsorption (AA) or emission spectroscopy. The procedure was conducted by following the EPA method that is briefly described below:

Approximately 50 g WTP residual cake sample was put into a plastic container and the particle size of sample was reduced to less than 0.5 mm by grinding. Then 4 grams of the residual was transferred to a 50 mL, round-bottom, narrow-neck centrifuge tube and

33 mL of 1.0 N sodium acetate solution was added. The tube was stoppered. It was shaken in a mechanical shaker for 5 min and centrifuged until the supernatant was clear. The liquid was decanted and this procedure was repeated two more times. 33 mL of 99 percent isopropyl alcohol was added, the tube was stoppered and shaken in a mechanical shaker for 5 min, and centrifuged until the supernatant was clear. This procedure was repeated three or more times. 33 mL of ammonium acetate solution was added, the tube was stoppered and shaken in a mechanical shaker for 5 min, and centrifuged until the supernatant liquid was clear. The washings were decanted into a 100-mL volumetric flask. This procedure was repeated two more times. The combined washing was diluted to the 100-mL mark with ammonium acetate solution and the concentration of sodium was determined by atomic absorption or emission spectroscopy.

3.3.2 Biodegradability

To evaluate the possibility of biogas production under a monofilled condition and to determine the biodegradation potential of the WTP residuals, the residuals were digested in three types of reactors. The first reactor consisted of 40 mL vials with Teflon coated rubber septa to sample gas. Three types of residuals, namely raw residual, conditioned residual, and dewatered residual, obtained from Jersey City WTP were put into those vials. Vials were placed in a constant temperature water bath at a temperature of 35°± 2 ° C. After two months, the composition of gas collected was determined.

The second type of reactors used was 150 mL anaerobic vials. One type of dewatered residuals was halved. The two halves were tested over a period of six months. One portion was added with 2 mL of seeds taken from an anaerobic digester in a local wastewater treatment plant and the other without seeds. Temperature was maintained at 35°C± 2 °C by placing all reactors in a water bath. The gas volumes were measured with syringe after 191 days.

A two liter batch reactor was utilized to investigate anaerobic biodegradation of WTP raw, conditioned, and dewatered residuals. The temperature was controlled at 35 °C± 2 °C and solid retention time was maintained at one month. During this test, the biogas production was measured and microorganisms were examined.

Gas chromatography (HP5890) with a thermal conductivity detector (TCD) was used to analyze carbon dioxide and methane. Stainless steel columns (2.0 m x4.0 mm ID) packed with 80/100-mesh Poropak Q was used in this experiment. Operating conditions were set at a column temperature of 95 °C, detector temperature of 100 °C, and with helium as the carrier gas at a flow rate of 35 mL/min. Quantitative analysis based on area normalization procedure was performed by using standard gas mixtures to determine response factors.

3.3.3 The Toxicity Characteristic Leaching Procedure (TCLP) Analysis for WTP Residuals

As a result of US EPA amendments by which hazardous wastes are identified under RCRA, a new extraction procedure, the toxicity characteristics leaching procedure (TCLP) was introduced. This test replaced the EP toxicity test and increased the number of contaminants regulated to 40 and lowered the acceptable limits of concentrations. In March 1990, the TCLP test was formally introduced.

The dewatered residuals were analyzed for volatile, semivolatiles, metals, organochlorine pesticides and herbicides by GC-MS with Purge-Trap Autosampler, GC-MS, ICP-MS, and GC-ECD, respectively. Before analyzing the regulated substances of dewatered residual samples, the extraction of contaminants was conducted using an extractor instrument. The flowchart of TCLP test is shown in Figure 3.11.

3.3.3.1 TCLP Extraction: EPA method 1311 (USEPA. 1986b) defines the extraction procedures. Since water treatment plant dewatered residuals contain significant amount of solids, which are usually between 15-35 percent on a dry basis, particle size reduction and liquid separation were performed if necessary. The solids are then extracted with 20 times extraction fluid (by weight). A 6-vessel rotary agitation apparatus (Associate Design and Manufacturing Company, Model 3740-6-BRE) was used to extract metals, pesticides, herbicides, BNA, and volatile compounds in this study. Following extraction, filtration was employed to separate liquid and solids.

The 6-Vessel rotary agitator with Zero-Headspace Extraction Vessel (Model 3745-ZHE) was used for residuals containing volatile organic compounds and Extraction Wide-Mouth Teflon bottles was used for residuals containing semivolatile and nonvolatile compounds.

Pressure filtration unit (Associated Design Model 3750-HWF) was used for separating solid samples and their extracts during the semivolatile and non-volatile tests in accordance with U.S. EPA Method 1311. The 142 mm and 90 mm Glass Fiber Filters, with an effective pore size of 0.7 μm, were used in the filtration.

There are two types of extraction fluids specified for TCLP. Fluid #1 was prepared by adding 5.7 mL glacial HOAc to 500 mL of the appropriate water, then adding 64.3 mL of 1.0 N NaOH and diluting to a volume of 1 liter. Fluid #2 was prepared in the same way as fluid #1 but without NaOH. The pH of fluid #2 was 2.88±0.05. The fluid #1 was employed to extract acidic samples, and fluid #2 was for extraction of basic samples.

In the extraction process, 100 g dewatered solids was first transformed into the extractor vessel, and 2 liter of appropriate extraction fluid was slowly added. The extractor bottle was closed tightly (with or without Teflon tape), and secured it in rotary extractor device. The extraction device was then rotated at a speed of 30±2 rpm for 18±2 hours at ambient temperature (22±3 ^{O}C).

After extraction, the material was filtered to separate liquid and solids with 0.6 to 0.8 µm glass fiber filter. For final filtration, the glass fiber filter might be changed. Following the collection of the extract, the pH was recorded. The extract was analyzed or preserved properly. Metals aliquots were acidified with nitric acid to pH ± 2 and all other aliquots were stored under refrigeration (4°C) until analyzed. The TCLP extracts were then further prepared and analyzed in accordance with the appropriate analytical method.

3.3.3.2 Determination of Trace Metals in WTP Residuals by ICP-MS: A liquid extract collected from TCLP extraction, was analyzed using ICP-MS for metals. First, liquid was digested by acids using hot plate or microwave, then diluted, and injected into ICP-MS for analysis.

Acid digestion. Two kinds of acid digestion methods were used in this test. One was Nitric-Acid-Hydrochloric Digestion (NAHD) and another was Microwave-Acid Digestion (MAD). Comparison of the results from two methods showed statistically no difference. Therefore Microwave-Acid digestion method was adapted as the main digestion method in subsequent testing.

In Nitric Acid-Hydrochloric Acid Digestion (EPA 3005), 100 mL sample was transferred into 150-mL Griffin beaker, 3 mL concentrated HNO_3 was added four or five times and cooked on the hot plate until the digestion was complete. The sample was diluted and filtered to remove silicates and other insoluble material that could clog the nebulizer.

In Microwave Acid-Digestion test, samples were placed in a closed Teflon PEA vessel and digested with a microwave heating device. MDS-81D microwave instrument, Teflon PEA vessels (120 mL size) with pressure relief valve, digestion turntable, and capping station were used in this acid digestion. During digestion 50 mL of sample was transferred into a vessel, and 5 mL of nitric acid was added. The cap was tightened and the vessel was placed in the turnable and attached to a venting tube. The DMS-81D exhaust fan was set to the maximum speed. The instrument time and power were programmed to digestion. After digestion, the sample was allowed to cool to room temperature and then shaken well. Blanks (Type II water and reagents) were carried through the entire sample preparation and analytical process. Duplicate samples were used for analysis to determine precision.

The apparatus was thoroughly cleaned and prerinsed with dilute HNO_3 before filtering or centrifuging. The final volume of the liquid was adjusted to 100 mL with Type II water.

Instrument and sample analysis. ICP-MS was used to determine the concentration of metals in extracts prepared from acid digestion. In this analysis, liquid sample was introduced by pneumatic nebulization into a radio frequency plasma where energy transfer processes cause desolvcation, atomization, and ionization. The ions were extracted from the plasma through a differentially pumped vacuum interface and separated on the basis of their mass-to-charge ratio by a quadrupole mass spectrometer having a minimum resolution capability of 1 amu peak width at 5 percent peak height

An aliquot of reagent water was treated exactly as a sample including exposure to all labware, equipment, solvents reagents, and internal standards that were used with other samples. The library reagent blank (LRB) was used to determine if method analysis or other interference were present in the laboratory environment, the reagents or the apparatus. A volume of ASTM type I water acidified with the same acid matrix present in the calibration standards was utilized for this test. Pure analyze(s) was added to a solution in known amount(s) and used to measured the relative responses

Four or five concentrations of the standards solutions were prepared based on the concentrations of samples from the semi-quantitative procedure analysis. The concentrations of standards covered the concentrations of samples by dilution factor.

Sample data was reported in unit of µg/L for aqueous samples or mg/kg dry weight for solid samples. Reported values were subtracted from calibration blank..

3.3.3.3 Determination of Pesticides-Herbicides in WTP Residuals Using GC-ECD: EPA method 8080 was used to determine the concentration of various organochlorine pesticides and polychlorinated biphenyls (PCBs). This method provides gas chromatographic conditions for the detection of ppb levels of seven regulated organochlorine pesticides. Samples, blanks, and standards were analyzed by direct injection.

HP 5890 GC Workstation with ECD detector, capillary column (SPB™ -608 coated 0.25 µm film in a 30x0.25 mm I.D.), and data analysis system were used in the analysis.

The TCLP extracts can be held up to 7 days prior to preparative extraction (EPA 3510). The preparative extracts can be held up to 40 days before the determinative analysis (GC-ECD analysis) begins. Direct injection of the sample into GC-ECD system with a 20-ul syringe was used. The detection limit was also determined. The system was calibrated by direct calibration.

Banks and duplicates were analyzed to assess the precision of the environmental measurements. All blanks and samples extracts were spiked with 1 mg/l of surrogate standard. Matrix spike standards spike one out of analytical group (ten samples) TCLP sample extracts as well as blank extracts, with 0.2 ppm of linden, 0.2 ppm of heptachlor, 0.2 ppm of aldrin, 0.5 ppm of dieldrin, 0.5 ppm of endrin and 0.5 ppm of 4,4'-DDT. The recovery of these standards was determined and the matrix was evaluated.

3.3.3.4 Determination of BNA in WTP Residuals Using GC-MS: EPA method 8270 was used to determine the concentration of semivolatile organic compounds in extracts prepared from EPA 3510. The twelve regulated compounds can be classified as basic/neutral and acidic organics. The HP 5980 gas chromatography equipped with 30-m x .25-mm I.D., 1-µm film thickness silicon-coated fused-silica capillary column (Supelco PTE-5) was used for this analysis. HP 5988 mass spectrometer was used for compounds identification. It is capable of scanning from 35 to 500 amu in every 1 sec or less, using 70 volts electron energy in the electron impact ionization impact ionization. This MS and GC-MS interface met all criteria when the tuning standard, 50 ng decafluorotriphenylphosphine (DFTPP) was injected through the GC.

The TCLP extracts can be held up to 7 days prior to preparative extraction (separatory funnel liquid-liquid extraction). The preparative extracts can be held up to 40 days before the determinative analysis (GC-MS analysis) begins. The extracts were stored at 4 °C, protected from the light in screw-cap vials with unpierced Teflon-lined septa.

The 1-mL extract obtained from sample preparation was spiked with 20 µL of 2000 mg/mL of internal standards to result in 40 ppm concentrations for each internal standard before GC-MS analysis. When the response of any quantitation ion exceeded the initial calibration curve range of the GC-MS system, an extract dilution was made. The diluted extract was added and the required 40 ppm of each internal standard was maintained. The diluted extract was reanalyzed.

Direct injection of the sample into GC-MS system with a 10-µl syringe was used. The detection limit was determined and the system was calibrated directly.

3.3.3.5 Determination of VOCs in WTP Residuals Using GC-MS with Purge-Trap:

Procedure specified in EPA 8240 was followed to determine volatile organic compounds in WTP residuals. The volatile compounds are concentrated in a purge-and-trap instrument and analyzed by gas chromatography and mass spectrometer, which was used to provide both qualitative and quantitative information.

HP 5980 gas chromatography with HP624 column, HP 5988A mass spectrometer, HP Teknivent Vector II MS Data System, and Teckmar purge-and trap with autosampler were used for the analysis.

TCLP volatile mix standard, VOA internal standard mix, surrogate spike mix, matrix spike mix, and volatile system performance check compound mix (purchased from SUPELCO) were used for calibration standards and QA-QC.

After receiving the samples, the TCLP extraction was carried out within 24 hours. For TCLP testing of VOC's, the liquid was separated from the WTP residual and extracted in Zero Headspace extractor for 18 ± 1 hours. Liquid and solids were then separated in the Zero Headspace (Associated Design 3745-ZHE). The liquid extraction was kept in the wide-mouth amber glass jar and stored in refrigerator (4 °C) until volatile organic compounds were analyzed by GC-MS with Purge-Trap. Other TCLP extraction liquid was extracted by regular TCLP extractor for TCLP semivolatiles, metals, organochlorine pesticides and herbicides testing.

The addition of matrix and surrogate spikes were employed for each analyte if (i) recovery of the compound from the TCLP extract was not between 50 and 150%; or (ii) if the concentration of the constituent measured in the extract was within 20% of the appropriate regulatory threshold.

3.3.4 The Total Analysis for WTP Residuals

Total amount of metals, pesticides, herbicides, semivolatile, and volatile organic compounds in the WTP residuals were conducted in the study.

3.3.4.1 Total Metals: Microwave assisted acid digestion (EPA method 3051) provided for the acid extraction of metals in WTP residual. One gram of dry sample was transferred into a Teflon vessel and 20 mL of 1:1 HNO_3:H_2O solution was also added. The extract was filtered, diluted, and analyzed by an ICP-MS (EPA method 200.8).

3.3.4.2 Total Pesticides, Herbicides, and BNAs: Soxhlet extraction (EPA method 3540A) was used to extract nonvolatile and semi-volatile organic compounds from the solids. The Soxhlet extraction process ensures intimate contact of the sample matrix with the extraction solvent. Ten grams of WTP dewatered residual were mixed with anhydrous sodium sulfate, placed in an extraction thimble, and extracted using a methylene chloride or other specified solvent in a Soxhlet extractor. The extract was then concentrated into 10 mL. The extract, as necessary, was exchanged into a hexane for pesticide quality analysis. The pesticides and herbicides were analyzed by a GC/ECD (EPA method 8080 and 8150) and the BNAs was tested by a GC/MS (EPA method 8270A).

3.3.4.3 Total Volatile Organic Compounds for WTP Dewatered Residuals: EPA method 8240 was used to determine volatile organic compounds in a variety of solid waste matrices. The volatile compounds were injected into the gas chromatography by the purge-and-trap method. The components were separated via the gas chromatography and detected using a mass spectrometer, which is used to provide both qualitative and quantitative information.

WTP dewatered residual sample and 5 mL reagent water were placed into purging chamber. An inert gas (Nitrogen) was bubbled through the solution at ambient temperature, and the volatile components were efficiently transferred from the aqueous phase to the vapor phase. The vapor was swept through a sorbent column where the volatile components were trapped. After purging was completed, the sorbent column was heated and backflushed with inert gas to desorb the components onto a gas chromatographic column. The components were then detected with a mass spectrometer.

3.3.5 Dewatering Tests

Tests were conducted to determine the quantities of water removed from WTP residuals by different dewatering mechanisms. Several procedures were adapted in the dewatering tests. Paint filter test was first conducted on 100 g of samples to determine the CPSC value. They were then dewatered mechanically by subjecting them to a pressure of 60±2 psi for a period of 80 minutes. The samples were then air dried until a constant solids content was obtained. The residuals were heated to a temperature of 104°C. The solids contents of residuals were determined at each stage.

3.4 Tests Methods for Metal Leaching Modeling of WTP Residuals

3.4.1 Sample Preparation

In leachability study, four WTP dewatered residuals were tested. The symbols for the WTP residuals tested are shown in Table 3.4. These samples represent three major types of WTP residuals, namely, alum, iron, and lime residuals.

3.4.2 Leaching Test Procedures

Several factors which should be considered in a leaching test for the development of a model include

1. Physical and chemical properties of WTP residuals;
2. Type of leachant;
3. Waste sample preparation for testing; and
4. Method of mixing waste with leachant.

Leaching tests used can be either batch test or continuous flow column test (Ham, et al. 1979). Single extraction or multiple extractions are in batch test. In this study, single batch extraction was used to study the effect of pH, buffering capacity, and particle size on leachability and multiple batch extractions was used on metals leaching rate. Feasibility of using test was also conducted.

3.4.2.1 Single Batch Leaching Tests: TCLP equipment and testing procedure was modified in the batch leaching tests (Federal Register, 1991). The fresh residual sample, which equals to 10 gram dry weight, were placed in Extraction Wide-Mouth Teflon bottles and 500 mL of 0.04 M acetic acid was added. This acetic acid provided an acidity 2 meg/g and a pH 5. The liquid to solid ratio was 20:1. The extraction bottles were sealed with Teflon lined cap and were placed in a rotary extractor device. The extraction device was then rotated at speed of 30±2 rpm for 24 hours at ambient temperature (22±3°C). After extraction, the leachate was separated from the solid by press liquid through a 0.45 μm filter with nitrogen gas. Filtered leachate was analyzed for pH, alkalinity, conductivity, and primary metals. Deionized (DI) water, sulfuric acid, and sodium hydroxide solution were also used as extracting solutions for comparison with results of tests.

3.4.2.2 Multiple Batch Leaching Tests: The multiple batch leaching test was performed by extracting the same solid sample several times with fresh leachant. This test is used to evaluate the leaching rate of a metals from a solid. After each extraction, leachate was filtered through a o.45 μm filter, and fresh acid leachant (500 mL) was added to the extractors for further extractions. Each filtered leachate was analyzed for pH, alkalinity, conductivity, and primary metals.

Ideally, a multiple batch leaching test should be conducted until no material leaches out. This is impractical because it may take years to complete.

The multiple batch leaching tests were performed with two leachants. One was acetic acid and another was DI water. The acetic acid solution was chosen to simulate acidic conditions in a sanitary landfill during the stage of solid wastes decomposition

(Perket and Webster, 1981) and WTP monofill during precipitation and runoff. The distilled water was used as leaching background study.

3.4.2.3 Column Leaching Tests: Column tests, in which the solid sample is packed in a column and the leachant solution is passed through, is a closer approximation of the landfill condition than batch test. The column test is usually used to study the waste-leachate contact and the rate of leachate migration.

Column leaching tests was tried in this study. Fresh WTP residual samples were compacted in the columns and leachant was applied. After one week, there was no leachate coming out. This is because WTP dewatered residuals has very low permeability, generally in the range of 10^{-6} to 10^{-8} cm/sec. Experiments was stopped after 3 months since no data can be collected.

3.4.2.4 Leachate Analysis: All chemical tests were analyzed in accordance with standard methods and EPA method as mentioned in 3.3.

3.4.3 Particle Size Analysis

Particle size of WTP residuals was analyzed by MasterSizer X (Malver Instruments LTD). This instrument is made of three parts, He-Ne laser, Sample System, and Detector. Particles dispersed in liquid suspension were exposed to the laser beam and the laser light scattered by the particles is related to the particle size. The detector gathers the scattered light over a range of solid angles of scatter to give the size distribution of the particles.

The derived diameter of particle is expressed in terms of equivalent spheres (Diffraction Reference Manual, 1992). The results of particle size distributions in the form of a surface area, length, or number distribution are as follows:

1) The diameter of 50% of the number of particles
2) The mean diameter over the volume distribution:

$$D(4,3)=\frac{\sum d_i v_i}{\sum v_i}$$

where d_i is the mean diameter of size band i, i=1 to 32;

v_i is the mean volume of size band i, i=1 to 32;

3) The mean diameter over the surface distribution (the Sauter Mean Diameter):

$$D(3,2)=\frac{\sum d_i s_i}{\sum s_i}$$

4) The specific surface area:

$$S_a=\frac{TotalArea}{TotalVolume}=\frac{6\sum \frac{v_i}{d_i}}{\sum v_i}$$

Table 3.1 Experimental methods and testing instruments employed

Test Parameter	Method	Instrument
TCLP	EPA 1311	Associated Design and Manufacturing Company, MODEL 3740-6-BRE
Pesticides/Herbicides	EPA 3510/8080	HP 5890 Workstation GC/ECD
BNA	EPA 3510/8270	HP 5890 GC/MS & Varian Star 3400/Saturn II
Metals	EPA 3510/200.8	VG Plasma Quad ICP/MS
VOCs	EPA 3510/8240	HP 5890 GC/MS with Teckmar purge & trap
Biodegradation Test		
Biogas Composition		HP 5890 GC/TCD
Microorganisms	EPA 1000	PZEISS Microscopy
Cation Exchange	EPA 9080	Perkin-Elmer AA-305B
Physical Examination	Standard Methods 200	
Paint Filter Test	EPA 9095	USEPA1996b

Table 3.2 Symbols for the water treatment plant residuals tested

Symbols	Description
ELD	Dewatered residual from drying bed in Ellwood City WTP, PA
FLR	Raw residual from backwashing in Bradenton WTP, FL
FLD	Dewatered residual from drying bed in Bradenton WTP, FL
FLDM	Dewatered residual from landfill in Bradenton WTP, FL
HWR	Raw residual from lagoon in Haworth WTP, NJ
HWD	Dewatered residual from drying bed in Haworth WTP, NJ
JCR	Raw residual from thickener in Jersey City WTP, NJ
JCC	Conditioned residual with lime before press filter in Jersey City WTP, NJ
JCD	Dewatered residual cake after press filter in Jersey City WTP, NJ
MQD	Dewatered residual from drying bed in Wanasquan WTP, NJ
MWD	Dewatered residual cake after centrifuge in Minneapolis Water Works, MN
NCD	Dewatered residual from drying bed in New Castle WTP, PA
PVR	Raw residual from thickener in Little Fall WTP, NJ
PVC	Conditioned residual with lime before press filter in Little Fall WTP, NJ
PVD	Dewatered residual cake after press filter in Little Fall WTP, NJ
RWA	Dewatered residual from drying bed (unfrozen) in West River WTP, CT
RWAF	Dewatered residual from drying bed (frozen) in West River WTP, CT
SLD	Dewatered residual from landfill in South County WTP, MO
WQD	Dewatered residual from air drying bed in Wanaque WTP, NJ
WQR	Raw residual from backwash in Wanaque WTP, NJ

Table 3.3 Information summary of water treatment facilities and residual samples

Sample	Residual Type	Name of the Facility	Water Source	
			Type	Name
JCD	Lime	Jersey City Water Treatment Plant at Boonton, New Jersey	Reservoir	Rockaway River and Boonton Reservoir
PVD	Lime	Little Falls Treatment Plant at Clifton, New Jersey	River	Passaic River
WQD	Ferric	Wanaque Water Treatment Plant at Wanaque, New Jersey	Reservoir	Wanaque Reservoir
MWD	Lime/Alum	Minneapolis Water Works, Minneapolis, Minnesota	River	Mississippi River
MQD	Alum	Manasquan Water Treatment Plant in Monmouth County, New Jersey	River/Reservoir	Manasquan River and Manasquan Reservoir
HWD	Alum	Haworth Water Treatment Plant at Harrington Park, New Jersey	Reservoir	Hackensack River , stored in four reservoirs
ELD	Lime	Ellwood City Treatment Plant in Ellwood City, Pennsylvania	River	Slippery Rock Creek
RWA/ RWAF	Ferric	West River Water Treatment Plant in Woodbridge, Connecticut	Reservoir	Glen, Watrous, and Dawson reservoirs
FLD/ FLDM	Alum	Bradenton Water Treatment Plant in City of Bradenton, Florida	Reservoir	Lake Manatee
SLD/ SLDF	Lime	South County Plant in St. Louis, Missouri	River	Meramec River

(continued)

Table 3.3 (continued)

	Impurities in Water Sources (Yearly Average Value) (Yearly Range)							
Sample	Turbidity (NTU)	Color (PCU)	Taste/Odor (Threshold Odor No.)	Iron (ppm)	Manganese (ppm)	Hardness ($CaCO_3$ mg/L)	Trihalomethane (ppb)	TOC (mg/L)
JCD	0.5-6	10-30	2M/3M	0.1	<0.02	40-70	Not Detected	3-4
PVD	5-75	25-100		1.3	0.11			
WQD	0.75-2.5	0-20		0.05-0.16	0.01-0.06	170-230		
MWD		10-100	<10	0.05-0.15		170-230		
MQD	6-300*	5-500*		0.7-5.0*	0-0.03*	60*	100-400* **	
	2-4**	60**		1.0**	<0.13**	30**		
HWD	3	25				120		
ELD	2-100		2	0.5	0.06	150	8.5	
RWA/RWAF	1.2	26	5	0.15	0.08	25	0-300	3.2
FLD/FLDM	1.5-25	176		0.25		71.9	400-600	15-25
		100-400				40-110		
SLD/ SLDF	21	12		0.169	0.013	167		

(continued)

Table 3.3 (continued)

Water Treatment Process and Chemicals Added(Yearly Average Value)					
Sample	Main Water Treatment Processes	Lime (ppm)	Alum (ppm)	Ferric Chloride (ppm)	Coagulant Aid and Others
JCD	Rapid mix, Flocculation, sedimentation, filtration, chlorination	1.0 - 3.0 ppm	3.0 - 6.0 ppm		Cationic polymer, (0.5 - 1.0)
PVD	Prechlorination, flocculation, sedimentation, multiple media filtration, chlorination		as coagulant		Polymer activated carbon
WQD	Pretreatment, coagulation, sedimentation, filtration, chlorination		Coagulant 10-12 ppm		Polymer, carbon, KMnO4
MWD	Lime softening, alum coagulation, carbon absorption, coagulation with ferric chloride, filtration, chlorination	170 ppm as $CaCO_3$	Coagulant 20 ppm	4 ppm	Powder activated carbon
MQD	Sedimentation, clarification, mixed media filtration, carbon adsorption, chlorination (NaOCl)		Coagulant		Polymer, KMnO4, GAC
HWD	Ozone contactor, flotation/skimmer, media filtration, disinfection		5 ppm		Cationic polymer
ELD	Presettling, coagulation, sedimentation, filtration, chloridation	Coagulant		20 ppm	PAC
RWA/RWAF	Oxidation, rapid mixing, filtration, chlorination			7.2 ppm	Cationic polymer
FLD/FLDM	Activated carbon addition, flocculation (alum, lime and polymer), sedimentation, postmixing (chlorine, ammonia and lime), mixed media filtration and chlorination		75 ppm		
SLD/SLDF	Softening, sedimentation, dual medial filtration, disinfection	94 ppm		12.8 ppm(sulfate)	

(continued)

Table 3.3 (continued)

Sample	Dewatering Process (Solids Content)	Conditioning Agent Added	Aging period for Sample Tested	Properties of Sample Tested		
				pH	Solids Content	Water Content
JCD	Thickener (1.5-2%), filter press (30-40%)	Lime (59%)		12.0	23.3%	329.2%
PVD	Thickener, Filter press (27-30%)	Lime (15%), (polyelectrolyte-occasionally)		12.0	26.2%	281.7%
WQD	Thickener, belt filter press (14%), Lagoon (0.5-1.5%), drying bed			6.5	5.4%	549.4%
MWD	Gravity thickner (1-2%), centrifuge (55-60%)			11.0	69.2%	44.5%
MQD	Lagoon, drying bed (30%)		Twelve months	7.8	32.6%	206.7%
HWD	Lagoon, drying bed			6.8	59.7%	67.5%
ELD	Lagoon, drying bed			6.2	39.9%	150.6%
RWA	Lagoon, drying bed		3.5 months	5.3	18.1%	452.5%
FLDM	Lagoon, drying bed			6.4	57.1%	75.1%
SLD	Lagoon		Twelve months	9.1	72.2%	38.5%
SLDF	Lagoon			9.5	38.2%	161.8%

(continued)

Table 3.3 (continued)

Sample	Location of Sample Collection	Description of Residual Sample Used for Geotechnical Testing	Sample Collection and Delivery
JCD	Outlet of dewatering machine	Cake form (about 2cm thick), composed of one inner black layer and two outer yellow layers, foul odor	Collected by NJIT research team
PVD	Outlet of filter press	Cake form (about 5cm thick), grey color, with strong foul odor	Collected by NJIT research team
WQD	Drying bed	Paste form, brown and black color, foul odor	Collected by NJIT research team
MWD	Outlet of dewatering machine	Paste form, gray color	Sent by the facility, standing water was observed on top of the container when sample was received
MQD	Drying pile (six month aged)	Lump form, brown, yellow and black color, hard and brittle	Collected by NJIT research team
HWD	Drying bed	Lump form, dark grey color	Collected by NJIT research team
ELD	Drying bed	Lump form, black, brown, and grey color, soft	Sent by the facility
RWA	Drying bed (unfrozen)	Paste form, black color, with foul odor	Sent by the facility, standing water was observed on top of the container when sample was received
FLDM	Drying bed prior to landfill	In pieces, coal black color, dry, hard and brittle. Res	Sent by the facility
SLD	Landfill (one year aged)	Paste form, greenish grey color	Sent by the facility
SLDF	Lagoon (fresh)	Mud form, greenish grey color	Sent by the facility, standing water was observed on top of the container when sample was received

Note: This information is based on data provided by individual treatment plants and may not be complete

* Manasquuan River; ** Manasquan Reservoir

Table 3.4 Symbols for the water treatment plant residuals tested in leaching modeling

Symbols	Description
ELD	Dewatered residual from drying bed in Ellwood City WTP, PA
PVD	Dewatered residual cake after press filter in Little Fall WTP, NJ
RWA	Dewatered residual from drying bed (unfrozen) in West River WTP, CT
WQD	Dewatered residual from air drying bed in Wanaque WTP, NJ

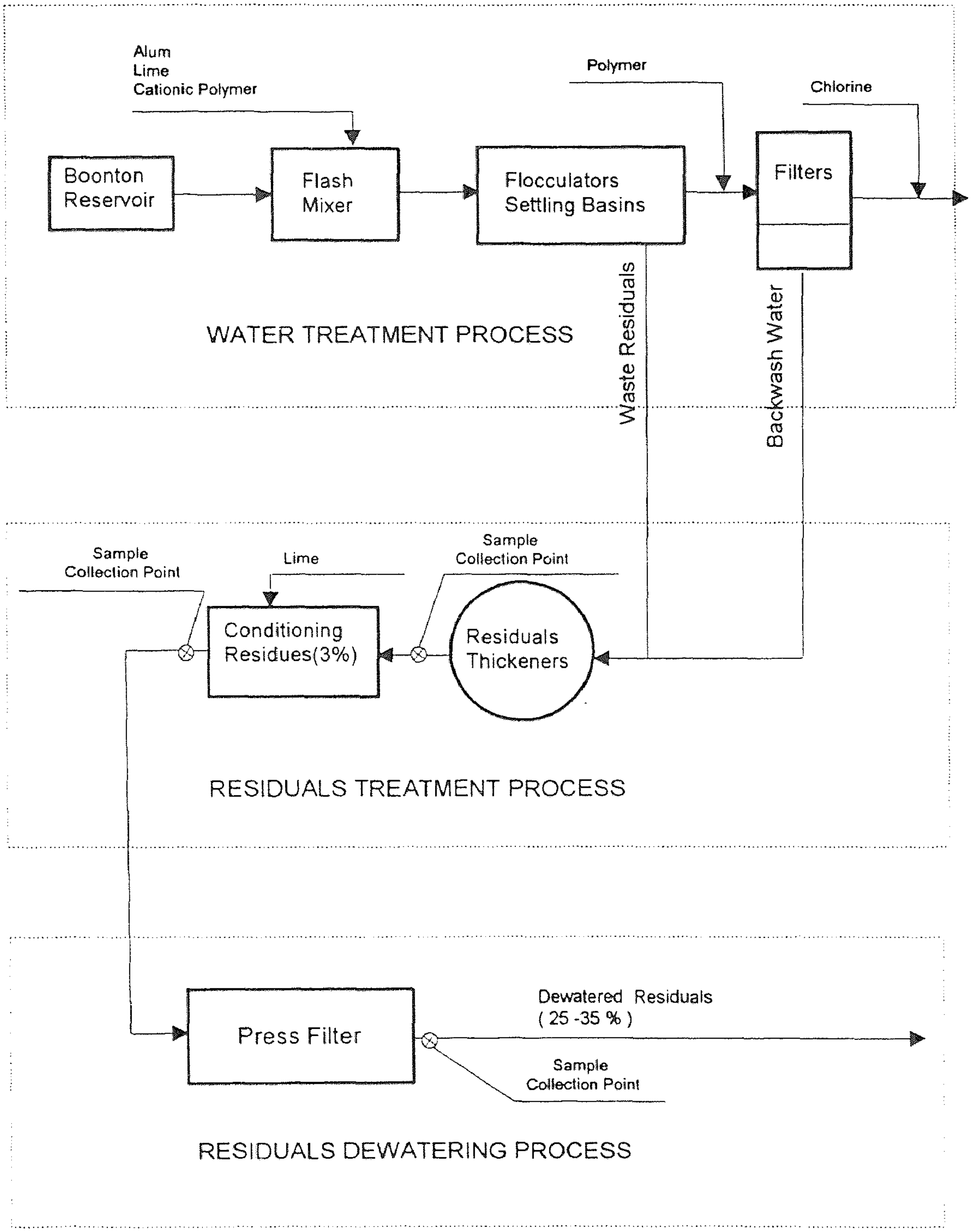


Figure 3.1 Flow diagram of Jersey City Water Treatment Plant, NJ

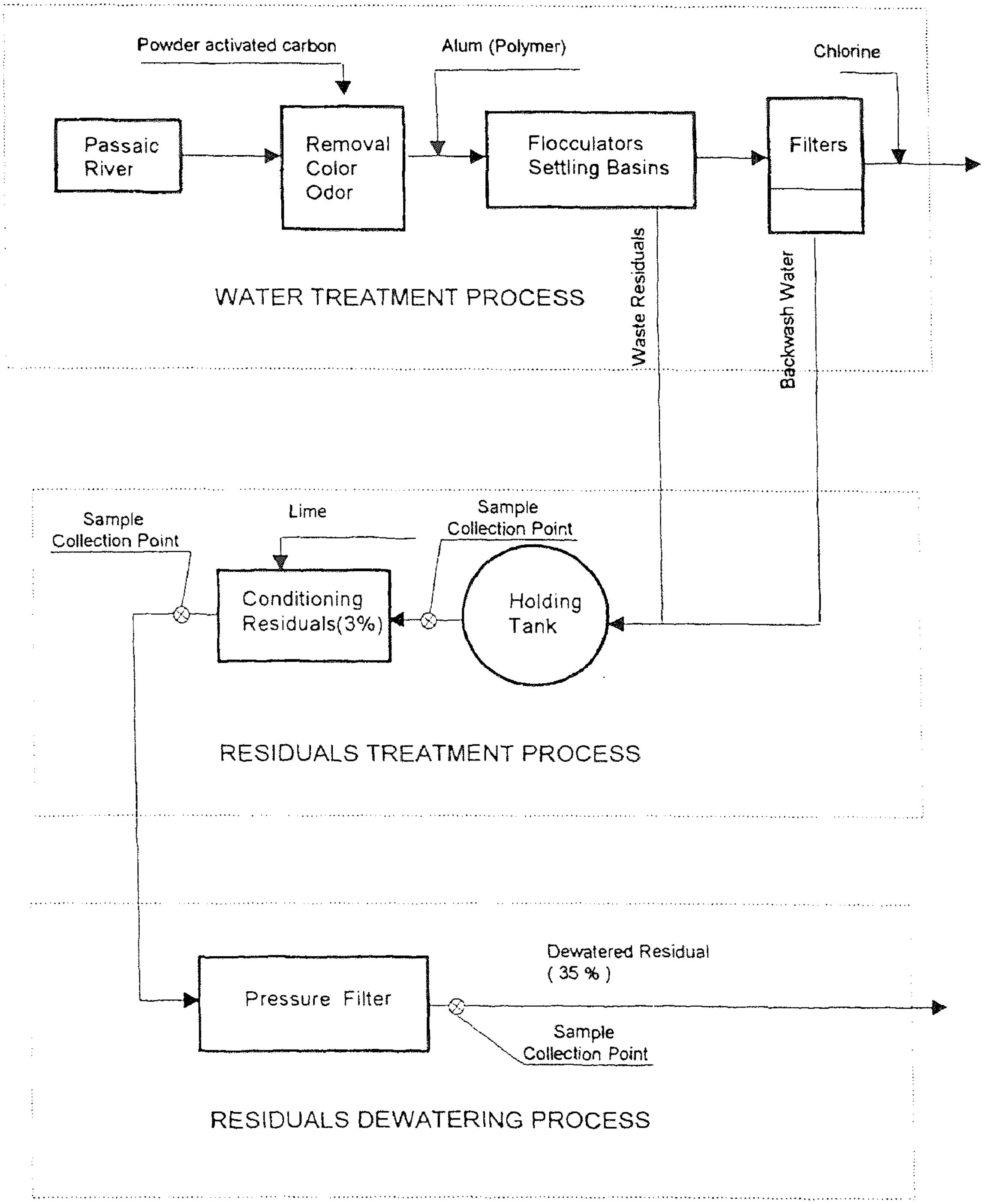


Figure 3.2 Flow diagram of Passaic Valley Water Treatment Plant, Little Fall, NJ

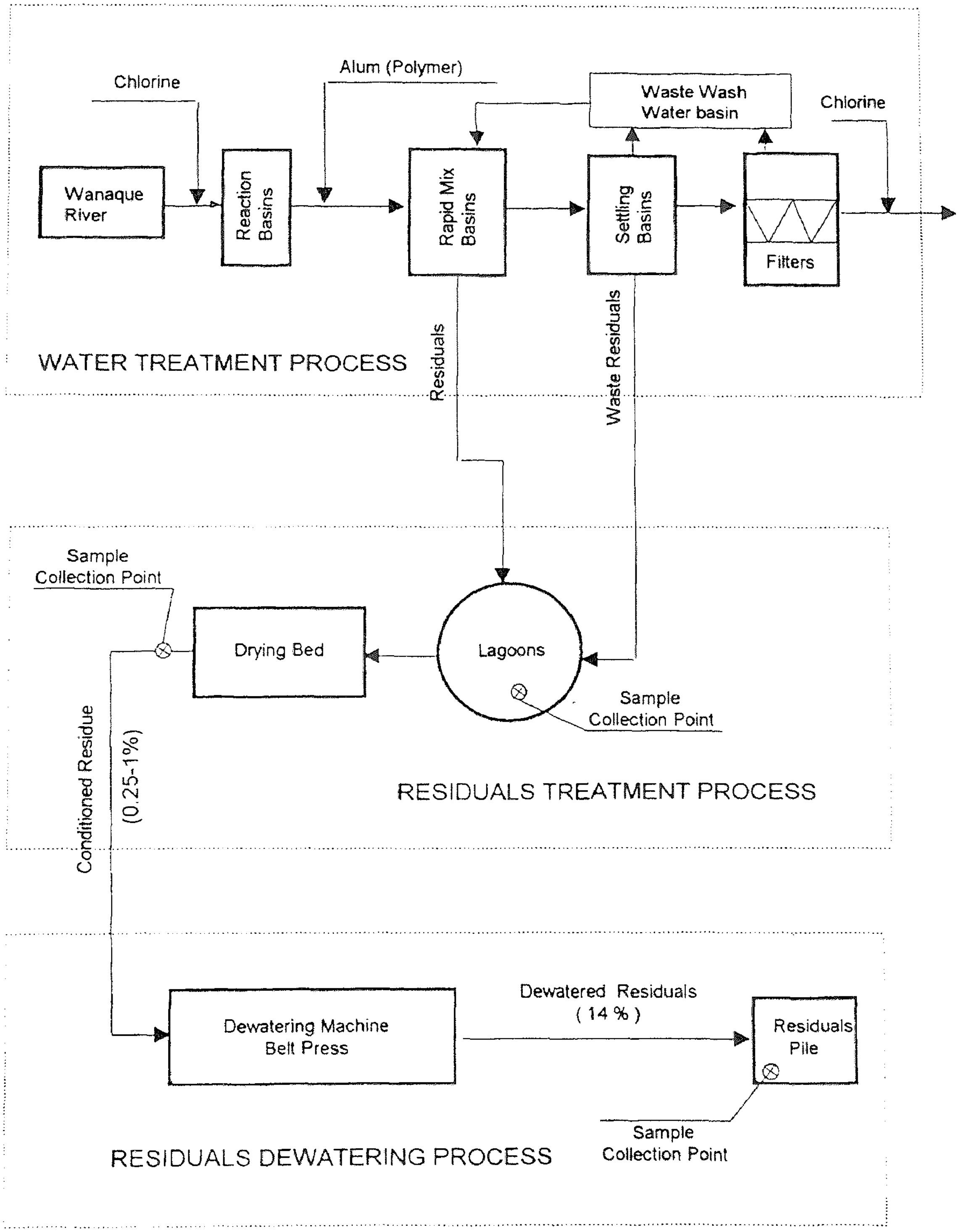


Figure 3.3 Flow diagram of water treatment plant of North Jersey District Water Supply Commission in Wanaque, NJ

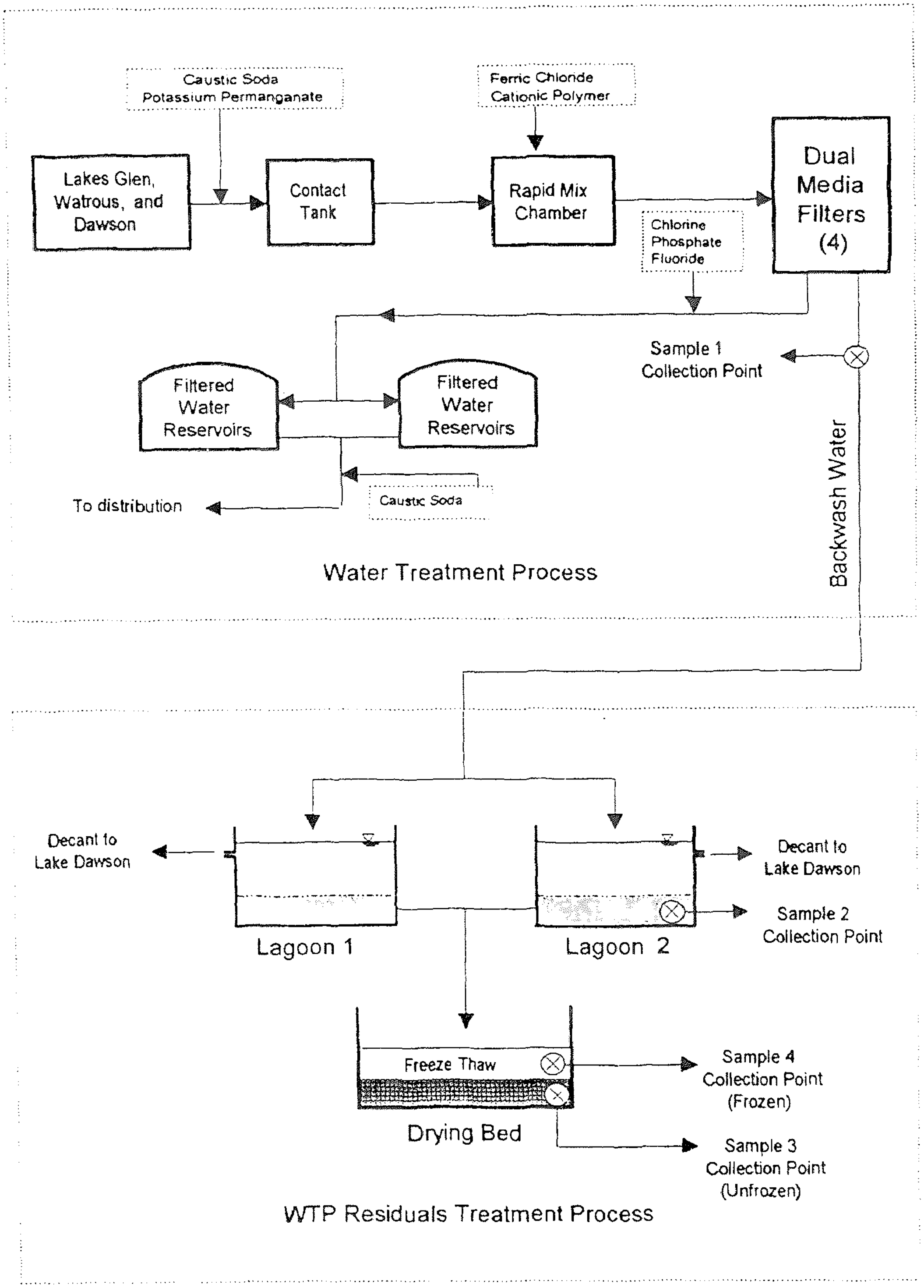


Figure 3.4 Flow diagram of West River Water Treatment Plant in Woodbridge, CT

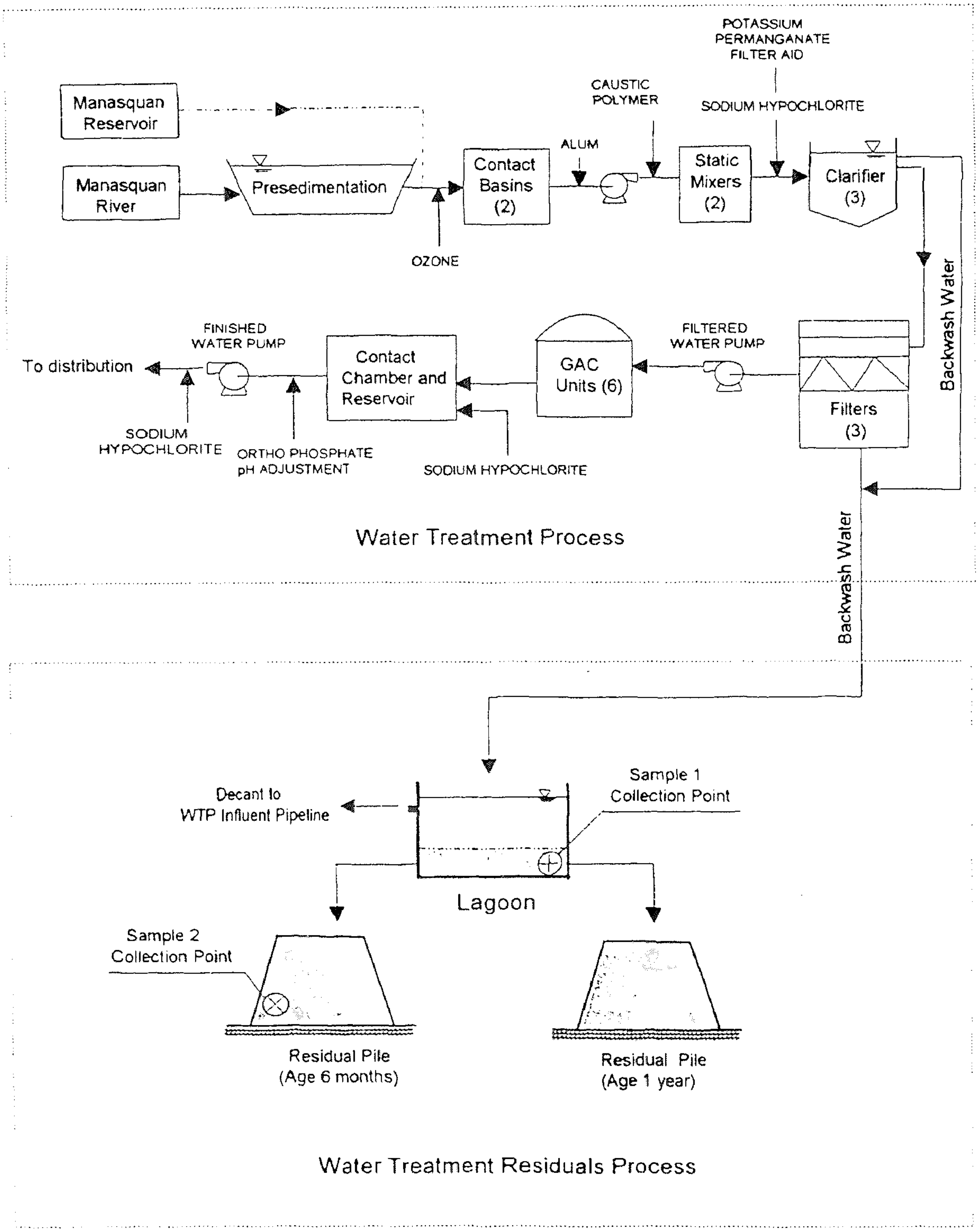


Figure 3.5 Flow diagram of Manasquan Water Treatment Plant in Monmouth County, NJ

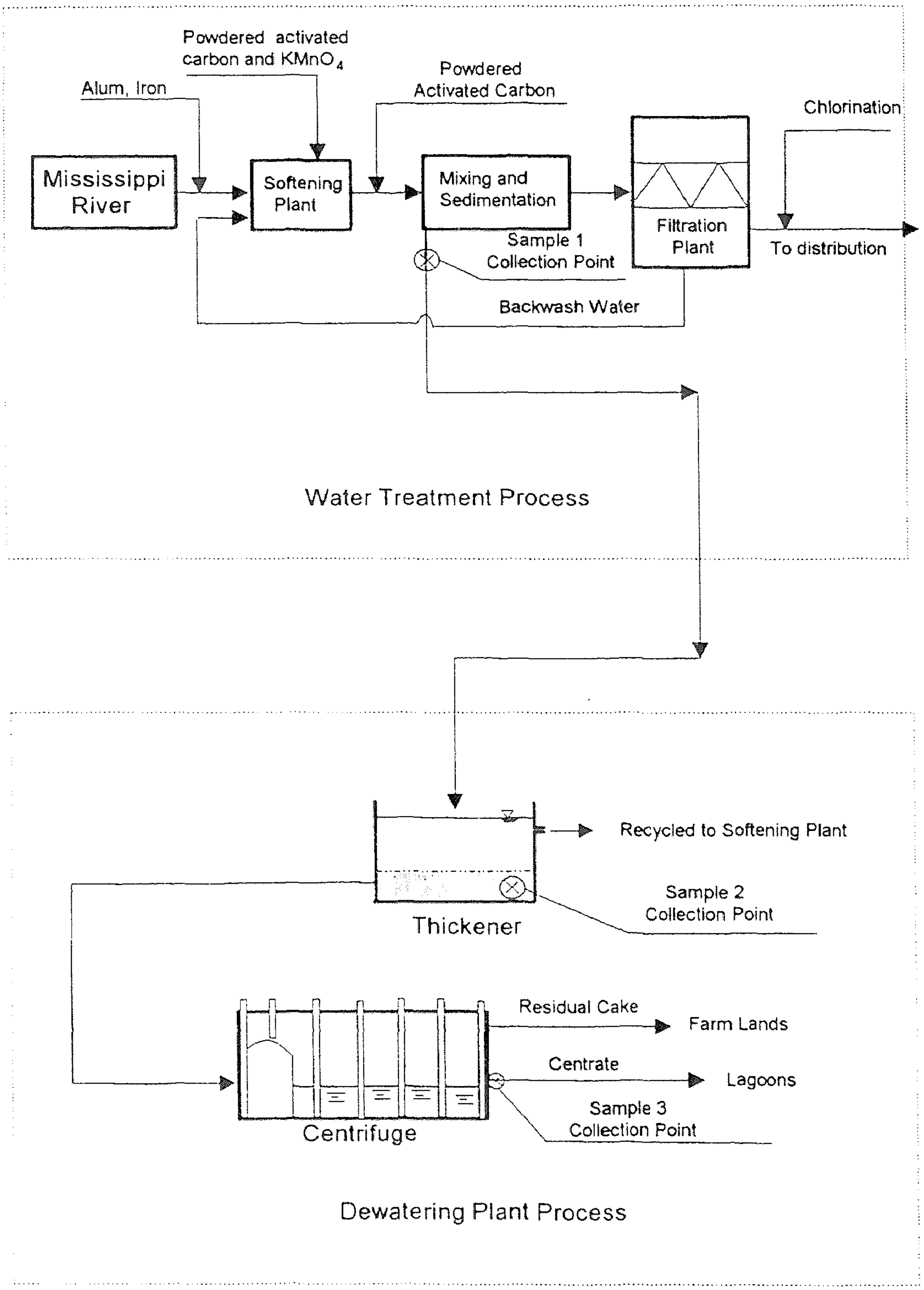


Figure 3.6 Flow diagram of Minneapolis Water Works Plant in Minneapolis, MN

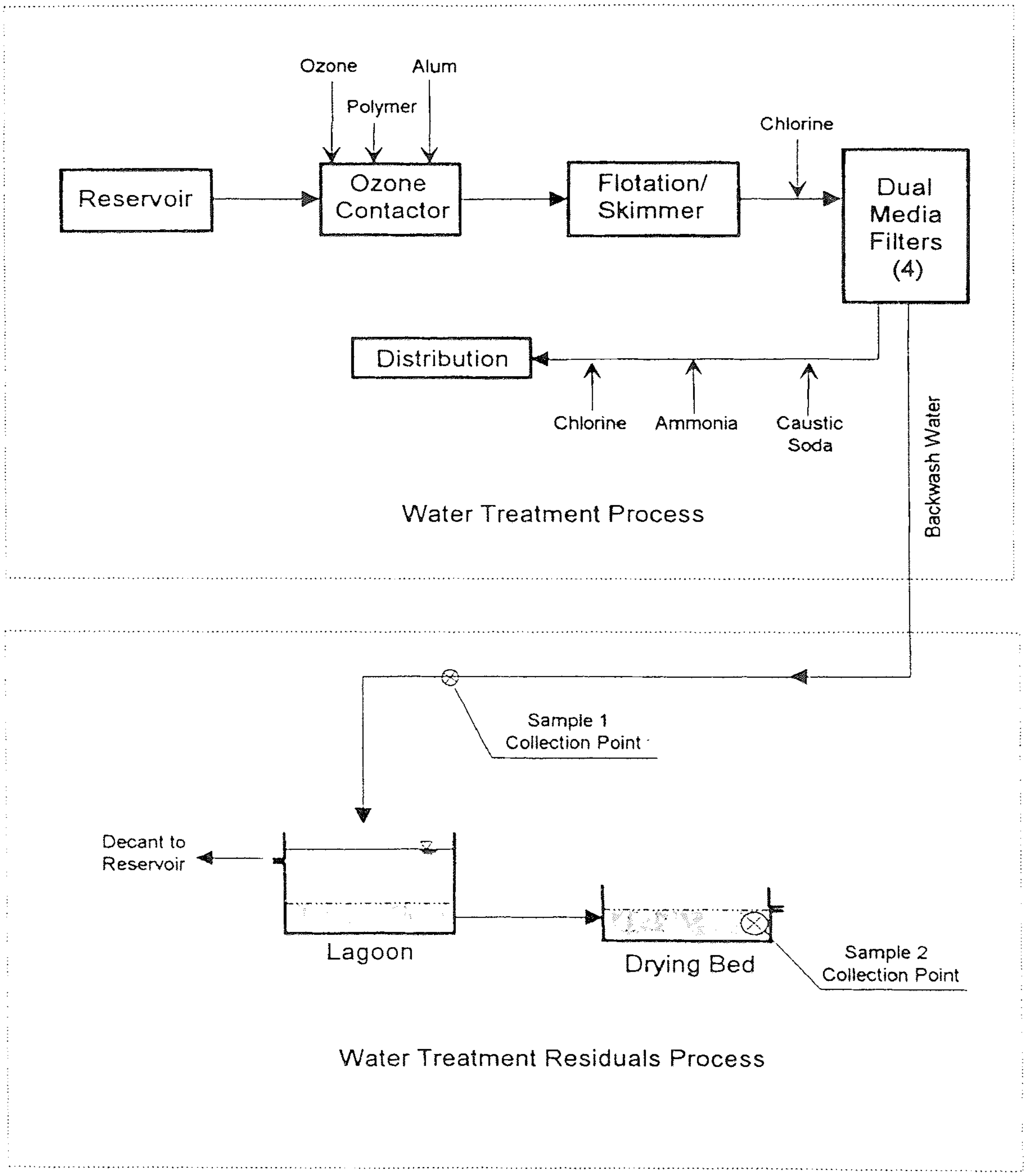


Figure 3.7 Flow diagram of Haworth Water Treatment Plant at Harrington Park, NJ

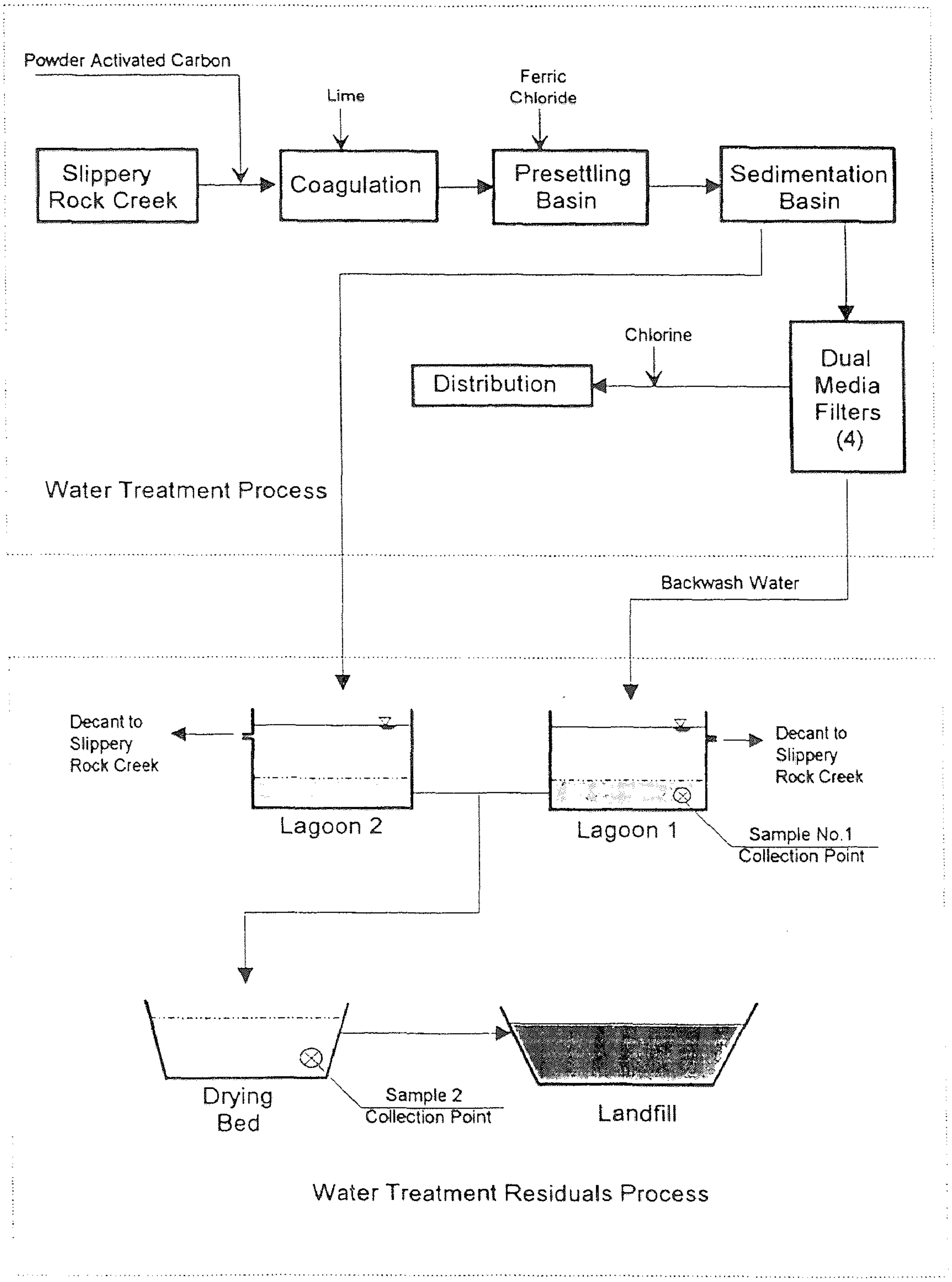


Figure 3.8 Flow diagram of Ellwood City Treatment Plant of Pennsylvania America Water Company Western Division in Ellwood City, PA

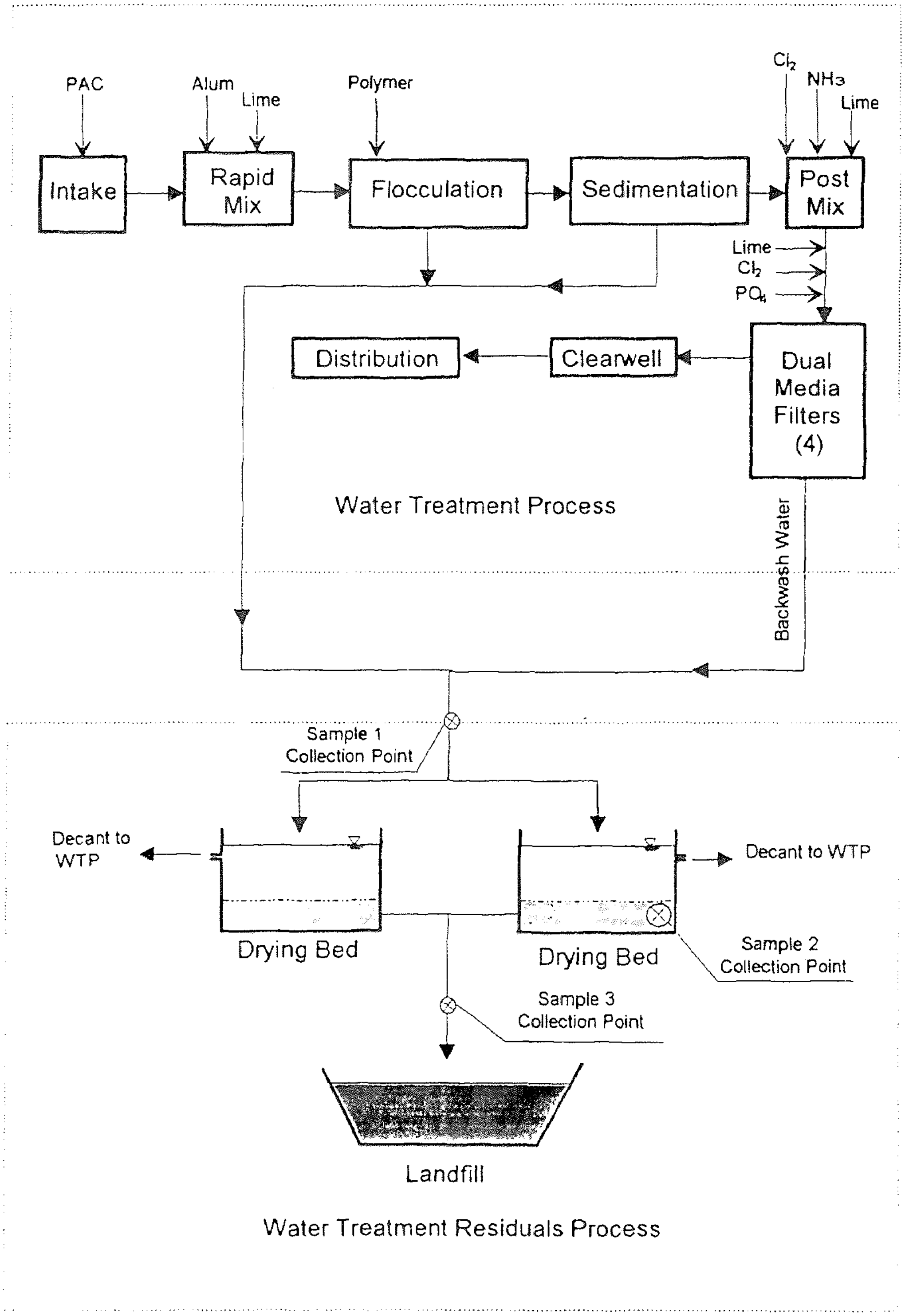


Figure 3.9 Flow diagram of Bradenton Water Treatment Plant of Manatee County Public Works Department, FL

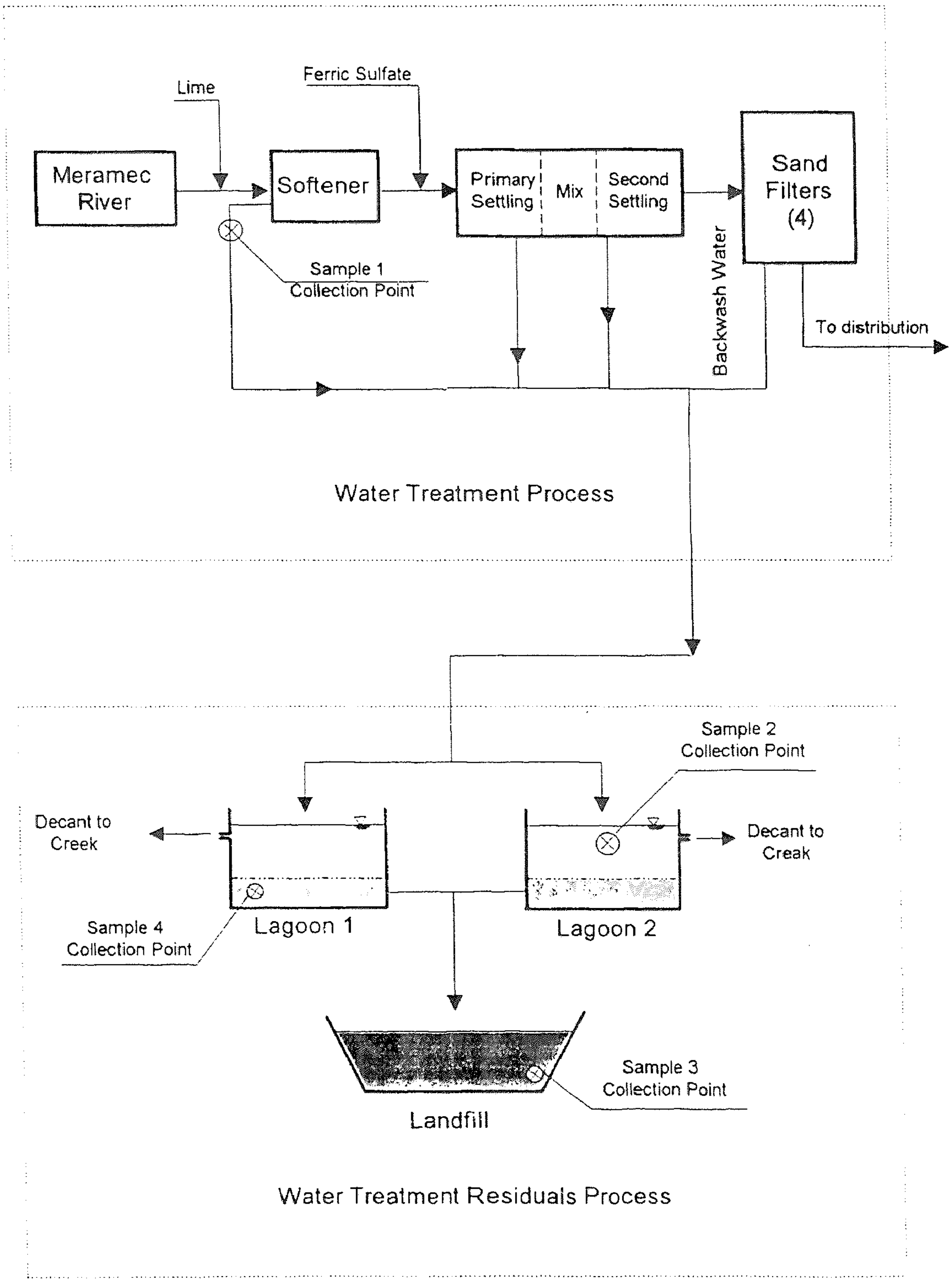


Figure 3.10 Flow diagram of South County Plant, St. Louis County Water Company in St. Louis, MO

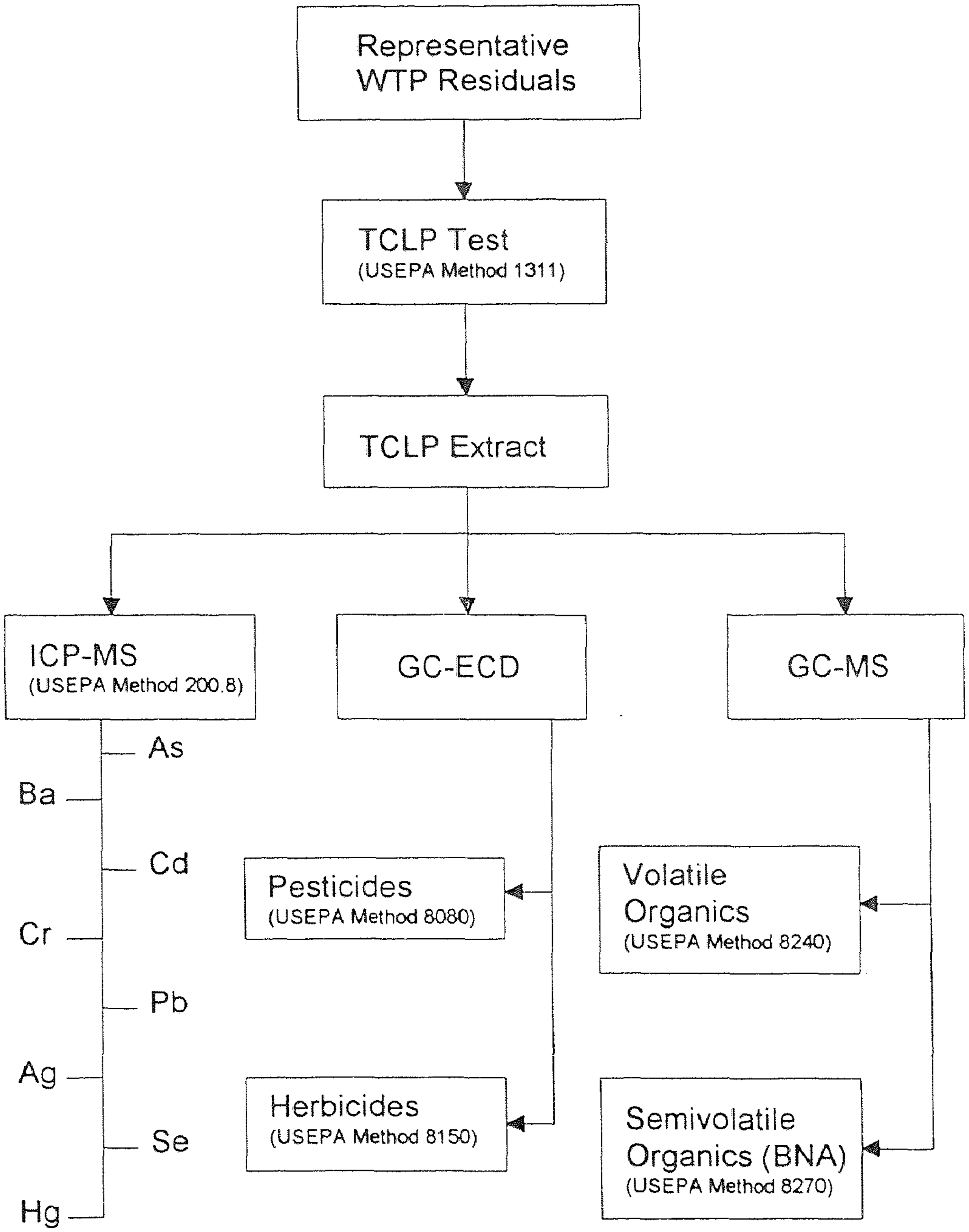


Figure 3.11 Flowchart of TCLP Tests for WTP Residuals

CHAPTER 4

ENVIRONMENTAL CHARACTERIZATION OF WTP RESIDUALS

4.1 Introduction

Environmental characteristics of WTP residuals are divided into four categories; general physical and chemical properties, biodegradation, TCLP test, and dewatering properties. For convenience of discussion, symbols are assigned to each of the samples collected. These symbols are listed in Table 3.2 and will be used throughout this Chapter. The samples include raw residual samples from gravity thickeners or lagoons, conditioned residual samples collected before entering dewatering machines, and dewatered residual samples.

4.2 General Physical and Chemical Properties

General physical and chemical properties of WTP residuals investigated for this study are paint filter test, solids content, pH, and metals composition.

4.2.1 Paint Filter Test

Paint filter liquid tests were conducted on all WTP residuals collected from the ten water treatment plants. As shown in Table 4.1, there was no liquid passing through the filter for any dewatered WTP residuals due to their high solids concentrations (14.8-82.0%). For the raw residuals obtained from backwash water in filters, sedimentation tanks, or lagoons, and the conditioned residuals obtained from chemical mixing chambers or thickeners, some liquid did pass through the filter for the solids concentration in the range of 0.38-4.08%.

4.2.2 Solids Content

Solids contained in WTP residuals include fixed solids and volatile solids. The volatile solids represent the organic contents. The fixed solids represent the concentrations of inorganics, such as clay, sand, silica in dewatered residual, and the chemicals present in the coagulants and the conditioning agents used in the water treatment processes. Solids contents determined for the dewatered WTP residuals obtained from ten water treatment plants are shown in Table 4.2.

The organics and inorganics contained in the WTP residuals were related to the water source and chemicals used in water treatment and dewatering processes. Tables 3.3 and 4.2 show that residuals obtained from treating river water contained low concentration of organics and high concentration of inorganics, for example, samples SLD, PVD, and MWD. On the other hand, WTPs treating water from reservoirs or lakes produced residuals having high organic contents and low inorganic contents (i.e. samples FLDM, RWAF, JCD, and WQD). Table 4.2 shows that the former residual samples had volatile solids content from 3.55% to 17.37%, and the volatile solids content of the latter ranged from 14.33% to 63.41%.

Organic content is one of main factors influencing the residual characteristics. As shown in Table 4.7, the volatile contents of ten samples were from 3.55% to 61.44%, the cation exchange capacities (CEC) were 21.96 to 134.8 (meq/100g). These results showed that high content of organics would increase the cation exchange capacity of residual by forming complex ions, which can retain the metals in the residual. High concentration of organics will also induce biodegradation to occur in the monofill, change structure of materials, and make the residual difficult to handle during placement, which will be further discussed in Chapter 5.

Chemical addition during the dewatering process and the method of dewatering also affected the composition of WTP residual. Table 4.3 presents the solids content of WTP residuals with samples collected from different process locations in five water treatment plants. Settled residual, thickened residual, and dewatered residual (filter press) samples were collected from two plants, PVD and JCD, where lime was used as conditioning reagent. Residuals from lagoons and drying beds were taken from three other plants, WQD, HWD, and FLD, where no chemicals were added. The samples PVR and PVC, which were from sedimentation tank and gravity thickener of Passaic Valley water treatment plant, had volatile solids of 25.49% and 20.92%, respectively, and fixed solids of 74.51% and 79.08%, which were in the same range. After lime was added as conditioning reagent, the volatile solids content decreased to 9.47% and fixed solids content increased to 90.53%. Three treatment plants used lagoons and drying beds in this group. By comparing the results of WQR and WQD, HWR and HWD, FLR and FLD, it can be seen that the percentage of volatile solids contents are all in the same range since chemicals were not added. Based on the results of the samples from Wanaque (WQR and WQD) and Haworth (HWR and HWD) treatment plants, the percentage of volatile solids contents of samples from the drying beds were found to be less than those from the lagoons. The reductions of organic matter in those residuals were probably caused by biodegradation in the lagoons during the six month stored period.

4.2.3 pH

The pH of the WTP residuals is an important factor to be considered in their disposal. It can affect leaching metals, biodegradation of organic matter, and growing vegetation if land application is considered. High pH will favor the immobilization of metals and inhibition of biodegradation (Bohn et al. 1985, USEPA 1986). If the pH is acidic (less than 5.0), metals may leach out from the residual and cause undesirable effects. Biodegradation occurs at neutral pH if other favorable environments (nutrients, temperature, and water) exist. It has been reported that if the pH of WTP residual is much greater than 7.0, the availability of some plant nutrients will be reduced.

The pH of the residuals obtained from ten water treatment plants is presented in Table 4.4. Generally, the pH of raw WTP residuals was in the range of 6.6-8.2, depending on the types of coagulants used. The pH of dewatered residuals was related to the type and amount of chemicals used during water treatment process. The lime residual had high pH ranging from 7.46 to 11.15. In places where large quantities of lime are added as conditioning agent during mechanical dewatering, such as the case in Jersey City (JCD) and Passaic Valley water treatment plants (PVD), the pH values increased after conditioning (pH = 9.94-11.82). The pH of ferric and alum residuals were from 6.25 to 7.81, as observed from the test results from samples of other treatment plants.

An experiment was performed to determine the resistance to change in the pH of various types of WTP residuals. Ten grams of dried WTP residual were added to 200 mL extraction fluid equal to 20 times its weight and placing the solution in a TCLP extraction vessel. The six different initial pH values of the extracting fluid, adjusted by sulfuric acid or sodium hydroxide, were 1.28, 2.01, 3.66, 5.55, 8.01, and 9.62, respectively. After 18 hours of mixing and extraction, the liquid and solids were separated and the pH of the liquid was measured. The results are shown in Figures 4.1 to 4.4.

The initial residual pH was 6.55 and the resulting pH was in the range of 4 to 7. In test 4, the initial pH of fluid was 3.66, which is similar to acid rain conditions. After reacting with alum residual, the pH of the solution became 6.97. In Test 1, the initial pH was 9.62 and it dropped to 6.99 after experiment. These results indicate that the residual has some pH buffering capacity to resist the change of pH in either acidic or basic conditions.

Variation of pH for a ferric residual (RWA) is shown in Figure 4.2. The initial pH of residual was 7.15. When the testing with extracting fluid of pH 3.66 to 9.62, the resulting pH was from 6.08 to 6.47. However, if the pH values of extracting solution were 2.01 and 1.28, the resulting pH values of the residuals dropped to 4.57 and 2.37 respectively. These indicate that ferric residual has the capacity of resisting pH change for pH values from 9.62 to 3.66. If pH was decreased to an extreme acidic condition, such as 2.01 and 1.28, the residual pH buffering capacity was not large enough to maintain the original pH of residuals.

Test results of PVD lime residual are shown in Figure 4.3. Initial pH of the residual was 11.15. The final pHs of residual were from 11.58 to 9.98 when tested with extraction fluids of pH 9.62 to 1.28. These results shows that the lime residual can maintain a high pH under all testing conditions.

An alum/lime residual (SLD) was also tested and the results are shown in Figure 4.4. The initial pH of residual was 9.15. The pH values of testing solution of residual were from 9.62 to 1.28 and the final residual pH values were 8.98 to 7.22. The results indicate that the lime/alum residual also had the capacity to resist pH change even in the extreme acidic condition, such as the one with an initial pH of 1.28.

Buffer capacity (β), is defined as the number of moles of strong acid or strong base required to change the pH by one pH unit. Assume the carbonic acid system is responsible the major buffering system, a mathematical expression can be used (Benefield et al., 1982):

$$\beta = 2.3\left[\frac{\alpha_1\left([Alk]_e - [OH^-] + [H^+]\right)\left([H^+] + \frac{K_1K_2}{[H^+]} + 4K_2\right)}{K_1\left(1 + \frac{2K_2}{[H^+]}\right)} + [H^+] + [OH^-]\right] \quad (1)$$

Where $\alpha_1 = \frac{K_1}{\left[H^+ + K_1 + \frac{K_1K_2}{[H^+]}\right]}$ (2)

β = buffer intensity (equivalents required per unit pH change per liter of solution)

$[OH^-]$ = hydroxyl ion concentration (moles/l)

$[H^+]$ = hydrogen ion concentration (moles/l)

$[Alk]_c$ = total alkalinity (eq/g)

K_1, K_2 = the equilibrium constants

Based on this equation, the buffer capacities of three WTP residuals were calculated as shown in Table 4.5. Lime residuals, such as JCD and PVD with high alkalinity and pH, had high buffer capacities (79.65 and 13.66×10^{-4} eq/g) and WQD with low alkalinity (0.41 meq/g) and pH (6.6) had low buffer capacity (5.67×10^{-4} eq/g).

Figure 4.5 illustrates the variation of β with respect to pH for a PVD sample system (Alkalinity = 5.88×10^{-3} eq/g, pH=11.15, and temperature (°C) = 24.5) as depicted by Equation 4-1. This shows that the maximum buffer capacity occurs around the point where the pH equals to 9.5, which is closed to pK_2 (10.34). The minimum buffer capacity occurs at pH values of 4.5 and 12, which are close to the first and third equivalence points. These are similar to what was observed in aquatic systems (Bennefield et al., 1982).

This equation can be helpful to estimate the pH change when a known concentration of acid is added to a WTP residuals.

The results of this experiment show that all WTP residuals have the buffering capacity for both acidic and basic conditions. The buffering capacity would affect the amount of metals being leached out depending on the initial pH and alkalinity of the residual. The pH would remain in neutral for alum and ferric residuals even when an acid is present, whereas the pH would remain to a value of above 8 for lime and alum/lime residuals under the same condition. The lime residual had high pH buffering due the presence of lime. This experiment also confirms the study reported by Cornwell et al. (1992), which stated that the buffering capacity of WTP residual may be related to its substantial alkalinity and $CaCO_3$ concentration.

4.2.4 Cation Exchange Capacity in WTP Residuals

The CEC of the WTP residual is defined as the sum of the exchangeable cations of a WTP residual expressed in milliequivalent for 100 gram of dry WTP residual. A high CEC generally denotes potentially a high capacity to retain contaminants in the residuals. The average CEC of soil is about 10 to 40 meq/100g. The CEC is related to the composition of the residuals. The most important considerations affecting the CEC value are the surface charges, hydroxyl-metals complexes, and organic matter.

The permanent charges are independent of pH and are associated with intrinsic characteristics of the clay mineral or colloid constituents. The presence of clay and colloids are related to the properties of water sources. Table 4.6 shows the ranges of CEC of some common clay minerals (Baize 1993).

The second factor affecting CEC is the presence of hydroxyl-aluminous complexes (alum residuals), hydroxyl-iron complexes (ferric residuals) or carbonate-calcium complexes (lime residuals), which are formed during the flocculation process. These complexes can have a large CEC value (Baize 1993).

The organic content in the residual also influences CEC due to the -COOH group and other peripheral active groups on the organic macromolecules from the water source or the organic coagulants. It is recognized that the CEC of organic matter is higher than that of clay minerals. The CEC of organic matter varies from 100 to 500 meq/100 g (Baize 1993).

The CEC of WTP residuals reflects the levels of clay, hydroxyl-metals complexes and organic matter in residuals and also indicates the ability of a WTP residual to hold metals by several mechanisms. When metals are tied-up in residuals they are not likely to cause metal toxicity to plants and soils organisms or to be taken up by plants.

As shown in Table 4.7, the CEC values of dewatered WTP residuals were generally from 22 to 135 meq/100g, which were greater than the CEC of soil and most clay minerals. The CECs of WTP dewatered residuals were related apparently to the organic content. Residuals with high organic contents also have high CECs, as it can be seen in FLD and WQD samples. This confirms Jacobs' observation (1981) that the CEC depends on the amount of organic matter present. The CEC values of all residuals except MQD are higher than 50 meq/100g. The low CEC of MQD residual can be attributed to the fact that this sample contained granular material that came out of underdrains in the lagoons during the sampling. The minimum value of CEC prescribed by New Jersey Department of Environmental Protection and Energy (1988) for clay liners for municipal solid waste landfills is 35 meq/100g (Woods, 1992, 1988). This information along with observation on the susceptibility of metals leaching out of residuals is the basis for our recommendation to eliminate the need to have clay liners for WTP residual monofills (Newman 1987).

4.2.5 Metals

Table 4.8 shows the metals present in dewatered residuals and the impurities in water sources from ten water treatment plants studied. From this data, it is seen that if the hardness was high, then the associated magnesium concentration in the dewatered residuals was high. The manganese concentration of dewatered residuals was also related to the manganese concentration in the water sources, as seen in samples RWAF, WQD, and ELD. High metal concentrations in residuals were mainly caused by water softening and coagulation processes utilized in water treatment. It is also interesting to see in Table 4.8 that the hardness of residual samples from river waters was higher than those from lakes and reservoirs for the samples studied.

The metals detected in dewatered residuals .and the chemical added in water purification and residuals treatment processes are shown in Table 4.9. These metals are closely related to the types and quantities of chemicals added. For example, in the dewatered cakes collected from Jersey City WTP (JCD), Little Fall WTP (PVC), and Minneapolis Water Works, where lime was used for softening or as dewatering conditioning reagent, the calcium concentrations were very high (10.06% - 17.97%). For iron residuals, iron concentrations in residuals range from 2.81% to 4.41%, which are much higher than those for other residuals. Table 4.9 also shows that the concentrations of aluminum in alum residuals WQD, HWD, MQD, FLD and FLDM were 0.75% to 3.30%. These are the highest among all samples tested in this study.

For residuals, such as JCD, PVD, MWD, and SLD, two chemicals were added, one was used as coagulant, and the other one was used for softening, as a conditioning reagent or as a second coagulant. Both elements were present in the residuals. For example, alum is used as coagulant in water purification process and lime is used as conditioning reagent in dewatering process in Jersey City WTP. Concentrations of both calcium and aluminum in sample JCD were both relatively high.

The results in Tables 4.8 and 4.9 show that only those metals leached from the residuals were present in the residuals. Types of the metals present in dewatered residuals and their concentrations are related to types of water sources, impurities of water sources, and chemicals addition in water purification and dewatering processes.

4.3 Biodegradation

In order to investigate the need for gas venting system and to evaluate the possibility of biogas production in monofills, a bench scale batch anaerobic respirometric study was conducted. The results obtained from the testing of three types of residuals are shown in Table 4.10. No methane was produced from dewatered cakes due to the high pH (10.5), low organic contents (Table 4.11), high metal contents, and relatively low water contents. The severe environmental conditions inhibited the growth of anaerobic bacteria. However, some methane, carbon dioxide, and other gases were produced in raw residuals. Microscopic examination indicated that there were no living microorganisms in dewatered cake but some microorganisms in raw residuals were observed.

The test was made to evaluate biodegradation of WTP residuals with or without seeds. The sample was placed in 150 mL vials under anaerobic condition. Temperature was maintained at 35°C by placing all reactors in a water bath. The gas volumes were measured after 191 days. The results are presented in Table 4.11.

Table 4.11 shows that in PVD dewatered cakes, even with the addition of seeds, bacteria would not grow in the high pH environment. For HWD dewatered residuals, which had very high solid content (60.06 %), some gas was produced. Based on the results of HWD samples, if a favorable growth environment and a high water content exist, biodegradation will occur.

Another 2 liter batch reactor was utilized to investigate anaerobic biodegradation of WTP raw, conditioned, and dewatered residuals. The temperature was controlled at 35 °C and solid retention time was maintained at one month. During this test, the biogas production was measured and the microorganisms were examined. Gas production and microorganisms were not observed. The pH was close to 10 (9.44-10.44) and that inhibited microbial activities.

4.4 TCLP Analysis for WTP Residuals

4.4.1 Analyses of Regulated Metals in WTP Residuals

The TCLP metals obtained for WTP dewatered residuals are shown in Table 4.12. It can be seen from these tables that metal contents detected were lower than regulated concentrations.

4.4.2 Analyses of Pesticides and Herbicides in WTP Residuals

Analyses of pesticides and herbicides of samples from the ten water treatment plants are presented in Table 4.12. All pesticides and herbicides had concentrations lower than regulated limits by careful examination of the corresponding peaks of the identified compounds with those in the standards.

4.4.3 Analysis of Base-Neutral and Acid in WTP Residuals

The results of WTP residuals are presented in Table 4.12. The BNA determination of samples revealed that the contents of regulatory compounds were below minimum detect limit (MDL) by GC-MS.

4.4.4 Volatile Organic Compounds in WTP Residuals

The results of VOCs from ten samples are shown in Tables 4.13. These observations indicate that the concentrations of volatile leaching out of the samples were below the regulatory concentrations.

4.5 Residual Dewatering

Solids content in dewatered residuals is affected by solids composition, dewatering characteristics, and dewatering processes. In the studies, the fixed and volatile solids will be tested. The WTP residuals samples will be collected from different dewatering processes including a drying bed, press filter, centrifuge. The dewatering tests will be conducted in the laboratory, such as paint filter, air drying, press filter, and thermal drying.

4.5.1 Solids Composition

The relationship between water content and dewatering method is shown in Table 4.3. In this table, six residual samples were obtained from the drying beds, four other samples were from dewatering machines, three of which were from filter presses and one from a centrifuge. The residual samples from the drying beds all had high solids contents (41.31 to 81.95%) compared with the samples from filter presses (14.83 to 29.28%). For the air drying bed samples, there was no relationship found between the solids content and volatile solids. For the mechanical dewatered samples, the dewatering properties appeared to be related to the organic content in the residuals. As shown in Table 4.3, in general, the solids content decreased with the decreasing volatile solids, lower organic content would improve the dewatering properties for WTP residuals.

4.5.2 Dewatering Characteristics of WTP Residuals

The dewatering characteristics of WTP residuals can be determined by various methods by which the water from residuals was removed in the laboratory, such as paint filter test, pressure filtration, air drying , and oven drying.

4.5.2.1 ***Paint Filter Test:*** The paint filter test can be used as an indicator for determining free water content in WTP residuals. Raw residual samples WQD, RWA, PVD (with lime), and PVD (without lime) were tested in this study for determination of the relationship between solids content and free water. Paint filter tests were conducted at different initial solids contents. Based on the experimental data, curves were plotted between volume of water removed and solids content as shown in Figure 4.6. A critical point, which is defined as the point beyond which water will not be removed by gravity, can be obtained. It may be noteworthy to mention that although a sample passes the paint filter test, it does not mean that water will not flow from it if even small amounts of energy is added. For example, the RWA sample at 18% solids passes the paint filter test, but this sample (and others) were received by NJIT with standing water in the container from the motion (vibration) of mere shipping i.e. passing the paint filter press does not mean a residual can be trucked without leakage. Graphical construction to obtain this critical point Cv is shown in Figure 4.6. the solids content at the critical point is referred to as CPSC (critical point solids content). The value of CPSC of the residual can represent the amount of free water. Different types of WTP residuals have different amount of free water, and, therefore different CPSC. If the solids content in a residual is equal to or greater than the value of the CPSC of the same residual, there will be no water coming out. Also, the higher the value of CPSC, the lower the quantity of free water present in a residual.

Figure 4.7 and Figure 4.8 are the curves for a fresh alum residual and 5-year old alum residual collected from a lagoon at Wanaque Water Treatment Plant. The values of CPSC of fresh and aged samples were 3.0% and 26.5%, respectively. Figure 4.9 presents the plot of the sample from the West River WTP, which was ferric residual (paste form). The corresponding CPSC value for this residual was about 4.5%. Two samples from Little Falls WTP were tested, as shown in Figures 4.10 and 4.11, one was from the thickener without the adding of lime, the other was from the mixing chamber, where lime was added. The CPSC values for these two samples were 4.6% and 8.0%, respectively. This indicates that the type of residual, or type of chemical added will affect the amount of free water in the residual.

Table 4.14 shows that the higher the CPSC value, the higher the solids content, and the more free water present in the residual. Higher CPSC values may also indicate that the water is easier to be removed by gravitational dewatering methods from the residual.

Table 4.14 also shows that the CPSC value may be affected by the age of the residual as in the case of WQD sample. The CPSC increased with the age. Some water, such as trapped water, capillary water, and surface adsorbed water would be vaporized due to the change of residual structure under weathering conditions. The CPSC value for 5-year old WQD sample (26.5%) was much higher than that of the fresh sample (3.0%). Also for the two PVD samples tested, the one with lime addition had a higher solids content. Addition of conditioning reagent also alters the residual characteristics so that the removal of water is more efficient.

4.5.2.2 Dewatering Test: In this test, five samples were tested. WQD1 fresh sample was taken from lagoon, and solids content was about 0.25-1%. WQD2 aged sample also was taken from lagoon, and solids content was about 4%. RWA sample (original sample #2) was taken from the lagoon, and solids content was about 2%. PVD1 sample was taken from the sample port before pressure filter, and solids content was about 3%. PVD2 sample was taken from holding tank, solids content was about 2%. The results are shown in Table 4.15. Several procedures were adapted in the dewatering test The results are shown in Table 4.15. These results can be used to differentiate the types of water that can be removed by various dewatering methods.

The CPSC value obtained from the paint filter test can be used for the measurement of free water since it indicates how much water can be removed by gravity. The difference between the solids contents obtained by mechanical dewatering and the CPSC value may indicate the quantity of trapped water that can be removed because of the structure change and void reduction by mechanical force. This may also indicate that a lower limit of water can be removed by aging and weathering. The difference in solids content between mechanical dewatering and air drying can be a measure of capillary water and some bound water. This may also indicate an upper limit of water that can be removed by aging and weathering.

The quantity of water left in the residual after air drying is a measure of the bound water.

4.5.3 Dewatering Methods for WTP Residuals

Two types of dewatering methods are commonly used for WTP residuals, air drying and mechanical dewatering. Sand drying beds and lagoons are most commonly used for air drying. The mechanical devices used include belt filter presses, centrifuges, pressure filter presses, and vacuum filters. Among the ten water treatment plants studied, six plants used drying beds, three plants utilize filter presses, and one used centrifuges to dewater residuals. The related information is presented in Table 4.16.

Table 4.16 shows that higher solids content can be obtained using drying beds. More water can be removed by air drying methods than mechanical dewatering machines. This is because free water and trapped water can be removed by dewatering machines when the residual floc changes. However, more water can be removed through vaporization and/or freeze/thaw in drying beds. Similar results were obtained from laboratory tests as shown in Table 4.15. However, it should be pointed out that these observations were based on the test of small quantities of sample in the laboratory. In field conditions, when large quantities are involved, the residual will have to be spread in relatively thin layers over a sufficient land area for a drying bed to be effective. The solids content of three types of WTP residuals through mechanical dewatering were from 12.08% to 23.82%, which are very close to the dewatered samples from WTP filter presses as shown in Table 4.15, except for five-year old sample WQD, which has the solids content of 44.78% because of the effects of freeze/thaw. The solids content of WTP residuals of the laboratory air drying samples can reach 83.66% to 88.65%, whereas the solids contents of WTP residuals obtained from drying beds were from 41.31% to 81.95% depended on the weathering conditions. As shown in Table 4.16, higher solids content can be obtained using the centrifuge. It is an effective dewatering machine.

Residual dewatering through the use of drying beds was not affected by the composition of residuals and types of water present in residuals, since all the solids content were high. On the other hand, mechanical dewatering seems related to the composition of residual as shown in Table 4.16. The dewatering efficiency increases with decreasing organic content and permeability.

4.6 Discussion and Conclusions

1. All the WTP residuals with solids contents greater than 10% can pass the paint filter test.
2. The organic content is one of the main factors to influence the geotechnical and environmental characteristics of residuals. High concentration of organics can increase the cation exchange capacity (CEC) which can retain metals in the residuals.
3. The pH of dewatered residuals is related to the type and amount of coagulant and conditioning agent added. Lime residual had a higher pH (7.46 to 11.82). The pH of ferric and alum residuals were from 6.25 to 7.81. All the WTP residuals had pH buffering capacity for both acids and bases. This buffering capacity affects the amount of metals being leached out due to the change of pH. The lime residual had higher pH buffering for its lime composition.
4. The CEC of WTP residuals were generally from 22 to 135 meq/100g. The CEC of the residual is related to the organic content. The higher the organic content, the larger the CECs.
5. The metals found in residuals were closely related to the type and quantities of chemicals added and impurities in water sources for the three types of residual tested. The metal leaching ability is related to the pH, particle size distribution, organic content, and cation exchange capacity. No direct relationship among these factors was determined.

6. None of the TCLP regulated constituents exceeded the TCLP regulatory limits for all the ten residuals tested. Most of these constituents were either not detected or below the minimum detection limit.
7. Dewatering properties of WTP residuals were related to solids composition, dewatering characteristics of residuals, and dewatering processes.
8. Biodegradation tests of WTP residuals were conducted. Gas production and microorganisms were not observed in dewatered cakes because of relatively low water contents, high metal contents, low organic contents, and high pH.
9. The value of CPSC of the residual can represent the amount of free water in WTP residuals. The higher the CPSC value, the lower the quantity of free water. Higher CPSC values may also indicate that the water is easier to be removed by gravitational dewatering methods from the residual.

Table 4.1 Paint filter test results of WTP residues

Sample name	Liquid through filter	Water content (%)	Solids content (%)
JCD	No	72.44	27.56
PVD	No	70.72	29.28
WQD	No	85.17	14.83
MWD	No	29.06	70.94
HWD	No	39.94	60.06
MQD	No	53.29	46.71
ELD	No	58.69	41.31
RWA	No	79.44	20.56
RWAF	No	54.52	45.48
FLD	No	71.26	28.74
FLDM	No	18.05	81.95
SLD	No	25.51	74.49
JCR	Yes	97.33	2.67
PVR	Yes	97.30	2.70
WQR	Yes	99.62	0.38
HWR	Yes	96.71	3.29
FLR	Yes	97.59	2.41
JCC	Yes	96.85	3.15
PVC	Yes	95.92	4.08

Table 4.2 Relationship between solids content and dewatering processes for WTP residuals

Sample name	Dewatering process	Solids content (%)	Volatile solids (%)	Fixed solids (%)
FLDM	Drying bed	81.95	63.41	36.59
SLD	Drying bed	74.49	3.62	96.38
HWD	Drying bed	60.06	14.33	85.67
MQD	Drying bed	46.71	3.55	96.45
RWAF	Drying bed	45.48	35.62	64.38
ELD	Drying bed	41.31	17.37	82.63
MWD	Centrifuge	70.94	15.02	84.98
PVD	Filter press	29.28	9.47	90.53
JCD	Filter press	27.56	34.45	65.55
WQD	Filter press	14.83	45.99	54.01

Table 4.3 Solids contents of WTP dewatered residuals from five water treatment plants

Sample name	Dewatering process	Concentration of chemicals added			Volatile solids (%)	Fixed solids (%)
		Alum (ppm)	Ferric (ppm)	Lime (ppm)		
JCR	Sedimentation	10	-	*	45.48	54.52
JCC	Gravity thickener	-	-	1	31.14	68.86
JCD	Filter press	-	-	-	34.45	65.55
PVR	Sedimentation	9.53	-	-	25.49	74.51
PVC	Gravity thickener	-	-	-	20.92	79.08
PVD	Filter press	-	-	50	9.47	90.53
WQR	Lagoon	10-12	-	-	51.82	48.18
WQD	Drying bed	-	-	-	45.99	54.01
HWR	Lagoon	5	-	-	29.88	70.12
HWD	Drying bed	-	-	-	14.33	85.67
FLR	Lagoon	75	-	25	61.19	38.81
FLD	Drying bed	-	-	-	61.44	38.56

- : No chemicals added

*: No information available

Table 4.4 Relationship with pH and primary metals concentration of WTP residuals

Sample	Residual	pH	Primary metals concentration (as dry basis) Aluminum (%)	Iron (%)	Calcium (%)
JCD	Lime	9.94	3.92	0.06	12.86
PVD	Lime	11.15	1.26	0.04	17.97
MWD	Lime	7.46	0.01	0.08	10.06
SLD	Lime	9.15	0.04	0.11	10.02
ELD	Lime	6.25	0.014	2.81	7.58
RWA	Ferric	7.15	0.34	3.87	0.27
RWAF	Ferric	6.71	0.07	4.41	0.35
WQD	Alum	6.55	5.80	0.32	1.16
HWD	Alum	6.82	0.82	0.006	0.100
MQD	Alum	7.83	2.02	0.16	0.98
FLD	Alum	6.62	3.03	0.16	0.13
FLDM	Alum	6.45	0.75	0.04	0.12

Table 4.5 Buffer capacities of WTP residuals

		PVD	JCD	WQD
Total alkalinity	(meq/g)	5.88	4.93	0.41
pH		11.1	9.9	6.5
Temperature	(°C)	24.5	24	24
pK_1		6.37	6.37	6.37
pK_2		10.34	10.34	10.34
K_1	(10^{-7})	4.28	4.25	4.25
K_2	(10^{-11})	4.62	4.57	4.47
α		0.13	0.72	0.60
Buffer capacity (β)	(10^{-4} eq/g)	13.66	79.65	5.67

Table 4.6 The CEC of some clay minerals

Minerals	CEC (meq/100 g)
Kaolinites	3-15
Smectites	80-150
Illites	10-40
Vermiculites	100-150
Chlorites	10-40
Attapulgite, palygorskite	3-15

Table 4.7 Relationship of CECs with volatile solids in dewatered WTP residuals

Sample	Water source	pH	Solids content (%)	Volatile solids (%)	CEC (meq/100g)
JCD	Reservoir	9.94	27.56	34.45	134.8
FLD	Reservoir	6.62	28.74	61.44	133.75
WQD	Reservoir	6.55	14.83	39.24	106.0
ELD	River	6.25	41.31	17.37	105.99
RWA	Reservoir	7.15	20.56	38.24	74.91
MWD	River	7.46	70.94	15.02	72.32
HWD	Reservoir	6.82	60.06	14.33	59.35
SLD	River	9.15	74.49	3.62	55.40
PVD	River	11.15	29.28	9.47	53.1
MQD	River/ Reservoir	7.83	46.71	3.55	21.96

Table 4.8 Relationship of metals with constituents from water sources

Sample	Type	pH	Concentration of primary metals (dry basis)				Constituents in water sources		
			Solid Content (%)	Mg (%)	Mn (%)	Zn (%)	Type	Hardness (as $CaCO_3$) (ppm)	Mn (ppm)
SLD	Lime	9.15	74.49	1.21	0.02	<0.001	River	167	0.013
MWD	Lime	7.46	70.94	1.04	0.01	<0.001	River	170-230	<0.01
PVD	Lime	11.15	29.28	0.23	0.13	0.008	River	112.4	0.11
ELD	Lime/Ferric	6.25	41.31	0.21	0.62	0.002	River	150	0.06
JCD	Lime	9.94	27.56	0.19	0.20	0.03	Reservoir	68.5	<0.02
HWD	Alum	6.82	60.06	0.172	0.148	0.002	Reservoir	120	NA
FLDM	Alum	6.45	81.95	0.081	0.006	0.001	Reservoir	40-110	NA
FLD	Alum	6.62	28.74	0.048	0.009	0.002	Reservoir	40-110	NA
WQD	Alum	6.55	14.83	0.046	1.55	0.009	Reservoir	47	0.045
MQD	Alum	7.83	46.71	0.024	0.111	0.047	Reservoir/river	30-60	0-0.13
RWAF	Ferric	6.71	45.48	0.017	1.30	0.01	Reservoir	25	0.08
RWA	Ferric	7.15	20.56	0.014	1.27	0.02	Reservoir	25	0.08

NA: Not available

Table 4.9 Relationship with primary metals and addition of chemical agents of residuals from ten water treatment plants

				Concentrations of primary metals (dry basis)			Concentration of chemical added in WTP process		
Sample	Type	pH	Solids content	Aluminum (%)	Iron (%)	Calcium (%)	Alum (ppm)	Ferric (ppm)	Lime (ppm)
JCD	Lime	9.94	27.56	3.92	0.06	12.86	10	-	1
PVD	Lime	11.15	29.28	1.26	0.04	17.97	9.53	-	50
MWD	Lime	7.46	70.94	0.01	0.08	10.06	20	-	170
SLD	Lime	9.15	74.49	0.04	0.11	10.02	-	12.8	94
ELD	Lime/Ferric	6.25	41.31	0.014	2.81	7.58	-	20	
RWA	Ferric	7.15	20.56	0.34	3.87	0.27	-	7.2	-
RWAF	Ferric	6.71	45.48	0.07	4.41	0.35	-	7.2	-
WQD	Alum	6.55	14.83	5.80	0.32	1.16	10-12	-	-
HWD	Alum	6.82	60.06	0.82	0.006	0.100	5	-	-
MQD	Alum	7.83	46.71	2.02	0.16	0.98		-	-
FLD	Alum	6.62	28.74	3.03	0.16	0.13	75	-	25
FLDM	Alum	6.45	81.95	0.75	0.04	0.12	75	-	25

-: Not applicable

Table 4.10 Composition of residual and gas produced from three types of WTP residuals (at Jersey City Water Treatment Plant in New Jersey)

WTP	pH	Solids Content (%)	Alkalinity (%)	Calcium (%)	Aluimium (%)	Carbon Dioxide (μg/L)	Methane (μg/L)	Carbon Monodioxide (μg/L)
Dewatered	9.94	27.56	4.937	12.86	3.92	ND	ND	0.11
Conditioned	10.44	3.15	6.707	13.53	4.68	73.4	1.65	ND
Raw Residual	6.96	2.67	1.276	1.22	12.25	0.66	2.96	0.18

Note: ND - Not Detected.

The number is the minimum attainable detection limit for the sample.

Table 4.11 Composition of WTP residuals with or without Seeds in biodegradation study

Sample	Type	pH	Solids Content (%)	Volatile Solids	Calcium (%)	Aluminum (%)	Biogas Production (ml/g)
PVD	Lime	11.15	29.28	9.47	17.93	1.25	ND
PVDs	Lime	11.15	29.28	9.47	17.93	1.25	ND
HWD	Alum	6.82	60.06	14.33	0.10	0.826	0.26
HWDs	Alum	6.82	60.06	14.33	0.10	0.826	0.25
HWL	Alum	6.82	3.29	29.88	0.25	2.13	0.16
HWLs	Alum	6.82	3.29	29.88	0.25	2.13	0.15

Note: ND - Not Detected.

PVD - The dewatered cake from the Little Fall Water Treatment Plant, New Jersey,

PVDs - The same sample with seeds,

HWD - The dewatered residual from air drying bed in Haworth Water Treatment Plant, New Jersey,

HWDs - The same sample with seeds,

HWL - The residual from the lagoon in Haworth Water Treatment Plant, New Jersey,

HWLs - The same sample with seed.

Table 4.12 TCLP analyses of metals, pesticides, herbicides, and BNA in water treatment plant residuals

Constituent	Regulatory* limit (mg/L)	Jersey City JCD (mg/L)	Little Falls PVD (mg/L)
Arsenic	5.0	0.069	< 0.069
Barium	100.0	2.566	2.695
Cadmium	1.0	0.052	< 0.052
Chromium	5.0	0.112	0.359
Lead	5.0	0.067	0.034
Mercury	0.2	< 0.019	< 0.019
Selenium	1.0	< 0.137	< 0.137
Silver	5.0	< 0.069	< 0.069
Chlordane	0.03	< 0.010	< 0.010
Endrin	0.02	< 0.008	< 0.008
Heptachlor	0.008	< 0.008	< 0.008
Heptachlor epoxide	0.008	< 0.008	< 0.008
Lindane	0.4	< 0.050	< 0.050
Methoxyclor	10.0	< 0.050	< 0.050
Toxaphene	0.5	< 0.050	< 0.050
2,4 D	10.0	< 0.010	< 0.010
2,4,5 TP (Silvex)	1.0	< 0.010	< 0.010
Hexachloroethane	3.0	< 0.010	< 0.010
o-Cresol	200.0	< 0.005	< 0.005
m-Cresol	200.0	< 0.005	< 0.005
p-Cresol	200.0	< 0.005	< 0.005
Cresol	200.0	< 0.005	< 0.005
1,4-Dichlorobenzene	7.5	< 0.005	< 0.005
2,4-Dinitrotoluene	0.13	< 0.005	< 0.005
Hexachlorobenzene	0.13	< 0.005	< 0.005
Hexachlorobutadiene	0.5	< 0.005	< 0.005
Nitrobenzene	2.0	< 0.005	< 0.005
Pentachlorophenol	5.0	< 0.005	< 0.005
Pyridine	5.0	< 0.005	< 0.005
2,4,5-Trichlorophenol	400.0	< 0.005	< 0.005
2,4,6 Trichlorophenol	2.0	< 0.005	< 0.005

(Continued)

Regulatory*	TCLP regulatory limit.
Sample, JCD	Dewatered residues (Cakes) from Jersey City Water Treatment Plant, New Jersey.
Sample, PVD	Dewatered residuals (Cakes) from Little Falls Water Treatment Plant, New Jersey.

Table 4.12 (Continued)

Constituent	Regulatory* limit (mg/L)	Wanaque	West River WTP	
		WQD (mg/L)	RWA	RWAF
			(mg/L)	
Arsenic	5.0	< 0.104	< 0.104	< 0.104
Barium	100.0	0.727	7.370	1.033
Cadmium	1.0	< 0.020	< 0.020	0.024
Chromium	5.0	0.078	0.100	0.053
Lead	5.0	< 0.062	0.284	0.453
Mercury	0.2	< 0.019	< 0.019	< 0.019
Selenium	1.0	< 0.189	< 0.189	< 0.189
Silver	5.0	< 0.007	< 0.007	< 0.007
Chlordane	0.03	< 0.001	< 0.001	< 0.001
Endrin	0.02	< 0.001	< 0.001	< 0.001
Heptachlor	0.008	< 0.001	< 0.001	< 0.001
Heptachlor epoxide	0.008	< 0.001	< 0.001	< 0.001
Lindane	0.4	< 0.001	< 0.001	< 0.001
Methoxyclor	10.0	< 0.001	< 0.001	< 0.001
Toxaphene	0.5	< 0.001	< 0.001	< 0.001
2,4 D	10.0	< 0.001	< 0.001	< 0.001
2,4,5 TP (Silvex)	1.0	< 0.001	< 0.001	< 0.001
Hexachloroethane	3.0	< 0.001	< 0.001	< 0.001
o-Cresol	200.0	< 0.005	< 0.005	< 0.005
m-Cresol	200.0	< 0.005	< 0.005	< 0.005
p-Cresol	200.0	< 0.005	< 0.005	< 0.005
Cresol	200.0	< 0.005	< 0.005	< 0.005
1,4-Dichlorobenzene	7.5	< 0.005	< 0.005	< 0.005
2,4-Dinitrotoluene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobenzene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobutadiene	0.5	< 0.005	< 0.005	< 0.005
Nitrobenzene	2.0	< 0.005	< 0.005	< 0.005
Pentachlorophenol	5.0	< 0.005	< 0.005	< 0.005
Pyridine	5.0	< 0.005	< 0.005	< 0.005
2,4,5-Trichlorophenol	400.0	< 0.005	< 0.005	< 0.005
2,4,6 Trichlorophenol	2.0	< 0.005	< 0.005	< 0.005

(Continued)

Regulatory*	TCLP regulatory limit.
Sample, WQD	Dewatered residues from Drying Bed in Wanaque Water Treatment Plant, New Jersey
Sample, RWA	Residues (unfrozen paste form) from Drying Bed in West River Water Treatment Plant
Sample, RWAF	Residues (frozen dry form) from Drying Bed in West River Water Treatment Plant

Table 4.12 (Continued)

Constituent	Regulatory*	Manasquan	St. Louis	
	Minneapolis limit (mg/L) (mg/L)	MQD (mg/L)	SLD (mg/L)	MWD
Arsenic	5.0	< 0.104	< 0.104	0.012
Barium	100.0	1.453	2.640	0.588
Cadmium	1.0	0.092	< 0.020	0.006
Chromium	5.0	0.062	0.288	0.035
Lead	5.0	< 0.062	0.144	0.044
Mercury	0.2	< 0.019	< 0.019	< 0.019
Selenium	1.0	< 0.189	< 0.189	< 0.189
Silver	5.0	< 0.007	< 0.007	< 0.007
Chlordane	0.03	< 0.001	< 0.001	< 0.001
Endrin	0.02	< 0.001	< 0.001	< 0.001
Heptachlor	0.008	< 0.001	< 0.001	< 0.001
Heptachlor epoxide	0.008	< 0.001	< 0.001	< 0.001
Lindane	0.4	< 0.001	< 0.001	< 0.001
Methoxyclor	10.0	< 0.001	< 0.001	< 0.001
Toxaphene	0.5	< 0.001	< 0.001	< 0.001
2,4 D	10.0	< 0.001	< 0.001	< 0.001
2,4,5 TP (Silvex)	1.0	< 0.001	< 0.001	< 0.001
Hexachloroethane	3.0	< 0.001	< 0.001	< 0.001
o-Cresol	200.0	< 0.005	< 0.005	< 0.005
m-Cresol	200.0	< 0.005	< 0.005	< 0.005
p-Cresol	200.0	< 0.005	< 0.005	< 0.005
Cresol	200.0	< 0.005	< 0.005	< 0.005
1,4-Dichlorobenzene	7.5	< 0.005	< 0.005	< 0.005
2,4-Dinitrotoluene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobenzene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobutadiene	0.5	< 0.005	< 0.005	< 0.005
Nitrobenzene	2.0	< 0.005	< 0.005	< 0.005
Pentachlorophenol	5.0	< 0.005	< 0.005	< 0.005
Pyridine	5.0	< 0.005	< 0.005	< 0.005
2,4,5-Trichlorophenol	400.0	< 0.005	< 0.005	< 0.005
2,4,6 Trichlorophenol	2.0	< 0.005	< 0.005	< 0.005

(Continued)

Regulatory*	TCLP regulatory limit.
Sample, MQD	Residues from Drying Bed in Manasquan Water Treatment Plant
Sample, SLD	Dewatered residues from Monofill in South County Plant of St. Louis Water Company
Sample, MWD	Centrifuge Cake from Minneapolis Water Department Dewatering Plant

Table 4.12 (Continued)

Constituent	Regulatory*	Ellwood WTP	HackenSack WTP	
	limit (mg/L) (mg/L)	NCD (mg/L)	HWR (mg/L)	HWD
Arsenic	5.0	< 0.104	0.139	0.145
Barium	100.0	3.4	0.590	2.285
Cadmium	1.0	< 0.020	0.108	0.081
Chromium	5.0	0.044	0.165	0.272
Lead	5.0	0.311	0.039	< 0.001
Mercury	0.2	< 0.019	< 0.019	< 0.019
Selenium	1.0	< 0.189	< 0.023	< 0.023
Silver	5.0	< 0.007	< 0.072	< 0.072
Chlordane	0.03	< 0.005	< 0.005	< 0.005
Endrin	0.02	< 0.001	< 0.001	< 0.001
Heptachlor	0.008	< 0.001	< 0.001	< 0.001
Heptachlor epoxide	0.008	< 0.001	< 0.001	< 0.001
Lindane	0.4	< 0.001	< 0.001	< 0.001
Methoxyclor	10.0	< 0.005	< 0.005	< 0.005
Toxaphene	0.5	< 0.001	< 0.001	< 0.001
2,4 D	10.0	< 0.005	< 0.005	< 0.005
2,4,5 TP (Silvex)	1.0	< 0.001	< 0.001	< 0.001
Hexachloroethane	3.0	< 0.001	< 0.001	< 0.001
o-Cresol	200.0	< 0.005	< 0.005	< 0.005
m-Cresol	200.0	< 0.005	< 0.005	< 0.005
p-Cresol	200.0	< 0.005	< 0.005	< 0.005
Cresol	200.0	< 0.005	< 0.005	< 0.005
1,4-Dichlorobenzene	7.5	< 0.005	< 0.005	< 0.005
2,4-Dinitrotoluene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobenzene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobutadiene	0.5	< 0.005	< 0.005	< 0.005
Nitrobenzene	2.0	< 0.005	< 0.005	< 0.005
Pentachlorophenol	5.0	< 0.005	< 0.005	< 0.005
Pyridine	5.0	< 0.005	< 0.005	< 0.005
2,4,5-Trichlorophenol	400.0	< 0.005	< 0.005	< 0.005
2,4,6 Trichlorophenol	2.0	< 0.005	< 0.005	< 0.005

(Continued)

Regulatory* TCLP regulatory limit.
Sample, NCD Dewatered Residues from Drying Bed in Ellwood City Treatment Plant in Pennsylvania
Sample, HWR Residues from Lagoon in Hackensack Water Company Harrington Park in New Jersey
Sample, HWD Dewatered Residuals from Drying Bed in Hackensack Water Company Harrington Park in New Jersey

Table 4.12 (Continued)

Constituent	Regulatory* limit (mg/L)	Bradenton WTP in Florida FLR (mg/L)	FLD (mg/L)	FLDM (mg/L)
Arsenic	5.0	0.032	< 0.007	0.145
Barium	100.0	0.151	1.650	0.796
Cadmium	1.0	0.058	0.030	0.043
Chromium	5.0	0.330	2.610	0.075
Lead	5.0	0.150	0.430	0.147
Mercury	0.2	< 0.019	< 0.019	< 0.019
Selenium	1.0	< 0.023	< 0.023	< 0.023
Silver	5.0	< 0.072	< 0.072	< 0.072
Chlordane	0.03	< 0.005	< 0.005	< 0.001
Endrin	0.02	< 0.001	< 0.001	< 0.001
Heptachlor	0.008	< 0.001	< 0.001	< 0.001
Heptachlor epoxide	0.008	< 0.001	< 0.001	< 0.001
Lindane	0.4	< 0.001	< 0.001	< 0.001
Methoxyclor	10.0	< 0.005	< 0.005	< 0.001
Toxaphene	0.5	< 0.001	< 0.001	< 0.001
2,4 D	10.0	< 0.005	< 0.005	< 0.001
2,4,5 TP (Silvex)	1.0	< 0.001	< 0.001	< 0.001
Hexachloroethane	3.0	< 0.001	< 0.001	< 0.001
o-Cresol	200.0	< 0.005	< 0.005	< 0.005
m-Cresol	200.0	< 0.005	< 0.005	< 0.005
p-Cresol	200.0	< 0.005	< 0.005	< 0.005
Cresol	200.0	< 0.005	< 0.005	< 0.005
1,4-Dichlorobenzene	7.5	< 0.005	< 0.005	< 0.005
2,4-Dinitrotoluene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobenzene	0.13	< 0.005	< 0.005	< 0.005
Hexachlorobutadiene	0.5	< 0.005	< 0.005	< 0.005
Nitrobenzene	2.0	< 0.005	< 0.005	< 0.005
Pentachlorophenol	5.0	< 0.005	< 0.005	< 0.005
Pyridine	5.0	< 0.005	< 0.005	< 0.005
2,4,5-Trichlorophenol	400.0	< 0.005	< 0.005	< 0.005
2,4,6 Trichlorophenol	2.0	< 0.005	< 0.005	< 0.005

Regulatory*	TCLP regulatory limit.
Sample, FLR	Residues from Lagoon of Bradenton Water Plant in Florida
Sample, FLD	Dewatered Residues from Drying Bed of Bradenton Water Plant in Florida
Sample, FLDM	Dewatered Residuals (Dry Form) from Outlet of Drying Bed of Bradenton Water Plant in Florida

Table 4.13 TCLP analyses of VOCs in water treatment plant residuals

Constituent*	Regulatory (mg/L)	SLD (mg/L)	FLD (mg/L)	MWD (mg/L)	RWA (mg/L)	JCD (mg/L)
Benzene	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Carbontetrachloride	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Chlorobenzene	100.0	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Chloroform	6.0	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
1,2 Dichloroethane	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
1,1 Dichloroethylene	0.7	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Methyl ethyl ketone	200.0	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Tetrachloroethylene	0.7	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Trichloroethylene	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Vinyl chloride	0.2	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010

(continued)

* All VOCs are determined by TCLP and purge-trap GC/MS,

Table 4.13 (continued)

Constituent*	Regulatory (mg/L)	HWD (mg/L)	PVD (mg/L)	MQD (mg/L)	WQD (mg/L)	NCD (mg/L)
Benzene	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Carbontetrachloride	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Chlorobenzene	100.0	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Chloroform	6.0	0.025	0.016	< 0.010	< 0.010	< 0.010
1,2 Dichloroethane	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
1,1 Dichloroethylene	0.7	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Methyl ethyl ketone	200.0	0.012	< 0.010	< 0.010	< 0.010	< 0.010
Tetrachloroethylene	0.7	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Trichloroethylene	0.5	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010
Vinyl chloride	0.2	< 0.010	< 0.010	< 0.010	< 0.010	< 0.010

Table 4.14 Critical points solids content (CPSC)for three types of WTP residuals

Sample name	Types	CPSC (%)	Age	Volatile solids (%)	Fixed solids (%)
WQD1	Alum	3.0	Fresh	48.23	51.77
WQD2	Alum	26.5	5-year	45.99	54.01
RWA	Ferric	8.5	1-year	38.24	61.76
PVD1	Lime	8.0	Fresh	20.92	79.08
PVD2	Without lime	4.6	Fresh	25.49	74.51

Table 4.15 Solids contents of three WTP residuals under different dewatering conditions

		Solids content (%)			
Sample	Type	Paint filter	Mechanical	Air dried	Oven dried
WQD1	Alum	3.0	30.05	83.86	100
WQD2	Alum	26.5	44.78	83.66	100
RWA	Ferric	8.5	12.08	86.61	100
PVD1	Lime	8.0	22.38	88.65	100
PVD2	Without lme	4.6	23.82	87.78	100

Table 4.16 Relationship with solids contents and dewatering processes of WTP residuals from ten water treatment plants

Sample name	Dewatering process	Solids content (%)	Volatile solids content (%)	Coefficient of permeability Kv,(cm/sec)
FLDM	Drying bed	81.95	63.41	7.33x10-8
SLD	Drying bed	74.49	3.62	2.60x10-6
HWD	Drying bed	60.06	14.33	3.30x10-6
MQD	Drying bed	46.71	3.55	5.49x10-6
RWAF	Drying bed	45.48	35.62	1.28x10-7
ELD	Drying bed	41.31	17.37	5.56x10-6
MWD	Centrifuge	70.94	15.02	3.28x10-6
PVD	Filter press	29.28	9.47	6.47x10-6
JCD	Filter press	27.56	34.45	5.53x10-7
WQD	Filter press	14.83	45.99	9.71x10-6

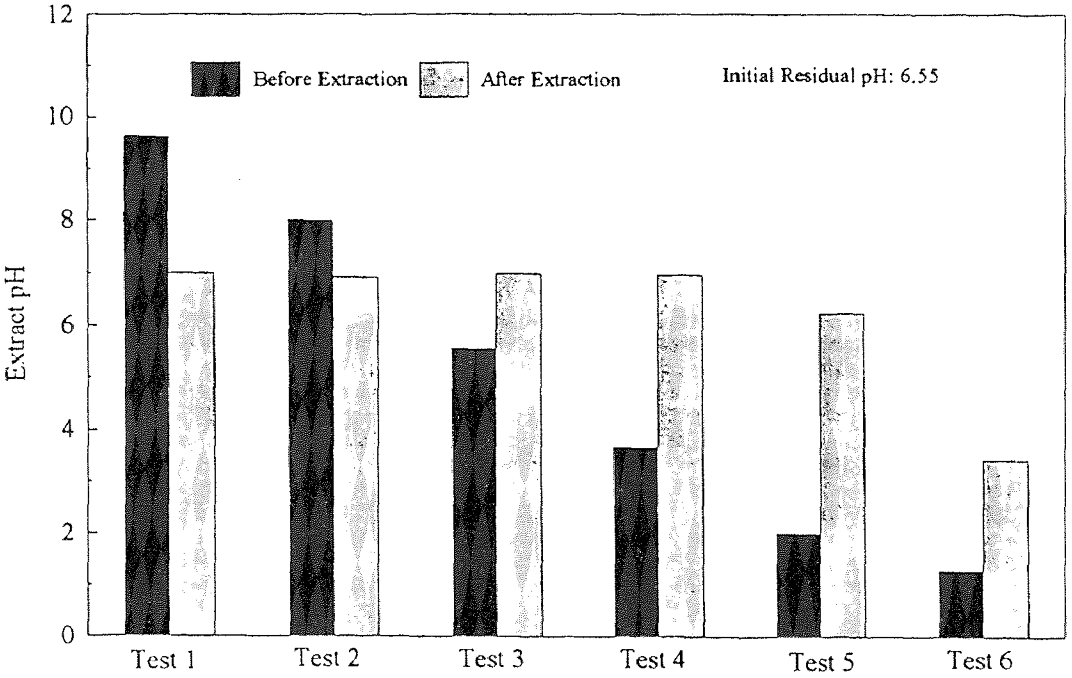


Figure 4.1 pH buffer capacity of ELD alum residual

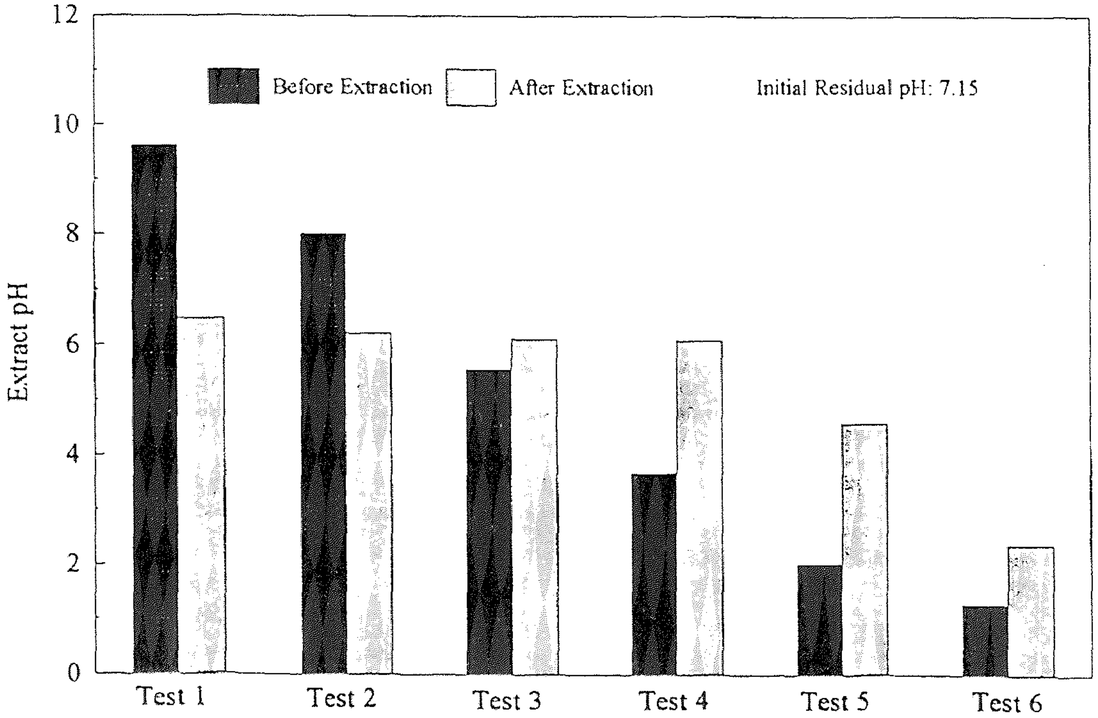


Figure 4.2 pH buffer capacity of RWA ferric residual

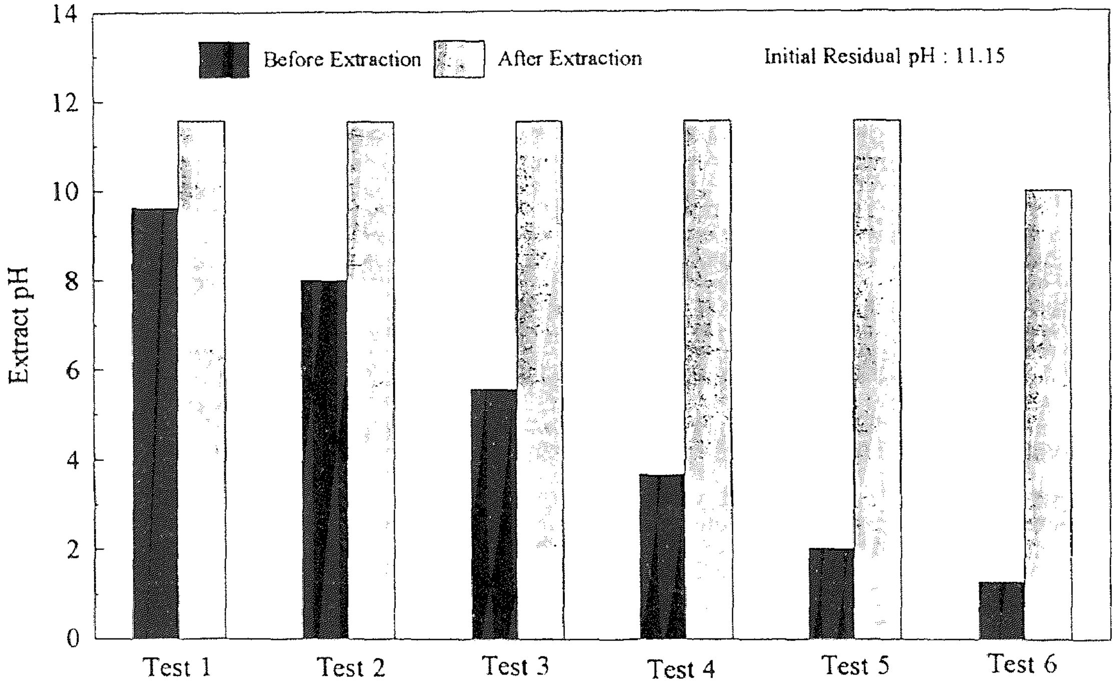


Figure 4.3 pH buffer capacity of PVD lime residual

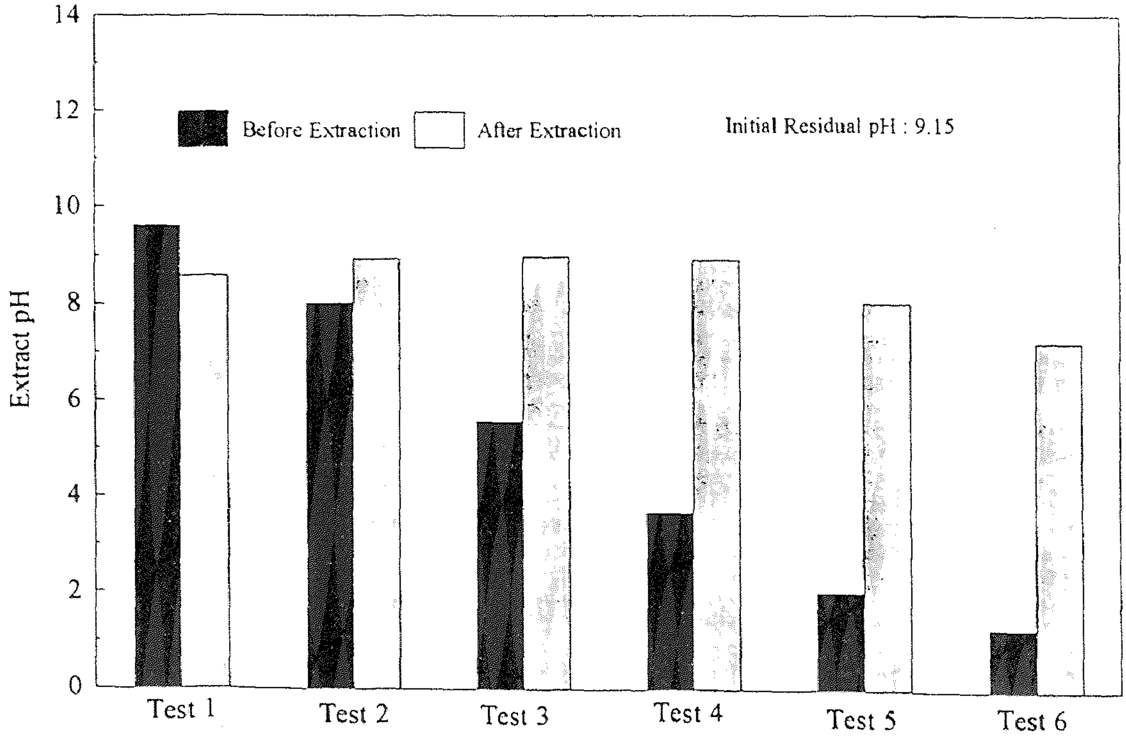


Figure 4.4 pH buffer capacity of SLD alum/lime residual

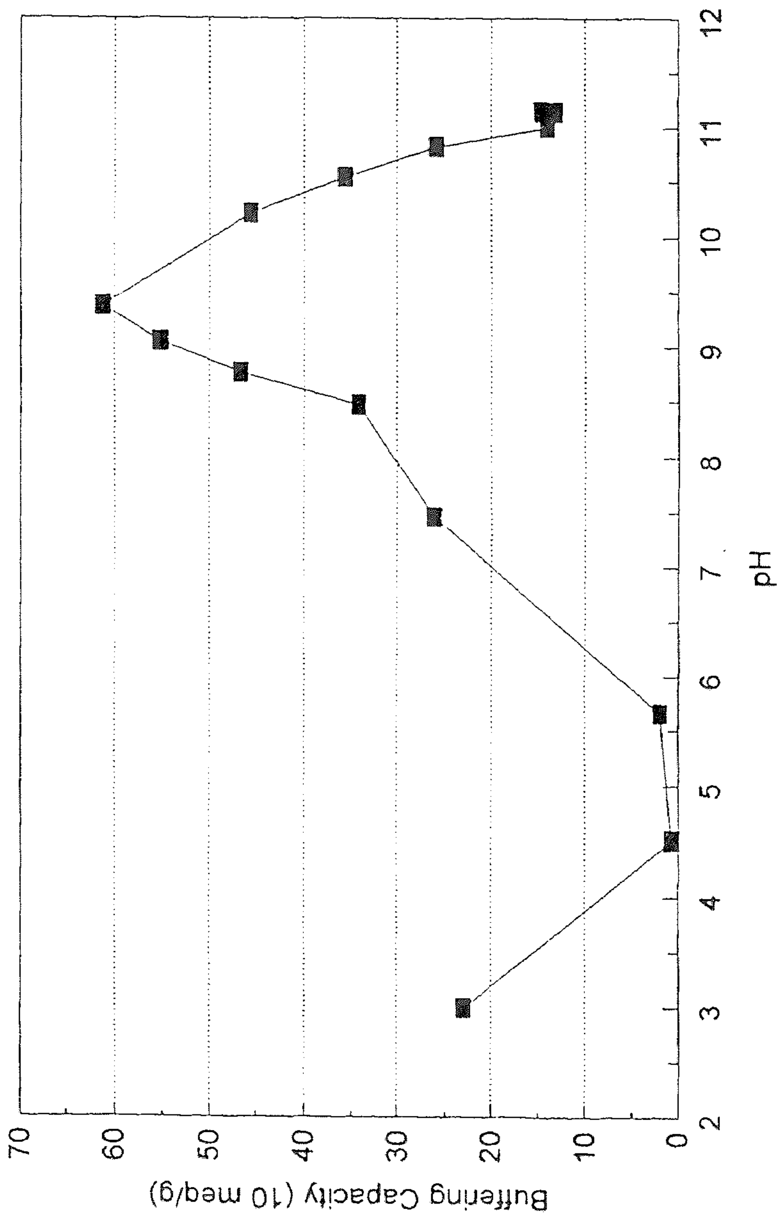


Figure 4.5 Variation of buffering capacity with pH for a PVD residual

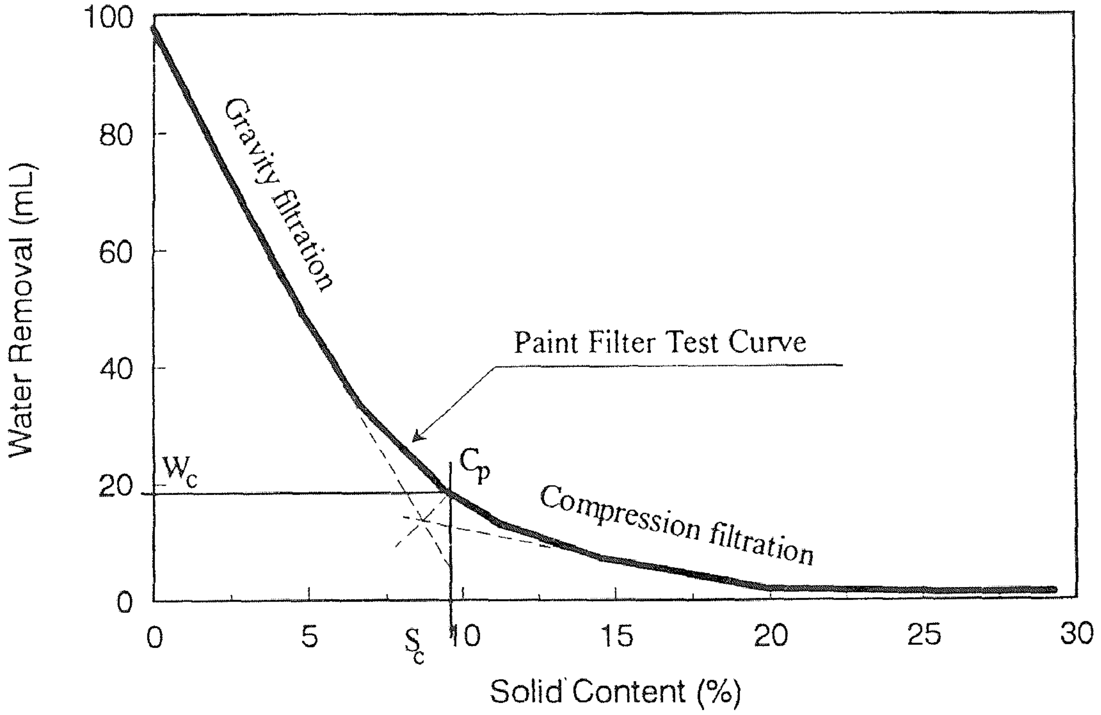


Figure 4.6 Graphical analysis of arbitrary paint filter test curve

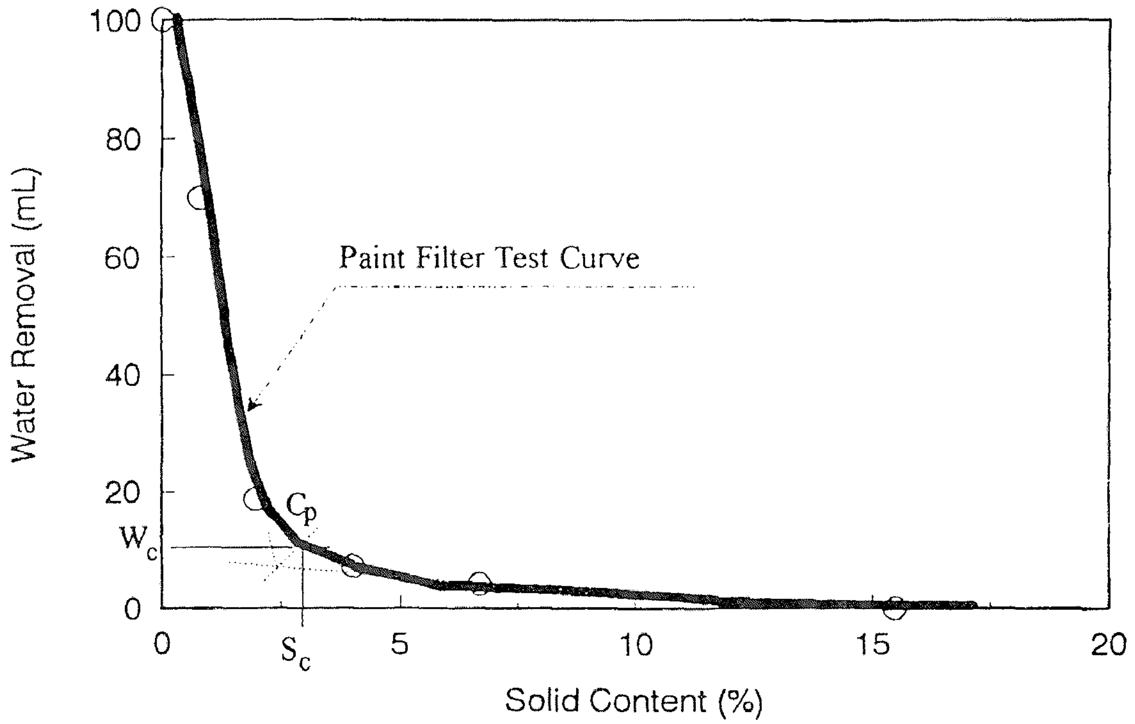


Figure 4.7 Paint filter test for determination of Wanaque WTP fresh residuals from lagoon

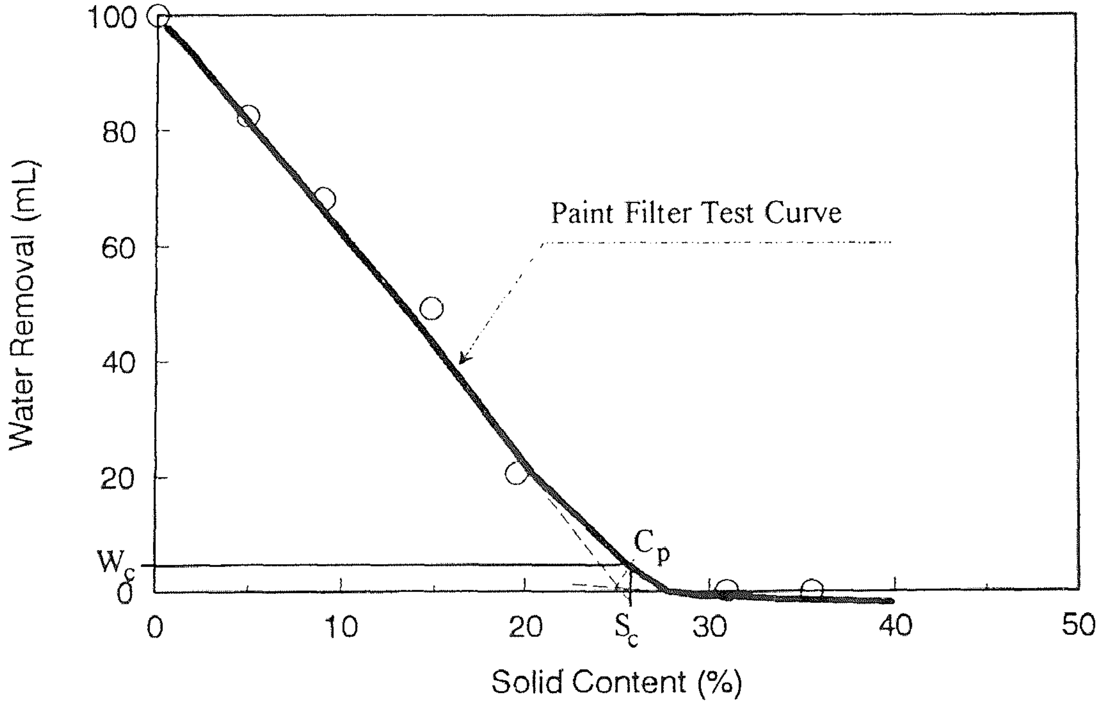


Figure 4.8 Paint filter test for determination of Wanaque WTP aged residuals from lagoon

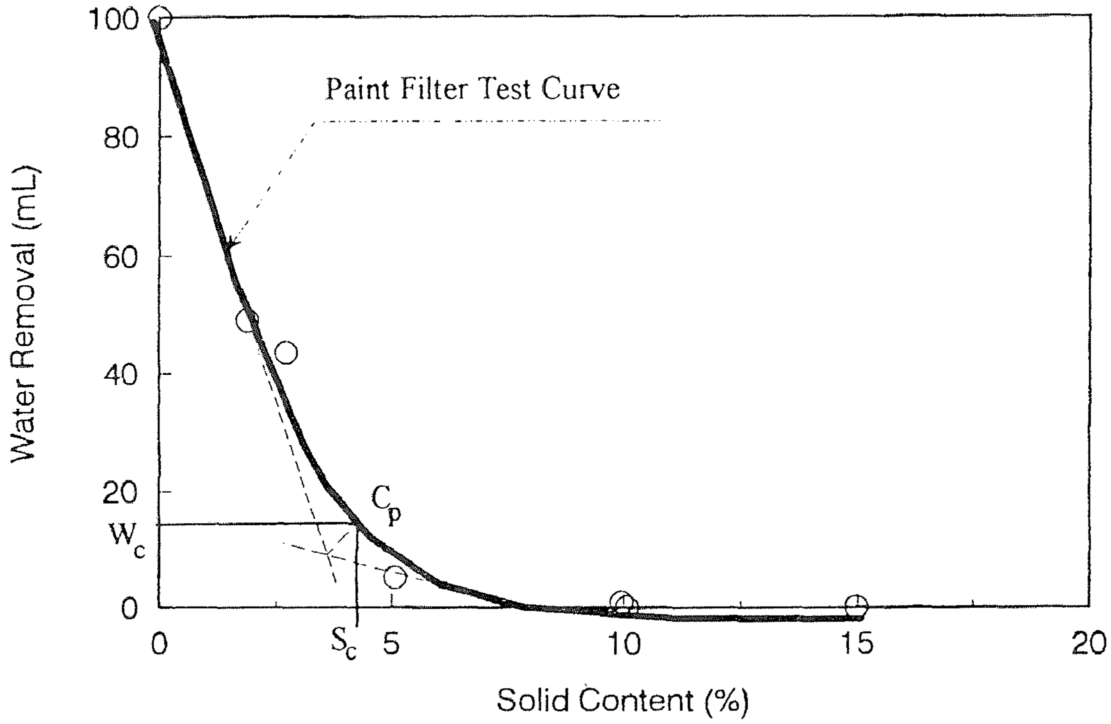


Figure 4.9 Paint filter test for determination of West River WTP residuals (paste form) from lagoon

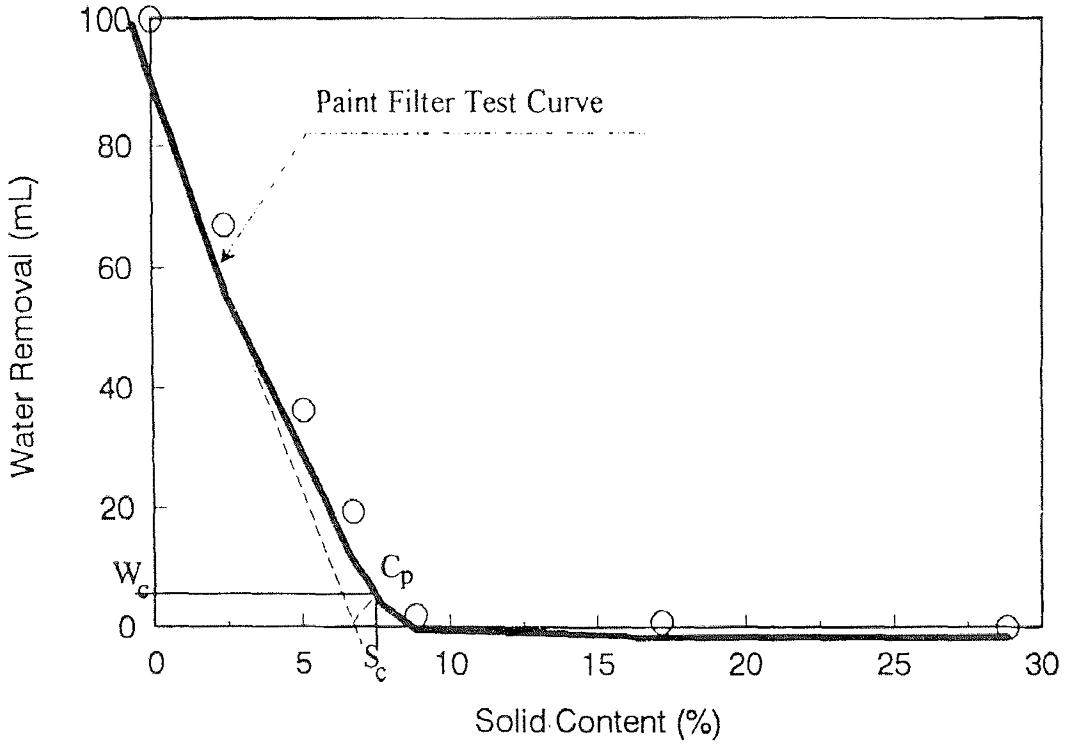


Figure 4.10 Paint filter test for determination of Little Falls WTP residuals with lime from the mixing chamber

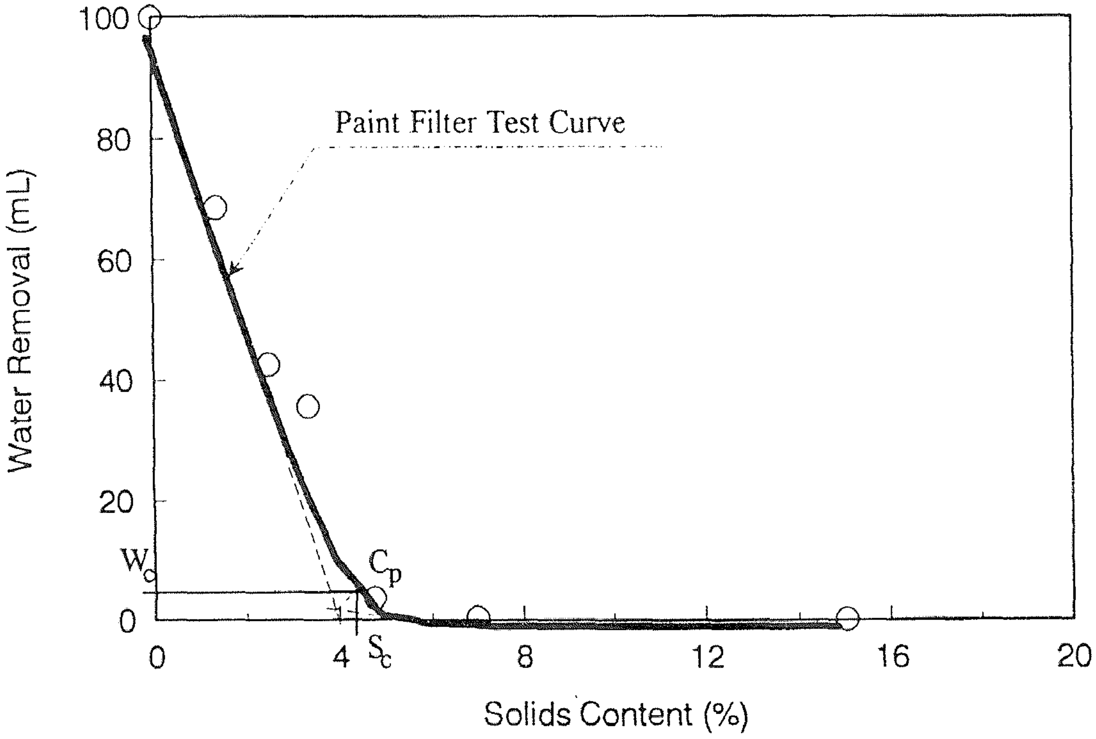


Figure 4.11 Paint filter test for determination of Little Falls WTP residuals without lime before the mixing chamber

CHAPTER 5

FIELD TESTS CONDUCTED IN A MINIMONOFILL

5.1 Introduction

Previous sections of this study discussed the characterization of WTP residual samples based on laboratory results. In order to develop criteria for the design, construction and operation of WTP residual monofills, it was necessary to observe a model monofill in operation (O'Leary et al, 1987a, 1987b). Laboratory tests can not provide complete and realistic answers to questions regarding leaching and biodegradation occurring from WTP residuals in monofills as time progresses. The effects of aging, weather, and environment on the characteristics of the WTP residuals such as permeability and strength can be studied more effectively by field testing.

In order to study the effects of aging and weathering on WTP residual in monofills, and to confirm and correlate the laboratory test results, field testing on a WTP residual was conducted at New Castle, Pa. After being approved by the Pennsylvania-American Water Company and the Bureau of Waste Management, Department of Environmental Resources, Pennsylvania, a mini monofill was constructed on October 19, 1993, at New Castle Water Treatment Plant, New Castle, Pennsylvania. Three phases of samples were collected from this model monofill during the period from October 19, 1993, to October 21, 1994. This chapter presents a summary of the results of the field tests and the geotechnical and environmental characteristics of the residual samples.

5.2 Sampling of WTP Residuals from the Monofill

5.2.1 New Castle WTP Monofill

New Castle residual waste monofill (Permit #101508) is solely dedicated to the disposal of water treatment plant residual. The WTP residuals generated at the adjacent New Castle Water Treatment Plant, along with residuals from the Ellwood City and Indiana treatment facilities, are disposed of at this facility.

The residuals generated at New Castle water treatment plant, were placed in minimonofill for field study. The flow diagram of New Castle Water Treatment Plant is shown in Figure 5.1. This facility has a production capacity of 8.4 mgd (31794 cubic meters per day) and it generates approximately 1,600 lb./day (727.3 Kg/day) of residual solids. The water treatment process consisted of two wood crib intakes, two flash mixing chambers, two flocculation-sedimentation basins, eight dual media filters, two clearwells and distributive pumping systems. Raw water is pumped from the Shenango River into a mixing chamber. At the inlet to the mixing chamber, alum, chlorine, polymer, potassium permanganate, carbon and lime, are added. Water then flows into the flocculation-sedimentation basin. Approximately 90% of the residuals are settled out in the sedimentation basins. The settled water flows into four mixed media filters, and to the clearwell where chlorine, fluoride and corrosion inhibitor are added at its inlet pipe. Water is then pumped to the distribution system.

The residual from the basins had a solids concentration of 0.5 to 4% and mainly consisted of river silt, aluminum hydroxide, lime residual, polymer, and carbon. It is pumped into the gravity thickener and is concentrated to about 3 to 4%. Polymer is added to the thickened residual before it enters the mixing tank to aid the belt press dewatering. The pressed residual cake had solids content of approximately 20 to 25 percent. It is then transported to the New Castle Water Treatment Plant Monofill for disposal.

5.2.2 Construction of Minimonofill

A minimonofill, 10 feet (3 meter) square, was installed in cell #2 in the New Castle Water Treatment Plant Monofill. The cross section, layout and other details of this small area are shown in Figure 5.2. The site was first cleaned and vegetation and top soil were stripped. All sharp materials and stones were removed. The surface was graded to a slope of 5 to 10% as shown in Figure 5.2. A 10 feet (3 meters) square and 30 mil thick PVC impervious liner was placed on subgrade and then 2 inches (50 mm) of drainage stone, 1/2 inch (12.5 mm) to 3/4 inch (19 mm) in size, was placed on top of the liner. A one and half inch (37.5 mm) diameter and 10 feet (3 meters) long perforated pipe was placed on the stone and connected to a leachate collection bottle. The pipe was then covered with 3 inches (75 mm) of stone and a drainage geotextile fabric. A layer of WTP residuals, about 3 inches (450 mm) in thickness, was placed on top of the drainage fabric. The model monofill was held in place by straw bales around its perimeter to prevent residuals from disturbance during field testing.

Personnel from the treatment facility were present during the construction of the mini monofill. They were requested to observe the leachate collection system periodically and to inform the NJIT research team if any unusual events had occurred.

5.2.3 Sampling from Minimonofill

Samples of water treatment plant residuals and leachate (called water sample in this Chapter) were collected periodically over a period of time from October 19, 1993, to October 21, 1994. WTP residual samples were then collected from the top of the monofill and at depth of one foot from the surface and water samples were collected from the leachate collection system, for geotechnical and environmental analyses. For convenience of discussion, symbols were assigned to each of the samples collected. These symbols are listed in Table 5.1 below and will be used throughout this Chapter. Samples include residual samples NCFRS for environmental tests and NCFG and NCFGW for geotechnical tests, water samples NCFWS, and blanks NCFTB and NCFFB. The three dewatered samples (NCFRS, NCFG and NCFGW)were collected on October 19, 1993, April 22, 1994, and October 21, 1994, respectively. The two leachate samples were collected on April 22 and October 21, 1994. Another sample was collected on August 2, 1994, for determination of VOCs.

Water samples were collected from the leachate collection bottle. The water in the bottle was from the same source as that in the container holding the bottle. The model monofill was located in a corner of the Monofill. In this corner, the topography is such that there is a drop in elevation of at least 8 to 10 feet (2.4 to 3.0 meters). This causes runoff down the slope and in the direction of the leachate collection pit. Hence it is our opinion that the water in the leachate container is due to surface run off and snow melt.

In collecting these samples, necessary procedures regarding trip and field blanks were followed. Since the top 3 inches (75 mm) of the monofill was in a dry powdery condition WTP residual samples were collected from a depth of about 3 to 4 inches (75 to 100 mm) from ground level.. WTP samples from the surface of the monofill were also obtained. All the samples collected were brought to the NJIT laboratory and stored for testing.

In order to prevent water from entering the leachate pit, the region around the pit was covered with a plastic membrane folded over eight times . The region around the plastic membrane was filled with soil and stone to stabilize it and to divert surface water away from the leachate collection system.

The sampling techniques for testing residuals and water for QA-QC tests were in accordance with EPA standards , which have been described in Chapter 3.

5.3 Results and Discussions

5.3.1 Physical and Chemical Characteristics

Field tests generally are used in evaluating the environmental characteristics of aged WTP dewatered residuals and leachates (O'Leary et al, 1986). TCLP tests were performed on WTP residuals and physical and chemical tests were conducted on both solid and leachate samples.

5.3.1.1 Paint Filter Test: Paint filter test was performed for all three samples collected during the installation of the minimonofill, six months after and one year later. It was found that there was no free liquid in all three samples. The solids content of the samples are shown in Table 5.2. The solids concentration of the samples were 19.0, 21.3, and 26.41%, respectively. All these samples possessed solids contents close to 20%, which has been observed in municipal solid waste (MSW) landfills.

5.3.1.2 Solids Content: Total solids, fixed solids, and volatile solids were determined in accordance with the Standard Methods (APHA, AWWA, and WEP 1992). The results of three residual samples are shown in Table 5.2. There was no significant change in solids content among the fresh, six-month, and one year old samples taken from the same layer. The total solids concentration of the one year old sample taken from the top layer was 66.7%, which is 2.52 times higher than that of the same aged sample present six inch below the top surface of minimonofill. The increase of solids content was probably caused by the freeze thaw effects and formation of ice during the winter season, which drove some water out from the residual. The weather had more effect on the residuals laid on the top 6-inches of the monofill

Table 5.2 shows that there was little change of the volatile solids concentration among all three samples. Most organics remained in the WTP residuals. Furthermore very little biodegradation took place and/or organic material leached out during the one year study period.

5.3.1.3 pH: The change of residual pH was observed and shown in Table 5.2. The pH values of the residuals were 6.4, 6.8, and 7.1, respectively, during the period of field test.

The reason for this slightly increased pH in the top portion of the model monofill is unknown.

5.3.2 TCLP

Test results of the thirty nine TCLP regulatory metals, pesticides, herbicides, BNA, VOCs and concentration limits are shown in Table 5.3. It can be seen from these tables that the concentrations of all of these metals and toxic organic compounds in all of the WTP residuals samples did not exceed TCLP limits. As shown in Table 5.3, the majority of regulated metals such as arsenic, cadmium, lead, mercury, selenium, and silver were not detected. Concentrations of other detected metals were much lower than the regulatory limits.

TCLP experiments conducted for the residuals collected from ten different treatment plants verified (as discussed in Chapter 4) that the WTP residuals were not hazardous. Cornwell et al. (1992) also carried out TCLP tests on three types of WTP residuals and found that they were not hazardous. Results of samples collected from the field during the one year study also confirmed the residuals to be nonhazardous. Some of the water treatment plants in the U.S. send their residual samples to certified laboratories for TCLP testing. Their results are similar to those obtained in this study.

The results of TCLP tests are not surprising since nontoxic chemicals are always added during the water treatment process. The only possibility of introducing toxic materials to the residuals are from the water sources and impurities in the coagulants or water softeners. This should be tested for and not allowed by the specifications. However, the processes of coagulation, water softening, sedimentation, conditioning, and dewatering are similar to the processes commonly used for solidification/stabilization (S/S) for hazardous wastes. Especially, lime, which is commonly used in coagulation, water softening, and chemical conditioning, is a solidifying agent used in S/S process. Flocculation, compaction, and chemical binding would trap all toxic chemicals, if any, inside the residual matrix and make them difficult to leach out.

Cornwell et al. (1992) proposed to use a relatively simple and inexpensive standard metals analysis. A similar test to determine the total amount of metals and organic compounds for the 39 regulated compounds and elements were conducted in this study for the alum residual (one year old) taken from the New Castle Water Treatment Plant (sample NCFRS3).

Total metals in the residuals were determined by extracting the residual samples by conducting microwave acid digestion procedures (EPA method 3051) and analyzing the extract with ICP/MS. BNA, pesticides, and herbicides were extracted by the soxhlet extraction method and analyzed by a GC (EPA method 3540A) and a GC/MS (EPA method 8270A). The total volatile organics in the samples were determined by direct purge-trap and GC/MS (EPA method 8240).

The results of these experiments are presented in Tables 5.4. Of the eight regulated metals, five of them were not detected. The total concentrations of these three metals present were much less than the TCLP regulated limits. For example , the concentration of barium in the WTP residual was 14.486 mg/L, which was only 14.049 percent of the regulated limit. The results in Table 5.4 also show that the total concentrations of pesticides, herbicides, BNA, and VOCs were all less than the method detection limits (MDLs) and much less than the TCLP regulatory limits.

Using dry solids as the basis, the absolute amount of maximum regulatory limits of 39 defined constituents were calculated. Total analysis, TCLP analysis for sample NCFRS3, and maximum regulatory limits were summarized in Table 5.5 on dry weight basis. Both total and TCLP analyses for sample NCFRS3 indicated that the concentrations of TCLP defined constituents present were much less than maximum regulatory limits.

5.3.3 Metals in WTP residuals

Metals are common constituents in raw water and certain coagulants and dewatering conditioning reagents. It has been mentioned in Chapter 4 that the concentrations of some metals in WTP residuals were relatively high. These include aluminum, iron, and calcium. However, one of the concerns is whether these metals will leach out during disposal or land application. Elliott et al. (1990) indicated that aluminum and iron hydroxides are strong adsorbents of inorganic phosphates. Application of WTP residuals to agricultural soils will have an impact on the availability of phosphates for plants.

Cornwell and Koppers (1990) investigated the presence of metals in residuals. from presedimentation tank. The source of water treated in this case was surface water. The average concentrations of metals were: 10.4% iron, 6.3% calcium, 1.4% aluminum, and 0.13% manganese. They also determined that the residuals from backwash water with groundwater as water source had average concentration of 25.5% iron, 6% calcium, 1.0%

aluminum, and 0.9% manganese in the Netherlands. Dempsey (1992) also indicated that concentrations of 10.32% aluminum and 2.96% iron were found in alum residuals. Total concentrations of iron and aluminum were 8.35% and 2.51% for iron residuals in the USA. The total metals in sample NCFRS3 (alum residual tested in this study) and from two other sources are shown in Table 5.6. The concentration of aluminum was 8.63%, which is comparable to what Cornwell et al. (1992) and Such et al. (1988) found. Other major metals detected were iron (4.20%), calcium (1.72%), manganese (0.73%), and magnesium (0.40%). Concentrations of trace metals detected such as copper, zinc, chromium, and barium were very low. Other trace metals such as arsenic, cadmium, lead, mercury, selenium and silver were less than detection limits, This analysis showed that that alum residuals were primarily composed of aluminum, iron, and calcium, which are 15.0% (by weight) of the total. Manganese and magnesium each makes up less than 1% (by weight). Trace metals account for less than 0.1% (by weight) of the total.

It was also found in the analysis that the total phosphorus was 1.59% which was more than the average value of 0.2% Dempsey (1992) reported. He analyzed 71 samples from 7 treatment plants; 4 plants used ferric coagulant and 3 used alum.

The primary metals such as aluminum, calcium, iron, manganese, and magnesium, in various WTP residuals were analyzed in this study. The results show in Table 5.7. The lime WTP residual has concentrations of calcium of 17.54%, iron of 2.76%, magnesium of 1.40%, manganese of 1.28%, and iron of 0.93% on dry weight basis, respectively. The very high calcium concentration was attributed to the use of lime for softening as a conditioning agent or coagulant in this residual. The alum residual has concentrations of aluminum of 10.27%, iron of 1.50%, calcium of 0.94%, magnesium of 0.21%, and manganese of 0.58%, respectively. High aluminum concentration can be due to the addition of alum as coagulant. The ferric residual had a concentration of iron of 21.36%, which can be attributed to the ferric coagulant. These analyses indicated that the composition of primary metals was related to addition in the WTP residuals of coagulants and conditioning agents. These metals are retained in the WTP residuals.

Leaching tests are generally used in evaluating the movement and the amount of contaminants released from a waste. Generally, there are three types of leaching tests used, extraction tests,(such as TCLP), lysimeter column tests, and field studies. In this study, the data from TCLP and total metal analysis were used to determine the amount of metals leached out. Leaching potential is defined as the ratio of metals detected in TCLP to the total metals detected by the acid digestion method (EPA method 3051). Leaching capacities for samples tested in this study are shown in Tables 5.8 and 5.9. It can be seen from this table that manganese has the highest leaching capacity, (more than 90%). This was followed by calcium (43.46%), sodium (35.75%), aluminum (35.75%), magnesium (22.33%), and iron (6.82%).

The amount of metals retained in residuals depends on the form of the metals and the anion group they are associated with. In other words, it depends on the compounds formed during the chemical reactions. Metals, such as Mn, Mg, Ca, and Fe, may associate with carbonates ($MnCO_3$, $MgCO_3$, $CaCO_3$, or $FeCO_3$), hydroxides ($Fe(OH)_3$ or $Mg(OH)_2$) or with organic matter. In water treatment residuals, metals to be observed for leaching are aluminum and iron. When placed in monofills, their leaching capacities are controlled, among other things, by their attenuation characteristics.

5.4 Metal Leaching Capability

5.4.1 Principle of Metal Leaching Capability

1. Attenuation of Aluminum

In order to study the attenuation of aluminum, it is necessary to know how aluminum is present in the residuals. Analysis of the amphoteric behavior of aluminum and the way alum treatment of wastewater caused flocculation may give an insight to this (Achari and Joshi 1995). Aluminum, an amphoteric metal, hydrolyzes to a large number of monomeric and polymeric compounds and many other compounds depending on the sample pH and other factors. (Hsu and Bates 1964). In most of the forementioned forms, aluminum exists as an ion in solution except when it is in the form of $Al(OH)_3$, where it may precipitate out, because the solubility of $Al(OH)_3$ in water is very low. In low pH solutions, aluminum hydroxide exists as cations and at high pH solutions it exists as anions (Bradley 1991). When aluminum sulfate is used for coagulation and flocculation of water purification, the pH is brought to a range between 4.5 and 8.0 by adding $Ca(HCO_3)_2$ so that aluminum hydroxide may form. Since the aluminum hydroxide is relatively insoluble in this range, it precipitates out causing the formation of flocs (Reynolds 1982). In the WTP monofill, the pH range was always between 4.5 and 8.0 and hence the aluminum hydroxide did not become soluble. This means that aluminum may not leach out.

The major attenuation mechanism of aluminum is precipitation. In a highly alkaline or near neutral environment, aluminum readily forms insoluble oxides, hydroxides and silicates. However, in a low pH environment, aluminum is quite soluble. The mobility of aluminum in a clayey environment is low (Griffin 1977). It was pointed out in Chapter 4 that WTP residuals possessed good buffering capacity. The pH, even under acidic rainfall conditions was near neutral. So it will be difficult to leach aluminum out under these conditions from the monofill. Moreover the insoluble substances formed will tend to act as a retarding layer for leachate flow inside a monofill.

In water treatment plants, alum is added as a flocculating agent and it forms an alkaline bulky precipitate of aluminum hydroxide. Therefore alum is most effective for water with pH values ranging from 5.5 to 7.2 (Packham 1962). Aluminum hydrolysis will produce highly charged molecules such as $Al_8(OH)_{20}^{+4}$, causing the aluminum hydroxide to precipitate as interparticulate bridges are formed. There is adequate evidence in the literature to suggest that the basic structure of an alum residual is crystalline. Weisner (1962) conducted X-ray diffraction studies of alumina floc and compared it with crystalline alumina, and found it to consist of hydrous mass of minute crystals of gamma-$Al_2O_3.H_2O$. Mackrle (1962) found that a precipitate was formed during coagulation with aluminum sulfate. Though the floc appears to be an amorphous material, it actually behaves as a crystalline material. This aluminum crystalline substance is a very strong sorbant and it has extremely low solubility, i.e. $Al(OH)_3$ ($K_{sp}=10^{-33.5}$) and $Al_3(OH)_4^{+5}$ ($K_{sp}=10^{-42.7}$). It has a stable structure and prevents most aluminum from coming out.

2. *Attenuation of Iron*

Precipitation, cation exchange, adsorption and biological uptake are the important attenuation mechanisms of iron. Below approximately neutral pH conditions, the solubility of divalent iron increases about 100 fold for each unit of decrease in pH (Lindsay 1972). Iron oxides have been found to be among the most significant factors influencing attenuation processes. In general, iron compounds appeared to be moderately attenuated in soil (Fuller and Korte 1976). Although Griffin et. al (1976a) found no significant correlation with cation exchange, Farquhar (1977, cited in Gebhard 1978) found that iron attenuation did increase with the cation exchange capacity of the soil.

Table 5.9 lists the solubility products of metals associated with carbonate, hydroxides, and sulfides, and the leaching capacity of the NCFRS. Leaching capacity and solubility product are placed in the descending order as follows:

Leaching Capacity: Mn > Ca > Al > Mg > Fe

Solubility Products (pK_{sp})of $(M^{n+})(OH^{-})^{n}$ Ca > Mg > Mn > Al > Fe

It can be seen that the release of metals is closely related to their solubility products. It is interesting also that this observation is very similar to that obtained from a test conducted earlier (Cornwell et al. 1992). In that study, the sequence of the seven metals tested by TCLP leaching test was found to be:

Mn > Ca > Na > Al > Mg > Fe

Several factors were considered to affect the release or remobilization of trace elements from WTP residuals. These include the change of pH, the existence of complex-forming substances, alteration of redox potential or microbiological activity, and salinity (Cornwell and Koppers 1990).

Reduction of the pH is a significant factor influencing the remobilization potential (Cornwell and Koppers 1990). Around pH 7, trace metals are insoluble. A reduction in the pH (pH < 5) may break down carbonates and hydroxides and release the elements. It has been noted in Chapter 4 that the WTP residuals have some pH buffering capacity, and that weather may only affect the top 6 inches. In reality, the pH of the residuals may not change very much in the field.

Change of redox potential (negative) may increase the release of arsenic, iron, and manganese (Cornwell and Koppers 1990). Disposal of WTP residual in lumps in a monofill and low permeability of the residual will make the contact of the reducing agent , if any, with residuals in the field difficult.

It has been shown that synthetic complexors such as the phosphates substitutes, NTA, ethylene diaminete traacetic acid (EDTA), and humic acid are capable of remobilizing the elements contained in WTP residuals. In the Netherlands and Germany, a policy of leaving a minimum distance of 3.3 ft (1.0 M) between the bottom of the landfill and the highest groundwater level limits the possible influence of humic acid contained in groundwater (Cornwell and Koppers 1990).

The leaching capacities of fresh, six-month, and one year old alum residuals are shown in Table 5.10. The leaching capacity of metals in descending sequence for these three samples are presented as follows:

Fresh sample: Mn > Al > Mg > Fe > Ca
Six-month old sample: Mn > Al > Ca > Mg >Fe
One-year old sample: Ca > Mn > Al > Mg > Ni >Cu > Zn > Fe

WTP monofill disposal is better than MSW landfill disposal based on the leachbility studies and leaching model prediction. Table 5.5 shows that the concentration of aluminum was 8.63% in the NCFRS sample, on dry weight basis. The TCLP tests showed that aluminum leached out immediately after placement, six months after placement, and after one year after placement were found to be 23.00, 23.65, and 18.39 mg/g respectively. These results indicate that approximately 75% of the aluminum remained in the residuals and that the total amount that can be leached out did not change much during the one year period of this study.

Ferrous hydroxide and ferrous carbonate may be the main forms of iron present in NCFRS sample tested in this study. The solubility of iron depends on the redox potential. $Fe(OH)_2$ ($K_{sp} = 10^{-15.2}$) is much more soluble than $Fe(OH)_3$ ($K_{sp} = 10^{-39.0}$).

5.4.2 Leachate Analysis

Two water samples were collected from the leachate collection system. The first one (NCFWS2) was taken on August 2, 1993 and the second (NCFWS3) on October 21, 1994. Concentrations of metals in these samples are shown in Table 5.11. The results show that concentrations of most of the heavy metals in the leachate were below detection limits. The concentrations of the chromium and nickel were very small, (close to field blanks or method detection limits).

Table 5.11 also shows the concentration of iron to be low. This is consistent with the data in Table 5.9, which indicates that iron leached out in very small quantities. Cornwell et al. (1992) also reported similar observations from their lysimeter column tests.

The results of analysis of pesticides, herbicides, BNAs, and VOCs are listed in Table 5.12. It can be seen from the table that concentrations of all organic compounds in leachate are below method detection limits.

5.5 Geotechnical Characteristics

Effects of Aging and Weathering on Geotechnical Properties of WTP Residuals

In this section, the effects of aging and weathering on geotechnical characteristics of the residual samples collected from the mini monofill at New Castle, Pennsylvania, were studied. The main items of interest from the geotechnical point of view were the durability characteristics (freeze-thaw and wet-dry) and the permeability characteristics.

The liquid limit of the sample NCFG3 was 125 when the test was conducted from wet to dry side. The corresponding value based on tests conducted from dry to wet side was

82. This sample in the air-dry condition behaved like a granular material due to the change in floc structure, loss of organic content and development of cementation upon drying. Hence its plastic limit could not be determined. From this observation, it can be inferred that when thoroughly dried, WTP residuals lose plasticity. However, as observed in samples NCFG2W and NCFG3W, this change can also be brought about by freeze and thaw at a much faster rate than by air drying or aging.

Handling characteristics of WTP residuals are very poor in the natural condition. They can be improved by increasing the solids content. The compactability of these materials is also very poor. It has been noticed that the optimum moisture contents of these materials that exhibit one hump type of compaction curve, based on wet to dry tests, are close to their respective plastic limit values. This means that if the natural water content of any given WTP residual is close to its plastic limit, the residual can be compacted easily in its natural condition. If the natural moisture content is greater than the plastic limit, it will be almost impossible to compact the material in its natural condition.

Based on durability test results, it was concluded that WTP residuals were susceptible to deteriorate under freeze-thaw and wet-dry conditions. Hence a final cover for WTP residual monofills, if required, can not be constructed with the residual material itself. For this purpose, a material not affected by weathering has to be utilized.

Table 5.1 Symbols for WTP residual and leachate samples from WTP mini-monofill

Symbols	Description	Date of Collection
NCFRS1	Residual sample collected from WTP mini-monofill in the first phase of study	10/19/93
NCFTB1	Trip blank in the first phase of study	10/19/93
NCFFB1	Field blank in the first phase of study	10/19/93
NCFRS2	Residual sample collected from WTP mini-monofill in the second phase of study	4/22/94
NCFWS2	Water sample collected from the leachate collection system in the second phase of study	4/22/94
NCFTB2	Trip blank in the second phase of study	4/22/94
NCFFB2	Field blank for the second phase of study	
NCFRS3	Residual sample collected from WTP mini-monofill in the third of phase study	10/21/94
NCFWS3	Water sample collected from the leachate collection system in the third phase of study	8/2/94
NCFTB3	Trip blank in the third phase of study	10/21/94
NCFFB3	Field blank in the third phase of study	10/21/94

Table 5.2 Physical and chemical characteristics of residuals in mini-monofill

Parameter	NCFRS1 (%)	NCFRS2 (%)	NCFRS3 (%)	NCFRT (%)
Solids Content	19.0	21.3	26.4	66.7
Volatile Solids	18.3	20.6	20.5	18.6
Fixed Solids	81.7	79.4	79.5	81.4
pH	6.4	6.8	7.1	7.3
CEC (meq/100g)	112.8	106.7	102.3	-

Note: NCFRT: One-year old residual collected from the top layer of the mini-monofill.

-: Not applicable

Table 5.3 Results of TCLP analysis on residual samples collected from mini-monofill

		TCLP regulated Metals			
Constituent	Regulatory (mg/L)	NCFRS1 (mg/L)	NCFRS2 (mg/L)	NCFRS3 (mg/L)	MDL* (mg/L)
Arsenic	5.0	< 0.069	< 0.069	0.010	0.069
Barium	100.0	2.566	2.695	12.023	0.078
Cadmium	1.0	ND	ND	ND	0.006
Chromium	5.0	0.112	0.359	0.344	0.044
Lead	5.0	ND	ND	ND	0.033
Mercury	0.2	ND	ND	ND	0.012
Selenium	1.0	ND	ND	ND	0.001
Silver	5.0	ND	ND	ND	0.033
Chlordane	0.03	< 0.010	< 0.010	< 0.010	0.010
Endrin	0.02	< 0.008	< 0.008	< 0.001	0.001
Heptachlor	0.008	< 0.008	< 0.008	< 0.001	0.001
Heptachlor epoxide	0.008	< 0.008	< 0.008	< 0.005	0.005
Lindane	0.4	< 0.050	< 0.050	< 0.001	0.001
Methoxyclor	10.0	< 0.050	< 0.050	< 0.010	0.010
Toxaphene	0.5	< 0.050	< 0.050	< 0.005	0.005
2,4 D	10.0	< 0.001	< 0.001	< 0.001	0.001
2,4,5 TP (Silvex)	1.0	< 0.001	< 0.001	< 0.001	0.001
Hexachloroethane	3.0	< 0.001	< 0.001	< 0.001	0.001
o-Cresol	200.0	< 0.005	< 0.005	< 0.200	0.200
m-Cresol	200.0	< 0.005	< 0.005	< 0.200	0.200
p-Cresol	200.0	< 0.005	< 0.005	< 0.200	0.200

(continued)

Table 5.3 (continued)

		TCLP regulated Metals			
Constituent	Regulatory (mg/L)	NCFRS1 (mg/L)	NCFRS2 (mg/L)	NCFRS3 (mg/L)	MDL* (mg/L)
Cresol	200.0	< 0.005	< 0.005	< 0.008	0.200
2,4-Dinitrotoluene	0.13	< 0.005	< 0.005	< 0.008	0.008
Hexachlorobenzene	0.13	< 0.005	< 0.005	< 0.001	0.001
Hexachlorobutadiene	0.5	< 0.005	< 0.005	< 0.001	0.001
Nitrobenzene	2.0	< 0.005	< 0.005	< 0.003	0.003
Pentachlorophenol	5.0	< 0.005	< 0.005	< 0.100	0.100
2,4,5-Trichlorophenol	400.0	< 0.005	< 0.005	< 0.400	0.400
2,4,6 Trichlorophenol	2.0	< 0.005	< 0.005	< 0.002	0.002
Benzene	0.5	< 0.001	< 0.001	< 0.005	0.005
Carbontetrachloride	0.5	< 0.001	< 0.001	< 0.005	0.005
Chlorobenzene	100.0	< 0.001	< 0.001	< 0.005	0.005
Chloroform	6.0	0.002	< 0.001	< 0.005	0.005
1,2 Dichloroethane	0.5	< 0.001	< 0.001	< 0.005	0.005
1,1 Dichloroethylene	0.7	< 0.001	< 0.001	< 0.005	0.005
Methyl ethyl ketone	200.0	< 0.005	< 0.005	< 0.025	0.025
Tetrachloroethylene	0.7	< 0.001	< 0.001	< 0.005	0.005
Trichloroethylene	0.5	< 0.001	< 0.001	< 0.005	0.005
Vinyl chloride	0.2	< 0.002	< 0.002	< 0.010	0.010
1,4-Dichlorobenzene	0.2	< 0.001	< 0.001	< 0.005	0.005

*MDL	Method Detection Limit
ND	Not detected

Table 5.4 TCLP defined constituents present in residual samples collected from mini-monofill

Constituent	TCLP Regulatory (mg/L)	NCFRS Total Conc. (mg/L)	NCFRS TCLP Conc.* (mg/L)	MDL** (mg/L)
Arsenic	5.0	0.671	<0.069	0.069
Barium	100.0	14.486	5.761	0.078
Cadmium	1.0	< 0.006	< 0.006	0.006
Chromium	5.0	2.488	0.272	0.044
Lead	5.0	< 0.033	< 0.033	0.033
Mercury	0.2	< 0.012	< 0.012	0.012
Selenium	1.0	< 0.001	< 0.001	0.001
Silver	5.0	< 0.033	< 0.033	0.033
Chlordane	0.03	< 0.010	< 0.010	0.010
Endrin	0.02	< 0.008	< 0.001	0.001
Heptachlor	0.008	< 0.008	< 0.001	0.001
Heptachlor epoxide	0.008	< 0.008	< 0.005	0.005
Lindane	0.4	< 0.050	< 0.001	0.001
Methoxyclor	10.0	< 0.050	< 0.010	0.010
Toxaphene	0.5	< 0.050	< 0.005	0.005
2,4 D	10.0	< 0.001	< 0.001	0.001
2,4,5 TP (Silvex)	1.0	< 0.001	< 0.001	0.001
Hexachloroethane	3.0	< 0.001	< 0.001	0.001
o-Cresol	200.0	< 0.005	< 0.200	0.200
m-Cresol	200.0	< 0.005	< 0.200	0.200
p-Cresol	200.0	< 0.005	< 0.200	0.200

(continued)

Table 5.4 (continued)

Constituent	TCLP Regulatory (mg/L)	NCFRS Total Conc. (mg/L)	NCFRS TCLP Conc.* (mg/L)	MDL** (mg/L)
Cresol	200.0	< 0.005	< 0.008	0.200
2,4-Dinitrotoluene	0.13	< 0.005	< 0.008	0.008
Hexachlorobenzene	0.13	< 0.005	< 0.001	0.001
Hexachlorobutadiene	0.5	< 0.005	< 0.001	0.001
Nitrobenzene	2.0	< 0.005	< 0.003	0.003
Pentachlorophenol	5.0	< 0.005	< 0.100	0.100
2,4,5-Trichlorophenol	400.0	< 0.005	< 0.400	0.400
2,4,6 Trichlorophenol	2.0	< 0.005	< 0.002	0.002
Benzene	0.5	< 0.001	< 0.005	0.005
Carbontetrachloride	0.5	< 0.001	< 0.005	0.005
Chlorobenzene	100.0	< 0.001	< 0.005	0.005
Chloroform	6.0	< 0.001	< 0.005	0.005
1,2 Dichloroethane	0.5	< 0.001	< 0.005	0.005
1,1 Dichloroethylene	0.7	< 0.001	< 0.005	0.005
Methyl ethyl ketone	200.0	< 0.005	< 0.025	0.025
Tetrachloroethylene	0.7	< 0.001	< 0.005	0.005
Trichloroethylene	0.5	< 0.001	< 0.005	0.005
Vinyl chloride	0.2	< 0.002	< 0.010	0.010
1,4-Dichlorobenzene	0.2	< 0.001	< 0.005	0.005

* Average value of TCLP concentration of three samples

**MDL Method Detection Limit

Table 5.5 Results of total analysis versus TCLP analysis on residual samples collected from mini-monofill

Constituent	Maximum* regulatory limit (mg/Kg)	NCFRS Total analysis** (mg/Kg)	NCFRS TCLP analysis*** (mg/Kg)
Arsenic	370.370	33.550	5.111
Barium	7407.407	724.300	426.741
Cadmium	74.074	< 0.300	< 0.444
Chromium	370.370	124.400	20.148
Lead	370.370	< 1.650	< 2.444
Mercury	14.815	< 0.600	< 0.889
Selenium	74.074	< 0.050	< 0.074
Silver	370.370	< 1.650	< 2.444
Chlordane	2.222	< 0.500	< 0.741
Endrin	1.481	< 0.400	< 0.074
Heptachlor	0.593	< 0.400	< 0.074
Heptachlor epoxide	0.593	< 0.400	< 0.370
Lindane	29.630	< 2.500	< 0.074
Methoxyclor	740.741	< 2.500	< 0.741
Toxaphene	37.037	< 2.500	< 0.370
2,4 D	740.741	< 0.050	< 0.074
2,4,5 TP (Silvex)	74.074	< 0.050	< 0.074
Hexachloroethane	222.222	< 0.050	< 0.074
o-Cresol	14814.815	< 0.250	< 14.815
m-Cresol	14814.815	< 0.250	< 14.815
p-Cresol	14814.815	< 0.250	< 14.815

(continued)

Table 5.5 (continued)

Constituent	Maximum * regulatory limit (mg/Kg)	NCFRS Total analysis** (mg/Kg)	 TCLP analysis*** (mg/Kg)
Cresol	14814.815	< 0.250	< 0.593
2,4-Dinitrotoluene	9.630	< 0.250	< 0.593
Hexachlorobenzene	9.630	< 0.250	< 0.074
Hexachlorobutadiene	37.037	< 0.250	< 0.074
Nitrobenzene	148.148	< 0.250	< 0.222
Pentachlorophenol	370.370	< 0.250	< 7.407
2,4,5-Trichlorophenol	29629.630	< 0.250	< 29.630
2,4,6 Trichlorophenol	148.148	< 0.250	< 0.148
Benzene	37.037	< 0.050	< 0.370
Carbontetrachloride	37.037	< 0.050	< 0.370
Chlorobenzene	7407.407	< 0.050	< 0.370
Chloroform	444.444	< 0.050	< 0.370
1,2 Dichloroethane	37.037	< 0.050	< 0.370
1,1 Dichloroethylene	51.852	< 0.050	< 0.370
Methyl ethyl ketone	14814.815	< 0.250	< 1.852
Tetrachloroethylene	51.852	< 0.050	< 0.370
Trichloroethylene	37.037	< 0.050	< 0.370
Vinyl chloride	14.815	< 0.100	< 0.741
1,4-Dichlorobenzene	14.815	< 0.050	< 0.370

* Maximum regulatory limit based on 27 percent of solids content

** Total analysis based on EPA methods

*** TCLP analysis based on EPA TCLP extraction methods

Table 5.6 Total metals in alum residual

Constituent		Such et al.[1]	Cornwell et al.[2]	NCFRS
Aluminum	(%)	14.5	10.70	8.63
Iron	(%)	1.8	4.85	4.20
Calcium	(%)	3.0	-	1.72
Magnesium	(%)	0.65	-	0.40
Manganese	(%)	0.99	0.12	0.73
Sodium	(%)	-	-	0.44
Zinc	(mg/g)	-	0.092	0.02
Copper	(mg/g)	-	0.168	0.04
Nickel	(mg/g)	-	0.024	0.03
Arsenic	(mg/g)	-	0.025	0.03
Barium	(mg/g)	-	0.030	0.72
Chromium	(mg/g)	-	0.120	0.12
Phosphorus	(mg/g)	-	-	1.59

1. Such et al. 1988.
2. Cornwell et al. 1992.
- Applicable.

Table 5.7 Total metals in WTP residuals

Constituent		Lime[1]	Alum[2]	Ferric[3]
Aluminum	(%)	2.76	10.27	1.17
Iron	(%)	0.93	1.50	21.36
Calcium	(%)	17.54	0.94	0.50
Magnesium	(%)	1.40	0.21	0.12
Manganese	(%)	1.28	0.58	0.62

1. Average value of JCD, PVD, MWD, and SLD
2. Average value of WQD, HWD, MQD, and FLD
3. Average value of RWA

Table 5.8 Metal leaching capacities of alum residual

Constituent	TCLP Metals* (mg/g)	Total Metals** (mg/g)	Leaching Capacity (%)
Manganese	6.73	7.33	91.81
Calcium	7.48	17.21	43.46
Sodium	1.58	4.41	35.75
Aluminum	21.68	86.33	25.11
Magnesium	0.90	4.03	22.33
Nickel	0.006	0.030	20.00
Iron	2.86	42.01	6.82
Zinc	0.74	0.22	-
Copper	0.18	0.04	-

* Average of TCLP regulated metals from fresh, six-month, and one-year old samples.

** Total metals from fresh sample.

\- Data not available.

Table 5.9 Relationship between leaching capacity and solubility

Constituent	Leaching Capacity (%)	Solubility Products of Metal (pK_{sp}) $(M^{n+})(OH^-)^n$	$(M^{2+})(CO_3^{2-})$	$(M^{2+})(S^{2-})$
Manganese	91.81	12.8	10.4	13.5
Calcium	43.46	5.1	8.42	
Aluminum	25.11	31.2	-	-
Magnesium	22.33	11.2	4.40	-
Nickel	20.00	15.7	-	-
Iron	6.82	39 (Fe^{+3})		
		15.2 (Fe^{+2})	10.7	18.1
Zinc	-	16.9	10.2	24.7
Copper	-	20.3	-	36.1

\- Data not available

Table 5.10 Leaching capacity of metals of alum residual in Mini-monofill

Constituent	Leaching Capacity NCFRS1 (%)	 NCFRS2 (%)	 NCFRS3 (%)
Manganese	109.55	90.18	75.72
Calcium	8.31	25.33	96.74
Sodium	9.07	28.12	70.07
Aluminum	26.64	27.40	21.30
Magnesium	24.57	21.84	20.42
Nickel	ND	ND	20.00
Iron	13.88	3.38	3.19
Zinc	-	-	10.45
Copper	-	-	13.64

ND: Not detected
- Data not available

Table 5.11 Analysis of metals* present in leachate

Constituent	NCFWS2 (mg/L)	NCFWS3 (mg/L)	TBLK (mg/L)	FBLK (mg/L)	MDL (mg/L)
Arsenic	< 0.006	ND	ND	ND	0.069
Barium	0.086	0.010	ND	ND	0.078
Cadmium	0.007	ND	ND	0.002	0.006
Chromium	0.041	0.003	ND	ND	0.044
Lead	0.068	0.102	0.005	0.011	0.033
Mercury	ND	ND	ND	0.020	0.011
Selenium	ND	ND	ND	0.026	0.001
Silver	ND	ND	ND	ND	0.033
Aluminum	0.901	3.285	0.120	0.135	0.069
Iron	0.465	0.879	0.055	0.108	0.019
Calcium	217.504	105.717	0.480	0.958	0.885
Sodium	4.470	12.325	0.286	0.391	0.203
Manganese	1.007	2.096	0.006	0.010	0.003
Zinc	< 0.044	0.023	ND	ND	0.044
Magnesium	0.440	21.915	0.004	0.020	0.003
Copper	0.127	ND	ND	ND	0.012
Nickel	ND	ND	ND	ND	0.025
Phosphorus	-	14.198	0.648	1.094	1.080
pH	6.35	6.91	6.26	6.16	

* All metals are extracted by EPA Standard Acid Digestion and determined by ICP-MS.

MDL Method detection limit

ND Not detected

Table 5.12 Analysis of pesticides, herbicides, NBA, and VOCs present in leachate from minimonofill

Constituent	NCFWS2 (mg/L)	NCFWS3 (mg/L)	TBLK (mg/L)	FBLK (mg/L)	MDL (mg/L)
Chlordane	< 0.005	< 0.010	< 0.010	< 0.010	0.010
Endrin	< 0.005	< 0.008	< 0.008	< 0.008	0.001
Heptachlor	< 0.005	< 0.008	< 0.008	< 0.008	0.001
Heptachlor epoxide	< 0.005	< 0.008	< 0.008	< 0.008	0.005
Lindane	< 0.005	< 0.050	< 0.050	< 0.050	0.001
Methoxyclor	< 0.005	< 0.050	< 0.050	< 0.050	0.010
Toxaphene	< 0.005	< 0.050	< 0.050	< 0.050	0.005
2,4 D	< 0.005	< 0.001	< 0.001	< 0.001	0.001
2,4,5 TP (Silvex)	< 0.005	< 0.001	< 0.001	< 0.001	0.001
Hexachloroethane	< 0.005	< 0.001	< 0.001	< 0.001	0.001
o-Cresol	< 0.100	< 0.005	< 0.005	< 0.005	0.200
m-Cresol	< 0.100	< 0.005	< 0.005	< 0.005	0.200
p-Cresol	< 0.100	< 0.005	< 0.005	< 0.005	0.200
Cresol	< 0.100	< 0.005	< 0.005	< 0.005	0.200
2,4-Dinitrotoluene	< 0.004	< 0.005	< 0.005	< 0.005	0.008
Hexachlorobenzene	< 0.001	< 0.005	< 0.005	< 0.005	0.001
Hexachlorobutadiene	< 0.001	< 0.005	< 0.005	< 0.005	0.001
Nitrobenzene	< 0.001	< 0.005	< 0.005	< 0.005	0.003
Pentachlorophenol	< 0.050	< 0.005	< 0.005	< 0.005	0.100
2,4,5-Trichlorophenol	< 0.200	< 0.005	< 0.005	< 0.005	0.400
2,4,6 Trichlorophenol	< 0.010	< 0.005	< 0.005	< 0.005	0.002
Benzene	< 0.001	< 0.005	< 0.001	< 0.001	0.005
Carbontetrachloride	< 0.001	< 0.005	< 0.001	< 0.001	0.005
Chlorobenzene	< 0.001	< 0.005	< 0.001	< 0.001	0.005
Chloroform	0.001	< 0.005	< 0.001	< 0.001	0.005
1,2 Dichloroethane	< 0.001	< 0.005	< 0.001	< 0.001	0.005
1,1 Dichloroethylene	< 0.001	< 0.005	< 0.001	< 0.001	0.005
Methyl ethyl ketone	< 0.005	< 0.005	< 0.005	< 0.005	0.025
Tetrachloroethylene	< 0.001	< 0.005	< 0.001	< 0.001	0.005
Trichloroethylene	< 0.001	< 0.005	< 0.001	< 0.001	0.005
Vinyl chloride	< 0.002	< 0.005	< 0.002	< 0.002	0.010
1,4-Dichlorobenzene	< 0.001	< 0.005	< 0.001	< 0.002	0.005

MDL: Method Detection Limit

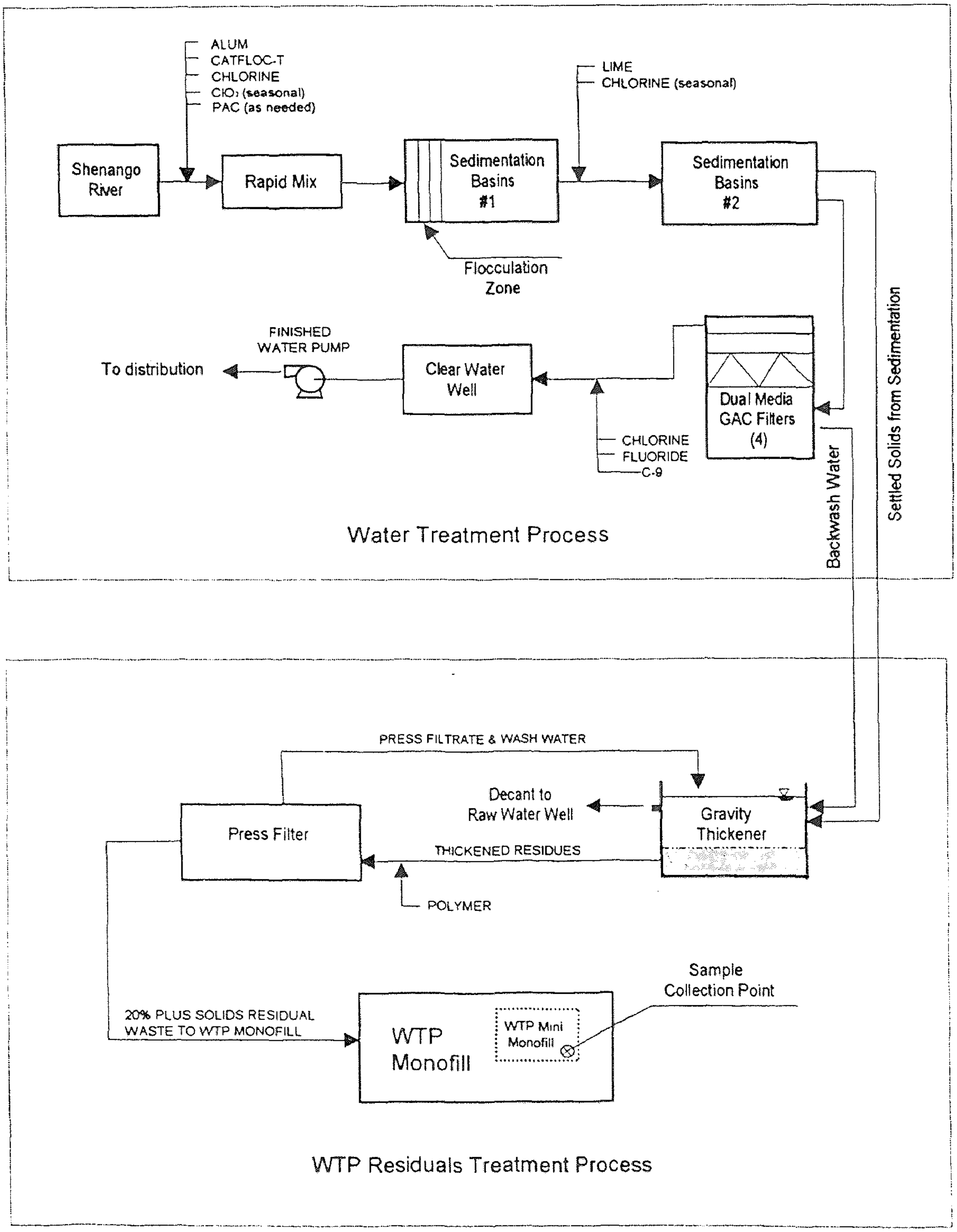
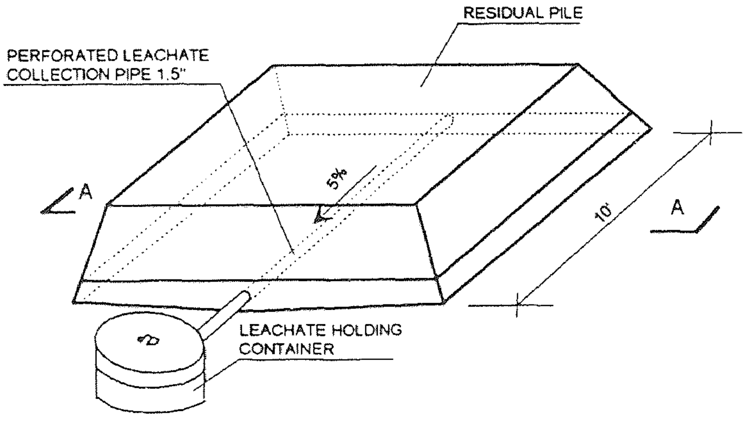


Figure 5.1 Flow diagram of New Castle Water Treatment Plant, Pennsylvania American Water Company at New Castle, PA



TOP-VIEW
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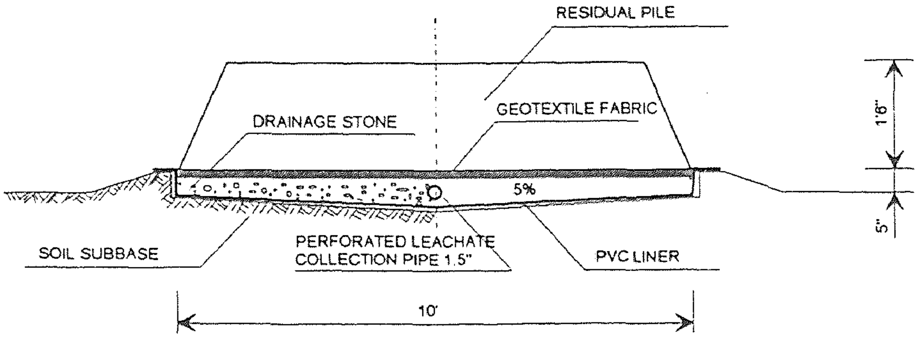

CROSS SECTION A-A
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Figure 5.2 The mini monofill and the leachate collection system

CHAPTER 6

METAL LEACHING MODEL DEVELOPMENT FOR WTP RESIDUALS

6.1 Introduction

Based on the laboratory and field TCLP study, the toxic organics and trace metals are not concentrated in WTP dewatered residuals and thus are not major constitutes in residuals. However the treatment of drinking water involves addition of alum, iron, or lime as coagulants to remove suspended solids, the resulting water treatment residuals consist primarily of the precipitated hydroxides or carbonates, such as ($Al(OH)_3$, $Fe(OH)_3$, and $Ca(CO)_3$. The major concern in water facilities is whether these primary metals, such as aluminum, iron, calcium, will leach out from the residuals. Since the leaching of such metals will impact on the environment and affect the methods of final disposal. Therefore, it is essential to develop metal leaching model for WTP residuals to understand their migration in WTP monofill or MSW landfill.

Leach models can be used to predict long-term leaching behavior in the environment based on laboratory leaching test results. Currently, leaching tests, such as the U.S. EPA Extraction Procedure (EP) or Toxicity Characteristic Leaching Procedure (TCLP), can only measure the amount of metals extracted and not the rate of leaching. Additional extraction tests are required to determine the leaching rate.

This chapter presents the development of a leaching model for major metal ions in WTP residuals using diffusion theory, and the experimental data from multiple batch extraction leaching tests are used to predict rates of metal leaching and leachate metal concentrations in WTP residuals.

6.2 Thermodynamics and Kinetics of Leaching

Most leaching models were developed by assuming equilibrium conditions existing in leaching process (Machiels and Pescatore 1982, Bishop, Gress, and Olofsson 1982). The second assumption made this study is that the residual is homogeneuens, that is the chemical concentration of any given species is the same anywhere in the solid, as well as on the solid surface. When the solid is placed in water, a flux species will move from the residual surface into water due to the concentration gradient. Once the species moves from the surface to water, the migration of species from the interior partion starting moving toward the surface. The third assumption is that solid and liquid is completely mixed. Based on this assumption, the diffusion coefficient and particle size can be obtained precisely and the problems of compaction and wall effect (Ham et al. 1979) can be avoided. But this is not real situation in the monofill or landfill. This is worst case situation of the leaching tests. The metal leaching rate may actually be much less in the field.

This driving force that moves a species from the surface of the solid into the water is proportional to the difference in chemical concentrations. In a batch extraction leaching test with no additional leachant added, as is the case with the EPA TCLP or EP tests, saturation of the aqueous solution with the species will eventually be achieved at equilibrium. The flux of the species diffused inside the solid can be described by Fick's first law of diffusion. In multiple extraction leaching test or continuous flow leaching test, a driving force is re-established or maintained between the solid and the liquid and moving of mass flux of species continues. The leaching rate is defined as the amount leached per unit surface area of the specimen and per unit time.

Several factors may affect the leaching rates. These include particle size, characteristics of residual, leachant composition, pH and flow conditions.

Particle size determine the surface area of the solids. Leaching of ions is a desorption process and it is controlled by the surface phenomenon.

Characteristics of residual determine the mobility of the contamination in a WTP residuals. As it is shown in Chapter 4, soluble species are mobile while precipitated forms of the same species or insoluble species are immobile.

Leachant composition will affects solubility of salt in residual. Its pH values, buffering capacity, the presence of complexing or chelating agents, and the redox potential of the system will all affect if the cations will precipitate.

pH probably has the most important effect on leaching. At higher pH, most metal cations form insoluble precipitates, while at lower pH these ions become solubilized.

Leachant flow conditions establishes the rate at which contaminants are washed out from the residuals.

For a monofill, distilled, deionized water or a synthetic rainwater can be used as a representative extracting agent. Leaching study can be performed by exposing residuals to these waters.

6.3 Development of Metal Leaching Model of WTP Residuals

6.3.1 Fick's Law

The Fick's law explains that the diffusion of a flux of species is proportional to concentration gradient in one-dimensional expression:

$$J = -D_e \frac{\partial C}{\partial X} \tag{1}$$

Where: J = flux of component in x-direction, [M/L^{2}T];

D_e = effective diffusion-dispersion coefficient [L^2/T];

C = concentration of component, mass per of volume solid, [M/L^3];

X = distance in x-direction [L];

6.3.2 General Leaching Model of WTP Residuals

The leaching model of WTP Residuals under WTP monofill or MSW landfill can be expressed by mass balance and is illustrated as in Figure 6.1.

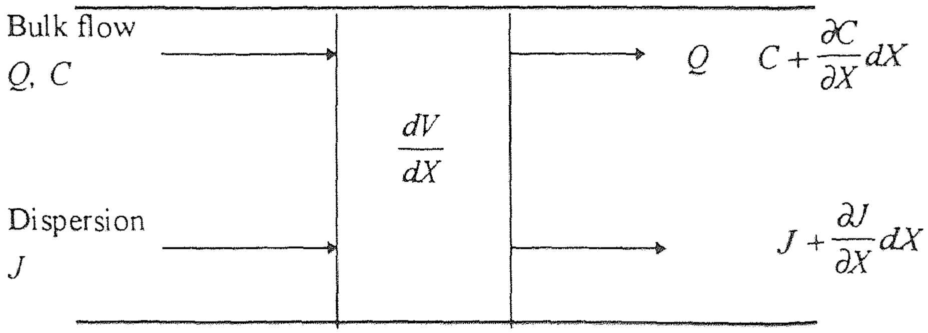

Figure 6.1 One dimensional model of mass balance of WTP residuals

The material balance can be formulated by a equation as follows:

Accumulation of Material = Amount of material entering

- Amount of material leaving

+ Net amount of material formed by reactions

$$dV\frac{\partial C}{\partial t} = QC + AJ - Q\left(C + \frac{\partial C}{\partial X}dX\right) - A\left(J + \frac{\partial J}{\partial X}dX\right) + rdV \qquad (2)$$

Where: C = concentration of component, [M/L^3];

dV = differential volume, [L^3];

t = leaching time [T];

Q = volumetric flowrate, [L^3/T];

X = distance, [L];

A= cross section area, [L^2];

J = flux of component, [M/L^{2}T];

r = reaction rate for component, [M/L^{3}T];

The equation can be simplified:

$$\frac{\partial C}{\partial t} = -u\frac{\partial C}{\partial X} - \frac{\partial J}{\partial X} + r \qquad (3)$$

Where: u = velocity (Q/A), [L^2/T];

Substituting Eq. (1) in Eq. (3) and resulting in

$$\frac{\partial C}{\partial t} = -u\frac{\partial C}{\partial X} - \frac{\partial}{\partial X}\left(-D_e\frac{\partial C}{\partial X}\right) + r \tag{4}$$

which is

$$\frac{\partial C}{\partial t} = -u\frac{\partial C}{\partial X} + D_e\frac{\partial^2 C}{\partial X^2} + r \tag{5}$$

6.3.3 Metal Leach Model of WTP Residuals

Since the composition of residual is homogeneous and that the convective velocity (u) is very small due to low permeabilities (as shown in Table 4.16), and if there is no chemical reaction occurred. The Eq. (5) can be simplified as:

$$\frac{\partial C}{\partial t} = D_e\frac{\partial^2 C}{\partial X^2} \tag{6}$$

As shown in Figure 6.2, if the constant *(L)* is defined as the distance from the edge of particle to its center, variable (X) is the distance from center of particle to outer surface, the initial condition and boundary condition can be expressed as follows:

Initial Conditions: $C(X,0)=C_0$ $\quad |X|<L$

$C(X,0)=0$ $\quad |X|>L$

Boundary Condition: $C(0,t)=0$ $\quad t\geq 0$

In Figure 6.2, *C(X,t)* represents the concentration profile of a given cations in the solid at distance *X* from the center of solid, at time *(t)*. The boundary conditions state that the concentration of the surface is equal to zero.

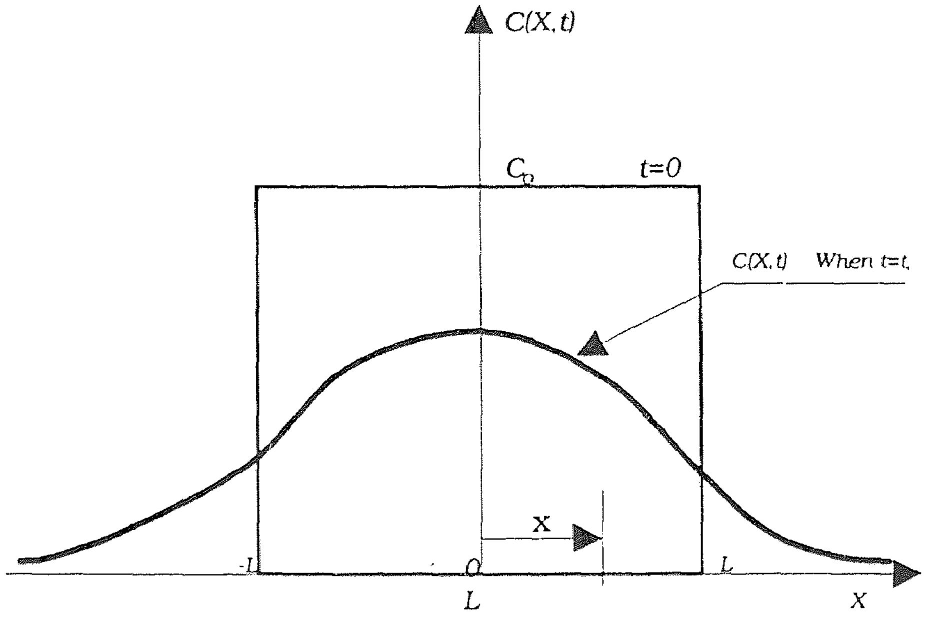

Figure 6.2 Solution C(X,t) of metals in solid phase for t=0 and t=t_i

The equation can be solved to give a profile of the concentration within the solids:

$$C(X,t) = C_0 \times erf\left(\frac{X}{(4D_e t)^{0.5}}\right) \tag{7}$$

Where:

C_0 = initial concentration of ions within solids when leaching starts, [M/L^3];

t = time [T];

erf is the error function defined by: (Fuller and Warrick, 1985)

$$erf(x) = \frac{2}{\sqrt{\pi}} \int_0^x e^{-x^2} dx \tag{8}$$

Equation (7) is useful in calculating the leached ions concentration from solid by integrating the concentration profile *(C(X,t))* across the entire particle. From Figure 6.2, the distance from center to particle edge *(L)*; the total ion mass in solid per unit surface area (M_o), the ion remained in solid per unit surface area (M_r), and ion leached at leaching time ($t=t_t$) can be expressed as follows:

$$\begin{aligned} M_o &= C_o \int_0^L dX \\ &= LC_o \end{aligned} \tag{9}$$

$$M_r = \int_0^L (C(X,t))dX \tag{10}$$

then, total ion leachant at time t is:

$$M_t = M_o - M_r = C_o \int_0^L dX - \int_0^L (C(X,t))dX \tag{11}$$

The fraction of contaminant leached from solid can be expressed as.

$$\frac{M_t}{M_0} = \frac{1}{LC_0}\int_0^L (C_0 - C(X,t))dX \tag{12}$$

Where: M_t/M_0 = fraction of contaminant leached at time t;

The solution of Eq. (12) is solved using Laplace Transform Method (Selby, 1975):

$$\frac{M_t}{M_o} = 1 - erf\left(\frac{L}{2\sqrt{D_e t}}\right) - 2e^{\left(\frac{L}{2\sqrt{D_e t}}\right)}\frac{\sqrt{D_e t}}{L\sqrt{\pi}} + 2\frac{\sqrt{D_e t}}{L\sqrt{\pi}} \tag{13}$$

Let $x = \dfrac{L}{2\sqrt{D_e t}}$

For large x, such as x >5, the simple approximation will be (Kreyszig, E. 1979)

$$erf\left(\frac{L}{2\sqrt{D_e t}}\right) = 1 - 2e^{\left(\frac{L}{2\sqrt{D_e t}}\right)}\frac{\sqrt{D_e t}}{L\sqrt{\pi}} \tag{14}$$

Then, Eq (13) can be simplified as:

$$\frac{M_t}{M_o} = 2\frac{\sqrt{D_e t}}{L\sqrt{\pi}} \tag{15}$$

If this model further assumes continual removal of the leachate from the system so that the concentration of ions do not build up in the leachate. This continuous process can be investigated with multiple batch or continuous flow leaching tests. In this study, multiple batch leaching tests were used, the model takes the form:

$$\frac{M_t}{M_o} = \frac{\sum_1^n a_n}{M_o} = \frac{2}{L}\left(\frac{D_e}{\pi}\right)^{0.5} t_n^{\,0.5} \tag{16}$$

Where a_n = metal loss in each leaching extraction, mg,

n = number of extraction,

D_e = effective diffusion coefficient, cm^2/sec,

t = leaching time, sec,

L = the length variable over the surface distribution can be expressed as follows:

$$D(3,2) = \frac{\sum d_i s_i}{\sum s_i}$$

If a WTP residuals is assumed to be a spherical, the length variable *(L)* can be expressed by the specific surface area *(S_a)*. The Equation (16) can be presented as:

$$\frac{\sum_1^n a_n}{M_o} S_a = \frac{1}{3}\left(\frac{D_e}{\pi}\right)^{0.5} t_n^{\,0.5} \tag{17}$$

where Sa The specific surface area: $S_a = \frac{Total\ Area}{Total\ Volume} = \frac{6\sum \frac{v_i}{d_i}}{\sum v_i}$

The effective diffusion coefficient (D_e) for metals leached from a WTP residuals can be calculated as a function of time and the length variable over the surface distribution of particles (L) by using Equation (16) if the particle size can be tested. If the specific surface area (S_a) of particle is available, the effective diffusion coefficient (D_e) can be determined by Equation (17).

Equation (16) has been recommended by American Nuclear Society (1981) for evaluation of leaching from solidified radioactive wastes. They state that a D_e value should be calculated for each of seven leaching periods and that the results be presented as a dimensionless "leachability index", LX:

$$LX = \frac{1}{7}\sum_{n=1}^{7} \log\left(\frac{1}{D_e}\right) \tag{18}$$

where n = Number of the leaching period,

The leachability index can be used to compare the relative mobility of different contaminants on a uniform scale that varies from 5 ($D_e = 10^{-5}$ cm^2/s, very mobile) to 15 ($D_e = 10^{-15}$ cm^2/s, immobile) (Cote and Hamilton, 1983).

The leaching model presented above can be modified to include the leachability index:

$$\left(\frac{\sum a_n}{A_o}\right) = 1.128(10^{-0.5LX})(t_n^{\ 0.5})\left(\frac{1}{L}\right) \tag{19}$$

6.4 Results and Discussion

6.4.1 WTP Residual Buffering Capacity

Most WTP dewatered residuals contain some alkalinity because of chemicals used in coagulation and inorganic substances present in water sources, particularly, in lime residuals.

Buffering capacity was determined for leachate collected in each sequential acid extraction until no alkalinity was detected. At the beginning of the extraction, 10 grams of equivalent dry weight of sample was placed in TCLP extractor and then 500 mL of 0.04 M acetic acid was added. This extracting liquid provides an acidity of 2 meq/g (USEPA 1986). The leachate was collected to test alkalinity after 24 hours. The extracting liquid was repeatedly added for 15 sequential extractions. Figure 6.3 shows the results of this test on PVD sample. The cumulative alkalinity of the samples was 3.65 meq/g. The initial pH of solid sample was 11.37 and the pH of the leachant was 3.5 and. In the first four extractions all acid was neutralized by the alkalinity in PVD sample and the final pH dropped to 7. During the next two extractions the pH dropped from 7 to about 4. pH values of the subsequent nine extractions were all close to 4.0, This indicates that little alkalinity was available after the 4th extraction. The total alkalinity of PVD sample was 12.24 (meq/g) as shown in Table 6.1. These results show that the cumulative alkalinity used for neutralizing acid was only 30 percent of the total alkalinity and 70 percent of alkalinity was not utilized. This could be due to the reason that some carbonic anions were associated with alkali metals to form the crystalline alkali metals (Weisner 1962), which could not be leached under the weak acid extraction conditions.

Types of alkalinity were determined and results are shown in Figure 6.4. This figure shows that approximately 90 percent of the leachable alkalinity in this residual was bicarbonate alkalinity and 10 percent was carbonate alkalinity.

As shown in Table 6.1, the initial pH and total alkalinity of RWA residuals were 7.15 and 1.02 meq/g, respectively. The results of buffering capacity of iron residual, RWA sample, during 15 days sequential extractions are shown in Figure 6.5. The pH of leachate immediately fell to about 4 in first acetic acid extraction and the total alkalinity of sample was less than 2 meq/g. In other words, all leachable alkalinity had been neutralized by acid in first extraction. When DI water was used as extracting fluid, the relationship between alkalinity and pH is shown in Figure 6.5. It is seen that the alkalinity was released gradually with extraction time. The total alkalinity released was approximately 0.07 meq/g, which is only 7 percent of the total alkalinity after 15 extraction.

Same experiment was performed for WQD residual sample. The total alkalinity of WQD sample was 2.18 (meq/g) and initial pH was 6.55. Test results are illustrated in Figure 6.6. As shown in Figure 6.6, the pH dropped to about 4 in the first acetic acid extraction. The cumulative alkalinity of WQD was about 0.09 meq/g after 15 extractions with DI water as extracting fluid. Type of alkalinity leached out from RWA and WQD samples are shown in Figures 6.7 and 6.8. Most of alkalinities of iron and alum residuals are from bicarbonic alkalinity.

These results indicate that lime residual has very high buffering capacity since 6 -12 times of alkalinity as iron and alum residuals. The iron and alum residuals have less alkalinity for acid neutralization. Most leachable alkalinity was consumed in the first extraction. The results also show that the buffering capacity of iron and alum residuals would decrease rapidly if there are large amount of acid presence, which is common in MSW landfill.

6.4.2 Metal Leaching As a Function of pH

Single batch leaching tests were performed on three types of WTP residuals to evaluate the effect of pH on metal leaching.

Figures 6.9 and 6.10 show the primary metals leached from PVD sample at different pHs of leachant. The final pHs of leachate were from 11.58 to 9.98 when leachant used were pH 9.62 to 1.28. Figure 6.9 is a logC-pH diagram to show metal variation with pH. Most of the metal concentrations in leachate did not change very much with pH, except aluminum, which increased rapidly from 3 mg/L to 65 mg/L after pH dropped below 2. Calcium leached out more than any other metals at all pH values because of its lower solubility product (Table 5.9). Figure 6.10 show that 99% of aluminum, 97% of iron , and 75% of lime remained in the residual during variation of pH. at high pH. The test results for other types of WTP residuals, RWA and WQD, are shown in Figures 6.11 to 6.14. As shown in Figure 6.11 for RWA sample, all metals leached out rapidly when pH dropped below 4.0, however those metals concentrations maintained approximately at the same level at pH from 4 to 10. For instance, Only 5% of aluminum, 40% of calcium, and 80%

of iron remained in residuals when pH dropped to 2 as shown in Figure 6.12. Figures 6.13 and 6.14 show similar pattern for WQD sample. More metals came out as pH dropped below 4 (Figure 6.13). Over 95% of aluminum and iron remained in WQD residual when pHs changed from 4 to 10 (Figure 6.14). These results indicate that primary metals in iron or alum residuals will leached out quickly with a pH less than 4.0. This is caused by insufficient buffering capacity for resisting pH change, which results in metals dissolving in leachant. Table 6.1 shows that the pHs of the iron and alum residuals are in the range of 6.55 to 7.15, which indicate that not much buffer capacity exists these type of residuals. The results shows that the lime residual can maintain a high pH and dissolved metal concentrations were not change very much under all different pH testing conditions.

6.4.3 Effect of Particle Size

Metal leaching rate usually would increases as particle size decreases due to an increase in available surface area for leaching (Bishop 1988). To investigate how particle size affects metal leaching rate, single batch leaching tests on three types of residual were developed.

Each residual has six sizes which were separated by gravity settling as shown in Table 6.2. Size range for PVD, RWA, and WQD samples was 20μm- 55μm, 17μm-26μm, and 37μm- 78μm, respectively. Sample with each size was then extracted with 50 ml of 1.0 N acetic acid for 24 hours and continuous for 72 hours. Acid was replaced every 24 hours. The particle sizes were analyzed by MasterSizer X and the particle size was expressed by D(3,2) which represents mean diameter over the surface area distribution.

Figure 6.15 shows information on material characteristics, such as largest and smallest particle sizes and relative proportions of fine sand, silt, and clay. This figure shows that three types of WTP residuals have approximately same size range and they belong to from fine sand to clay. 80% to 95% of residuals are silt. 15% of RWA and 20% of WQD samples were fine sand. Comparison the results of surface area of three types residuals in Table 6.1 with the values of d_{50} in Figure 6.15, d_{50} of WQD is about 30 μm with surface area of 0.30 m^2/cm^3, and the values of d_{50} of PVD and WQD are 13 μm with 0.68 m^2/cm^3 and 0.74 m^2/cm^3, respectively. The observation shows that The particle size distribution of residuals is corresponding to the surface area quite well. The surface area is increased with particle size decreased.

Figures 6.16 and 6.17 show effect of particle size on iron and calcium leaching. Surprisingly, metals leached out were not significantly varied with particle sizes. Iron released from PVD and RWA residuals were almost at same level. The concentrations of iron and calcium in leached from WQD residual increased slightly with increasing particle size. The results are opposite to what was expected. It is also different from the previous research for stabilized/solidified hazardous wastes (Bishop 1988 and Brown et al. 1986). Brown et al. (1986) reported in leaching test for stabilized/solidified hazardous wastes that smaller sized particles leached considerably less metals initially than larger particles because of the increasing rate of alkalinity leaching, which neutralized acid in the leachant. Another study (Bishop, 1988) showed that metal leaching rates decreased with decreasing particle size. The explanation is that metals were bound to the particle by sorption mechanism. Larger surface area in smaller particles would have greater sorption capacity and thus release less metal concentrations in leachate. Another reason could be that WTP residuals are "floc" structure (explained detailed in Chapter 2), and 75% to 85% of moisture are trapped in this loose structure, which are different from dry solids, such as stabilized/solidified hazardous wastes. In "floc" structure particle, the diffusion of metals in moisture phase play more important role than in solid phase because the 75 to 85% of space is occupied by moisture.

6.4.4 Primary Metal Binding Mechanisms

Multiple batch leaching tests were performed on WTP residuals to elucidate metal binding mechanisms. It is believed that metals may be trapped in the pores of the residual matrix, adsorbed onto pore walls, or chemically complexed with oxidized compounds.

Figure 6.18 shows the concentrations of primary metals (Al, Fe, and Ca) leached from the PVD lime residual sample as a function of the acid added in a 15-day multiple batch leaching test. The release of Al from residuals did not occur until approximately 12 meq acid/g solid were added (6 days of leaching time). Al concentration in the leachate then increased gradually when the pH of the leachate fell below 4.5, indicating that the alkalinity could no longer neutralize all the acid added. Calcium leached out rapidly right started from the beginning of the extraction. Comparing Figures 6.4 and 6.12, it can be seen that the leaching rate of calcium is correspondent to the leaching rate of alkalinity. These means that most leachable calcium was associated with carbonate and bicarbonate alkalinity and they leached together. Not much iron came out because of its low solubility and its low concentration in PVD sample.

Figures 6.19 and 6.20 illustrate the cumulative metals leaching out from RWA and WQD residuals, subjected to 0.04 M acetic acid multiple batch extractions. The amount of primary metals leaching out is in the order of calcium, aluminum, and iron, which matches the solubilities of those metals, as shown in Table 5.9. Only 4% of calcium leached from PVD and RWA before the first 4 days, and there were no more calcium released after that. This indicates that most calcium remains in the solid and leachable calcium was bound with the solid in the pores or adsorbed on pore surfaces, it would be released only when the alkalinity is consumed. Iron was the least soluble metal and is known to be tightly adsorbed. Iron release was the smallest among all the metals tested.

A substantial amount of the primary metals still remained in the residuals even after 15 extractions and neutralization all alkalinity. This indicates that binding mechanisms other than simple metal-hydroxide formation were involved. Most metals were probably bound to the crystalline structure and were not solubilized. Analyses of metals remaining in the PVD residuals show that 99.7% of iron, 98% of aluminum, and 95.5% of calcium still in residual after 15 extractions. Of all the metals present in RWA sample, 99.6% of iron, 97% of aluminum, and 96% of calcium remained in the residuals. For WQD sample, 99.4%, 98% of aluminum, and 94% of calcium remained after the extraction.

It is apparent from these results that the WTP residuals bind the primary metals very effectively in solid structure, thus making them difficult to leach out.

6.4.5 Determination of Leaching Rates

The metal leaching model, as shown in Equations 6.16 and 6.19, can be used to determine leaching rates for the WTP residuals. This model accounts for most of the variables discussed earlier which include diffusion coefficient (D_e), particle size (V and S), and initial primary metal concentration (A_o).

Particle sizes of WTP residuals were analyzed by Mastersize X and the results of particle are presented in Table 6.2. This information along with the results obtained from multiple batch extraction are used to determine rates of the leaching. These rates can then be compared with the field leaching rates and used for prediction over extended time period of time.

Effective diffusion coefficients (D_e) for primary metals leached from WTP residuals is a function of time and can be calculated using Equation 6.16. Leachability indexes *(LXs)* are calculated by Equation 6.19. The results of De and LX for three types of residuals are presented in Tables 6.3, 6.4, and 6.5. Effective diffusion coefficients (D_e) for three types of residuals are plotted in Figures 6.21, 6.22, and 6.23. Those data show that all three metals, iron, aluminum, calcium, are immobile in WTP residuals based on the criteria and definition of a leachability index or effective diffusion coefficients defined by the American Nuclear Society (American Nuclear Society, 1980). These Figures also show that the calcium is most mobile among three metals, and iron is the least mobile. These results are in agreement with the previous discussion in Chapter 5. It is seen from these Figures that the primary metals effective diffusion coefficient or dissolution rate for particular metal varies with time as the residual is contacted with the continuous addition of acid. The effective diffusion coefficients for aluminum and iron increased with time. This is probably due to the decreasing of buffering capacities left in residuals. The increasing addition of acid in leachant dissolves metals and washes them out.

6.4.6 Prediction of Metal Leaching Rates in WTP monofill or MSW Landfill

6.4.6.1 The procedures of prediction of metal leaching: The diffusion-based leaching model can be used for prediction of metal leaching rates in WTP monofill or landfill. The following procedures are developed to predict the leaching rate.

1) The effective diffusion coefficients (Equation (16)) or the leachability indices (Equation (19)) for WTP residuals is determined by a multiple extraction test first.

2) Determine the acidity of the leachant. Actual acidity measurement would be the best, but if this information is not available the following values may be used as approximations (Bishop, 1986):

   rainwater acidity = 100 μeq/l (5 mg/l as $CaCO_3$)

   groundwater acidity = 1000 (μeq/l) (50 mg/l as $CaCO_3$)

   Landfill leachate acidity = 100 meq/l (5000 mg/l as $CaCO_3$)

3) Determine the projected leachate velocity through the in-place residual. As a worst case, assume that the residual has crumbled and that all rainwater impacting the surface of the fill will penetrate through it. It is highly likely that the actual leachant velocity will be considerably less than this because much of residual will probably not disintegrate totally and because much of

the rainwater will run off the surface and away from the residual rather than penetrate through it.

4) Analyze a mean particle size *(L)* or a specific surface area *(S_a)*. Initially, the value of a mean particle size *(L)* is very small or a specific surface area *(S_a* is very high. But for in-situ condition, compaction and weathering conditions will cause the particle size to increase or specific surface area to decrease. A more realistic value may be estimated by considering the field situation.

5) Determine the expected LX of WTP residuals at given time intervals based on the predicted rate of the leachant acid addition. In other words, Determine the predicted LXs with long term period based on the experimental LXs with short term period.

6) Determine the predicted annual metal leaching rate from the WTP residuals using the appropriate time, LX and L values and Equation 6.19 to obtain the cumulative leaching rates for two consecutive years. The difference between these is the predicted amount of metal leached during that year.

7) Determine the metal concentration in the leachate based on the annual amount of metal leached and the amount of percolating water during that year.

8) Determine the resulting metal concentration in groundwater underlying the site by assuming some appropriate dilution factor.

6.4.6.2 Application of LXs: The Leaching Index *(LX)* for long term leaching period can be determined by short term experiments. This method is based upon the amount of acid added in the laboratory being equivalent to the acid added in leachant or rainfall in landfill or monofill. Table 6.6 shows the long term leaching time equivalent to the extraction time in laboratory multiple extraction leaching tests. For instance, three-day multiple extraction test in laboratory, with 6 meg/g acetic acid addition, is equivalent to 10,440 years of leaching due to precipitation (0.5 meq/L rainwater acidity, 1 m/year rainfall intensity) or 52 years leaching caused by MSW landfill. Figure 6.24, 6.25, and 6.26 show long term LXs values for three types of WTP residuals disposed in MSW landfill with leachant acidity of 100 meq/L.

The example of calculation of prediction of metal leaching rates of WTP residuals disposed in MSW landfill or a WTP monofill is presented in Appendix B.

6.4.7 Predicted Leaching Results for WTP Residuals Disposed in MSW Landfill or Monofill

Tables 6.7, 6.8, and 6.9 list the predicted leaching concentrations of alum, iron, and lime metals for PVD samples, disposed in a typical WTP monofill or a MSW landfill for a period of 200 years. Table 6.7 shows that if WTP residuals are disposed of in MSW landfill, LX values decreased with leaching time due to high acidity generated from leachant. This will result in higher concentrations of alum, iron, and lime leaching from WTP residuals compared with residuals disposed of in WTP Monofill. Taking the aluminum leached from PVD residual as an example (shown in Table 6.7), the values of LX for PVD residuals in MSW landfill decreased from 19.90 to 17.30, the predicted results show total 21.63% of aluminum leached out in 100 years, and 45.3 mg/L of aluminum exists in leachate in the year 100. If PVD residual disposed of in WTP monofill, LX will maintain constant in 200 year due to lower acidity in rainwater. The accumulative aluminum leached out is only 4.88%, which is about 25% of amount of aluminum leached out in MSW landfill in the same leaching time period. The concentration of aluminum in leachate was only 10.6 mg/L during year 100. This result shows that less metals will leached out if WTP residuals are disposed of in a monofill than in a MSW landfill because of the high concentration of acidity generated in a MSW landfill, which will speed up the metals leaching from WTP residuals. If WTP residuals are disposed of in a monofill, the LXs are constant and the predictive curves can be plotted based on Equation (19). The results are shown in Figures 6.27, 6.28, and 6.29. These figures illustrate the fractions of cumulative primary metals leached from lime, iron, and alum WTP residuals disposed of in a monofill from 1 year to 1000 years. Take the PVD residuals (Figure 6.27) as an example, up to 10 year, about 34% calcium will leach out and most calcium will leach out before 100 years. For aluminum in the same sample, less than 1% of aluminum will leach out in three years. Only 5% of aluminum will be released in 100 years period. The iron has lowest solubility, less than 1% of iron will be leached during 100 years. This graphical method will be useful in predicting the primary metals leaching from WTP residuals disposed of in a WTP monofill. The concentrations of projected primary metals that can be leached out from all three types of WTP residuals in WTP monofill, MSW landfill, and groundwater for the long term period are presented in Appendix A.

Table 6.1 Properties related to primary metals leaching from WTP residual samples

Sample Name	Type	pH	Alk	Particle Size				Ion Concentration		
			(meg/g)	d(0.5)[1] (µm)	D(4,3)[2] (µm)	D(3,2)[3] (µm)	Sa[4] (m^2/cm^3)	Al (mg/g)	Fe (mg/g)	Ca (mg/g)
PVD	Lime	11.15	12.24	1.87	24.68	8.76	0.6846	47.78	16.64	139.10
RWA	Iron	7.15	1.02	3.41	85.19	8.05	0.7455	11.68	213.55	4.97
WQD	Alum	6.55	2.18	10.94	62.81	19.81	0.3029	137.50	14.83	10.30

1 - d(0.5) -d_{50} The diameter of 50% of number of particles

2 - D(4,3)The mean diameter over the volume distribution: $D(4,3)=\dfrac{\sum d_i v_i}{\sum v_i}$

where d_i is the mean diameter of size band i, i=1 to 32;
v_i is the mean volume of size band I, I=1 to 32;

3 - D(3,2)The mean diameter over the surface distribution (the Sauter Mean Daimeter): $D(3,2)=\dfrac{\sum d_i s_i}{\sum s_i}$

4 - Sa The specific surface area: $S_a=\dfrac{TotalArea}{TotalVolume}=\dfrac{6\sum \frac{v_i}{d_i}}{\sum v_i}$

Table 6.2 Particle sizes of three types of WTP residuals

Types of Residual	Particle Size (μm)					
PVD	20.47	21.86	23.9	32.16	47.84	54.82
RWA	18.15	18.34	20.28	21.75	25.59	26.49
WQD	36.66	51.36	52.31	53.25	52.99	78.63

Table 6.3 Effective diffusion coefficients and leachability indecs for leaching from PVD samples

Extraction Time (Days)	D_e (cm/s)			*LX*		
	Al	Fe	Ca	Al	Fe	Ca
1	2.10×10^{-20}	1.11×10^{-21}	1.23×10^{-17}	19.68	20.96	16.91
2	4.19×10^{-20}	2.22×10^{-21}	2.46×10^{-17}	19.38	20.65	16.61
3	3.54×10^{-19}	2.95×10^{-21}	3.43×10^{-17}	18.45	20.53	16.47
4	3.30×10^{-19}	3.90×10^{-21}	5.67×10^{-17}	18.48	20.41	16.25
5	3.22×10^{-19}	3.44×10^{-21}	5.78×10^{-17}	18.49	20.46	16.24
6	3.22×10^{-19}	3.74×10^{-21}	6.22×10^{-17}	18.49	20.43	16.21
7	1.03×10^{-18}	3.30×10^{-21}	5.67×10^{-17}	17.99	20.48	16.25
8	2.36×10^{-18}	3.79×10^{-21}	5.01×10^{-17}	17.63	20.42	16.30
9	3.47×10^{-18}	5.08×10^{-21}	4.46×10^{-17}	17.46	20.29	16.35
10	4.49×10^{-18}	6.63×10^{-21}	4.02×10^{-17}	17.35	20.18	16.40
11	5.16×10^{-18}	7.43×10^{-21}	3.66×10^{-17}	17.29	20.13	16.44
12	5.66×10^{-18}	7.88×10^{-21}	3.35×10^{-17}	17.25	20.10	16.47
13	5.99×10^{-18}	9.96×10^{-21}	3.09×10^{-17}	17.22	20.00	16.51
14	6.22×10^{-18}	11.60×10^{-20}	2.87×10^{-17}	17.21	19.93	16.54
15	6.23×10^{-18}	20.80×10^{-20}	2.68×10^{-17}	17.21	19.68	16.57
"ANS LX'				19.56	20.56	16.42

Note: "ANS LX" is equal to the average LX for the first seven leaching periods.

Table 6.4 Effective diffusion coefficients and leachability indecs for leaching from RWA samples

Extraction Time (Days)	D_e (cm/s)			*LX*		
	Al	Fe	Ca	Al	Fe	Ca
1	1.55×10^{-18}	6.27×10^{-21}	4.17×10^{-17}	17.81	20.20	16.38
2	3.78×10^{-18}	1.93×10^{-20}	4.88×10^{-17}	17.42	19.72	16.31
3	5.93×10^{-18}	3.62×10^{-20}	5.44×10^{-17}	17.23	19.44	16.26
4	8.83×10^{-18}	5.31×10^{-20}	7.02×10^{-17}	17.05	19.28	16.15
5	9.48×10^{-18}	7.01×10^{-20}	5.62×10^{-17}	17.02	19.15	16.25
6	1.01×10^{-17}	9.03×10^{-20}	4.68×10^{-17}	17.00	19.04	16.33
7	1.06×10^{-17}	1.06×10^{-19}	4.01×10^{-17}	16.98	18.97	16.40
8	1.10×10^{-17}	1.28×10^{-19}	3.51×10^{-17}	16.96	18.89	16.45
9	1.08×10^{-17}	1.46×10^{-19}	3.12×10^{-17}	16.97	18.83	16.51
10	1.10×10^{-17}	1.69×10^{-19}	2.81×10^{-17}	16.96	18.77	16.55
11	1.12×10^{-17}	1.91×10^{-19}	2.55×10^{-17}	16.95	18.72	16.59
12	1.11×10^{-17}	2.13×10^{-19}	2.34×10^{-17}	16.95	18.67	16.63
13	1.17×10^{-17}	2.38×10^{-19}	2.16×10^{-17}	16.93	18.62	16.67
14	1.22×10^{-17}	2.67×10^{-19}	2.01×10^{-17}	16.91	18.57	16.70
15	1.21×10^{-17}	2.89×10^{-19}	1.87×10^{-17}	16.92	18.54	16.73
"ANS LX'				17.22	19.40	16.30

Note: "ANS LX" is equal to the average LX for the first seven leaching periods.

Table 6.5 Effective diffusion coefficients and leachability indecs for leaching from WQD samples

Extraction Time (Days)	D_e (cm/s)			*LX*		
	Al	Fe	Ca	Al	Fe	Ca
1	1.66×10^{-18}	4.52×10^{-20}	1.62×10^{-16}	17.78	19.34	15.79
2	3.22×10^{-18}	5.81×10^{-20}	1.35×10^{-16}	17.49	19.24	15.87
3	1.19×10^{-17}	1.41×10^{-19}	1.44×10^{-16}	16.92	18.85	15.84
4	1.91×10^{-17}	1.96×10^{-19}	1.41×10^{-16}	16.72	18.71	15.85
5	2.32×10^{-17}	2.20×10^{-19}	1.43×10^{-16}	16.64	18.66	15.85
6	2.23×10^{-17}	2.87×10^{-19}	1.47×10^{-16}	16.65	18.54	15.83
7	2.41×10^{-17}	3.42×10^{-19}	1.49×10^{-16}	16.62	18.47	15.83
8	2.26×10^{-17}	4.16×10^{-19}	1.53×10^{-16}	16.65	18.38	15.82
9	2.42×10^{-17}	5.83×10^{-19}	1.57×10^{-16}	16.62	18.24	15.80
10	2.69×10^{-17}	8.73×10^{-19}	1.71×10^{-16}	16.57	18.06	15.77
11	3.08×10^{-17}	1.15×10^{-18}	1.79×10^{-16}	16.51	17.94	15.75
12	3.12×10^{-17}	1.49×10^{-18}	1.88×10^{-16}	16.51	17.83	15.73
13	3.32×10^{-17}	1.76×10^{-18}	1.94×10^{-16}	16.48	17.75	15.71
14	3.16×10^{-17}	2.06×10^{-18}	1.99×10^{-16}	16.50	17.69	15.70
15	3.35×10^{-17}	2.47×10^{-18}	2.14×10^{-16}	16.48	17.61	15.67
"ANS LX'				16.97	18.83	15.84

Note: "ANS LX" is equal to the average LX for the first seven leaching periods.

Table 6.6 Effective Diffusion Coefficients and Leachability Indecs for Leaching from WQD Samples

Extraction Time (Days)	D_e (cm/s)			*LX*		
	Al	Fe	Ca	Al	Fe	Ca
1	1.66×10^{-18}	4.52×10^{-20}	1.62×10^{-16}	17.78	19.34	15.79
2	3.22×10^{-18}	5.81×10^{-20}	1.35×10^{-16}	17.49	19.24	15.87
3	1.19×10^{-17}	1.41×10^{-19}	1.44×10^{-16}	16.92	18.85	15.84
4	1.91×10^{-17}	1.96×10^{-19}	1.41×10^{-16}	16.72	18.71	15.85
5	2.32×10^{-17}	2.20×10^{-19}	1.43×10^{-16}	16.64	18.66	15.85
6	2.23×10^{-17}	2.87×10^{-19}	1.47×10^{-16}	16.65	18.54	15.83
7	2.41×10^{-17}	3.42×10^{-19}	1.49×10^{-16}	16.62	18.47	15.83
8	2.26×10^{-17}	4.16×10^{-19}	1.53×10^{-16}	16.65	18.38	15.82
9	2.42×10^{-17}	5.83×10^{-19}	1.57×10^{-16}	16.62	18.24	15.80
10	2.69×10^{-17}	8.73×10^{-19}	1.71×10^{-16}	16.57	18.06	15.77
11	3.08×10^{-17}	1.15×10^{-18}	1.79×10^{-16}	16.51	17.94	15.75
12	3.12×10^{-17}	1.49×10^{-18}	1.88×10^{-16}	16.51	17.83	15.73
13	3.32×10^{-17}	1.76×10^{-18}	1.94×10^{-16}	16.48	17.75	15.71
14	3.16×10^{-17}	2.06×10^{-18}	1.99×10^{-16}	16.50	17.69	15.70
15	3.35×10^{-17}	2.47×10^{-18}	2.14×10^{-16}	16.48	17.61	15.67
"ANS LX'				16.97	18.83	15.84

Note: "ANS LX" is equal to the average LX for the first seven leaching periods.

Table 6.7 Projected aluminum concentrations in leachate and groundwater as a function of leaching time for PVD sample which was disposed in MSW landfill or WTP monofill

Leaching Time (years)	LX	Aluminum Leached (cumulative fraction of initial Al)	(fraction per year)	Al leached per year (g Al/m^2 yr.)	Leachate Conc. (Al mg/L)	Groundwater Conc. (Al mg/L)
MSW Landfill Leachate						
1	19.90	0.0049				
2	19.90	0.0069	0.00202	83.78	83.8	0.84
9	19.80	0.0164				
10	19.80	0.0173	0.00089	36.83	36.8	0.37
49	18.50	0.1707				
50	18.50	0.1724	0.00173	72.04	72.0	0.72
99	18.60	0.2163				
100	18.60	0.2173	0.00109	45.29	45.3	0.45
119	18.00	0.4731				
120	18.00	0.4751	0.00198	82.45	82.5	0.82
149	17.50	0.9413				
150	17.50	0.9445	0.00315	131.09	131.1	1.31
Rainwater Leachate (WTP Monofill Leachate)						
1	19.90	0.0049				
2	19.90	0.0069	0.00202	83.78	83.8	0.84
9	19.90	0.0146				
10	19.90	0.0154	0.00079	32.89	32.9	0.33
49	19.90	0.0341				
50	19.90	0.0344	0.00035	14.69	14.7	0.15
99	19.90	0.0485				
100	19.90	0.0488	0.00025	10.59	10.6	0.11
149	19.90	0.0595				
150	19.90	0.0598	0.00022	9.34	9.3	0.09
199	19.90	0.0689				
200	19.90	0.0691	0.00020	8.40	8.4	0.08

Assumptions:

S/V (cm^{-1}) = 6.846 x10^3
Initial Al concentration (g/kg) = 47.78 (47780 ppm)
LX decrease over time
Percolating water (m^3 /m^2 -yr.) = 1
WTP residual material (kg/m^2 surface (2 m thick) = 870
Leachate acidity (meq/L) = 100
Groundwater concentration (meq/L) = 0.01 x leachate concentration 1
Groundwater dilution factor = 100

Table 6.8 Projected iron concentrations in leachate and groundwater as a function of leaching time for PVD sample which was disposed in MSW landfill or WTP monofill

		Iron Leached				
Leaching Time (years)	LX	(cumulative fraction of initial Fe)	(fraction per year)	Fe leached per year (g Fe/m 2 yr.	Leachate Conc. (Fe mg/L)	Groundwater Conc. (Fe mg/L)
		MSW Landfill Leachate				
1	21.40	0.0009				
2	21.39	0.0012	0.00037	5.39	5.4	0.05
9	21.20	0.0033				
10	21.20	0.0034	0.00018	2.56	2.6	0.03
49	20.50	0.0171				
50	20.50	0.0172	0.00017	2.51	2.5	0.03
99	20.40	0.0272				
100	20.40	0.0274	0.00014	1.99	2.0	0.02
149	20.40	0.0334				
150	20.40	0.0335	0.00011	1.62	1.6	0.02
199	20.20	0.0486				
200	20.20	0.0487	0.00012	1.77	1.8	0.02
		Rainwater Leachate (WTP Monofill Leachate)				
0	21.40	0.0000				
1	21.40	0.0009	0.00087	12.53	12.5	0.13
9	21.40	0.0026				
10	21.40	0.0027	0.00014	2.03	2.0	0.02
49	21.40	0.0061				
50	21.40	0.0061	0.00006	0.89	0.9	0.01
99	21.40	0.0086				
100	21.40	0.0087	0.00004	0.63	0.6	0.01
149	21.40	0.0106				
150	21.40	0.0106	0.00004	0.51	0.5	0.01
199	21.40	0.0122				
200	21.40	0.0122	0.00003	0.44	0.4	0.00

Assumptions:

S/V (cm^{-1}) =	6.846 x10^3
Initial Fe concentration (g/kg) =	16.64 (47780 ppm)
LX decrease over time	
Percolating water (m^3 /m^2 -yr.) =	1
WTP residual material (kg/m^2 surface (2 m thick) =	870
Leachate acidity (meq/L) =	100
Groundwater concentration (meq/L) = 0.01 x leachate concentration	1
Groundwater dilution factor =	100

Table 6.9 Projected calcium concentrations in leachate and groundwater as a function of leaching time for PVD sample which was disposed in MSW landfill or WTP monofill

		Calcium Leached				
Leaching Time (years)	LX	(cumulative fraction of initial Ca)	(fraction per year)	Ca leached per year (g Ca/m 2 yr.)	Leachate Conc. (Ca mg/L)	Groundwater Conc. (Ca mg/L)
		MSW Landfill Leachate				
1	17.20	0.1089				
2	17.20	0.1541	0.04512	5460.34	5460.3	54.60
9	17.10	0.3667				
10	17.10	0.3865	0.01983	2400.24	2400.2	24.00
14	17.00	0.5131				
15	17.00	0.5311	0.01801	2179.45	2179.4	21.79
19	16.90	0.6707				
20	16.90	0.6881	0.01742	2108.55	2108.6	21.09
34	16.70	1.1295				
35	16.70	1.1460	0.01649	1995.57	1995.6	19.96
39	16.60	1.3573				
40	16.60	1.3746	0.01729	2092.55	2092.5	20.93
		Rainwater Leachate (WTP Monofill Leachate)				
1	17.20	0.1089				
2	17.20	0.1541	0.04512	5460.34	5460.3	54.60
9	17.20	0.3268				
10	17.20	0.3445	0.01768	2139.21	2139.2	21.39
19	17.20	0.4748				
20	17.20	0.4872	0.01233	1492.74	1492.7	14.93
29	17.20	0.5866				
30	17.20	0.5966	0.01003	1213.59	1213.6	12.14
39	17.20	0.6803				
40	17.20	0.6889	0.00867	1048.76	1048.8	10.49
49	17.20	0.7625				
50	17.20	0.7703	0.00774	936.85	936.8	9.37

Assumptions:

S/V (cm^{-1}) =	6.846 x10^3
Initial Ca concentration (g/kg) =	139.1 (47780 ppm)
LX decrease over time	
Percolating water (m^3/m^2-yr.) =	1
WTP residual material (kg/m^2 surface (2 m thick) =	870
Leachate acidity (meq/L) =	100
Groundwater concentration (meq/L) = 0.01 x leachate concentration	1
Groundwater dilution factor =	100

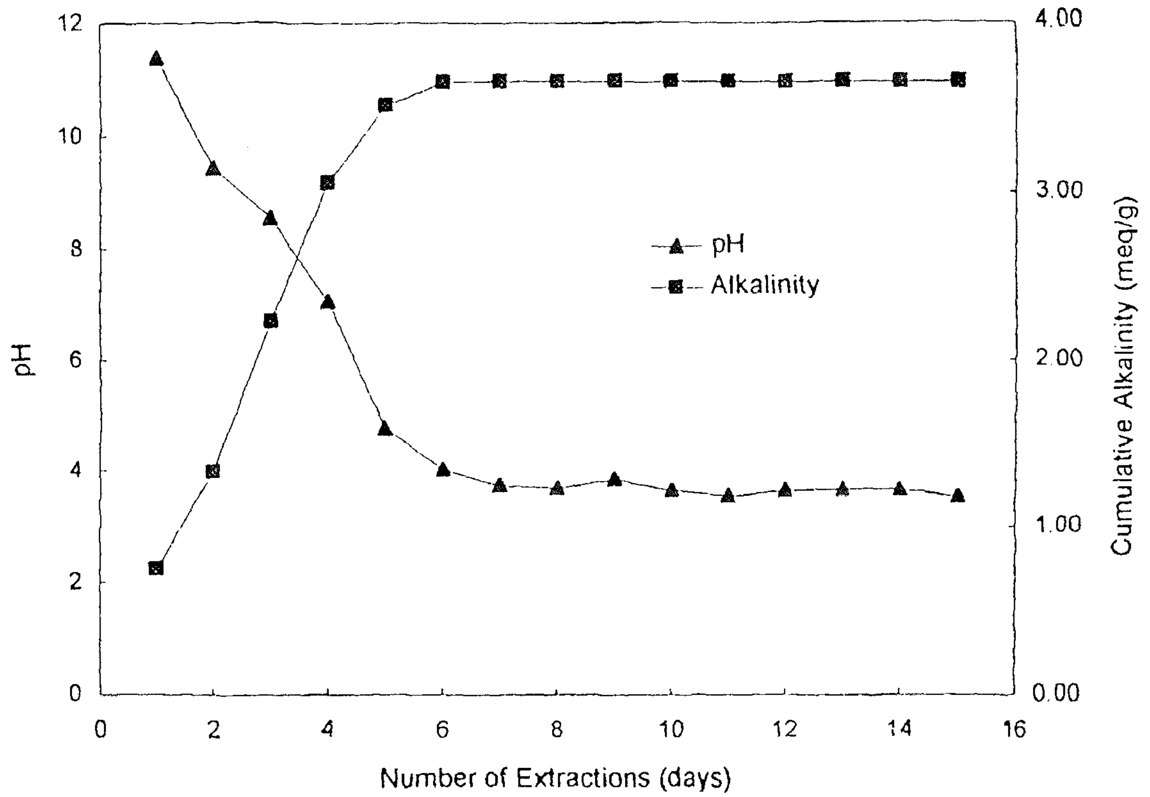

Figure 6.3 pH values and cumulative alkalinity in leachate collected from PVD sample in multiple extractions with 0.04 M Acetic Acid

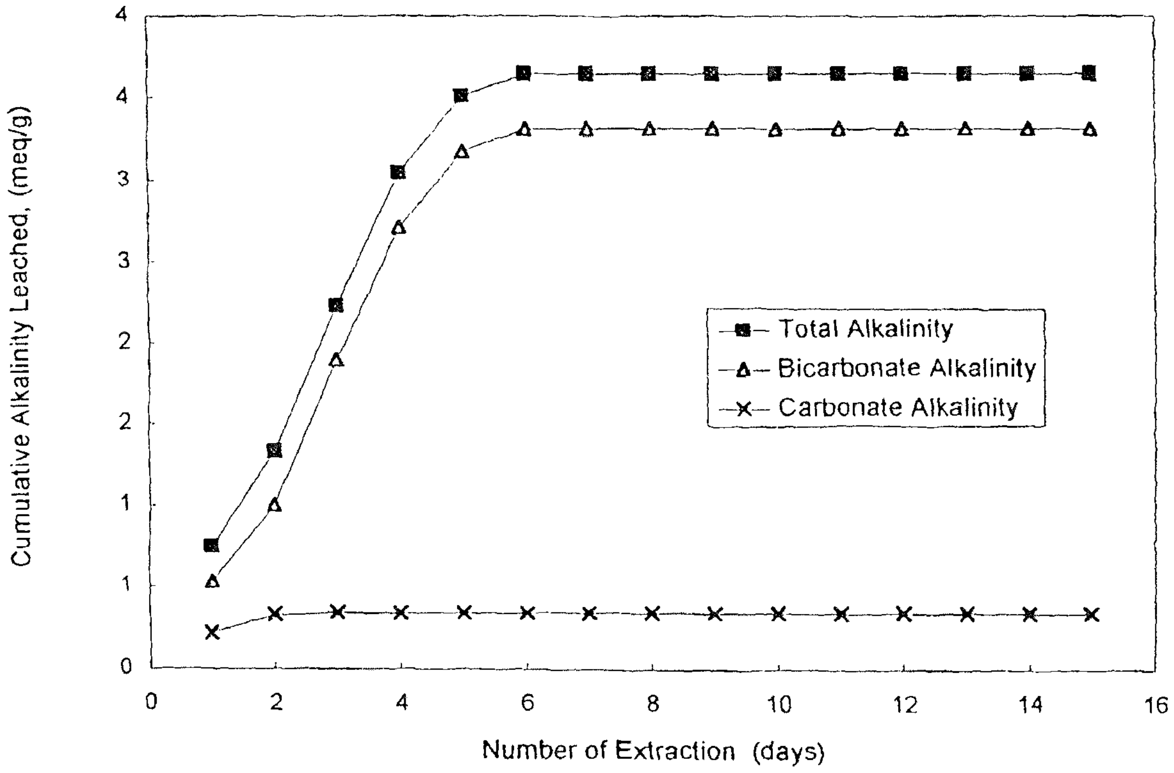

Figure 6.4 Types of alkalinities in PVD sample during multiple extraction with 0.04M acetic acid

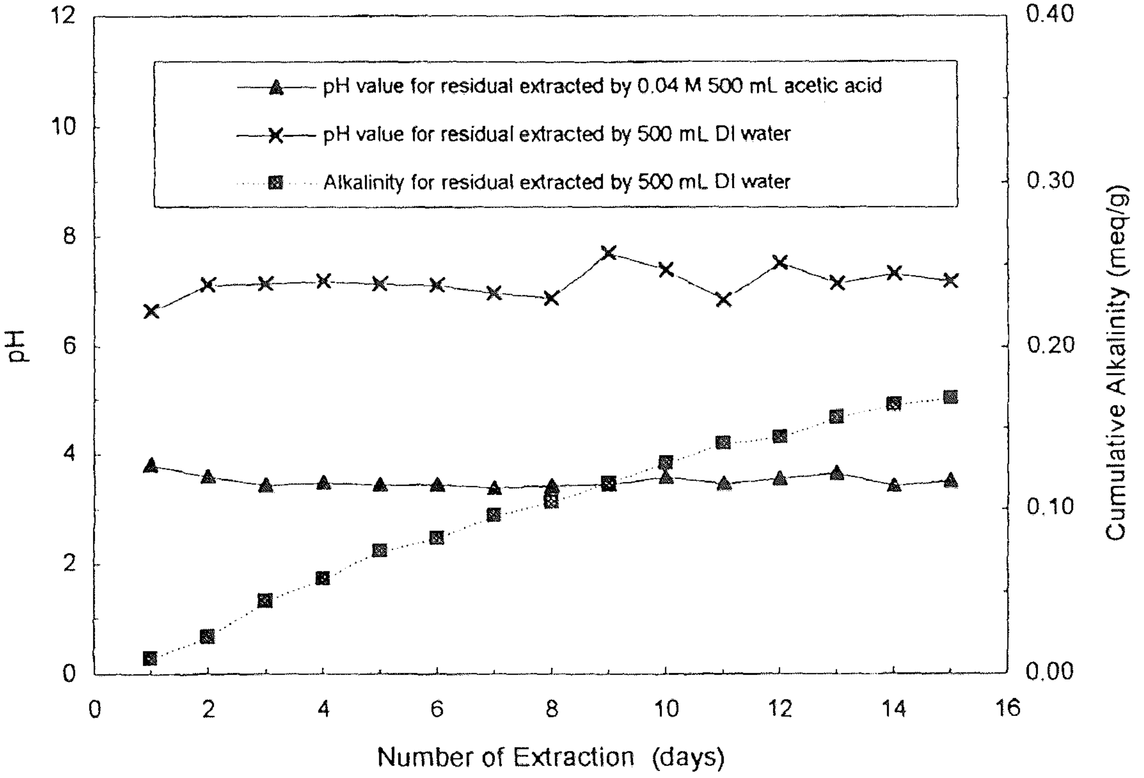

Figure 6.5 pH values and cumulative alkalinity in leachate collected from RWA sample in multiple extractions with 0.04M Acetic Acid or 500 mL DI water

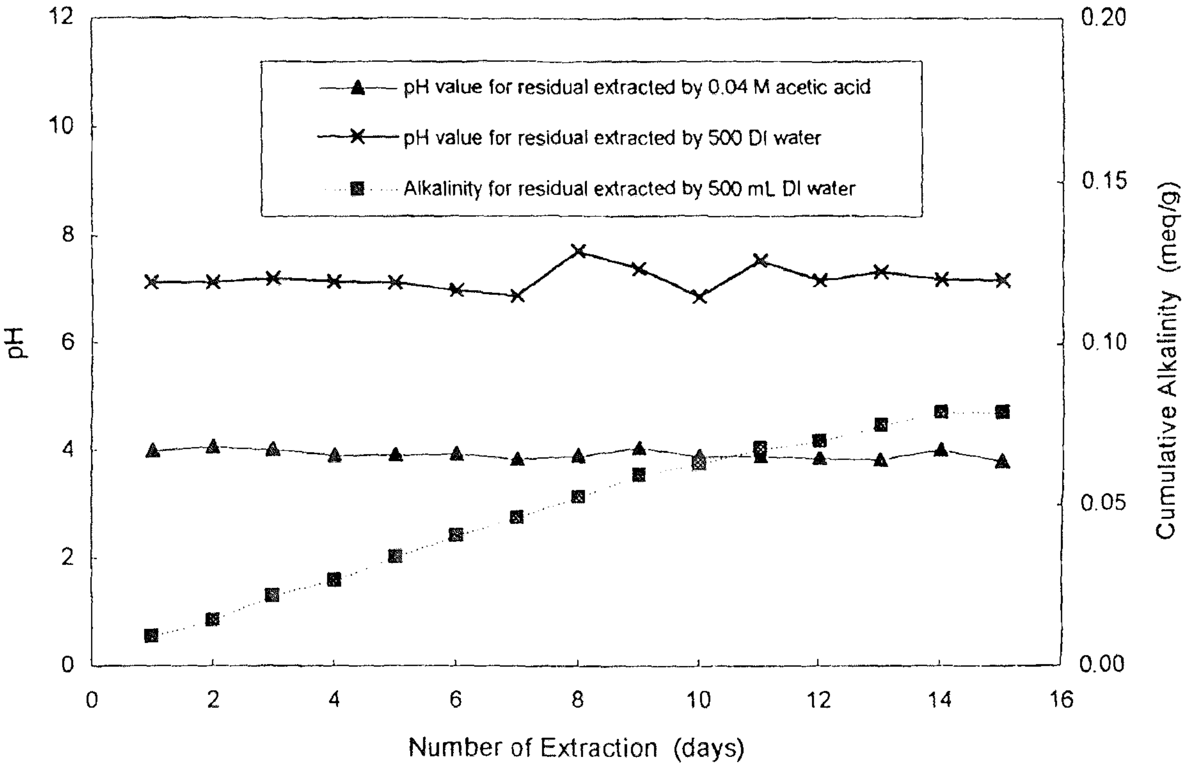

Figure 6.6 pH values and cumulative alkalinity in leachate collected from WQD sample in multiple extractions with 0.04M Acetic Acid or 500 mL DI water

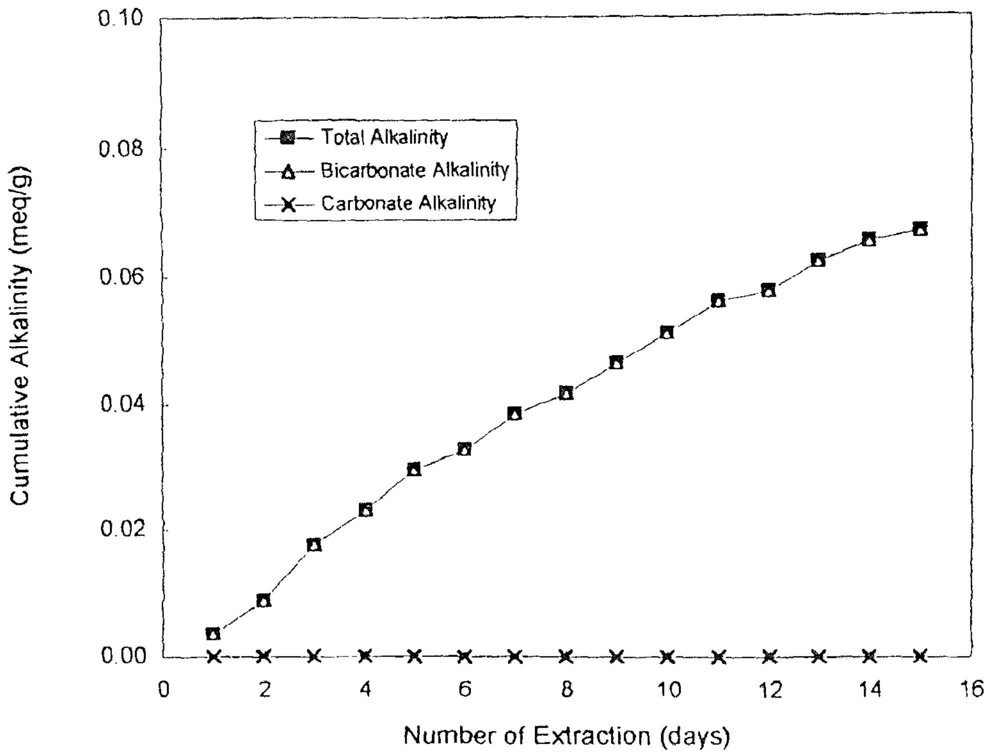

Figure 6.7 Types of alkalinity Leached from RWA sample in multiple extraction with 500 mL DI water

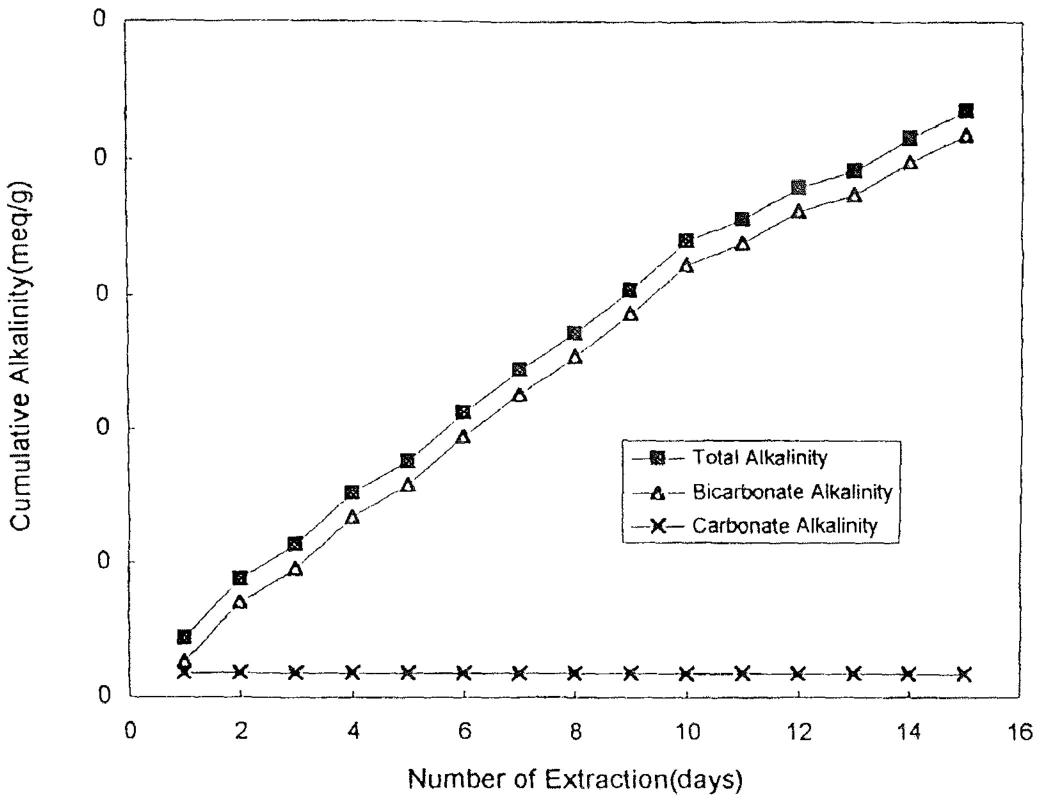

Figure 6.8 Types of alkalinity leached from WQD Sample in multiple extraction with 500 mL DI water

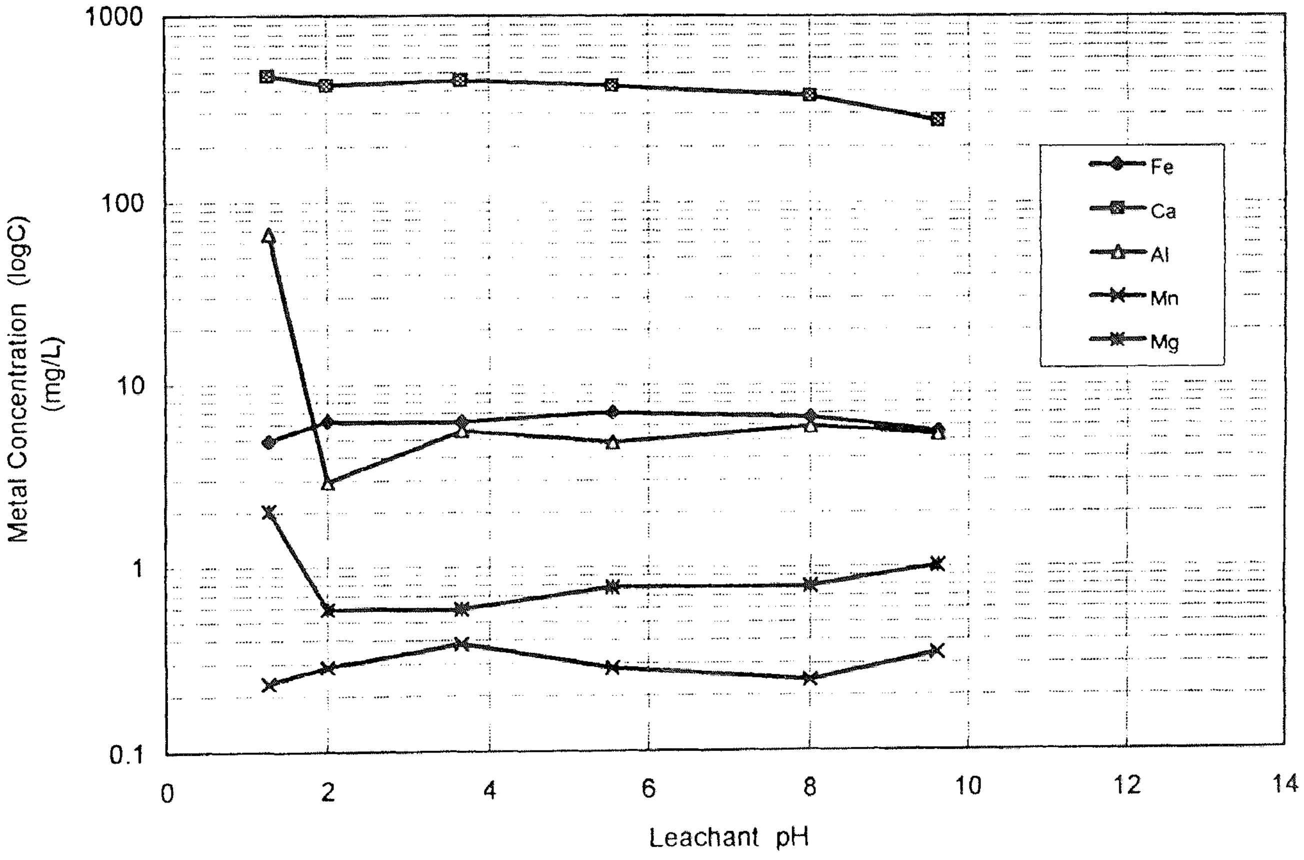

Figure 6.9 Metals concentration at various pH for PVD residuals

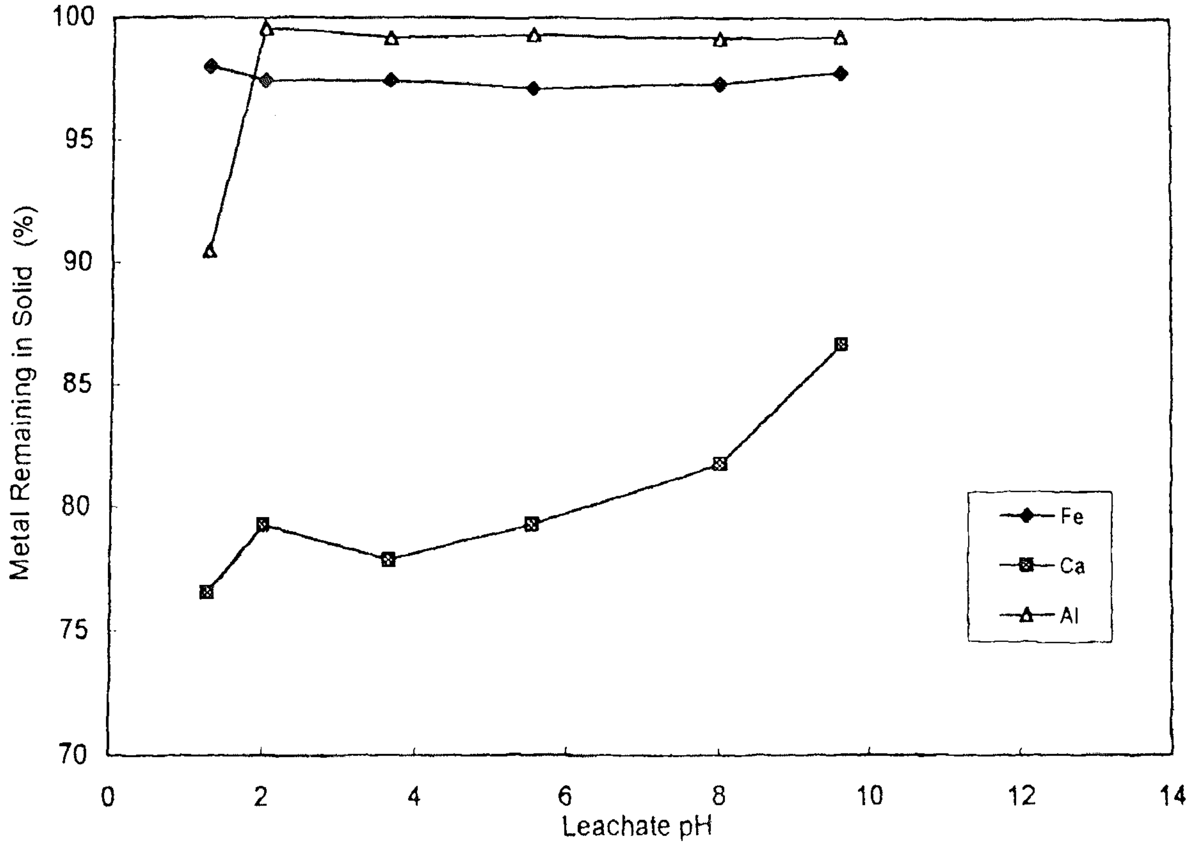

Figure 6.10 Percent metals remained in residuals at various pH in single batch Leaching test for PVD residuals

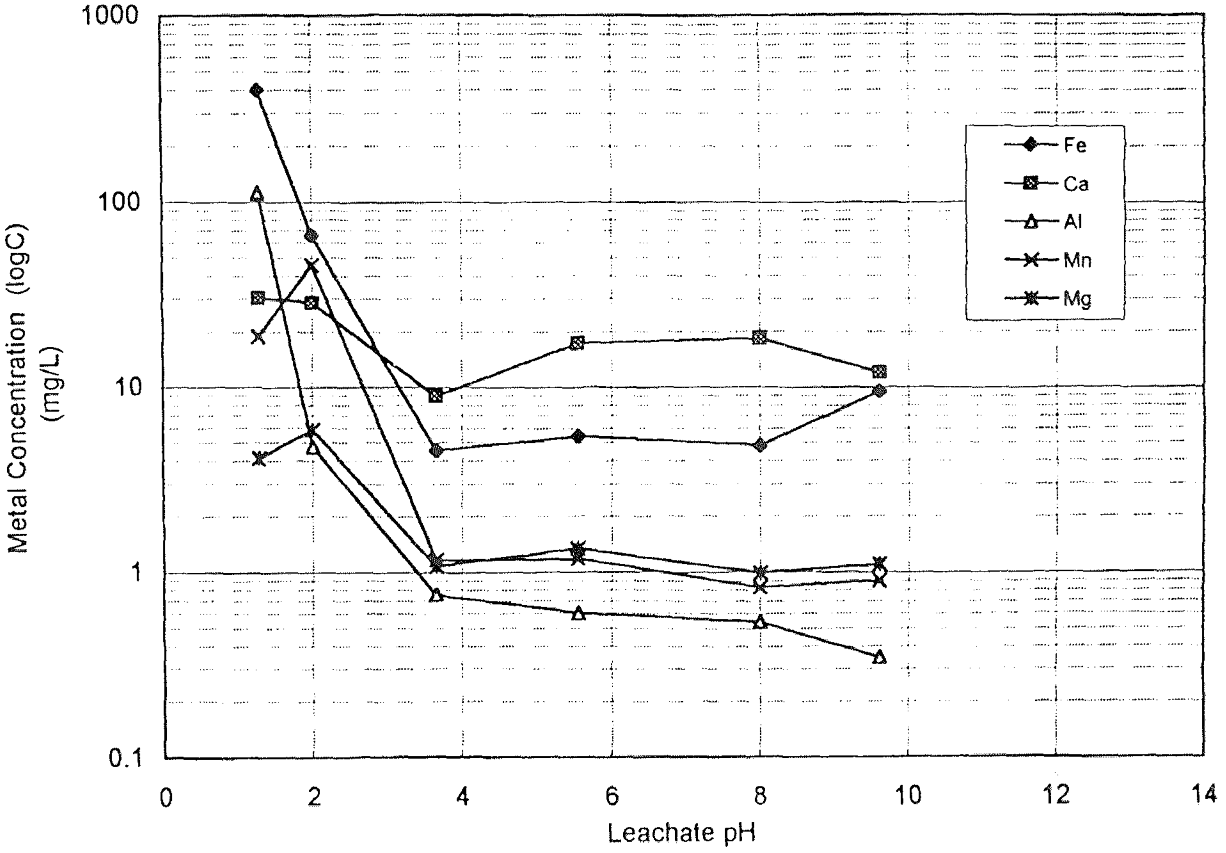

Figure 6.11 Metals concentration at various pH for RWA residuals

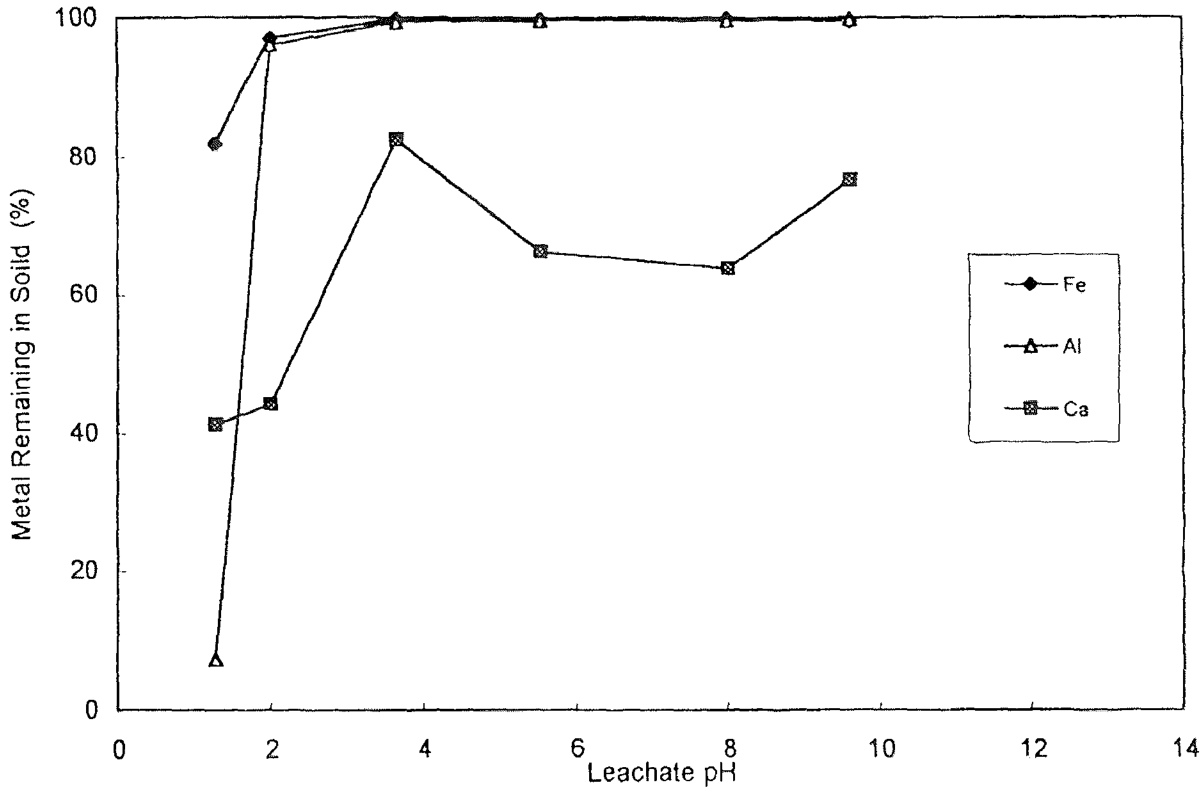

Figure 6.12 Percent metals remained in residuals at various pH in single batch leaching test for RWA residuals

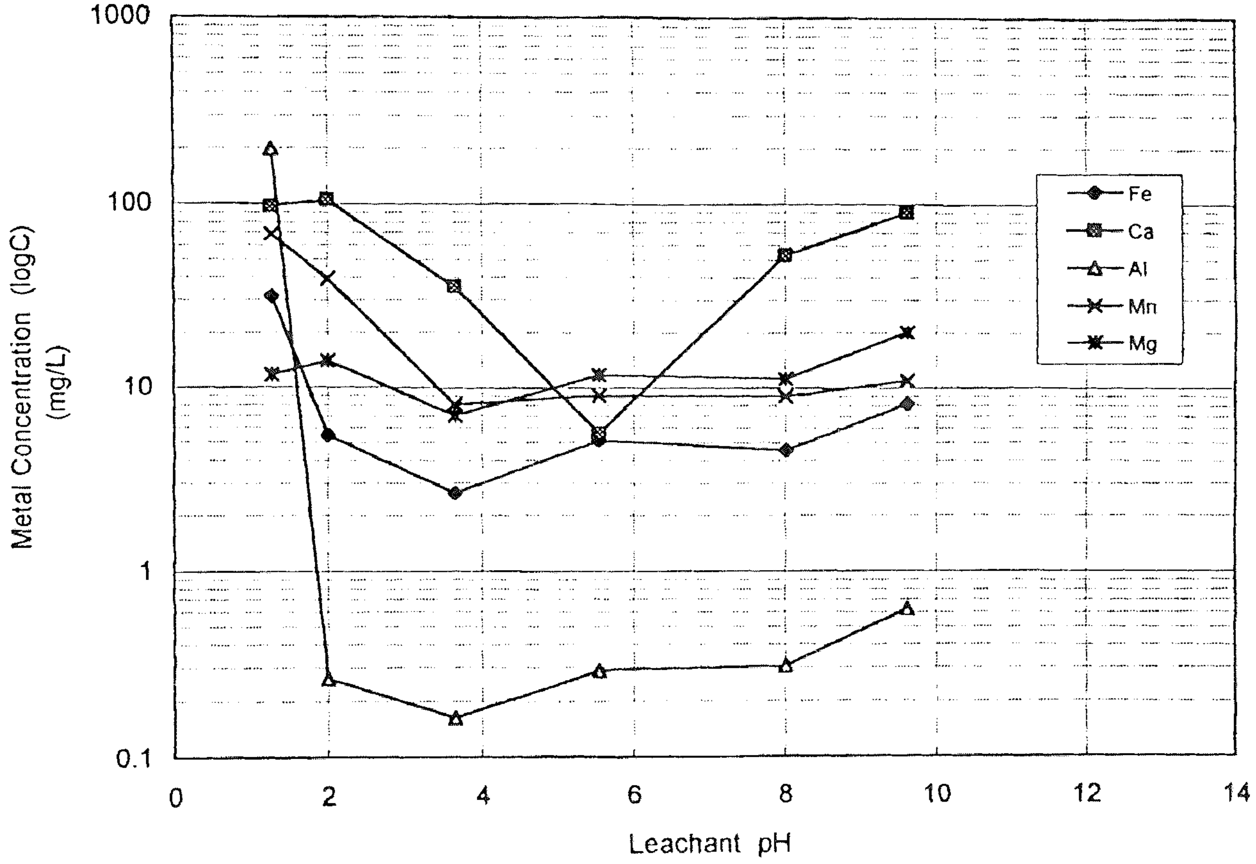

Figure 6.13 Metals concentration at various pH for WQD residuals

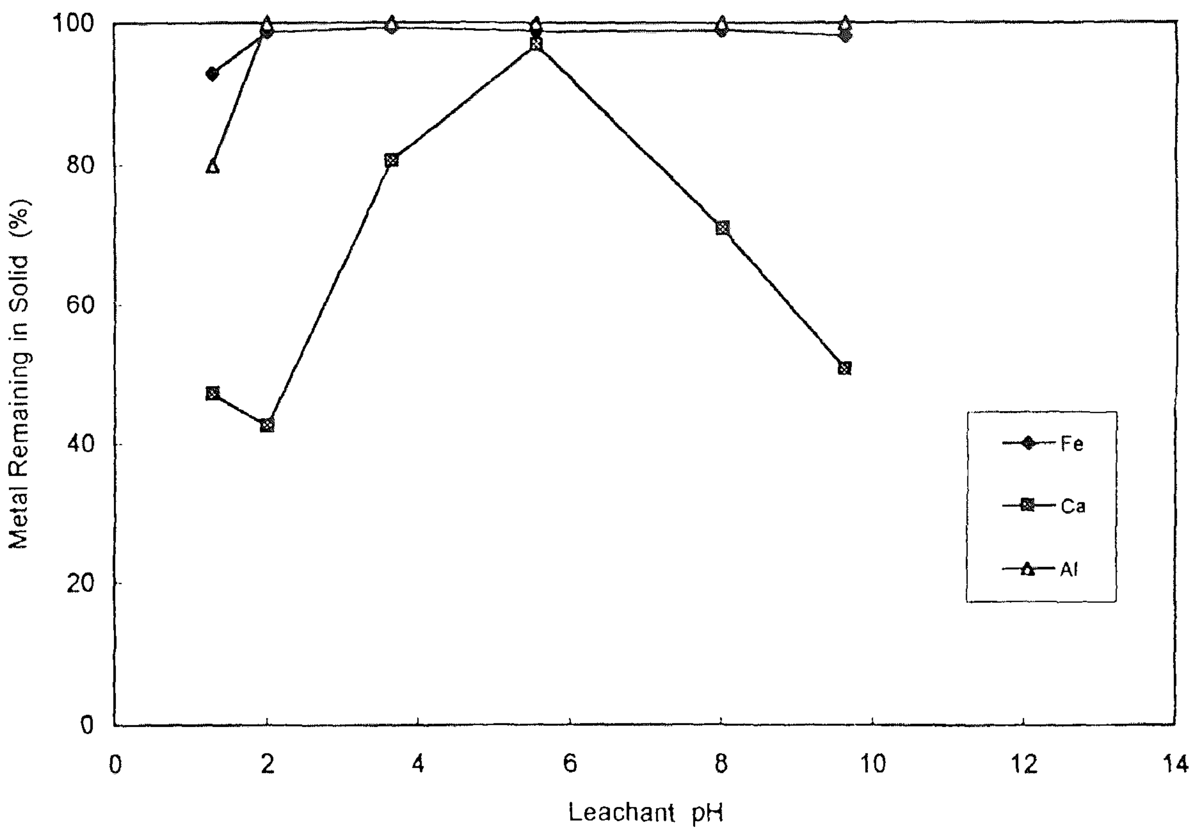

Figure 6.14 Metals remained in residual at various pH in single batch leaching test for WQD residuals

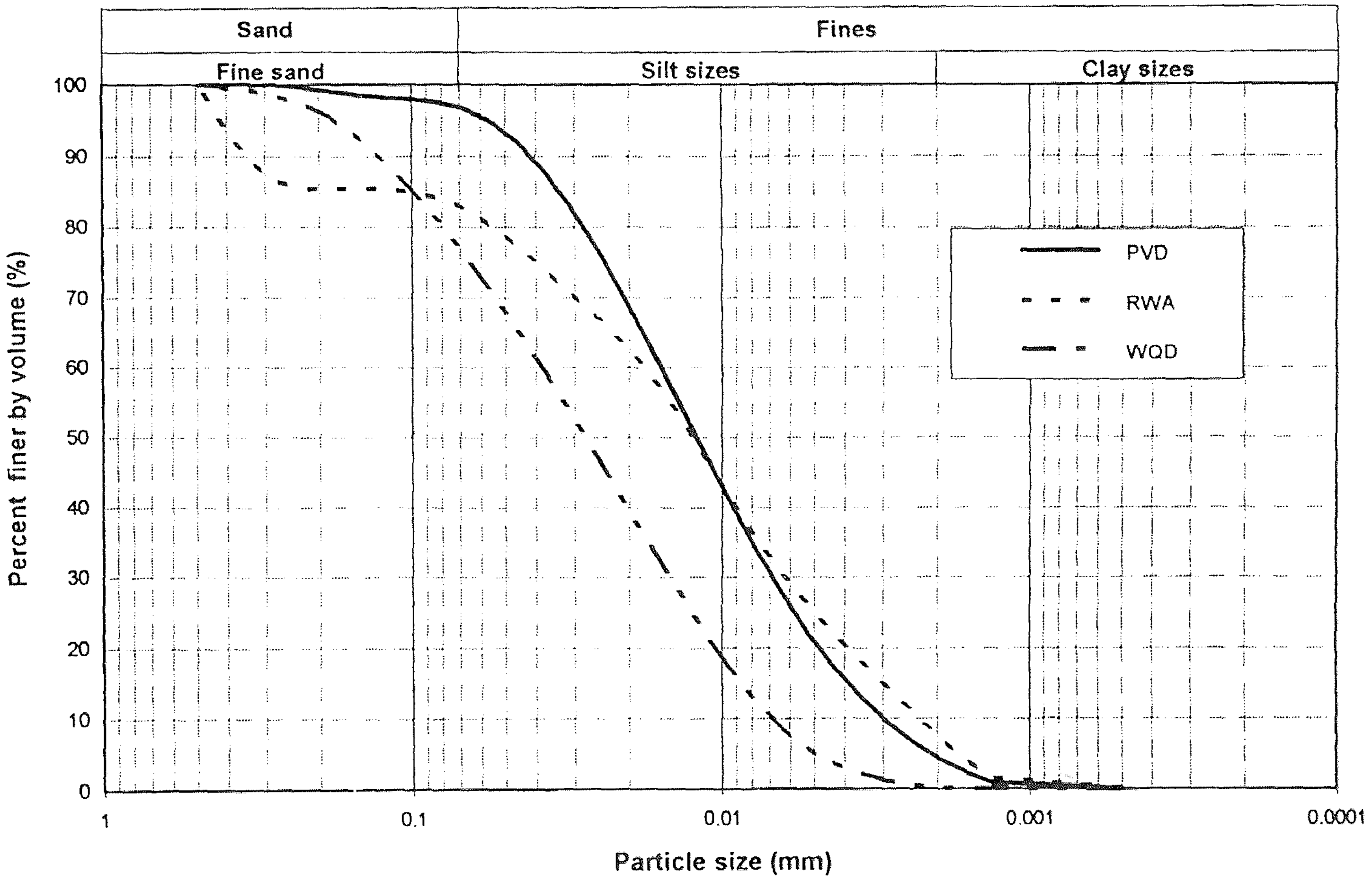

Figure 6.15 Particle size distribution for sample PVD residuals

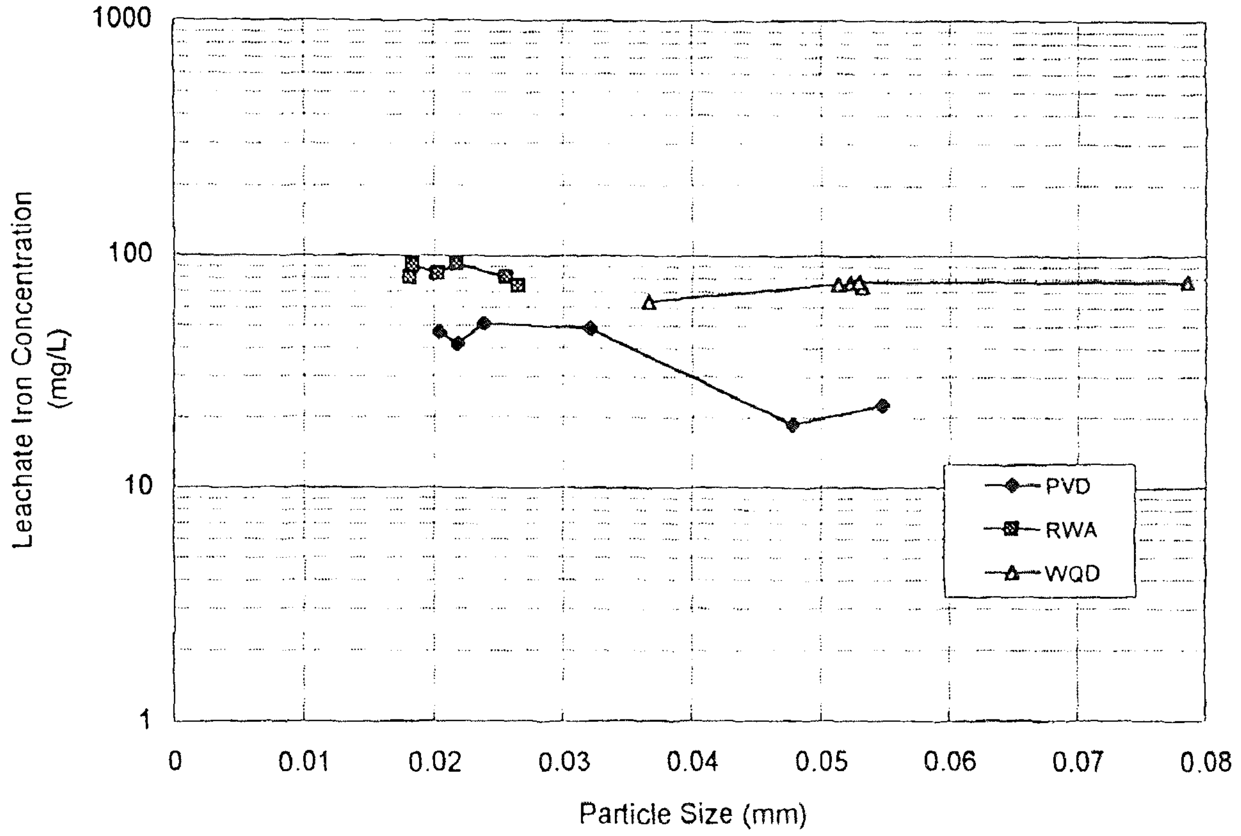

Figure 6.16 The effects of the particle size and metal concentration on the leaching of iron from WTP residuals in a single batch extraction procedure

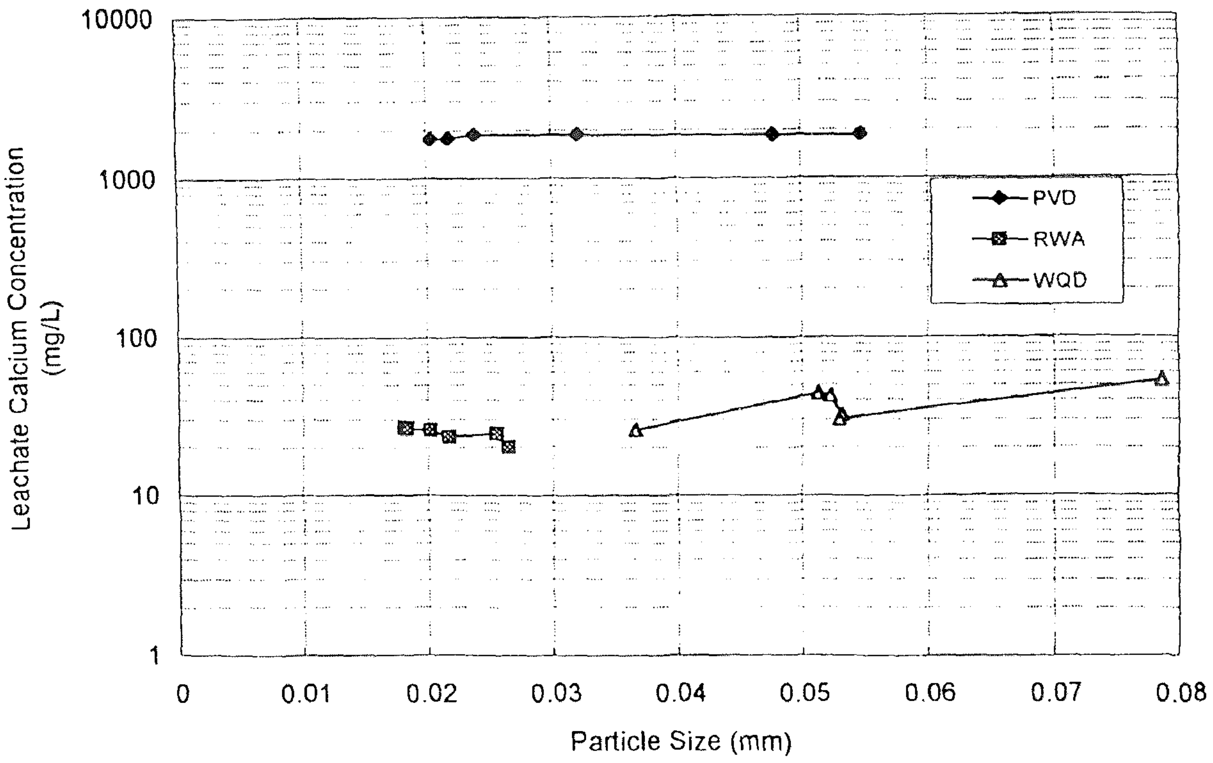

Figure 6.17 The effects of the particle size and metal concentration on the leaching of calcium from WTP residuals in a single batch extraction procedure

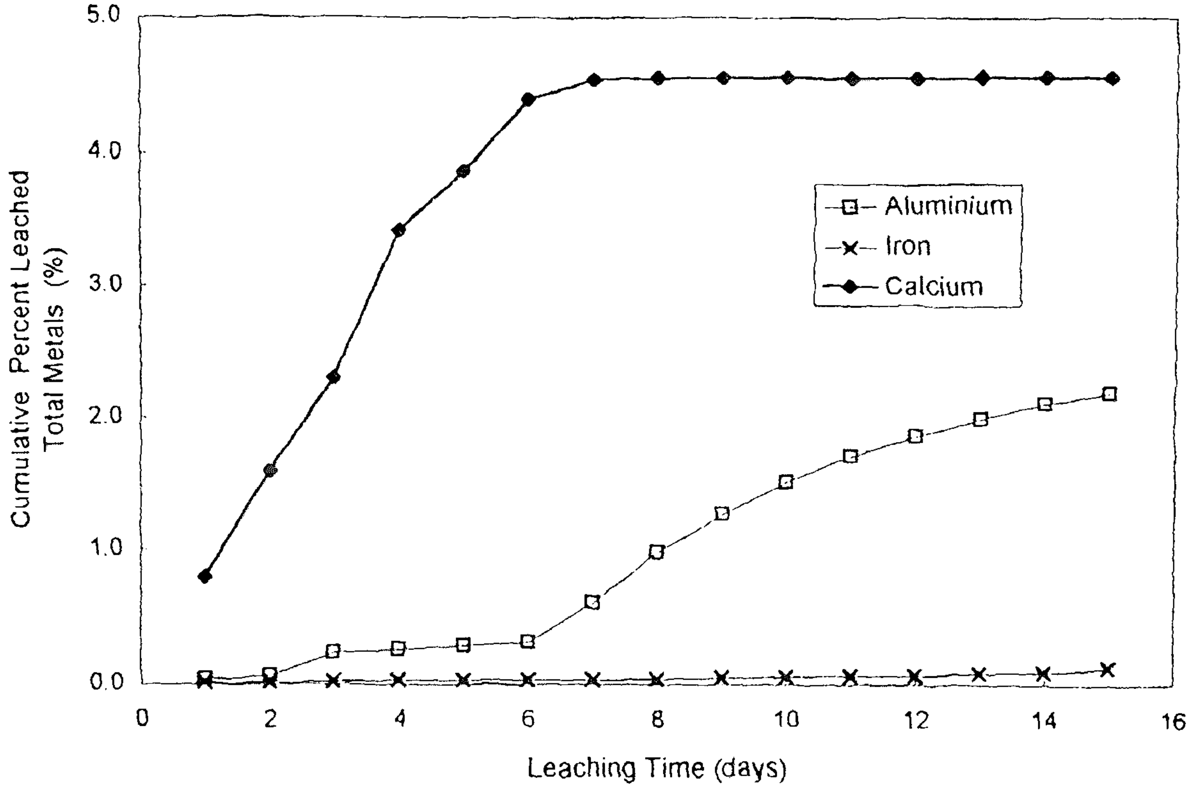

Figure 6.18 The Cumulative metals leached from PVD samples subjected to the multiple batch extraction procedure (0.04 M acetic acid extraction)

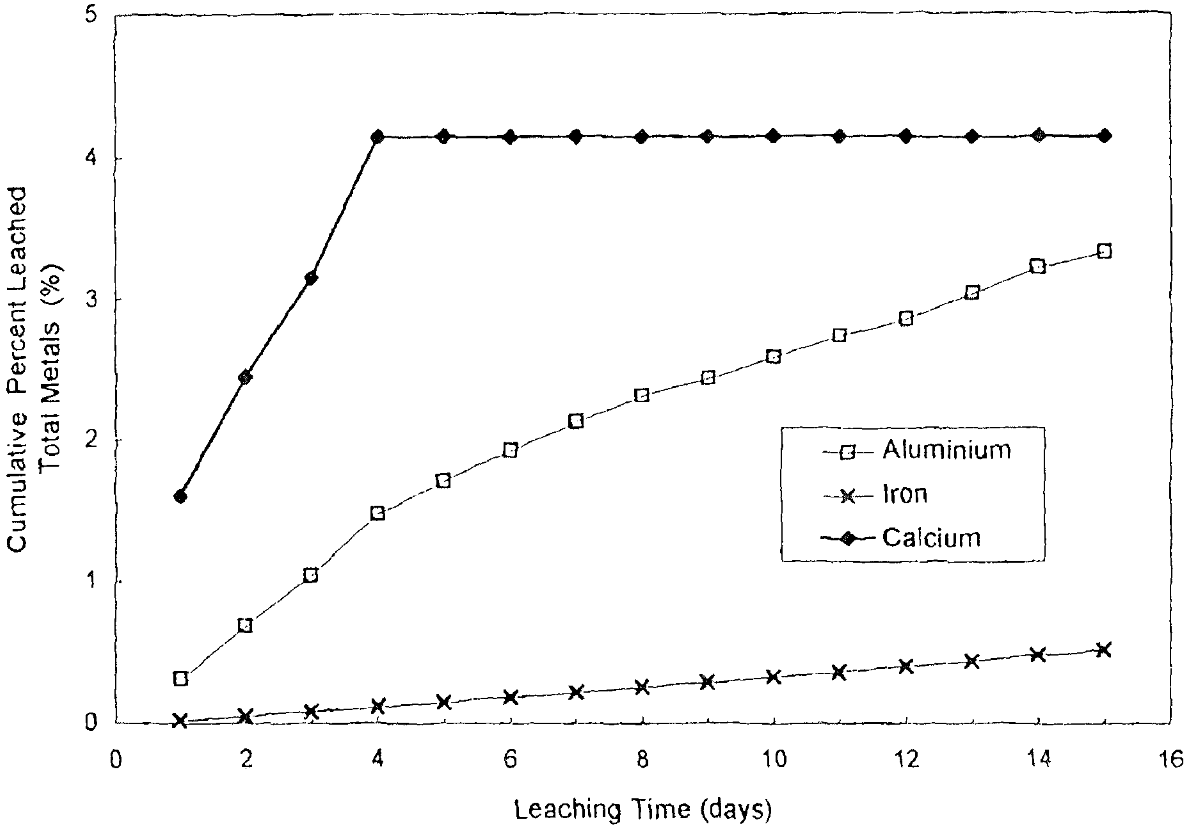

Figure 6.19 The cumulative metals leached from RWA samples subjected to the multiple batch extraction procedure (0.04M acetic acid extraction)

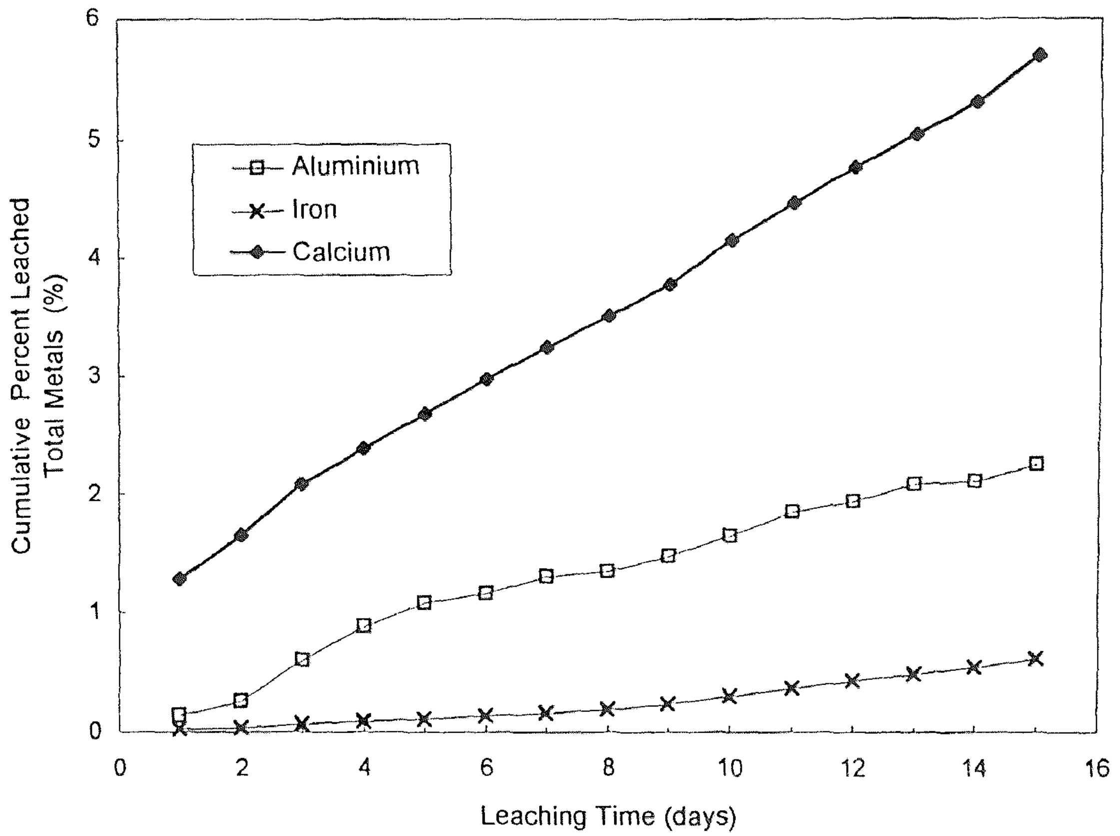

Figure 6.20 The cumulative metals Leached from WQD samples subjected to the multiple batch extraction procedure (0.04M acetic acid extraction)

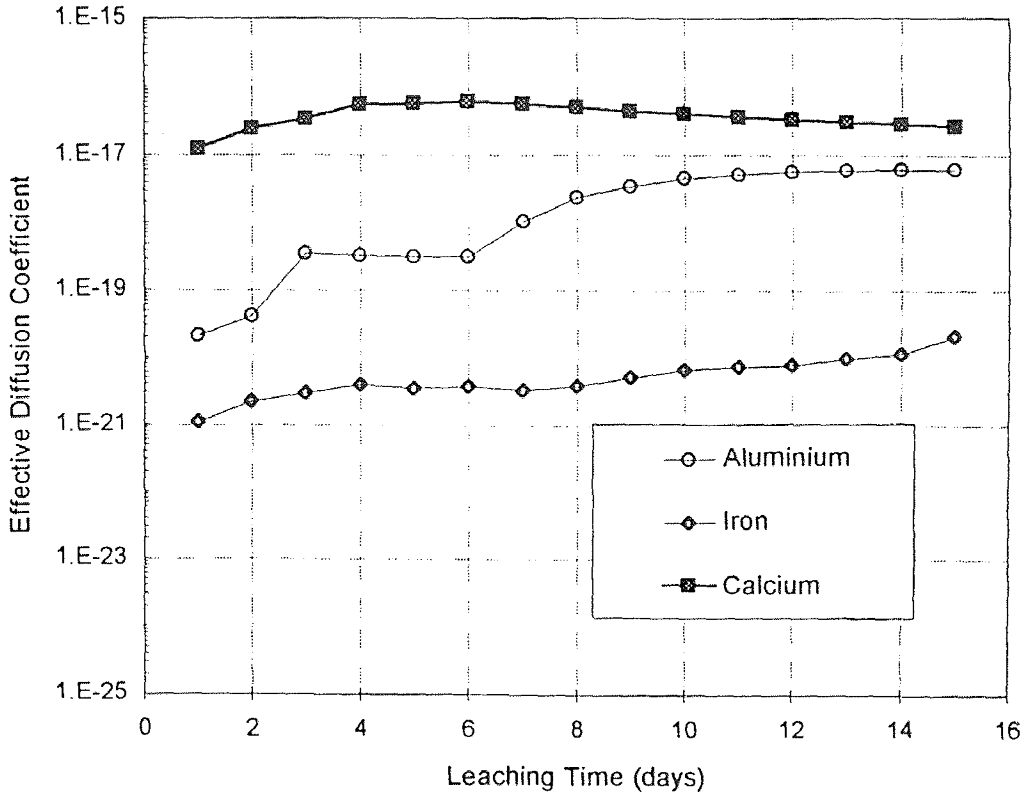

Figure 6.21 Effective diffusion coefficients for primary metal leaching from PVD samples using the multiple batch extraction procedure

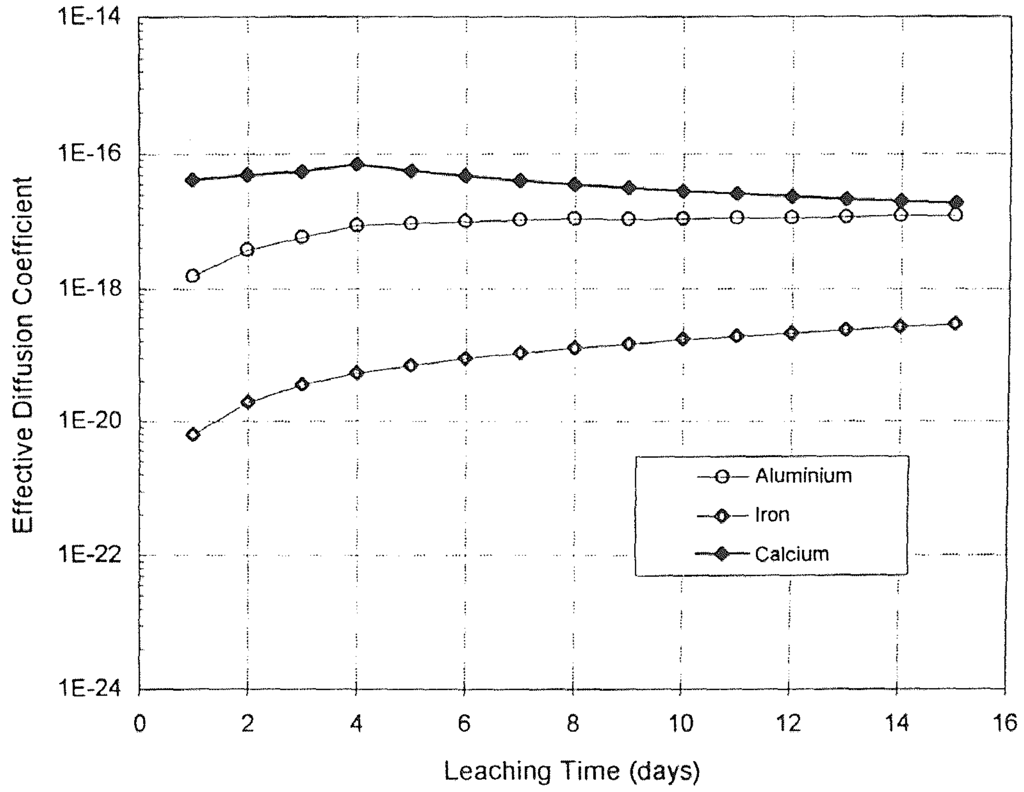

Figure 6.22 Effective diffusion coefficients for primary metal leaching from RWA samples using the multiple batch extraction procedure (0.04M acetic acid extraction)

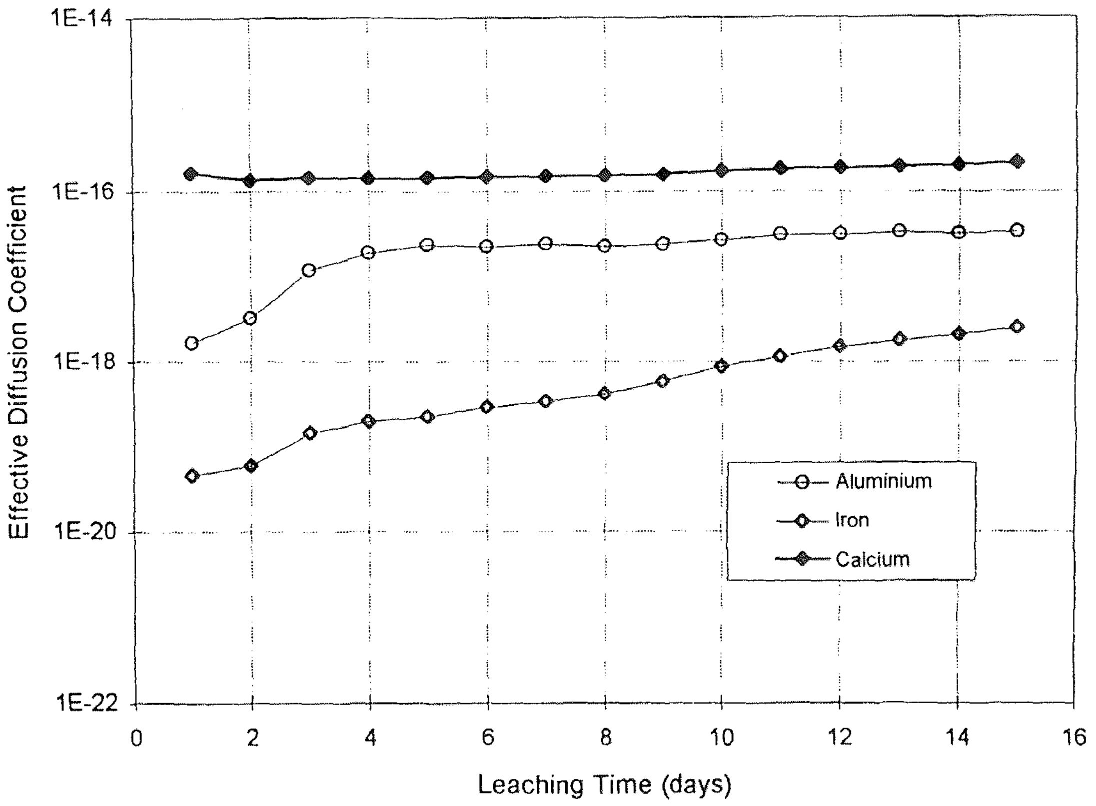

Figure 6.23 Effective diffusion coefficients for primary metal leaching from WQD samples using the multiple batch extraction procedure (0.04M acetic acid extraction)

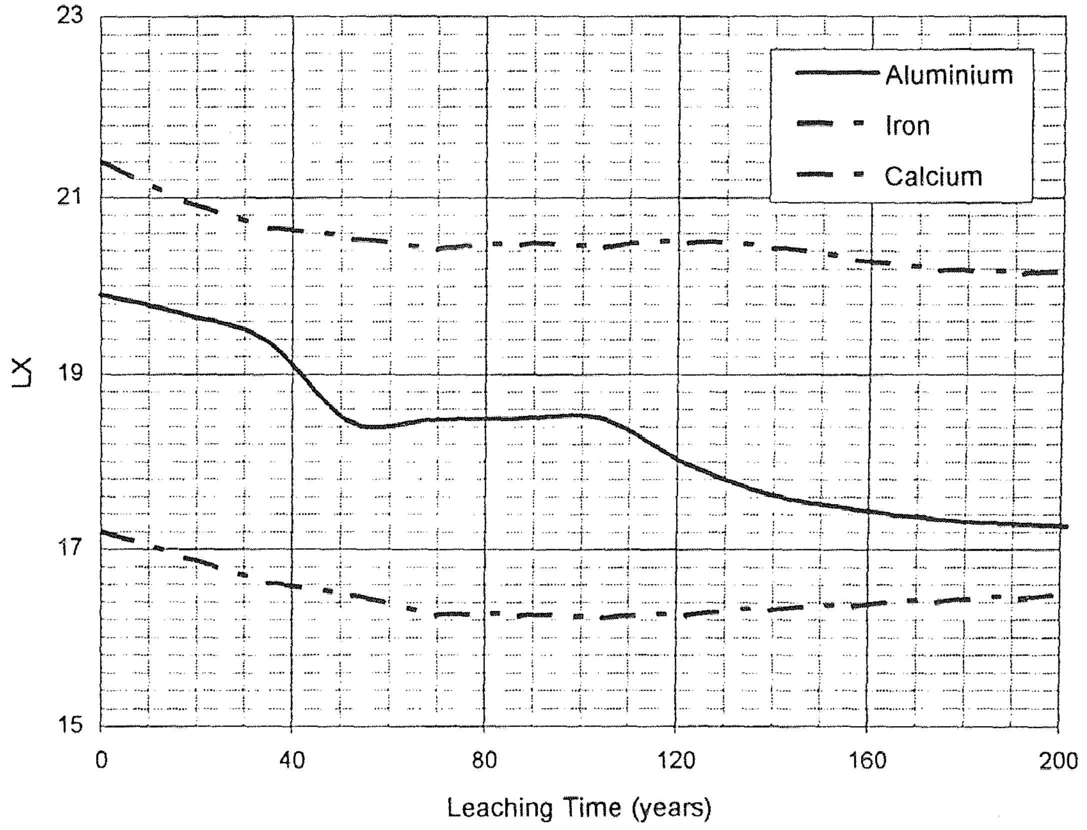

Figure 6.24 Leachability indexes for primary metal leaching from PVD sample disposed of in MSW landfill (leachant acidity =100 meq/L)

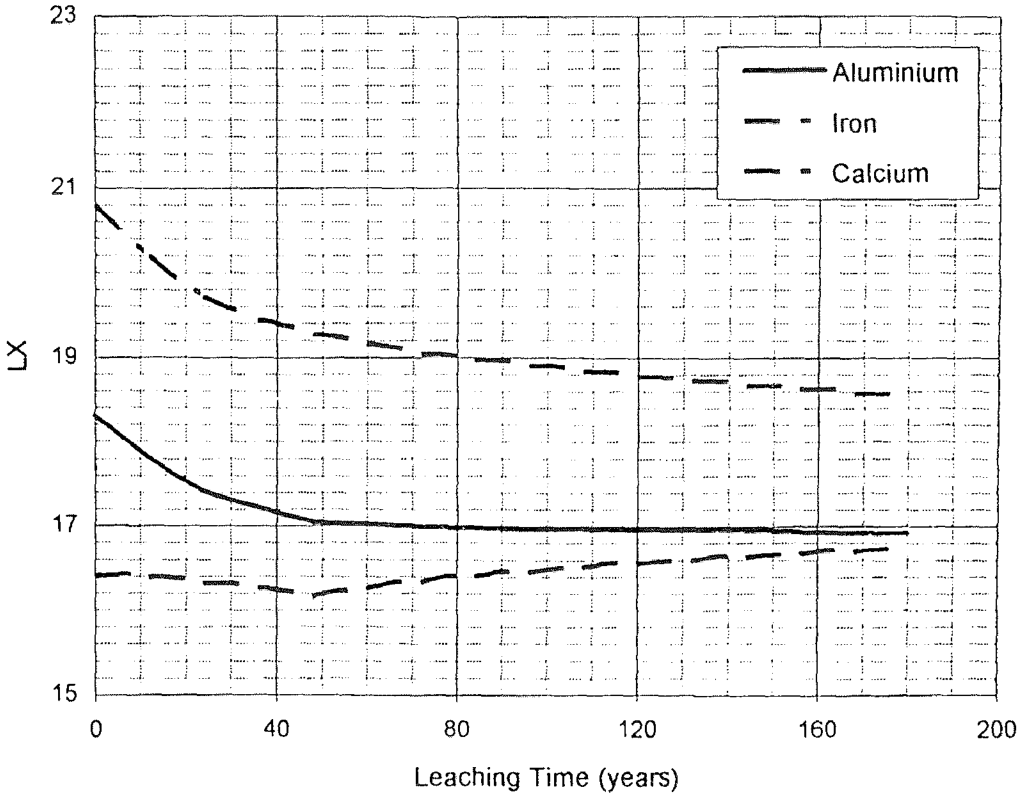

Figure 6.25 Leachability indexes for primary metal leaching from RWA sample disposed of in MSW landfill (leachant acidity =100 meq/L)

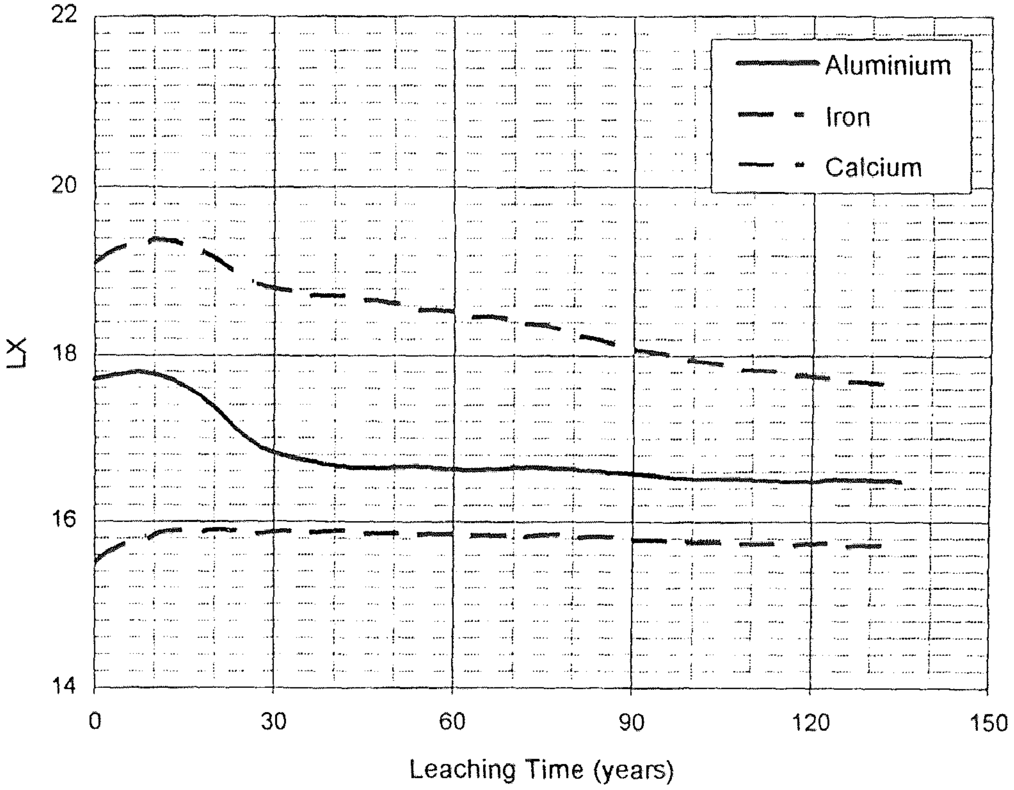

Figure 6.26 Leachability indexes for primary metal leaching from WQD sample disposed of in MSW landfill (leachant acidity =100 meq/L)

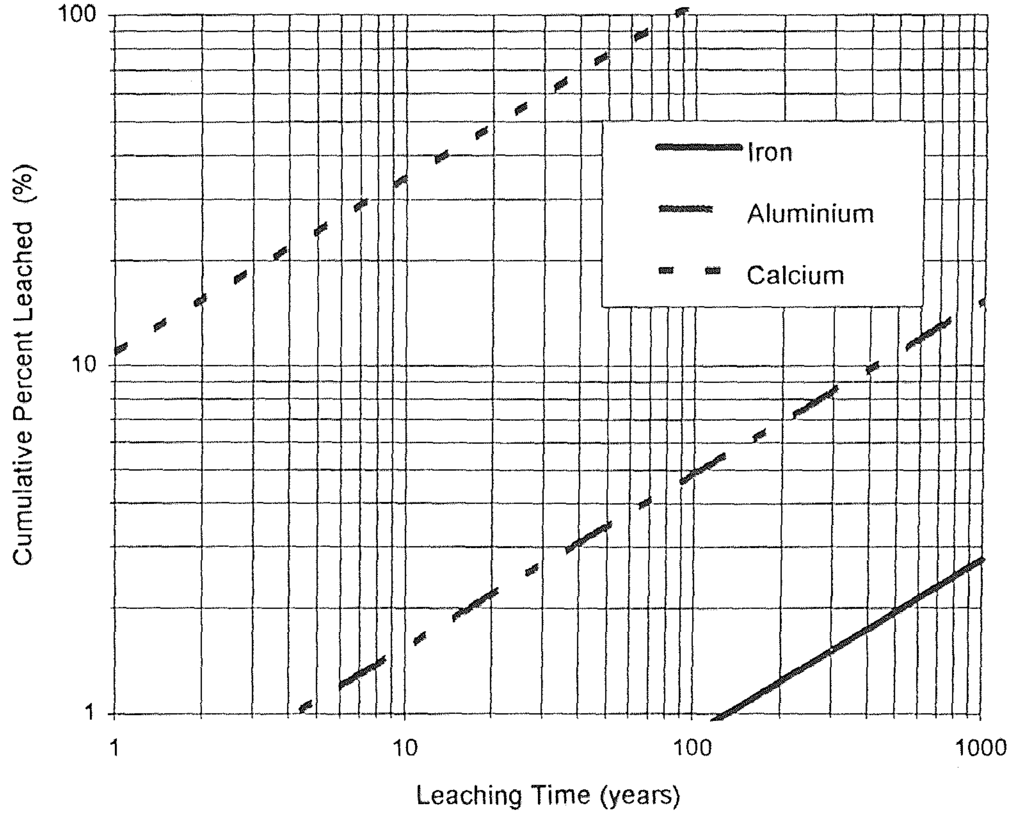

Figure 27 Predicted cumulative percent primary metals leaching from PVD samples disposed of in WTP monofill as a function of leaching time

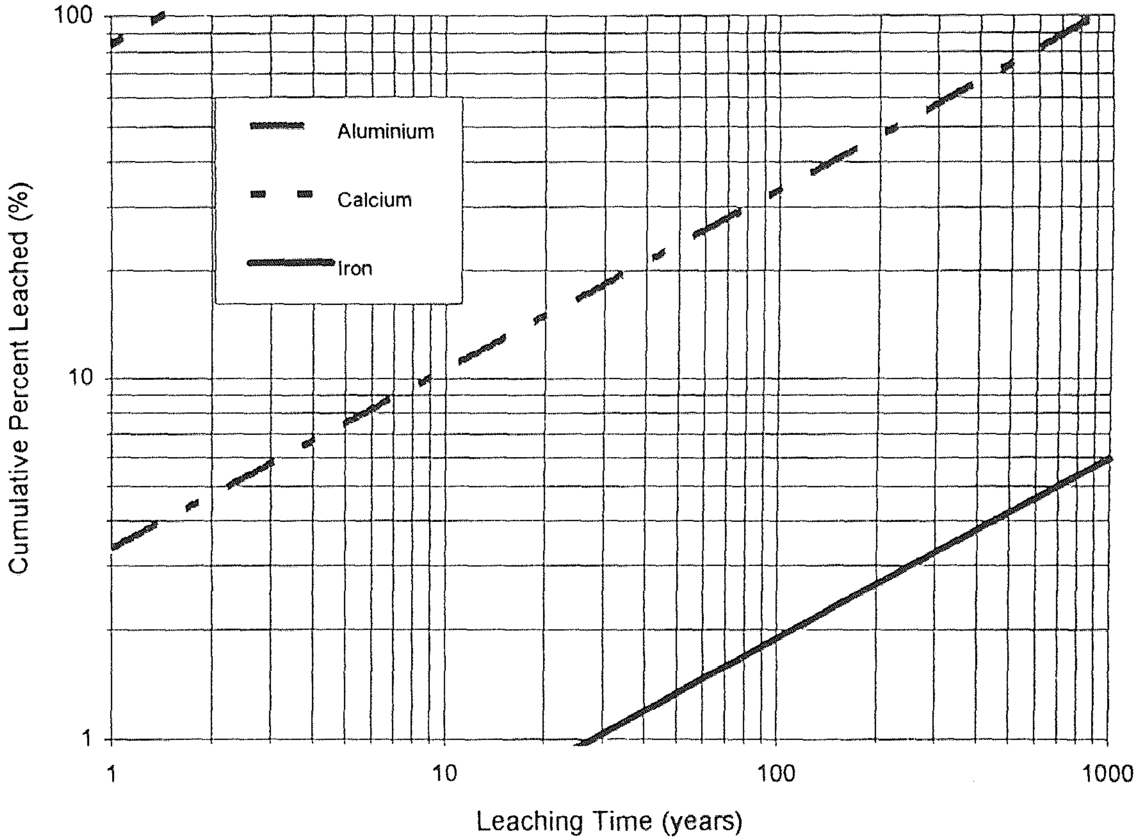

Figure 28 Predicted cumulative percent primary metal leaching from RWA samples disposed in WTP monofill as a function of leaching time

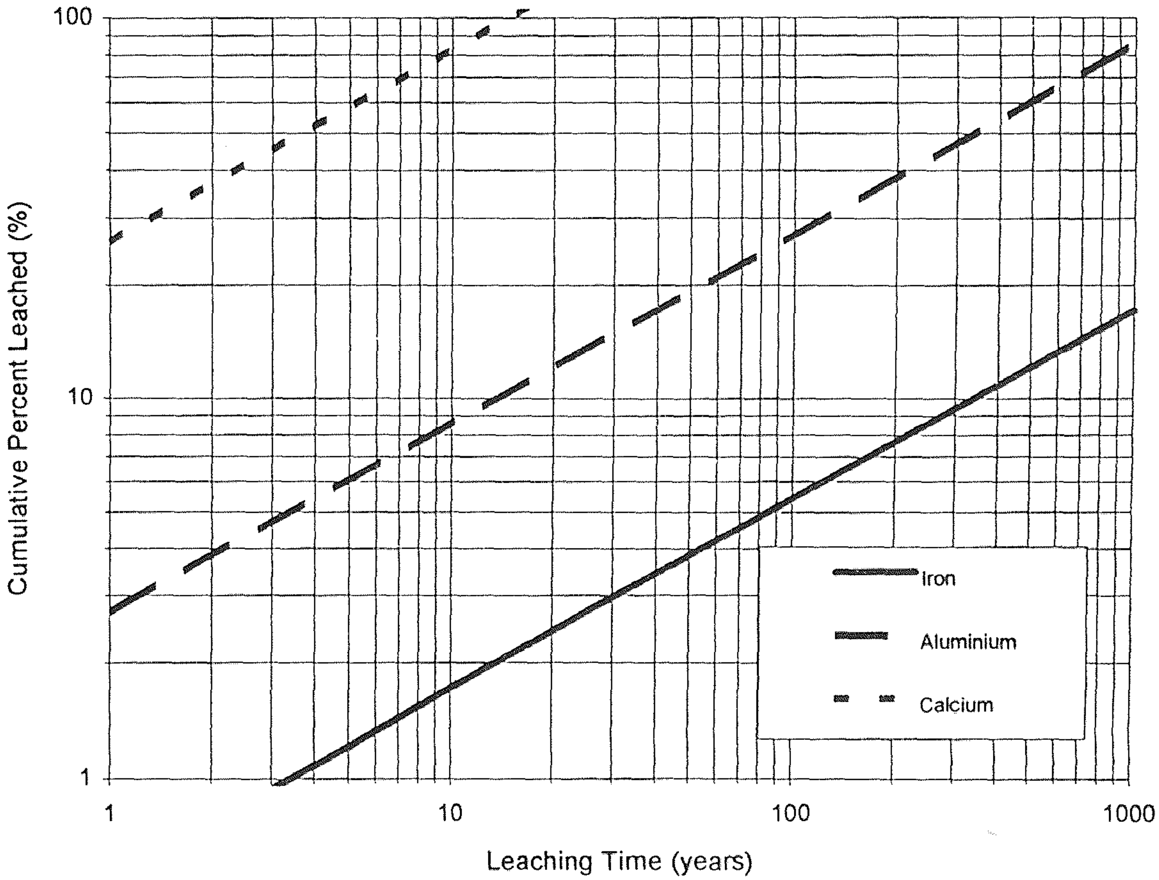

Figure 29 Predicted cumulative percent primary metal leaching from WQD samples disposed of in monofill as a function of leaching time

CHAPTER 7

SUMMARY

Disposal of water treatment plant (WTP) residual has always been an important consideration for the water industry. Disposal these residuals have raised great concerns due to the stringent water quality standards and environmental regulations, large quantities to be disposed of, and limited availability of land for ultimate disposal. The use of monofills may be one of the most economical, simplified, and feasible options.

This investigation was divided into three major phases, they are study of physical and chemical properties of WTP residuals, field testing of WTP residuals in monofills, and development of metal leaching model for WTP residual monofills.

Based on the experimental results and theoretical analysis, the following conclusions may be drawn:

7.1 Laboratory Study of Physical and Chemical Properties of WTP Residuals

Average solids content of raw residuals was 1.5 % and solids content of dewatered residuals varied from 15% to 82% in this study. It was noted that if the solids content of the residuals were below 15%, the residuals would usually fail the paint filter tests. According to environmental regulation, residuals failing this test can not be disposed of in landfills. Generally, the samples collected from Water Treatment Plants, dewatered by filter press, centrifuge, and drying bed, could pass the paint filter tests.

The pH of WTP residuals mainly depends on the coagulant added. The pH of alum and ferric residuals were generally neutral and the pH of lime residuals ranged from 9 to 12. Most residuals have high buffering capacity due to the presence of high alkalinity. The pH value and buffer capacity of lime residual were high and thus prevent metals from leaching out.

A high cation exchange capacity (CEC) means high capacity to maintain contaminants in the residuals. The CEC of most residuals analyzed in this study were close to those of clays and were generally from 20 to 35 meq/100g. The CECs of WTP dewatered residuals were found to be related to the organic content. The higher the organic content of residuals, the larger the CECs.

There is a direct relationship between what is trapped inside the residual and what can be leached out. Metal contents, pH, and CEC in dewatered residuals are related to types of water sources, impurities of water sources, and chemicals added during water purification and dewatering processes. It was also observed that as the residual ages, there is less potential leaching of metals. This could be attributed to the fact that as the solids content increases, some of the metal ions present in the residual become attached to the floc by cementation.

The toxicity characteristics leaching procedure (TCLP) analysis for WTP dewatered residuals from ten water treatment facilities were conducted. Analytical results show that the concentrations of the TCLP regulated compounds in all the samples were below the regulation limits. This indicates that the WTP residual materials tested in this study were non hazardous.

Total analysis of primary metals for various WTP residuals showed that the lime WTP residuals had 10.50%-21.15% of calcium content. This relatively high calcium concentration was attributed to use of lime as a conditioning agent or coagulant in treatment processes. The alum residuals had aluminum content from 5.84% to 13.75%. This relatively high aluminum concentration was due to the addition of alum as coagulant. The iron residual had an iron content of 21.36% because of the use of iron as coagulant agent. These analysis indicate the composition of WTP residuals are directly related to the coagulants and conditioning agents added.

Alum residual is primarily composed of aluminum, iron, and calcium, (15.5% by weight). Manganese and magnesium each makes up less than 1% (by weight) of residual. Minor trace metals account for less than 0.01% (by weight).

From leaching test results, it was found that only 30 percent of the total alkalinity in lime residual was used for acid neutralized if leachate is present, whereas iron and alum residuals had less alkalinity for acid neutralization. The test results also show that the buffering capacity of iron and alum residuals would decrease rapidly if there are large amounts of acid present, which is common in MSW landfill.

7.2. Field Study of WTP Residuals in a WTP Monofill

The results of the field study can be summarized as follows: The paint filter test was employed to test the aged samples. There was no free liquid observed in six-month, and one-year aged WTP dewatered samples taken from minimonofill. This could be attributed to increased solids content.

There was no significant change in solids content among the fresh, six-month, and one year-old samples taken from the same layer. The total solids content of samples taken from top layer was much higher than that of the sample below the top layer (about 6 inches). The increase of solids content was probably caused by freezing and formation of ice during the winter season, which drove some water out from the residual.

The volatile solids did not change very much (18.32% to 20.48%), during the period of study. The results indicated that most organics remained in WTP residuals and there was no significant biodegradation noticed.

The WTP residual had a good pH buffering capacity. This was attributed to the substantial alkalinity and $CaCO_3$ concentrations of the WTP residual. The pH buffering capacity and the insoluble substances (retarding layer) formed inside the monofill prevented significant leaching of metals from the minimonofill residual.

Results of total analysis of the TCLP 39 defined constituents indicate that the total concentrations of metals, pesticides, herbicides, BNA, and VOCs were less than the method detection limits (MDLs) and much less than the TCLP regulatory limits.

7.3. Development of Metal Leaching Modeling for WTP Residual Monofills

The metals leaching model was developed based on diffusion, metals solubility, and adsorption mechanisms. Several factors may affect the leaching characteristics. These include the characteristics of residual, particle size, leachant composition, pH and flow conditions.

The small particle size of WTP residuals results in larger surface area. This is one of reasons for absorption of metals and to prevent them from leaching, but this also will increase the diffusion effect. The leaching model shows that the cumulative fraction of metals leached at any time is directly related to particle surface area. Consequently, larger particles leach less than smaller ones in the long run.

Leaching is affected by solubility. The higher the solubilities, the more metals will be leached. The amount of calcium leached, for example, which has the highest solubility constant, was the highest. Most of iron and aluminum remained in solid form because the their solubility constants are relatively smaller.

The values of pH of residuals and leachant will affect the leaching rates. The results indicate that primary metals in iron or alum residuals will leach out quickly with a pH less than 4.0. Lime residual can maintain a high pH and dissolved metal concentrations did not changed very much under the pH values from 2 to 12.

The diffusion-based leaching model was utilized for prediction of metal leaching rates in WTP monofill or MSW landfill. The calculation procedure was implemented by computer program to predict the leaching rate. The results of leachability index (LX) for three types of WTP residuals showed that iron, aluminum, calcium, are immobile in WTP residuals. WTP monofill disposal is better than MSW landfill disposal based on the leachbility studies and leaching model prediction.

APPENDIX A

QUALITY CONTROL AND QUALITY ASSURANCE

Determination of Trace Elements in WTP Residuals by Inductively Coupled Plasma - Mass Spectrometry (ICP -MS)

ICP-MS is used to determine the concentration of metals in extracts prepared from acid digestion. ICP-MS instrument operating conditions for precision and recovery data listed in Table A1.

A reference standard solution used to determine acceptable instrument performance prior to calibration and sample analyses were employed and is shown in Table A2.

This solution contains known concentrations of method analyses which is used to fortify an aliquot of Library Reagent Blank (LRB) matrix. The Quality Control Standard (QCS) prepared by Marine Analytical Chemistry Standard Program was used to check laboratory performance. The minimum detect limit (MDT) is determined by the analysis of laboratory blanks. The average blanks and 3 times standard deviation was used as the Method Detection Limit. An example is shown in Table A3.

Determination of Pesticides-Herbicides in Extracts of WTP Dewatered Residuals Using GC-ECD Work Station

A formal quality control program was implemented in this analysis. This program consists of an initial demonstration of instrument capability and an ongoing analysis of spiked samples to evaluate data quality. Records are maintained to document the quality of the data generated.

If the analytical result is below the value of blank average plus 3 times standard deviation, then it will be reported as the average blank plus three standard deviations.

The demonstration of interferences under control from analytical system, glassware, and reagents was performed through the analysis of a reagent water blank. The blank samples were carried out through all stages of sample preparation and measurement steps.

The GC-ECD system is evaluated using performance evaluation mix to ensure that minimum average response factors (RFs) are met before the calibration curve was used. For pesticides, the System Performance Evaluation Mix (SPEMs) are: 10 ppm of α-Cyclohexane (BHC), 10 ppm of β-Cyclohexane (BHC), and 10 ppm of γ-Cyclohexane(BHC). The results for system performance check compounds are presented in Table A4.

To assess accuracy, reagent blank, matrix spike, and matrix spike duplicate were analyzed for each analytical batch (up to a maximum of 20 samples /batch) on a ongoing basis.

To determine acceptable accuracy and precision limits for surrogate standards by which the efficiency and recovery of preparative extraction procedure and instrument are evaluated, by conducting to the following procedures:

The percent recovery of each surrogate in each analyzed sample and blank is calculated. Once a minimum of thirty samples of the same matrix had been analyzed, the average percent recovery (p) and standard deviation of the percent recovery (s) for each of surrogates were calculated. An example of test result is shown in Table A5.

The upper and lower control limit for method performance for each surrogate standard were calculated by the following equations:

Upper Control Limit (UCL) = p + 3s

Lower Control Limit (LCL) = p - 3s

If recovery does not fall within limits, the instrument performance should be checked and the extract should be reanalyzed.

If none of the above is a problem or the data is flagged as "estimated concentration", the sample is recalculated and reanalyzed.

Determination of Semivolatile Organic Compounds in Extracts of WTP Dewatered Residuals Using Gas Chromatography-Mass Spectrometry

A formal quality control program was implemented in this analysis. This program consists of an initial demonstration of instrument capability and an ongoing analysis of spiked samples to evaluate data quality. Records were maintained to document the quality of the data generated. The demonstration of interferences under control from analytical system, glassware, and reagents under control was performed through the analysis of a reagent water blank. The blank samples were carried out through all stages of the sample preparation and measurement steps.

The GC-MS system was tuned to meet the criteria for a 50-ng injection of DFTPP as shown in Table A6.

The GC-MS tuning standard was also used to assess GC column performance and injection port inertness. Degradation of DDT to DDE and DDD should not exceed 20 percent. Benzidine and pentachlorophenol were present at the normal responses, and no peak tailing was visible. Corrective actions were taken prior to any analysis. The internal standards selected below were used for the quantitation.

The average RF and percent relative standard deviation (%RSD = 100[SD/RF] were calculated for each compound. The %RSD should be less than 30% for each compound. However, the %RSD for each individual calibration check compound (CCCs) must be less than 30 percent. The calibration check compounds, hexachlorobutadiene, pentachlorophenol and 2,4,6-trichlorophenol, are used for the daily GC-MS calibration. The relative retention time of each compound in each calibration run should agree within 0.06 relative retention time unit.

A system performance check was performed to ensure that minimum average response factors (RFs) were met before the calibration curve was used. An example of the results for the System Performance Check Compounds (SPCCs) is shown in Table A7.

For semivolatiles, the System Performance Check Compounds (SPCCs) are 2,4-dinitrophenol and 4-nitrophenol. The accuracy concentration and retention time of each compound met the criteria. The minimum acceptable average RF for these compounds SPCCs is 0.050. The response factors for all compounds met this SPCC criterion. Then, the linear calibration curve consisting of five levels for each target compound is made. Prior to analysis of samples, the GC-MS tuning standard, 50-ng DFTPP was analyzed and resulted in a mass spectrum which met the criteria. Calibration standards at mid-level concentration containing the regulated semivolatile analyte include internal standards and surrogate standards. The response factor data from the standards are compared with the average response factor from the initial calibration as per the system performance check compounds (SPCCs) and calibration check compounds (CCCs) criteria. A system performance must be made by using system performance check compounds (SPCCs). If the SPCCs criteria are met, a comparison of response factors is made for all compounds. If the minimum response factor is not met with, the system must be evaluated, and corrective action must be taken prior to sample analysis. The minimum RF for semivolatile SPCCs is 0.050. After the performance check is met, calibration check compounds (CCCs) are used to check the validity of the initial calibration. The internal standard responses and retention times in the calibration standards must be evaluated immediately after or during data acquisition. If the retention time for any internal standard changes by more than 30 sec from the last check calibration (12 hr earlier), the chromatographic system must be inspected for malfunctions and corrections must be made. In addition, if the EICP area for any of the internal calibration standard changes by a factor of two (-50% to +100%) from the last daily calibration standard check, the mass spectrometer must be inspected for malfunctions and corrections must be made.

To generate acceptable accuracy and precision, the following operations were performed. To assess accuracy, a reagent blank, a matrix spike, and a matrix spike duplicate were analyzed for each analytical batch (up to a maximum of 20 samples /batch) on an ongoing basis. To determine acceptable accuracy and precision limits for surrogate standards the efficiency and recovery of preparative extraction procedure and instrument is evaluated according to the following procedures. The percent recovery of each surrogate in each analyzed sample and blank is calculated. Once a minimum of thirty samples of the same matrix has been analyzed, the average percent recovery (p) and standard deviation of the percent recovery (s) for each of surrogates are calculated, as shown in Table A8.

The upper and lower control limits were calculated for method performance for each surrogate standard by the following equations:

$$\text{Upper Control Limit (UCL)} = p + 3s$$

$$\text{Lower Control Limit (LCL)} = p - 3s$$

Determination of Volatile Organic Compounds in WTP Dewatered Residuals Using Gas Chromatography/Mass Spectrometry with Purge-Trap Device

GC-MS system met BFB key ion abundance criteria after hardware-tuning for a 100-ng purging of 4-bromofluorobenzene (2-µL injection of the BFB standard). An example of test results is shown in Table A9.

The calibration standards at five concentration levels and regression relative coefficients were determined to evaluate the calibration curve. The detection limit was used as a minimum concentration level of standard (0.010 mg/L). Each compound was calculated with internal standard method to correct for instrument drift and physical interferences.

A system performance check was made before each standard curve was used. Four compounds (the System Performance Check Compounds or SPCCs) were checked for a minimum average response factor. These compounds are chloromethane, 1,1-dichloroethane, bromoform, and chlorobenzene. The average RF for these compounds should be greater than 0.300 (0.25 for bromoform) which was the minimum acceptable average RF as shown in Table A10.

A reagent blank, TCLP blank, and a matrix spike were analyzed for each analytical batch to assess accuracy. The percent recovery (Ps) of each analyte had fallen in the advisory limit range 30 to 130 percent. The surrogate standards added into each sample were determined, and the percent recovery of each surrogate in the sample was within the acceptable limit range (30 to 130 percent). An example of test results is shown in Table A11.

Table A1 ICP-MS Operating conditions

Instrument	VG PlasmaQuad Type II
Plasma forward power	1.35 kW
Coolant flow rate	13.5 L/min
Auxiliary flow rate	0.6 L/min
Nebulizer flow rate	0.78 L/min
Solution uptake rate	0.6 ml/min
Spray chamber temperature	15 °C
Data Acquisition	
Detector mode	Pulse and Analog counting
Replicate integration	3
Mass range	8-240 amu
Total acquisition time	3 minutes per sample

Table A2 Reference standard solution for ICP-MS

	Laboratory Fortified Sample Matrix				
Element	Mass		Certified Conc. (ppb)	Tested Conc. (ppb)	RPD (%)
Magnesium	Mg	24	1510 ± 130	1502	0.53
Aluminum	Al	27	84.4 ± 3.4	84.8	0.47
Barium	Ba	138	13.8 ± 0.3	13.4	2.90
Chromium	Cr	52	0.45 ± 0.07	0.363	19.3
Cobalt	Co	59	0.063 ± 0.012	0.073	15.87
Copper	Cu	65	2.76 ± 0.17	2.28	17.39
Iron	Fe	57	129 ± 7	165	27.91
Lead	Pb	208	0.129 ± 0.011	0.140	8.53
Manganese	Mn	55	10.1 ± 0.3	13.953	38.15
Nickel	Ni	60	1.03 ± 0.10	1.141	10.78
Strontium	Sr	88	27.3 ± 0.4	25.782	5.56
Zinc	Zn	66	3.33 ± 0.15	2.204	33.81

RPD--Relative percent deference between certified and tested concentration.

Table A3 Method detection limit (MDT) in the batch system

Element		Blank1 (ppb)	Blank2 (ppb)	Blank3 (ppb)	Ave. Blk+3S (ppb)
Arsenic	As	1.78	2.51	1.64	3.12
Barium	Ba	0.111	0.197	0.254	0.364
Cadmium	Cd	0.0815	0.499	0.0815	0.811
Chromium	Cr	1.02	1.67	0.936	2.193
Lead	Pb	0.266	0.416	0.262	0.53
Mercury	Hg	0.452	0.424	0.855	1.168
Selenium	Se	4.42	3.28	6.74	9.132
Silver	Ag	0.152	0.122	0.319	0.458

Table A4 System performance check compounds

Compounds	Average Percent (%)	Standard Deviation (%)
α-Cyclohexane (BHC)	93	8
β-Cyclohexane (BHC)	90	10
γ-Cyclohexane (BHC)	102	15

Table A5 Accuracy and precision for pesticides in the samples

Parameter	Sample Percent Recovery (%)	Status*
2,4,5,6-Tetrachloro-M-xylene	84	OK
Decachlorobiphenyl	39	OK
Lindane	90	OK
Heptachlor	81	OK
Aldrin	33	OK
Dieldrin	56	OK
Endrin	85	OK
4,4'-DDT	87	OK

* Advisory surrogate limits (20-150 %).

Table A6 DFTPP key Ions and ion abundance criteria

Mass	Ion Abundance Criteria	Met Percent (%)
51	30-60% of mass 198	35.67
68	<2% of mass 69	0
70	<2% of mass 69	0
127	40-60% of mass 198	42.27
197	<1% of mass 198	0
198	Base peak, 100% relative abundance	100
199	5-9% of mass 198	6.19
275	10-30% of mass 198	31.34
365	>1% of mass 198	5.15
441	present but less than mass 443	9.28
442	>40% of mass 198	63.71
443	17-23% of mass 442	16.82

Table A7 Response factors for system performance check compounds

Compounds	RF	Criteria
N-nitroso-N-propyl-1- propamine	0.263	0.05
1,2,3,4,5,5-Hexachloro-1,3-cyclopentadiene	0.194	0.05
2,4-Dinitrophenol	0.210	0.05
4-Nitrophenol	0.372	0.05

Table A8 Surrogate and matrix spike recovery limits for the samples

Compound	Recovery limits (%)	Recovery of sample (%)
Nitrobenzene-d5	35 - 114	60
2-Fluorobiphenyl	43 - 116	76
p-Terphenyl-d14	33 - 141	80
Phenol-d6	10 - 94	100
2-Fluorophenol	21 - 100	101
2,4,6-Tribromophenol	10 - 123	98

Table A9 BFB key ions abundance criteria and met percent

Mass	Ion Abundance Criteria	Met Percent (%)
50	15 to 40% of mass 95	15.87
75	30 to 60% of mass 95	39.64
95	Base Peak, 100% Relative Abundance	100
96	5 to 9% of mass 95	8.97
173	<2% of mass 174	0
174	>50% of mass 95	98.71
175	5 to 9% of mass 174	8.81
176	>95% but < 101% of mass 174	98.91
177	5 to 9 % of mass 176	8.96

Table A10 Response factors for VOCs system performance check compounds

Compounds	RF	Criteria
Chloromethane	1.237	0.300
1,1-dichloroethane	2.708	0.300
Bromoform	0.260	0.250
Chlorobenzene	1.071	0.300

Table A11 Accuracy and precision for VOCs in the samples

Parameter	Blanks		Aqueous Sample	
	Average Percent Recovery	Standard Deviation (%)	Average Percent Recovery	Standard Deviation (%)
1,2-Dichloroethane-d4	90	5	115	24
Tolune-d8	97	8	95	65
4-Bromoflurobenzene	97	4	106	24
1,1-Dichloroethylene	120	20	120	26
Benzene	109	16	101	20
Trichloroethylene	107	16	96	19
Chlorobenzene	91	9	81	20

APPENDIX B

EXAMPLE OF CALCULATION OF PREDICTION OF METAL LEACHING RATES IN WTP MONOFILL AND MSW LANDFILL

Examples of Prediction of Metal Leaching Rates in WTP Monofill and MSW Landfill

PVD sample had high concentration of aluminum (4.78%) in this study. Aluminum is used as an example in this model. Two leachant are considered - typical rainwater and landfill leachate. These simulate a WTP residual disposed in a monofill or a MSW landfill. Acidities of 0.5 meq/l and 100 meq/l are used in the leachant. An acidity of 0.5 meq/l and 100 meq/l are equivalent to 25 and 5000 mg/l as $CaCO_3$, respectively.

In this example, PVD residuals is placed in a monofill or MSW landfill. The residual is 2.0 m thick, the rainfall intensity is 1.0 m/year and all rain penetrates the residuals evenly rather than runs off. Obviously, these are extremely conservative assumptions. The solids contents of PVD is 29% and a residual density of 1500 kg/m^3 is used.

The following analysis uses the procedures outlined above.

1) Determine the effective diffusion coefficients and leachability indices for WTP residuals.

a) Multiple batch extraction tests have been performed on residual. The results of the effective diffusion coefficients and leachability indices for PVD is presented in Tables 6.4.

2) The acidity of the expected leachant.

a) Two leachant acidities are considered:

rainfall acidity = 0.5 meq/l

landfill leachant acidity = 100 meq/l

3) Determine projected leachant velocity

a) Assume the residuals are above the ground water table and are fractured, and that there is no clay cover materials so that all rainwater impacting the site penetrates through the WTP residuals. This is obviously the worst case situation.

b) Rainfall = 1.0 m/year.

c) Therefore, leachant velocity = 1.0 m/year

$= 1.0\ m^3/m^2$ surface area/year.

4) Analyze the particle size or the S/V ratio.

Results for PVD residual is listed in Table 6.2.

5) Determine the LX of WTP residuals in real leaching time based on the short term LX results for 15 batch extractions and the mass balance of acid added.

a) Determine the amount alkalinity leached from the waste and the resulting leachant pH versus amount of acid added.

i) Perform an alkalinity titration. titration results os PVD samples is shown in Table 6.2.

ii) In each extraction, as shown in Table 6.7, 2 meq acid/g waste/day was added.

iii) Acidity of leachate (100 meq/l) caused by decomposition in MSW landfill.

$= (100\ meq/l)(1.0\ m^3/m^2\ /yr)(1000\ l/m^3)$

$= 100{,}000\ meq\ acid/m^2/yr$

iiii) Acidity of leachate (500 μeq/l) due to rainfall

$= (0.5 \text{ meq/l})(1.0 \text{ m}^3/\text{m}^2/\text{yr})(1000 \text{ l/m}^3)$

$= 500 \text{ meq acid/m}^2/\text{yr}$

iiiii) Assume residual is laid 2.0 m thick, the a solids content is 29%, and it has a density of 1500 kg/m^3

The amount of residuals per surface area (g/m^2)

$= 2.0 \text{ (m)} \times 1500 \text{ (kg/m}^3) \times 0.29 \times 1000 \text{ (g/kg)}$

$= 8.7 \times 10^5 \text{ (g/m}^2)$

acidity added from by landfill leachate

$$= \frac{100{,}000 meq / m^2 / yr}{(8.7 \times 10^5 g / m^2)}$$

$= 0.115 \text{ meq/g waste/yr}$

acidity added by rainfall

$$\frac{500 meq / m^2 / yr}{(8.7 \times 10^5 g / m^2)}$$

$= 5.747 \times 10^{-4} \text{ meq/g waste/yr}$

b) Determine LX of the WTP residuals in real leaching time

Based on the above results and acid mass balance, each extraction performed in the multiple batch extraction test is equivalent to the following real field leaching times.

$$\text{landfill leachate} = \frac{2\ meq / g}{(0.115\ meq / g / yr)} = 17\ yr.$$

$$\text{rainfall leachate} = \frac{2\ meq/g}{(5.7471x10^{-4}\ meq/g/yr)} = 3480\ yr.$$

Table 6.7 shows the relationship between extraction time in multiple batch extraction conducted in the labaratory and the real the leaching time in a monofill (rainwater) or MWS landfill (acidic leachant).

c) Determine LX of the WTP residuals for the leaching time calculated above in a specific monofill (0.5 meq/l rainwater) and a MSW landfill (100 meq/l acidic leachant) using LX obtained from multiple batch extraction tests and leaching time listed in Table 6.7. Those calculated results are summarized in Figures 6.25, based on regressional analysis.

6) Determine the annual amount of metal leaching rates of PVD sample disposed in WTP monofill

or a MSW landfill

a) Use Equation 6.16 with appropriate LX, which can be found in Figure 6.25 with specific leaching time and S/V values in Table 6.2.

b) The following calculation is based on a S/V value of 6846 and a LX value of 19.83:

Cumulative fraction of aluminum leached to year 9.

$$\frac{\sum a_9}{A_o} = 1.128 \times 10^{(-0.5\times 19.83)} \times \left(9yr \times \frac{3153600\sec}{yr}\right)^{0.5} \times \left(6846cm^{-1}\right)$$

$= 0.01582$

Cumulative fraction of aluminum leached in year 10

$$\frac{\sum a_{10}}{A_o} = 1.128 \times 10^{(-0.5\times 19.83)} \times \left(10yr \times \frac{3153600\sec}{yr}\right)^{0.5} \times \left(6846cm^{-1}\right)$$

$= 0.01668$

Fraction of initial aluminum leached from ninth to tenth year period

$= 0.00086$

Initial Al concentration = 47.784 (g/kg waste)

For a 2.0 m thickness residual with a solids content of 29% and a density of 1500 kg/m^3,

Total amount of dry residuals = 870 (kg/m^2 surface)

Al = (47.784 g/kg)(870 kg/m^2)=41,572 (g Al/m^2 surface)

Amount of aluminum leached from ninth to tenth year period

= (0.00086)(41,572 g/m^2)

= 35.75 (g Al/m^2 surface)

7) Determine aluminum concentration in the leachate.

a) Leachate volume = 1.0 m^3/m^2 surface/yr

b) For year 10:

$$\text{Al concentration in landfill leachate} = \frac{35.75\, g\ / m^2}{(1.0\, m^3 / m^2 / yr)} = 35.75 \text{ mg/l}$$

8) Determine the aluminum concentration in groundwater

a) Assume a dilution factor for leachate in groundwater of 100:1 and apply this to the results in step 7.

$$\text{Al concentration in groungwater (mg / l)} = \frac{35.75 \text{ mg / l}}{100} = 0.36 \text{ mg / l}$$

Table B1 Projected iron concentrations in leachate and groundwater as a function of leaching time for RWA sample which was disposed in MSW landfill or WTP monofill

Leaching Time (years)	LX	Iron Leached				
		(cumulative fraction of initial Fe)	(fraction per year)	Fe leached per year (g Fe/m 2 yr.	Leachate Conc. (Fe mg/L)	Groundwater Conc. (Fe mg/L)
		MSW Landfill Leachate				
0	20.80	0.0000				
1	20.80	0.0019	0.00188	1204.43	1204.4	12.04
9	20.70	0.0063				
10	20.70	0.0067	0.00034	219.30	219.3	2.19
49	20.32	0.0229				
50	20.32	0.0231	0.00023	148.75	148.7	1.49
99	19.90	0.0527				
100	19.90	0.0530	0.00027	170.15	170.2	1.70
149	19.58	0.0935				
150	19.58	0.0938	0.00031	200.65	200.6	2.01
199	19.40	0.1329				
200	19.40	0.1333	0.00033	213.69	213.7	2.14
		Rainwater Leachate (WTP Monofill Leachate)				
0	20.80	0.0000				
1	20.80	0.0019	0.00188	1204.43	1204.4	12.04
9	20.80	0.0056				
10	20.80	0.0059	0.00031	195.45	195.5	1.95
49	20.80	0.0132				
50	20.80	0.0133	0.00013	85.60	85.6	0.86
99	20.80	0.0187				
100	20.80	0.0188	0.00009	60.37	60.4	0.60
149	20.80	0.0229				
150	20.80	0.0230	0.00008	49.25	49.3	0.49
199	20.80	0.0265				
200	20.80	0.0266	0.00007	42.64	42.6	0.43

Assumptions:

S/V (cm^{-1}) = 7.455 x10^3
Initial Fe concentration (g/kg) = 213.55 (213550 ppm)
LX decrease over time
Percolating water (m^3 /m^2 -yr.) = 1
WTP residual material (kg/m^2 surface (2 m thick) = 3000
Leachate acidity (meq/L) = 100
Groundwater concentration (meq/L) = 0.01 x leachate concentration 1
Groundwater dilution factor = 100

Table B2 Projected aluminum concentrations in leachate and groundwater as a function of leaching time for RWA sample which was disposed in MSW landfill or WTP monofill

Leaching Time (years)	LX	Aluminum Leached				
		(cumulative fraction of initial Al)	(fraction per year)	Al leached per year (g Al/m^2 yr.)	Leachate Conc. (Al mg/L)	Groundwater Conc. (Al mg/L)
		MSW Landfill Leachate				
0	18.30	0.0000				
1	18.30	0.0334	0.03343	1171.45	1171.5	11.71
9	18.20	0.1125				
10	18.20	0.1186	0.00609	213.30	213.3	2.13
49	17.88	0.3795				
50	17.88	0.3834	0.00385	135.02	135.0	1.35
99	17.55	0.7888				
100	17.55	0.7928	0.00397	139.25	139.2	1.39
149	17.55	0.9677				
150	17.55	0.9710	0.00324	113.60	113.6	1.14
199	17.55	1.1184				
200	17.55	1.1212	0.00281	98.34	98.3	0.98
		Rainwater Leachate (WTP Monofill Leachate)				
0	18.30	0.0000				
1	18.30	0.0334	0.03343	1171.45	1171.5	11.71
9	18.30	0.1003				
10	18.30	0.1057	0.00543	190.10	190.1	1.90
49	18.30	0.2340				
50	18.30	0.2364	0.00238	83.25	83.3	0.83
99	18.30	0.3326				
100	18.30	0.3343	0.00168	58.72	58.7	0.59
149	18.30	0.4081				
150	18.30	0.4095	0.00137	47.90	47.9	0.48
199	18.30	0.4716				
200	18.30	0.4728	0.00118	41.47	41.5	0.41

Assumptions:

S/V (cm^{-1}) = 7.455 x10^3
Initial Al concentration (g/kg) = 11.68 (11680 ppm)
LX decrease over time
Percolating water (m^3/m^2-yr.) = 1
WTP residual material (kg/m^2 surface (2 m thick) = 3000
Leachate acidity (meq/L) = 100
Groundwater concentration (meq/L) = 0.01 x leachate concentration 1
Groundwater dilution factor = 100

Table B3 Projected calcium concentrations in leachate and groundwater as a function of leaching time for RWA sample which was disposed in MSW landfill or WTP monofill

Leaching Time (years)	LX	Calcium Leached (cumulative fraction of initial Ca)	(fraction per year)	Ca leached per year (g Ca/m 2 yr.)	Leachate Conc. (Ca mg/L)	Groundwater Conc. (Ca mg/L)
		MSW Landfill Leachate				
0	16.40	0.0000				
1	16.40	0.2736	0.27362	4079.69	4079.7	40.80
9	16.40	0.8209				
10	16.40	0.8653	0.04440	662.04	662.0	6.62
14	16.42	1.0005				
15	16.42	1.0356	0.03512	523.57	523.6	5.24
19	16.42	1.1655				
20	16.42	1.1958	0.03028	451.46	451.5	4.51
34	16.40	1.5955				
35	16.40	1.6188	0.02329	347.30	347.3	3.47
39	16.42	1.6699				
40	16.42	1.6911	0.02127	317.18	317.2	3.17
		Rainwater Leachate (WTP Monofill Leachate)				
0	16.40	0.0000				
1	16.40	0.2736	0.27362	4079.69	4079.7	40.80
9	16.40	0.8209				
10	16.40	0.8653	0.04440	662.04	662.0	6.62
19	16.40	1.1927				
20	16.40	1.2237	0.03098	461.97	462.0	4.62
29	16.40	1.4735				
30	16.40	1.4987	0.02519	375.58	375.6	3.76
39	16.40	1.7088				
40	16.40	1.7305	0.02177	324.57	324.6	3.25
49	16.40	1.9153				
50	16.40	1.9348	0.01945	289.93	289.9	2.90

Assumptions:

S/V (cm^{-1}) = 6.846 x10^3
Initial Ca concentration (g/kg) = 4.97 (4970 ppm)
LX decrease over time
Percolating water (m^3/m^2-yr.) = 1
WTP residual material (kg/m^2 surface (2 m thick) = 3000
Leachate acidity (meq/L) = 100
Groundwater concentration (meq/L) = 0.01 x leachate concentration 1
Groundwater dilution factor = 100

Table B4 Projected aluminum concentrations in leachate and groundwater as a function of leaching time for WQD sample which was disposed in MSW landfill or WTP monofill

		Aluminum Leached				
Leaching Time (years)	LX	(cumulative fraction of initial Al)	(fraction per year)	Al leached per year (g Al/m^2 yr.)	Leachate Conc. (Al mg/L)	Groundwater Conc. (Al mg/L)
MSW Landfill Leachate						
0	17.70	0.0000				
1	17.70	0.0271	0.02710	11179.84	11179.8	111.80
9	17.73	0.0785				
10	17.73	0.0828	0.00425	1752.65	1752.6	17.53
49	17.80	0.1691				
50	17.80	0.1708	0.00172	708.12	708.1	7.08
99	17.60	0.3026				
100	17.60	0.3041	0.00152	628.78	628.8	6.29
149	17.20	0.5883				
150	17.20	0.5903	0.00197	812.99	813.0	8.13
199	16.80	1.0776				
200	16.80	1.0803	0.00270	1115.41	1115.4	11.15
Rainwater Leachate (WTP Monofill Leachate)						
0	17.70	0.0000				
1	17.70	0.0271	0.02710	11179.84	11179.8	111.80
9	17.70	0.0813				
10	17.70	0.0857	0.00440	1814.24	1814.2	18.14
49	17.70	0.1897				
50	17.70	0.1916	0.00193	794.53	794.5	7.95
99	17.70	0.2697				
100	17.70	0.2710	0.00136	560.40	560.4	5.60
149	17.70	0.3308				
150	17.70	0.3319	0.00111	457.18	457.2	4.57
199	17.70	0.3823				
200	17.70	0.3833	0.00096	395.76	395.8	3.96

Assumptions:

S/V (cm^{-1}) = 3.029 x10^3
Initial Al concentration (g/kg) = 137.5 (137500 ppm)
LX decrease over time
Percolating water (m^3 /m^2 -yr.) = 1
WTP residual material (kg/m^2 surface (2 m thick) = 3000
Leachate acidity (meq/L) = 100
Groundwater concentration (meq/L) = 0.01 x leachate concentration 1
Groundwater dilution factor = 100

Table B5 Projected iron concentrations in leachate and groundwater as a function of leaching time for WQD sample which was disposed in MSW landfill or WTP monofill

Leaching Time (years)	LX	Iron Leached				
		(cumulative fraction of initial Fe)	(fraction per year)	Fe leached per year (g Fe/m 2 yr.	Leachate Conc. (Fe mg/L)	Groundwater Conc. (Fe mg/L)
MSW Landfill Leachate						
0	19.10	0.0000				
1	19.10	0.0054	0.00541	240.59	240.6	2.41
9	19.15	0.0153				
10	19.15	0.0161	0.00083	36.86	36.9	0.37
49	19.32	0.0294				
50	19.32	0.0297	0.00030	13.27	13.3	0.13
99	19.31	0.0423				
100	19.31	0.0425	0.00021	9.47	9.5	0.09
149	19.06	0.0691				
150	19.06	0.0694	0.00023	10.30	10.3	0.10
199	18.80	0.1078				
200	18.80	0.1080	0.00027	12.03	12.0	0.12
Rainwater Leachate (WTP Monofill Leachate)						
0	19.10	0.0000				
1	19.10	0.0054	0.00541	240.59	240.6	2.41
9	19.10	0.0162				
10	19.10	0.0171	0.00088	39.04	39.0	0.39
49	19.10	0.0379				
50	19.10	0.0382	0.00038	17.10	17.1	0.17
99	19.10	0.0538				
100	19.10	0.0541	0.00027	12.06	12.1	0.12
149	19.10	0.0660				
150	19.10	0.0662	0.00022	9.84	9.8	0.10
199	19.10	0.0763				
200	19.10	0.0765	0.00019	8.52	8.5	0.09

Assumptions:

S/V (cm^{-1}) =	3.029 $x10^3$
Initial Fe concentration (g/kg) =	14.83 (14830 ppm)
LX decrease over time	
Percolating water (m^3/m^2-yr.) =	1
WTP residual material (kg/m^2 surface (2 m thick) =	3000
Leachate acidity (meq/L) =	100
Groundwater concentration (meq/L) = 0.01 x leachate concentration	1
Groundwater dilution factor =	100

Table B6 Projected calcium concentrations in leachate and Groundwater as a function of leaching time for WQD sample which was disposed in MSW landfill or WTP monofill

		Calcium Leached				
Leaching Time (years)	LX	(cumulative fraction of initial Ca)	(fraction per year)	Ca leached per year (g Ca/m 2 yr.)	Leachate Conc. (Ca mg/L)	Groundwater Conc. (Ca mg/L)
		MSW Landfill Leachate				
0	15.50	0.0000				
1	15.50	0.3412	0.34120	10543.14	10543.1	105.43
9	15.60	0.9123				
10	15.60	0.9616	0.04935	1524.86	1524.9	15.25
14	15.62	1.1119				
15	15.62	1.1510	0.03903	1205.93	1205.9	12.06
19	15.63	1.2805				
20	15.63	1.3138	0.03327	1027.92	1027.9	10.28
34	15.75	1.4919				
35	15.75	1.5137	0.02178	673.04	673.0	6.73
39	15.76	1.5796				
40	15.76	1.5997	0.02012	621.80	621.8	6.22
		Rainwater Leachate (WTP Monofill Leachate)				
0	15.50	0.0000				
1	15.50	0.3412	0.34120	10543.14	10543.1	105.43
9	15.50	1.0236				
10	15.50	1.0790	0.05537	1710.92	1710.9	17.11
19	15.50	1.4873				
20	15.50	1.5259	0.03864	1193.87	1193.9	11.94
29	15.50	1.8374				
30	15.50	1.8688	0.03141	970.61	970.6	9.71
39	15.50	2.1308				
40	15.50	2.1580	0.02715	838.78	838.8	8.39
49	15.50	2.3884				
50	15.50	2.4127	0.02425	749.28	749.3	7.49

Assumptions:

S/V (cm^{-1}) = 3.029 x10^3
Initial Ca concentration (g/kg) = 10.3 (10300 ppm)
LX decrease over time
Percolating water (m^3/m^2-yr.) = 1
WTP residual material (kg/m^2 surface (2 m thick) = 3000
Leachate acidity (meq/L) = 100
Groundwater concentration (meq/L) = 0.01 x leachate concentration 1
Groundwater dilution factor = 100

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