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The Water Pollution Potential From Demolition-Waste Disposal

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Division of Water Pollution Control
Massachusetts Water Resources Commission
Contract Number 76-10 (2)

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DEPARTMENT OF CIVIL ENGINEERING
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PREFACE

This report is a reproduction of Mr. David W. Ferguson's Masters Thesis, which was directed by Dr. James W. Male. The research that is reported herein concerns the pollution potential of leachate from landfills containing predominantly demolition waste. The report concentrates on three areas (1) field measurement and analysis, (2) laboratory lysimeter studies, and (3) possible leachate control strategies.

The authors would like to acknowledge the contributions made by a number of people and thank them for their assistance. Professors Donald Dean Adrian and O. Thomas Zajicek offered valuable suggestions throughout the course of the study. The cooperation of Alan Weinberg and Peter Mokrezecky, Department of Environmental Quality Engineering, Western Massachusetts District, and Thomas Pregman, Department of Environmental Protection, Connecticut, is gratefully acknowledged. In addition, Dotty Pascoe, Andrew Barr and Robert Drake assisted in the preparation of the final report.

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ABSTRACT

This study investigates the water pollution potential of the landfill disposal of demolition waste. Demolition waste is defined as any waste resulting from the demolition of buildings and other structures and any material left over from the construction, repair, or remodeling of structures. Three aspects of the problem are presented: (1) field studies, (2) laboratory lysimeter studies, and (3) possible control strategies.

Four landfill sites were selected in Southern New England which had received solely (or largely) demolition waste and which had identifiable water courses in contact with the waste. Upstream and downstream samples were analyzed to show that, in general, the pH decreased and alkalinity, hardness, and some metals increased in concentration.

In the laboratory studies, four lysimeters were used to study the effect of different types of waste (predominantly masonry and predominantly wood) under different hydraulic loadings (saturated and percolating). Analyses of samples withdrawn from the lysimeters showed that concentrations of constituents in the saturated waste were much higher. In addition, noticeable differences were obvious between the saturated masonry based and saturated wood based wastes, especially for pH values. The pH also affected the concentration of metals in the samples.

The final phase of the study investigated a leachate control strategy utilizing the presence of masonry in the waste. The lime in the masonry caused an increase in the pH resulting in a decrease in the concentration of metals in the leachate. Studies were done on different sizes of limestone and concrete as potential landfill base material, under both aerobic and anaerobic conditions. The concrete aggregate provided better removal of manganese and showed potential as a means for controlling manganese in leachate.

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CHAPTER I

INTRODUCTION

Overview

The disposal of solid waste with a minimum of damage to the environment has become an important issue. A number of occurrences have been reported linking surface and groundwater contamination to the leachate generated by sanitary landfills. Although municipal landfill leachates have been studied for a number of years (7,11,28), very little attention has been given to the leachate resulting from demolition waste (18). Demolition waste is one of the many categories of solid waste and constitutes a significant portion of the total solid waste load. It is considered by many to be clean, solid-fill, with no serious water pollution problems resulting from demolition waste landfills being reported.

Because of the characteristics of demolition waste and the practice of using it as fill material, the question may be raised regarding the suitability of the landfill requirements proposed by the Resource Conservation and Recovery Act (RCRA) of 1976 (27). Due to the waste's relatively inert composition, direct comparisons between demolition waste and municipal waste should not be made. For these reasons this research attempted to establish the potential effects of leachate resulting from the landfill disposal of demolition waste.

Research Objectives

The objective of this study was to investigate the water pollution potential of the landfill disposal of demolition waste and to review the characteristics of landfill components, operation, and design that affect the release of leachate. Of particular concern was the potential contamination of ground and surface waters by demolition waste leachate.

The study has been conducted in three phases. First, selected field sites were characterized and sampled for potential degradation of water quality. In the second phase, laboratory lysimeters were constructed and maintained for 40 weeks to simulate demolition waste landfills. The third phase, drawing from the results of the first two phases, studied a possible control strategy in an attempt to minimize the strength of leachate released from demolition waste landfills.

This research attempted to answer the following questions:

1. What water quality parameters are typical of demolition waste disposal and what are the maximum concentrations to be expected?
2. Does demolition waste present a leachate of any significance, and should this waste continue to be considered clean, solid-fill, relatively inert, with little potential for deleterious environmental effects?

3. Is the degradation of water quality evident from the monitoring of field sites, and is there supporting evidence from the assessment of the laboratory study?
4. If demolition-waste disposal poses a threat to water quality, what control strategies might be used to minimize this potential for contamination?
5. What recommendations should be made concerning site selection, design, and operation of demolition waste disposal sites?

CHAPTER II

LITERATURE REVIEW

Composition of Demolition Waste

One of the most useful classifications of solid waste is based on the kinds of materials: garbage, rubbish, ashes, street refuse, dead animals, abandoned automobiles, industrial waste, demolition waste, construction waste, sewage solids, and hazardous and special wastes (1). Demolition waste is the debris from the destruction of buildings or other structures. Construction waste is the waste material from the construction, repair, or remodeling of buildings or other structures. The primary components of both groups of waste are similar and are usually handled in the same manner. Most available data are reported for a combination of both groups. For this reason, the category of demolition waste will be used throughout this study to include both classifications.

The typical components of demolition waste include concrete and masonry rubble, lumber, plaster, roofing, insulation, wiring, piping, and related products. The waste can be divided into combustible and non-combustible groups, with the combustible portion averaging a much smaller percentage of the total. The combustible fraction, mostly wood, has an average density of 350 to 450 pounds per cubic yard (lb/cu yd), and the non-combustible portion can have a density of over 1800 lb/cu yd (18). The average overall density is approximately 1000 lb/cu yd (600 kg/cu m).

Quantities of Demolition Waste in the United States

Wilson (38) reports on attempts to correlate the generation rates of demolition waste as related to several base variables. These variables include: total population of the community, number of construction and demolition employees, number of construction and demolition permits, and quantities of construction and demolition debris estimated from structure characteristics. The difficulty in estimating generation rates is that the quantities of demolition waste are highly variable. Fluctuations in the load for disposal have no reliable seasonal pattern and depend largely on urban-renewal activity.

Although the amount of waste varies greatly for different buildings and structures, estimates based on structure characteristics have been made by the U. S. Department of Health, Education, and Welfare (34). Table 1 lists the waste production in cubic yards and tons for a number of different structures.

The most comprehensive information on component quantities of solid waste is that gathered by the 1968 National Survey of Community Solid Waste Practices (36). It was based on 6,259 inquiries and covered a population of 92.5 million. The pounds per capita per day (pcd) for

TABLE 1

Wastes Produced by Demolition Operations (34)

Type of Structure	Waste Production Cu.Yd.	 Tons*
1-family frame (25 x 100 ft)	160	56
1-family brick (brick salvaged)	160	56
2-family frame	200	70
2-family brick (brick salvaged)	200	70
6-family tenement frame	800	280
6-family tenement brick (brick salvaged)	800	280
100 x 200 ft commercial or factory structure	4200	1470
1 to 3 story hotel, apartment, commerical complex (100 x 100 ft)	4000	1400

*Density assumed to be 700 lb/cu yd.

selected regions and for urban areas has been summarized in Table 2 and Table 3. From the 1968 survey, it is estimated that demolition waste amounts to 0.72 pcd in urban areas and 0.66 pcd as a national average. The majority of this load is disposed of as landfill material and amounts to approximately 0.16 acre-feet (200 cubic meters) of fill volume per 1,000 population annually (1). This constitutes a significant volume of landfill requirement.

Disposal Alternatives

In the demolition business, salvage was once an important factor. Salvage depends on the type of construction, the local market for the materials, and the time limitations on the demolition job. Even under ideal conditions, more than half of the materials in a demolished building have no value. For these reasons, and the increase in labor costs, salvage has less of an influence, and most everything is disposed of. Collection of these wastes and transport to disposal is usually the responsibility of the contractor. In some cities, the contractors are allowed to dump refuse at the municipal facilities (1).

The proportion of the mixture of wood, rubble, and other components in building debris depends upon the construction of the building. If the building is wood frame construction, the debris is largely wood. This fraction could easily be burned, and it is a nuisance material in a sanitary landfill.

TABLE 2

Analysis of Selected Classes of Solid Waste Collected in Selected Regions (36)

	Pounds per Capita per Day		
	Combined Household & Commercial Refuse	Demolition Refuse	Tree & Landscape
National Average	4.05	0.66	0.18
New England	4.60	0.84	0.21
Southeast Region	3.48	0.16	0.81
Southwest Region	3.20	0.69	0.40
Great Lakes Region	3.73	1.16	0.13
Pacific Coast Region	9.28	0.12	0.34

TABLE 3

Solid Wastes Collected in the United States (36)

	Pounds per Capita per Day	
	Total	Urban
Combined household & commercial refuse	4.05	4.29
Industrial refuse	1.86	1.90
Demolition & construction refuse	0.66	0.72
Sewage treatment plant solids	0.47	0.50
Street & alley cleanings	0.25	0.25
Institutional refuse	0.24	0.16
Tree & landscape refuse	0.18	0.18
Park & beach refuse	0.16	0.15
Catch basin refuse	0.04	0.04

In the past, it was common practice to burn this material on the site. Now, with Federal air pollution standards, this practice of open burning is widely prohibited. The use of conical-shaped burners, "tepees", was tried for this purpose. At best, this method is controlled open burning with inadequate control of combustion and poor pollution control. It is not considered a satisfactory method, although several kinds of supplementary aids are available to improve performance (1).

Several large cities, including New York and Philadelphia, have explored the feasibility of using shredding devices to convert combustible demolition waste into a form suitable for incineration. In Norwalk and Stanford, Connecticut, bulky combustible refuse and demolition lumber are incinerated without shredding. Problems with separation of rubble, damage to chipping equipment, and overall economics discourage this practice. The net result is the predominant current practice of disposing of demolition waste as landfill material.

Demolition Waste Leachate

The Resource Conservation and Recovery Act (RCRA) of 1976 (27) requires that landfills be designed, constructed, and operated so that discharges of leachate are minimized or do not occur. Leachate is defined as the liquid which has percolated through solid waste and has extracted dissolved or suspended material from it (8). Disposal sites can be sources of ground and surface water contamination because of the generation of leachate. Precipitation falling on a site either becomes runoff, returns to the atmosphere, or infiltrates the landfill. Generally, the more water that flows through the waste material, the more pollutants will be leached out; thus, contamination problems are more likely to occur in humid areas. Municipal landfill leachate is generally a highly mineralized fluid containing such constituents as iron, manganese, lead, copper, zinc, sodium, nitrate, chloride, and a variety of organic chemicals.

From available information on solid waste, only the operational problems of demolition waste have been mentioned (1). These include poor compaction characteristics, problems of differential settling, and damage to landfill equipment. Lee, et al. (18), concluded that the elemental or pollutant constituents of the waste are unknown, that the general composition of construction and demolition waste indicates that pollutants do not seem to be a significant problem, and that the wastes may be used as landfill material without subsequent deleterious environmental effects. Sweet and Fetrow (32), on the other hand, demonstrated that strictly wood wastes can result in leachate with high levels of lignin-tannin and volatile organic acids causing reducing conditions that dissociated iron and manganese from the soil in high concentrations. They cited contamination of at least eleven domestic water supply wells, rendering the water non-potable. This finding is important because demolition waste often contains large quantities of wood.

Another major component of demolition waste is concrete and masonry rubble. In an EPA report on the impacts of construction activities in wetlands, Darnell et al. (6) report that concrete surfaces leach out chemical substances, mostly carbonates and hydroxides of calcium and magnesium. These leachings are carried into the water courses, appear in neighboring wetlands, and affect the chemical composition of the water. The authors state that the greatest leaching occurs during and immediately after construction, but long-term leaching undoubtedly takes place.

Lea and Desch (17) state that pure water decomposes set cement compounds, dissolving the lime from them, and to some extent the alumina; continued leaching eventually leaves only a residue of incoherent hydrated silica, iron oxide, and alumina. They explain that this action on a mortar or concrete is so slow as to be negligible unless water is able to pass continuously through the mass, and that the degree and rate of attack increases greatly with increased acidity or the presence of sulphates.

These studies imply that concrete, and therefore masonry rubble, can be long-term sources of the leaching of their chemical constituents. These chemical constituents are fairly well defined because roughly 95 percent of cement production in the United States is Portland cement (29). Portland cement is produced by the high temperature burning of calcareous material, argillaceous material, and siliceous materials. The percentages of oxides present are lime(CaO) 60-66, silica (SiO_2) 19-25, alumina (Al_2O_3) 3-8, and iron oxide (Fe_2O_3) 1-5, with magnesium oxide (MgO) limited to 4 percent (20). The different types of Portland cement are produced, in part, by changing the relative proportions of its four predominant chemical compounds.

To date, no studies or references are available reporting the contamination of ground or surface water from demolition waste disposal. The remaining literature reported here is of importance to this study, but refers largely to municipal landfills.

Factors Affecting Leachate Production

One report (8) written for the U. S. EPA states that whenever water comes into direct contact with solid waste, it will become contaminated. Cameron (4) states that because water infiltrating the site is essential for leachate production, not all sites will generate leachate. Remson et al. (26), Apgar and Langmuir (2), and Caffrey and Ham (3) state that water which infiltrates into a landfill must bring the refuse to field capacity before large quantities of leachate will be formed. At field capacity, the refuse, by definition, can hold or absorb no more water. In general, the primary source of water, which in turn is the primary source of leachate, is precipitation.

Caffrey and Ham (3) list the factors that affect leachate production in a soil covered landfill to be type of soil and its slope, surface drainage, vegetation, air temperature and humidity, initial moisture contents of refuse and soil cover, and the location and flow of groundwater. The factors that affect the leachate production of uncovered refuse are precipitation, particle size and homogeneity, compaction, air temperature and humidity, surface drainage, initial refuse moisture, and the location and flow of groundwater. The main difference is that uncovered refuse allows almost no runoff, and, hence, all precipitation must either evaporate or infiltrate to form leachate. Caffrey and Ham conclude that it is important to consider the water pollution potential of all solid waste disposal sites, but that most of the important factors that control leachate production are at least partially controllable by man. Therefore, most leachate problems can be reduced or eliminated by proper selection, design, and operation of disposal sites.

Leachate Strength

Caffrey and Ham (3) relate leachate strength to decomposition. They explain that contaminants in the refuse are released to percolating water through physical and chemical leaching and biological decomposition. Physical and chemical leaching is basically the rinsing or dissolving of matter by the flow of water through the refuse and biological decomposition is the degradation of refuse into leachate matter or gas. The authors state that in general, the amount of matter being released to the water increases as the rate of decomposition increases. This decomposition can be aerobic (commonly referred to as composting) or anaerobic. Aerobic decomposition is a relatively rapid process, generates high temperatures, and creates no odor problems. Anaerobic decomposition occurs in the absence of oxygen, is a much slower process, generates lower temperatures, and may create odor problems. The authors report that the tolerable pH range for anaerobic decomposition is 5.0 to 9.0 with 6.7 to 7.2 being the optimum.

Anaerobic decomposition actually occurs in two stages. The first is a non-methane producing stage identified by acid production and low pH values. The second stage is the methane producing stage and accounts for most of the decomposition. Hydrogen sulfide, noted for its obnoxious odor, may be produced. Since anaerobic decomposition is a relatively slow process, Caffrey and Ham (3) explain that several years are required in landfill conditions to reach relative stability.

In addition to the leachate processes generally acknowledged, Cameron (4) reported that a considerable portion of leachate strength may be attributable to the textiles, rubber, leather, wood, paper, and cardboard present in the refuse. These components, to a limited extent, can be present in demolition waste. In a preliminary study, Cameron illustrated quite high concentrations of a variety of metals resulting from complete acid digestion of a variety of paper samples. Although total acid digestion is not representative of the concentrations that would be present in leachate where water is the solvent, the point remains that refuse constituents such as paper, textiles, and leather are potentially significant contributors to the contaminant load. Also of importance, is that the solvent action of water changes quite significantly during movement through the fill and with age of the fill. Cameron reported the results of test landfill cells and indicated that the concentration of different materials peak at different times, have different rates of decrease, and have different ratios of peak height to initial and final concentrations. The conclusions reached concerning peak concentrations are that increased ambient temperature results in higher and earlier peaks, increased refuse depth results in lower and later peaks, increased precipitation results in lower and earlier peaks, and higher initial moisture content has little effect.

Lowenbach (19) states that pH is one of the most important variables controlling leachate composition. Because dissolution occurs in the aqueous phase, pH may be considered as a master variable of the system; that is, any reaction which involves either H^+ or OH^- will be affected by the pH of the medium. Löwenbach goes further and explains how the distribution of metals between the solid and aqueous phases by the formation of precipitates is a function of pH.

Fuller, et al. (11) evaluated the chemical characteristics of municipal solid waste leachate and determined that the solubility of the components of the leachate changes markedly as a result of pH fluctuations, aeration, dilution by water, drying and wetting, freezing and thawing, and stages of biodegradation. They reported that exposure of natural leachate to air increased the pH and reduced the solubility of a number of leachate constituents, with iron and manganese being two of the most prominent.

An extensive analysis of both organics and inorganics present in leachate samples collected from landfills located in different parts of the United States was reported by Chain and DeWalle (5). The composition of leachate samples from different sanitary landfills showed a large variation. The authors presented the ranges of leachate composition and showed that the age of the landfill, and thus the degree of solid waste stabilization, has a significant effect on the composition of the leachate. They also mentioned other factors that contribute to the variation of data, such as: the composition and size of the waste, degree of compaction, the moisture content and degree of rainwater infiltration, temperature, sampling, and analytical methods.

The EPA procedures manual for groundwater monitoring at solid waste disposal facilities (35), compiles in one volume an array of valuable information. In part, the manual includes the purposes and objectives of monitoring, types and limitations of monitoring, indicators (toxic and those not usually considered toxic), the assessment of existing land disposal sites, determination of existing water quality, and determination of the pollution potential of a landfill.

Leachate Movement

Remson, et al. (26), state that knowledge of the occurrence and movement of moisture in a sanitary landfill is basic to the knowledge of the generation and movements of water-borne contaminants. They presented moisture-routing methods based upon the equation of continuity. This method requires knowledge of the hydraulic characteristics of unsaturated permeable materials. By first determining the available water storage capacity of soil and landfill layers, Remson, et al. applied the technique to a hypothetical landfill. They concluded that installation season and initial moisture content can affect the hydrologic development of a given landfill in a given climate, and that a number of other techniques are available to control the moisture regimen of a sanitary landfill.

A generalized method for the theoretical determination of the concnetrations of some easily extractable materials leached from sanitary landfills was presented by Qasim and Burchinal (25). They applied the theory of column operation to the leaching of chloride during the vertically downward movement of water, and established families of curves from experimental results. These curves can be directly applied to estimate the concentrations of various components in the leachate from sanitary landfills. Physical factors such as depth of the refuse and cumulative volume of percolation into the fill need to be predetermined.

Tirsch and Jennings (33) state that leachate problems may not be confined to the area immediately under and adjacent to the landfill, and that given certain rates of leachate production and hydrogeological conditions, leachate may enter the groundwater and travel significant distances. They cited a number of studies which have traced leachate plumes.

The potential effects of leachate on surface waters and groundwater is presented by Cameron (4). The study states that the nature of the receiving water and its dilution capacity will have significant effects on the potential for damage caused by leachate discharge. Therefore, each case should be assessed on a site specific basis. Cameron illustrated the path that a leachate might follow under various conditions.

An EPA manual (35) emphasizes that leachate is not diluted with the entire body of groundwater but tends to remain as an intact body with only slight dispersion and diffusion along the edges, that leachate constituents actually move faster than the average groundwater velocity because of hydrodynamic dispersion, and that the path of a leachate plume will follow the direction of groundwater flow.

Leachate Attenuation

Of major importance to the release of leachate and the movement of pollutants into groundwater is the ability of soil and earth materials in retarding or preventing such movement. Soil attenuation can be defined as the decrease of the maximum concentration of a pulse of solute as it moves for some fixed time or distance (10). Soil attenuation as a mechanism is poorly understood and can vary greatly for different leachate constituents and from soil to soil.

Tirsch and Jennings (33) reported on laboratory studies of the leachate attenuation capacity of soil; the concern being that if soil attenuation is not effective, leachates released from solid waste disposal sites pose a serious threat to groundwater quality. They performed batch equilibrations over a wide range of leachate mass to soil mass ratios as well as column studies simulating saturated and unsaturated flow through soil. They concluded that the sand they studied was ineffective in reducing the total ionic strength of leachate and that active sorptive removals were found to be reversible. Fuller (10), on the other hand, concluded that the amounts of elements retained by soils against subsequent extraction with water and 0.1 N HCl suggest substantial permanent retention capacity for soils, although total organic carbon (TOC) and chemical oxygen demand (COD) were not significantly retained by any soils.

Frost and Griffin (9) studied the removal (exchange-adsorption plus precipitation) of a number of metals from a municipal landfill leachate by kaolinite and montmorillonite clays. They used removal versus pH data to construct adsorption isotherms at several pH values and found that sorption increased with increasing pH values and with increasing concentration of heavy metals. Precipitation contributed significantly to the removal of heavy metals from the leachate at pH values above 6. They concluded that the migration of heavy-metal ions through clay materials is predicted to be much greater in landfill leachate than in comparable aqueous solutions of lower ionic strength.

Control Strategies for Leachate

Fuller (10) reported on a laboratory study using eleven soils from seven major orders throughout the United States and municipal landfill leachate alone and spiked with such levels of metals as might be found in the most highly polluted leachates from combined municipal and industrial wastes. The study demonstrated that the attenuation of leachate by the soils is, in part, a function of pH. This was done by overlaying one centimeter of ground limestone in the soil columns studied. Fuller explains that although a pH effect can be shown in laboratory studies, the utility of this information, by itself, in the field is limited. In locations where the pH of soil at a proposed site is unfavorable the only option will be adjustment of pH at the refuse-soil interface or in a relatively shallow depth of soil below the landfill. Because of cost and the need for effectiveness over a period of time without reapplication, limestone or some lime-based material will likely be used if pH-adjustment is attempted. Although the increase in pH will not increase attenuation for all elements, Fuller maintains that the effect was significant enough in the laboratory study to warrant some further study of the use of limestone as a means of minimizing pollutant release from landfills.

Frost and Griffin (9) were in agreement with this possible control strategy. They stated that the pH of a landfill leachate will normally be the factor determining what concentration of heavy-metal cation can exist. They suggested that the pH is probably low in situations where high concentrations of heavy metal ions exist in a leachate and that the removal of these ions may be achieved by raising the pH sufficiently high to precipitate them by the use of crushed limestone.

It has been proposed by Geswein (14) that it may be possible to build a liner composed of several layers of different soils that will act as a treatment facility as the leachate percolates through the soil. Geswein stated that the construction of such a liner is only a concept and that much additional work in this area is needed before a full scale system can be built.

The possibility of using concrete rubble or crushed limestone as a lime-based material for controlling the strength of leachate release from a demolition waste landfill is the basis for the third phase of this research.

Laboratory Studies Simulating Municipal Landfills

Qasim and Burchinal (24) reported on the leaching of pollutants from municipal refuse beds of three different heights. The purpose of the study was to evaluate the characteristics of leachate from different heights of refuse columns containing similar fill materials and operating under similar conditions of percolation and leaching. The simulated landfills consisted of concrete cylinders three feet (0.9 meters) in diameter, mounted over a six inch (15.2 cm), thick concrete base. A layer of clinker was placed on the bottom of the cylinders to insure free runoff, and a 0.75 inch (1.9 cm) diameter steel pipe carried the drainage water to an exterior valve. Fill materials consisted of mixed municipal refuse, and after completion, the cylinders were covered with plywood to exclude natural precipitation.

An initial water volume equivalent to 16.67 inches (42.3 cm) of precipitation was applied to all cylinders. This was the amount of water sufficient to saturate the refuse in the cylinder of greatest height. During the remaining test period, water equivalent to three inches (7.6 cm) of precipitation was applied to all the cylinders at each two-week interval. The water applied was allowed to percolate and collect in the bottom of the cylinders to be drained after the first week and again prior to the application of the next batch at the end of the second week. From the results, Qasim and Burchinal calculated the quantities of extracted material and used these values to assess the quantities of extracted material per acre-foot of landfill.

Fungaroli (12) reported on laboratory sanitary landfill and field sanitary landfill studies. The laboratory sanitary landfill was contained in a lysimeter consisting of a fiberglass-lined steel tank, 13 feet high (4.0 m) and six by six feet (1.8 x 1.8 m) in cross section. The lysimeter size and the size of the refuse components were such as to insure the validity of collected data. The lysimeter functioned as a closed system which permitted the contained landfill to be representative of the center of a large municipal sanitary landfill. Lysimeter leachate and gas samples were analyzed, and temperatures were monitored on a routine basis. The top of the lysimeter was closed and temperatures and water input were adjusted to the average monthly conditions for the locality. The water input was distilled water added as a light "rain" over the surface, on a weekly basis, to represent the excess of precipitation over evapotranspiration. The results of the laboratory study were then compared to the field sanitary landfill facility consisting of a 50 foot by 50 foot (15.2 x 15.2 m) site with similar soil cover and depth of refuse.

The importance of the studies by Qasim and Burchinal (24) and Fungaroli (12) to the research on demolition waste is not the leachate data, but rather, the apparatus and methodology used in the laboratory experiments.

CHAPTER III

FIELD STUDIES

Methodology

The field studies involved the sampling of local demolition waste disposal sites. The fundamental objective of monitoring land disposal sites is to serve as a check on potential leachate contamination (35). It is important to protect against water contamination at existing sites a well as new sites. An assessment of the leachate from existing demolition disposal sites must be made to determine the need for, and type of, control strategies of future sites.

Four demolition waste disposal sites were chosen for study. All the sites are located in Southern New England and, in general, have characteristics resembling much of the region. Their selection was based on the fact that they are largely or solely demolition disposal landfills, and they are also all in contact with water through percolation, stream flow, or direct placement beneath the water table. In two cases some organic waste was present prior to the demolition waste disposal, illustrating the difficulty of finding a "pure" demolition waste disposal site. Because the sites vary greatly in size, configuration, content, operation, type of water percolation, and resulting levels of pollutants, a brief description of the characteristics of each site follows.

Sampling Sites. Site No. 1 consists of a ravine into which approximately 2000 cubic meters of demolition waste has been disposed. The waste now occupies a space about 15 meters wide, 12 meters long and 10 meters deep. Flowing directly from the base of the waste is a stream with a flow rate of 0.3 to 3.0 liters per second. Within these limits, the stream flow is constant and originates from the overflow of a large pond approximately 150 meters upstream. Samples were taken from the stream above the site, from the stream directly below the site, and 100 and 800 meters downstream.

The second site is a privately owned landfill which accepts demolition waste, tree refuse, and large bulky items, such as major appliances. Prior to 1972, the site resembled an open dump and received a large variety of material. Since its re-design as a "sanitary landfill", the only organic waste reported is the disposal of some paper mill sludge. The importance of this contribution is unknown and may be significant. The site was chosen because it is large (5-6 hectares), but receives no household waste or garbage. It is also constructed over a stream which is fed by springs and storm sewers. The stream has an average flow rate of 2.7 liters per second and is piped under the entire landfill, a length of over 300 meters. The pipe stops at the edge of the landfill, and the stream flows approximately 800 meters until it enters a large river. During construction, it was required that the pipe be covered with a minimum of

1.22 meters (four feet) of clean fill to conform to state regulations. Stream samples were taken at the pipe effluent, and 100 and 800 meters downstream. An additional sample was collected from a standing pool of water, in a marsh 100 meters downgrade of the site. Because of the branching pipe network beneath the landfill, no adequate upstream sampling was available.

The third site is a deep, brownstone quarry that has been filled in with demolition waste. Prior to being filled, and to some extent during construction, the site had a stream passing directly through it, and was polluted from the sewage of neighboring homes. These sewage connections were corrected, and the stream was rerouted around the perimeter of the site. For several years since its completion, the site has been open to the public for receipt of demolition waste, leaves, grass, and road cleanings. Samples were taken from the stream, upstream and downstream from the site.

The fourth site is privately owned and appears to receive strictly demolition waste. No direct water contact is evident, but one corner of the site slopes down into a low marsh land. From the disposal area a leachate stream trickles downhill about 15 meters through the marsh and enters a small stream. Samples were taken of the leachate from the stream above and below the site, and from standing water in a bulldozer hole adjacent to the disposal area.

Analyses. Each landfill was sampled for local water quality including pH, conductivity, alkalinity, hardness, chemical oxygen demand (COD), iron, manganese, zinc, copper, cadmium, chromium, and lead. The method of analysis included: pH at 25°C, conductivity in micromhos/cm using a portable, self-contained conductance meter, alkalinity by potentiometric titration to an equivalence point of 4.5, EDTA hardness method 309 B "Standard Methods" (30), COD method 508 "Standard Methods", and total metals acidified and anaylzed using atomic absorption spectrophotometry. These parameters were selected because of the composition of demolition waste.

Where possible, background water quality was sampled. Upstream samples and flow rates were taken in the cases where a stream passed through or adjacent to the site. All sampling was performed on a weekly basis between the months of September and November. All samples collected were grab samples taken in one liter plastic sample bottles. Any parameter not tested immediately upon return to the laboratory was preserved according to "Standard Methods".

Results

The results of the analysis are summarized in Table 4. The downstream versus upstream samples for sites 1 and 3 show a decrease in pH and an increase in alkalinity, hardness, and metals. For site 1 the values of alkalinity and hardness doubled, while the iron, manganese, and zinc levels are four to seven times the upstream concentrations. At

TABLE 4

Average Concentrations of Water Samples

Site Number	1				2*				3		4			
Sample Location	Stream Above Site	Stream Below Site	100 m Downstream	800 m downstream	Marsh Below Site	Stream Below Site	100 m Downstream	800 m Downstream	Stream Above Site	Stream Below Site	Dump Face Leachate	Groundwater Above	Stream Above Site	Stream Below Site
pH	7.00	6.85	7.25	7.35	6.45	7.00	7.25	7.95	7.75	7.00	7.95	6.75	7.20	7.20
Conductivity µMHOS	--	--	--	--	750	800	800	700	300	420	1500	100	130	130
Alkalinity ($CaCO_3$) mg/ℓ	25	51	50	37	220	330	340	290	110	210	530	14	33	33
Hardness ($CaCO_3$) mg/ℓ	29	55	54	42	270	350	360	320	160	250	1100	50	58	59
COD mg/ℓ	<10	<10	<10	<10	--	60	60	60	<10	33	270	10	20	20
Iron (Fe) mg/ℓ	0.5	5.9	2.3	0.5	28.2	13.0	12.2	3.4	0.5	2.1	7.0	0.6	0.6	0.7
Manganese (Mn) mg/ℓ	*0.11*	*0.81*	*0.65*	*0.11*	*2.38*	*2.16*	*2.58*	*2.03*	*0.13*	*0.55*	*0.6*	*1.4*	*0.18*	*0.23*
Zinc (Zn) mg/ℓ	<.01	0.04	0.03	<.01	0.05	0.16	0.12	0.05	0.07	0.06	0.13	0.02	0.02	0.02
Flow Rate ℓ/sec	0.26	0.26	0.26	0.26	--	2.8	2.8	2.8	34	34	--	--	62	62
# of Samples Averaged	3	3	2	2	5	5	5	5	3	3	3	3	3	3
Sampling Period (Weeks)	2	2	2	2	4	4	4	4	4	4	4	4	4	4

*No upstream samples reported.

site 3 the conductivity and COD was also increased, while the increase in iron and manganese was four times the upstream value. Site 2, which had no good basis for upstream comparison, exhibited much higher levels of alkalinity, hardness, conductivity, COD, iron, manganese, and zinc than did sites 1 or 3. The standing pool of water just downstream of site 2, had a concentration of iron as high as 40 mg/ℓ during the second week of sampling. Site 4 had the only leachate that was sampled directly, before dilution in a stream. The sample had an average conductivity of 1500 μmhos/cm, hardness of 1100 mg/ℓ, COD of 270 mg/ℓ, and iron and manganese concentrations of 7.0 and 8.6 mg/ℓ, respectively. The stream flowing near this site presents a very large dilution factor, and no contamination was detected except a minute increase in iron and manganese.

In addition to the parameters listed in Table 4, the levels of lead, copper, cadmium, and chromium were measured for all sites. All concentrations were found to be below normal detectable levels; values of Pb<0.5, Cu<0.10, Cd<0.25, and Cr<0.10 mg/ℓ.

Discussion

The field results continually showed high concentrations of conductivity, alkalinity, hardness, iron, and manganese. As a reference, the National Technical Advisory Committee (23) reports that the maximum contaminant level of raw surface water for public water supplies is 0.3 mg/ℓ for iron and 0.05 mg/ℓ for manganese. In the field study the limit of manganese was exceeded by as much as 180 times, and the iron limit was exceeded by over 130 times.

The implications of these results should be considered carefully in view of both site selection and design of demolition waste landfill sites. All four field sites are examples of poor site selection. The location of site 1 allows stream water to percolate directly through the waste material. Site 2 uses an unacceptable technique of simply piping an existing stream and drainage system underneath the entire landfill. At site 3, a quarry was completely filled in with demolition waste. Site 4 is also a poor location because of its proximity to a local stream.

From these studies, demolition waste has demonstrated the need, similar to municipal landfills, for proper site selection and design to reduce the possible degradation of both ground and surface water quality. Because of the variability between the field site there is a need for controlled laboratory studies on more defined components of demolition waste under different water percolation conditions in order to better assess the pollution potential.

CHAPTER IV

LABORATORY STUDIES

Methodology

A laboratory study was conducted because of the difficulty in exercising satisfactory control over important variables in the field, such as: fill constituents, percolation, precipitation, and the mechanics of leaching. Laboratory lysimeters were used to study two extreme variations in demolition waste composition under two different water percolation conditions; thus, serving as a basis to predict the water pollution potential. The laboratory test was not designed to evaluate any changes in leachate quality due to dilution or infiltration through soils. The major concern of the laboratory study was the determination of the parameters leached from different compositions of demolition waste, an indication of the maximum concentrations of these parameters, and the total amounts released.

Apparatus. Four lysimeters were used in this study. The lysimeters were constructed out of two steel drums joined together top to bottom. The overall height was 1.77 m (70 inches) and the inner diameter was 0.57 m (22.3 inches). Each lysimeter was lined with two, 6-mil thick, polyurethane liners. A 7.6 cm (three inch) deep filter base was placed on the bottom in two layers. The top half of the filter consisted of 2.38 to 6.35 mm gravel and the lower portion 6.35 to 12.7 mm gravel. The gravel base acted as a filter to allow only the finest grain size, colloidal and suspended solids to pass through. Samples were collected from the center of the bottom through a 5.08 cm (two inch) drain connected to a 1.27 cm (0.5 inch) PVC pipe. The apparatus is illustrated in Figure 1.

Two of the four lysimeters were filled with 50 percent masonry waste and only 12 percent wood waste on a volume basis. The other two lysimeters were filled with 50 percent wood waste and only 12 percent masonry waste on a volume basis. The remaining volume for each was a duplicate make-up of the other elements of demolition waste including plaster, roofing, floor tile, ceiling tile, insulation, upholstery, electrical wiring, cast iron, iron, galvanized steel, lead, aluminum, and copper. The actual weight and percentages by weight of each component were recorded in detail (see Table 5).

The waste was evenly distributed in the lysimeters and compacted to a final height of 1.60 m (63 inches), a volume of 0.386 m^3 (0.500 cu yds). The dry weight of the masonry-based waste was 275.2 kg (567.0 lbs) with a density of 670 kg/m^3 (1134 lbs/cu yd). The wood-based waste weighed 127.7 kg (281.5 lbs) and had a density of only 332 kg/m^3 (563 lbs/cu yd).

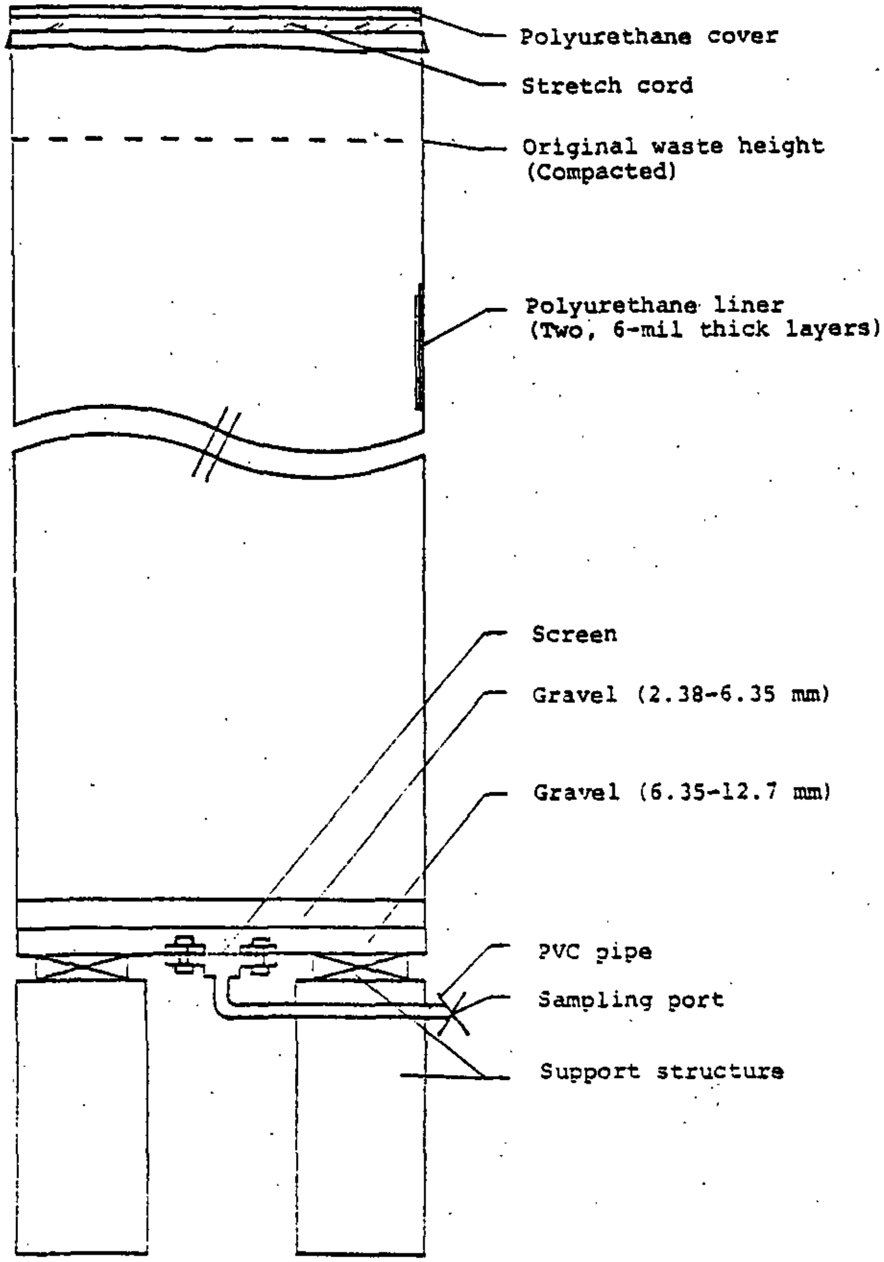

Figure 1. Laboratory Lysimeter

TABLE 5

Composition of Waste from Laboratory Study

Component	Masonry-Based Waste		Wood-Based Waste	
	Weight (kg)	% of Total	Weight (kg)	% of Total
Masonry rubble	186.0	72.5	45.4	35.5
Masonry associated steel	10.8	3.9	1.5	1.2
Wood	6.8	2.7	27.2	21.3
Plaster	15.9	6.2	15.9	12.4
Cast iron	9.1	3.6	9.1	7.1
Asphalt shingles	7.3	2.8	7.3	5.7
Iron and steel	3.9	1.5	3.9	3.1
Floor tile	3.6	1.4	3.6	2.8
Ceiling tile	3.2	1.2	3.2	2.5
Insulation	1.9	0.7	1.9	1.5
Galvanized steel	1.4	0.6	1.4	1.1
Tar paper	1.4	0.5	1.4	1.1
Lead	1.2	0.5	1.2	0.9
Rubber	1.1	0.4	1.1	0.9
Foam rubber	1.0	0.4	1.0	0.8
Electrical wiring	0.9	0.4	0.9	0.7
Rug	0.9	0.4	0.9	0.7
Copper	0.4	0.1	0.4	0.3
Aluminum	0.3	0.1	0.3	0.3
Rug mat	0.1	0.1	0.1	0.1
Total Weight (kg)	257.2		127.7	

One lysimeter of each type of waste was maintained under unsaturated flow conditions with a weekly application of a rainwater leaching solution. A volume equivalent to 2.54 cm (1.00 inch) of distilled deionized water acidified to a pH of 4.5, was used to simulate local rainwater. Samples were extracted from the bottoms of the lysimeters on a one or two week predetermined schedule. The gravel base acted as a collection and storage reservoir for the time between rainwater application and leachate removal.

The remaining two lysimeters, one of each type of waste, were maintained under saturated conditions. A sufficient volume of local groundwater was added to bring each of the two types of waste to saturation. With the same time schedule, samples were extracted from the bottom of the saturated lysimeters. The weekly volume of liquid removed was 6.5 liters, that volume equivalent to the "rainwater" added in the other two lysimeters. After the samples were extracted, groundwater was added to bring the liquid level back to the surface of the waste.

Analysis. The analysis included all of the pollution parameters tested for in the field studies with the addition of total carbon, lignin-tannin, and solids. The total carbon was determined using a combustion-infared carbon analyzer. Lignin-tannin was measured as tannic acid or "tannin-like compounds", using the spectrophotometric method 513, "Standard Methods" (30). The types of solids determined were "total residue", "total volatile", "fixed residue", and "total suspended matter"; method 208, "Standard Methods". The common names used for these types of solids are: total solids, volatile solids, fixed residue, and suspended solids.

Results

For the two unsaturated lysimeters the leachate that accumulated in the gravel base was drained prior to the rainwater application of the following week. The volume of leachate was recorded and analysis performed. Leachate appeared in the wood-based waste after the first week; however, no leachate passed through the masonry-based waste until the third week. The initial volumes were quite small, approximately two percent of the volume added each week. These values increased steadily until on the sixth week both leachates were about half of the volume of rainwater applied. At that time the sampling period was extended to two week intervals. With this schedule the leachate was allowed to accumulate in the bottom of the lysimeters for two weeks. After six more weeks it was realized that this volume of accumulated leachate exceeded the capacity of the gravel base and therefore was saturating the bottom of the waste column. To alleviate this problem the leachate was once again drained every week. On even numbered weeks the entire analysis was performed and on odd numbered weeks only the volume was recorded.

The results of all samples are presented in Appendix B. For selected parameters, pH, conductivity, total solids, volatile solids, alkalinity, hardness, COD, iron, and manganese, the results are illustrated in the subsequent sections.

Water Input and Leachate Volumes. For the two saturated lysimeters 6.5 liters of leachate was extracted each week for the first six weeks and 13.0 liters every two weeks for the remainder of the forty week test period. Following the extraction of samples, sufficient groundwater was added to bring the liquid level back to the surface of the waste. The volume of water input and the volume of leachate sampled are listed for the two saturated lysimeters in Table 6.

The volume of groundwater required to initially saturate each waste was 261 liters for the masonry-based waste and 293 liters for the wood-based waste. In addition to replacing the extracted volume of leachate during the first ten weeks, 20 liters of water was added to the wood-based waste and 5 liters to the masonry-based waste to balance absorption and maintain saturation. Over the last thirty weeks of the test period both wastes absorbed less than 2 liters of additional water.

TABLE 6

Cumulative Volume of Applied Groundwater and Extracted Volume of Leachate from Saturated Lysimeters

	Masonry-Based Waste		Wood-Based Waste	
Time (Weeks)	Cumulative Volume Applied*	Cumulative Volume Leachate	Cumulative Volume Applied**	Cumulative Volume Leachate
2	15.14	13.00	30.29	13.00
4	29.14	26.00	45.29	26.00
6	42.84	39.00	58.99	39.00
8	56.84	52.00	72.49	52.00
10	70.04	65.00	85.49	65.00
12	83.34	78.00	98.49	78.00
14	96.34	91.00	111.49	91.00
16	109.34	104.00	124.49	104.00
18	122.64	117.00	137.49	117.00
20	135.84	130.00	150.49	130.00
22	148.84	143.00	163.49	143.00
24	161.84	156.00	176.49	156.00
26	174.84	169.00	189.49	169.00
28	187.84	182.00	202.49	182.00
30	201.14	195.00	215.49	195.00
32	214.44	208.00	228.79	208.00
34	227.84	221.00	242.09	221.00
36	240.84	234.00	255.09	234.00
38	253.84	247.00	268.09	247.00
40	-	260.00	-	260.00

All values in liters.

*This value excludes the initial addition of 261 liters of groundwater to bring the waste to saturation.

**This value excludes the initial addition of 293 liters of groundwater to bring the waste to saturation.

For the unsaturated wastes there was an initial wetting period which approached equilibrium by week 20 as the waste reached field capacity. At field capacity the waste can hold or absorb no more water and the amount of leachate approximately equaled the volume of water added. The volume of water input and the quantity of leachate sampled are listed in Table 7.

pH. One of the most important variables examined was pH. The pH of a leachate will normally be the factor determining what concentration of heavy metals can exist. Due to the effect of pH on the solubility of metals and other reactions, pH may be considered as a master variable of the system.

The results are illustrated in Figure 2 and show significant variations between the different lysimeters. The saturated masonry-based waste had an extremely high pH with an initial value of 11.40 which increased steadily to a value of 12.15. The pH of the saturated wood-based waste increased from 6.30 to 6.85 within the first ten weeks and maintained that approximate level throughout the test period.

With the onset of substantial volumes of leachate, the unsaturated masonry-based waste had an average pH of 7.50 for weeks four, five and six. The pH then increased to 11.50 by week ten. This increase was due to the two week sampling interval and the effective saturation of the

TABLE 7

Cumulative Volume of Applied Rainwater and Extracted Volume of Leachate from Unsaturated Lysimeters

Time (Weeks)	Masonry-Based Waste		Wood-Based Waste	
	Cumulative Volume Applied*	Cumulative Volume Leachate	Cumulative Volume Applied*	Cumulative Volume Leachate
2	13.00	-	13.00	0.87
4	26.00	1.36	26.00	5.62
6	39.00	6.72	39.00	12.42
8	52.00	13.47	52.00	20.37
10	65.00	21.57	65.00	29.37
12	78.00	30.77	78.00	38.67
14	91.00	40.97	91.00	48.37
16	104.00	49.76	104.00	58.34
18	117.00	60.79	117.00	68.14
20	130.00	72.38	130.00	78.61
22	143.00	84.26	143.00	89.76
24	156.00	95.99	156.00	100.76
26	169.00	108.19	169.00	112.24
28	182.00	120.45	182.00	123.88
30	195.00	133.05	195.00	135.60
32	208.00	145.67	208.00	147.40
34	221.00	158.04	221.00	159.34
36	234.00	170.4	234.00	171.09
38	247.00	182.87	247.00	183.24
40	260.00	195.37	260.00	195.20

All values in liters.

*This value represents the volume of rainwater up to, but not including, the week in question.

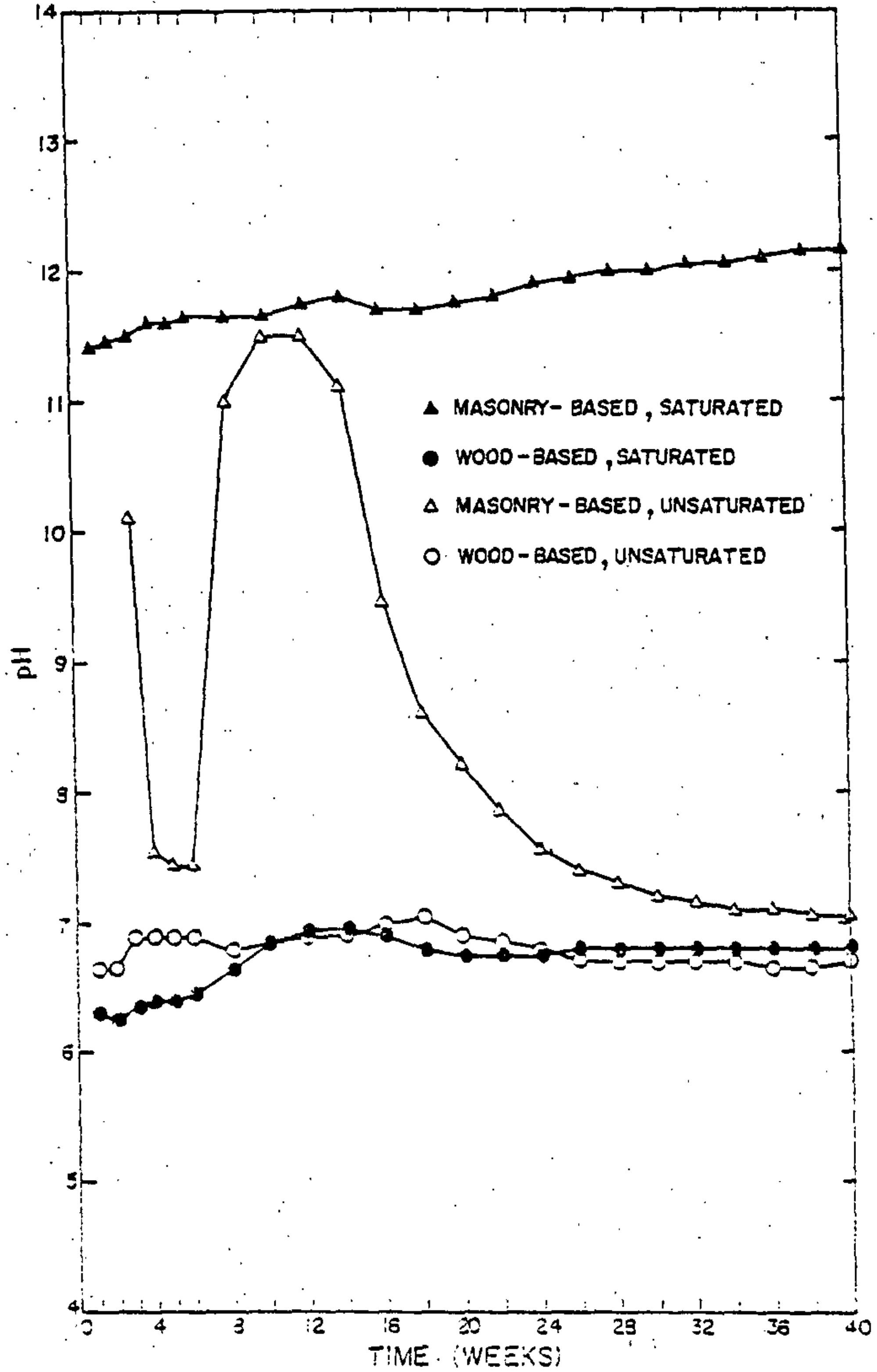

Figure 2. pH Values for Lysimeter Leachates

bottom of the waste. When this problem was alleviated (on week 14) by draining the leachate each week, the pH decreased. The initial decrease was quite dramatic and then tapered down to a pH of 7.05 by the end of the test period.

In the unsaturated wood-based waste the pH of the leachate varied between 6.65 and 7.05 for the first half of the test period and the leveled off at approximately 6.70. Unlike the masonry-based waste, there was no apparent effect on the pH from the difference in sampling intervals.

Conductivity. The conductivity of a solution is a measure of the ability to carry an electrical current and varies with the number and type of ions in solution. Conductivity is at least as good a criterion of the degree of mineralization as the more commonly used "total dissolved solids" (30).

The conductivity results are given in Figure 3. The results show very high initial values followed by slight increases over the first few weeks. The values were recorded to the nearest 100 μmhos/cm and as such, were relatively consistant.

The conductivity of the saturated masonry-based leachate peaked on the fourth week and remained at approximately 4200 μmhos/cm throughout the test period. For the saturated wood-based waste the conductivity peaked on week 12 and leveled off for the last 20 weeks at

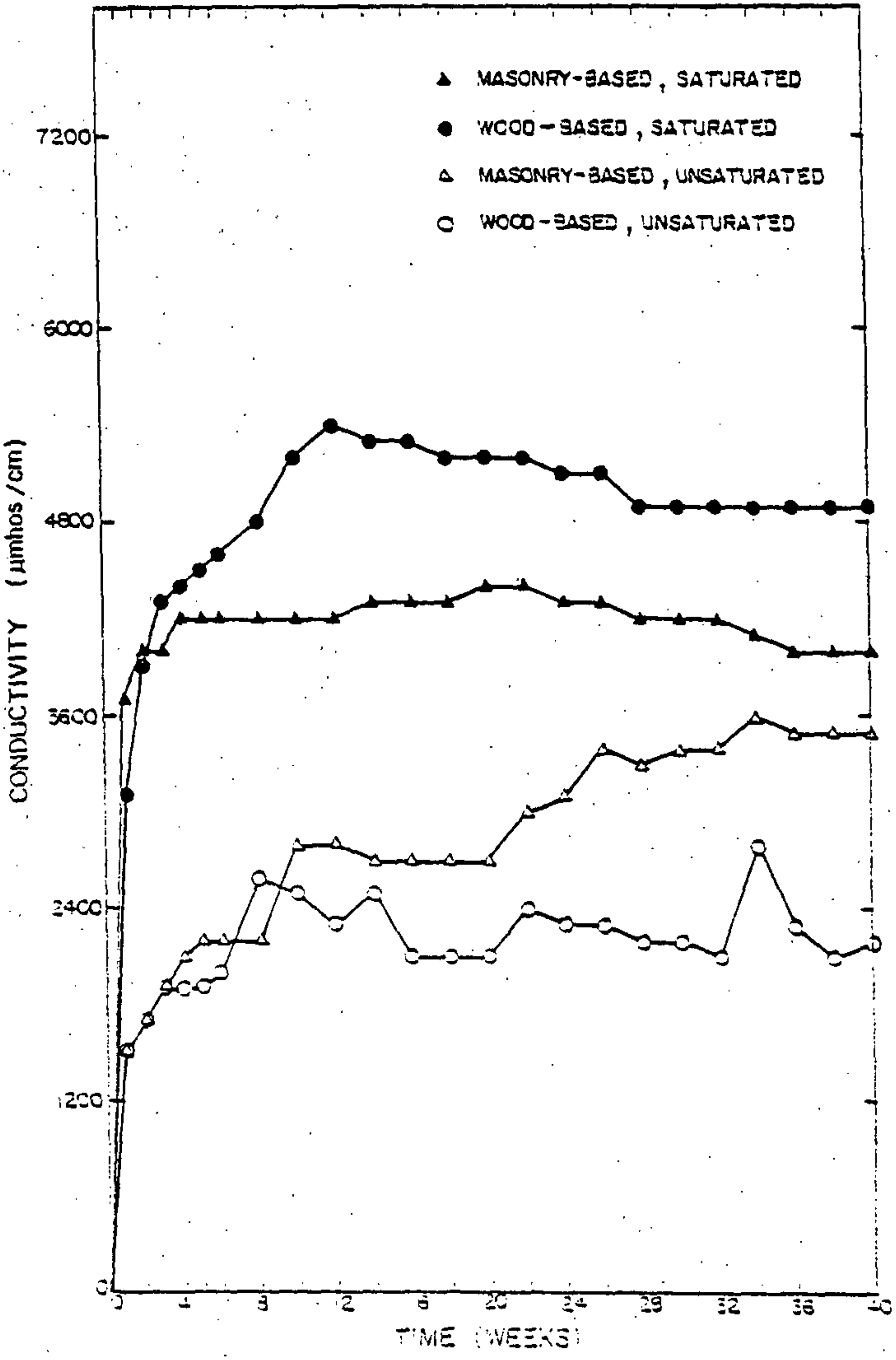

Figure 3. Conductivity Values for Lysimeter Leachates

4900 μmhos/cm.

In the unsaturated waste leachates the values for conductivity were more variable. The conductivity of the unsaturated masonry-based leachate increased through most of the first 26 weeks and approximately leveled off at 3500 μmhos/cm. In the unsaturated wood-based leachate the conductivity increased to 2600 μmhos/cm by week eight and then varied within a few 100 μmhos/cm of 2300 throughout the test period.

Solids. The types of solids determined were total, volatile, fixed residue, and suspended. The total solids measurement is an excellent parameter for evaluating the pollution level of leachate in that it is a summation of all the solids present (13). The volatile solids fraction offers a rough approximation of the amount of organic matter present and the fixed residue is simply the difference between the total and volatile. The suspended solids generally removed in the field by underlying soils, proved to be extremely low in the lysimeter leachates due to filtration through the gravel base.

The values of total solids and volatile solids are illustrated in Figures 4 and 5, The total solids for the saturated waste leachates were more than double the level for the unsaturated waste leachates, and the volatile fraction for the saturated waste leachates was about four times greater. The results indicated very high initial

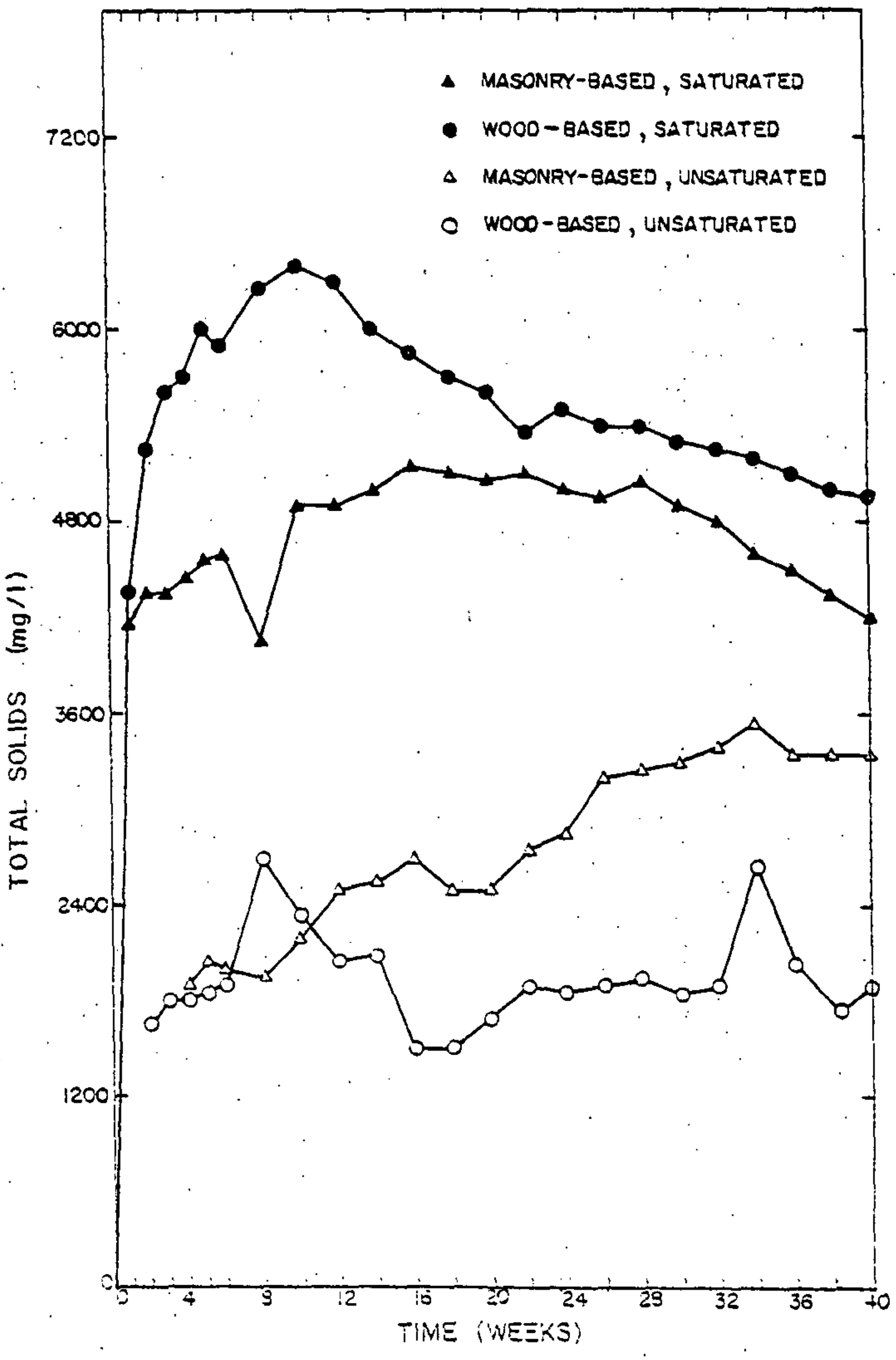

Figure 4. Total Solids for Lysimeter Leachates

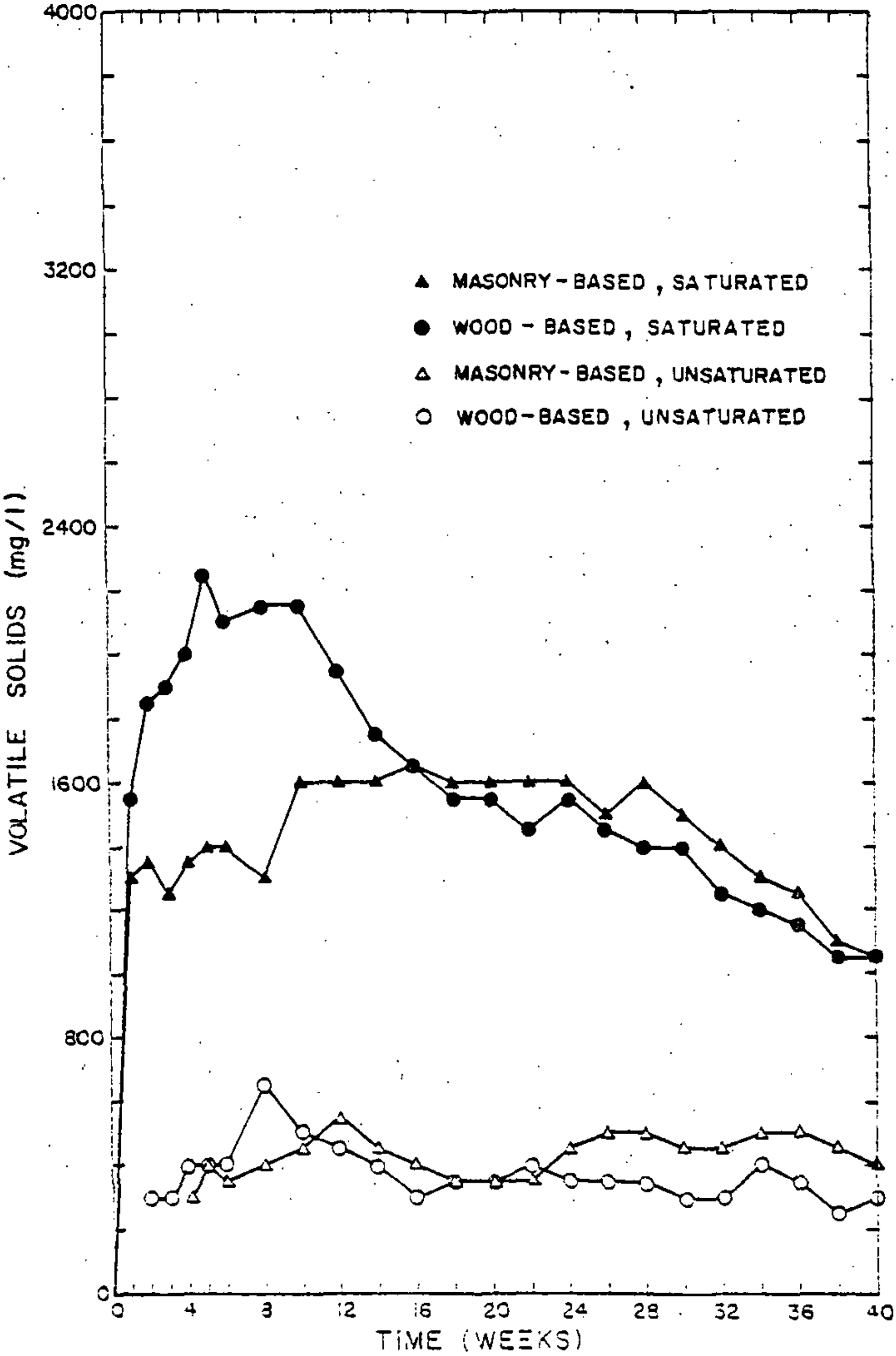

Figure 5. Volatile Solids for Lysimeter Leachates

values with slight trends throughout the test period. The total solids and volatile solids for the saturated waste leachates peaked during the first half of the test period and then declined steadily. In comparison, the total solids for the unsaturated masonry-based waste actually increased throughout the test period.

Color and Odor. The color of each leachate during sampling was recorded and any obnoxious odor was noted. This was done to determine gross differences in the condition of the leachates, and no attempt was made to quantify the color or analyze leachate gases. The color and odor in this study may give an indication of the state of biodegradation and the level of reducing conditions.

It has been recognized for some time that the typical clear straw color of freshly sampled municipal landfill leachate is modified to a murky dark green or black by exposure to air (16). The obnoxious odor of leachate and landfill gas is also reported by a number of researchers (3,16,18). The anaerobic decomposition of refuse produces methane, carbon dioxide, and a number of other gases often including hydrogen sulfide. Hydrogen sulfide is noted for its unpleasant "rotten-egg" odor which is detectable at very low concentrations and actually quite toxic at high concentrations. The detection of this odor is an indication of the anaerobic decomposition of sulfur-containing organic matter (22).

The results of color and odor were quite different between the wood-based and masonry-based wastes. The saturated masonry-based waste generated a bright yellow leachate while the unsaturated masonry-based leachate was faint yellow. Both leachates had relatively consistant color throughout the test period and no strong recognizable odor. The saturated wood-based waste, on the other hand, generated a leachate with an extremely potent odor, very identifiable as hydrogen sulfide. The color of this leachate was dark gray the first few weeks, black through week ten, and yellowish gray or pale yellow for the rest of the test period. The unsaturated wood-based leachate had an odor detectable as hydrogen sulfide but much less potent than that of its saturated counterpart. For the first few weeks the color of this leachate was yellowish-brown and progressed to gray and then black by week 14. Then over the following few weeks the color progressed back to gray and remained as such throughout the test period. The peak at week 14 could once again be attributed to the saturation of the bottom of the waste by the two week sampling interval from week six to week 14.

Lignin-Tannin. Lignin and tannin are both plant constituents and their detection can be used as a measure of the degradation of wood wastes. The determination is a general measure of hydroxylated aromatic compounds and is measured as "tannic acid".

The results show levels approximately an order of magnitude higher for the saturated wastes than the unsaturated wastes. For the wood-based waste the values in mg/ℓ ranged from 90 to 680 for the saturated waste and 11 to 65 for the unsaturated waste. For the masonry-based waste the values in mg/ℓ were 51 to 83 for the saturated waste and 7 to 17 for the unsaturated waste.

Alkalinity. The alkalinity of a water is a measure of the capacity to neutralize acids. When the chemical composition of a sample is unknown, as with leachate, the test only indicates general properties and not specific substances.

The results are illustrated in Figure 6. The trend, in general, was a gradual increase within the first ten weeks, after which the saturated wastes leveled off while the unsaturated wastes varied. The unsaturated masonry-based waste initially dropped after week ten, then increased gradually until leveling off after week 30. The values for the unsaturated wood-based waste peaked during weeks 14 and 16 and then generally declined for the remainder of the test period.

Hardness. The hardness of a water was originally understood to be a measure of the capacity of the water for precipitating soap (30). Hardness is caused chiefly by the calcium and magnesium ions commonly present in water,

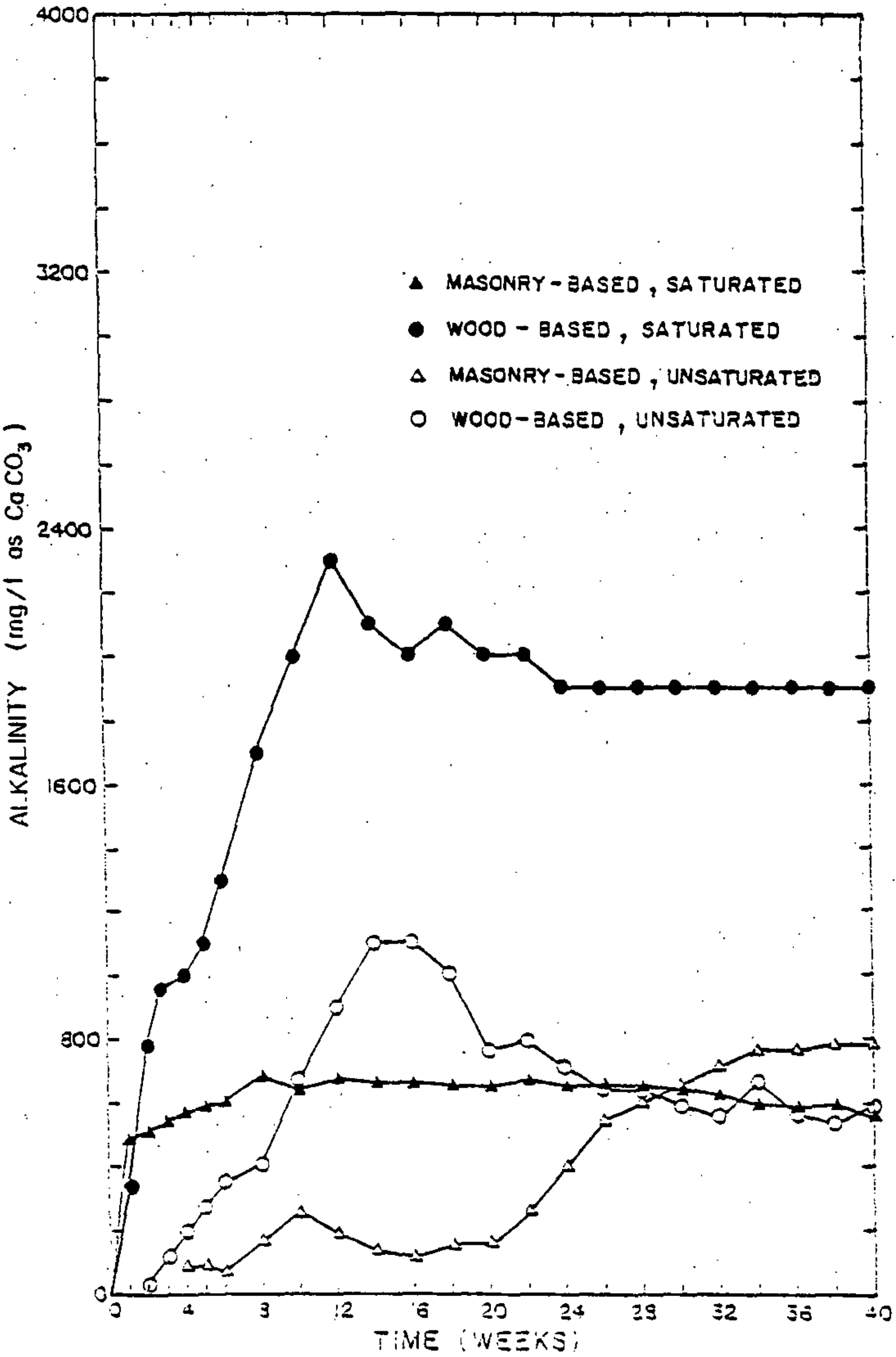

Figure 6. Alkalinity for Lysimeter Leachates

but also by ions including iron, manganese, zinc, and others. The method used for the hardness determination was the ethylenediamine tetraacetic acid (EDTA) titration, which measures the calcium and magnesium ions.

The hardness results are shown in Figure 7. The trends almost identically parallel those for the conductivity values. Both showed very high initial values followed by slight increases over the succeeding few weeks and then steady trends. The magnitude of conductivity in μmhos/cm and hardness in mg/ℓ as $CaCO_3$ were in the ratio of about 2 to 1.

COD. The chemical oxygen demand (COD) test is widely used to measure the pollutional strength of leachates. The test is a measure of the amount of chemically oxidizable substances dissolved or suspended in a sample. For leachate the test indicates the amount of organic matter and reduced inorganic compounds. The method fails to include some degradable organic compounds, such as acetic acid, while including other compounds that are not readily degradable, such as cellulose.

The results, as shown in Figure 8, demonstrate high initial values followed by gradual trends. The COD for the saturated wastes were significantly higher than the unsaturated wastes. The COD for the saturated masonry-based waste peaked during the middle of the test period and then declined, while the peaks for all the other leachates were early in the test period followed by gradual

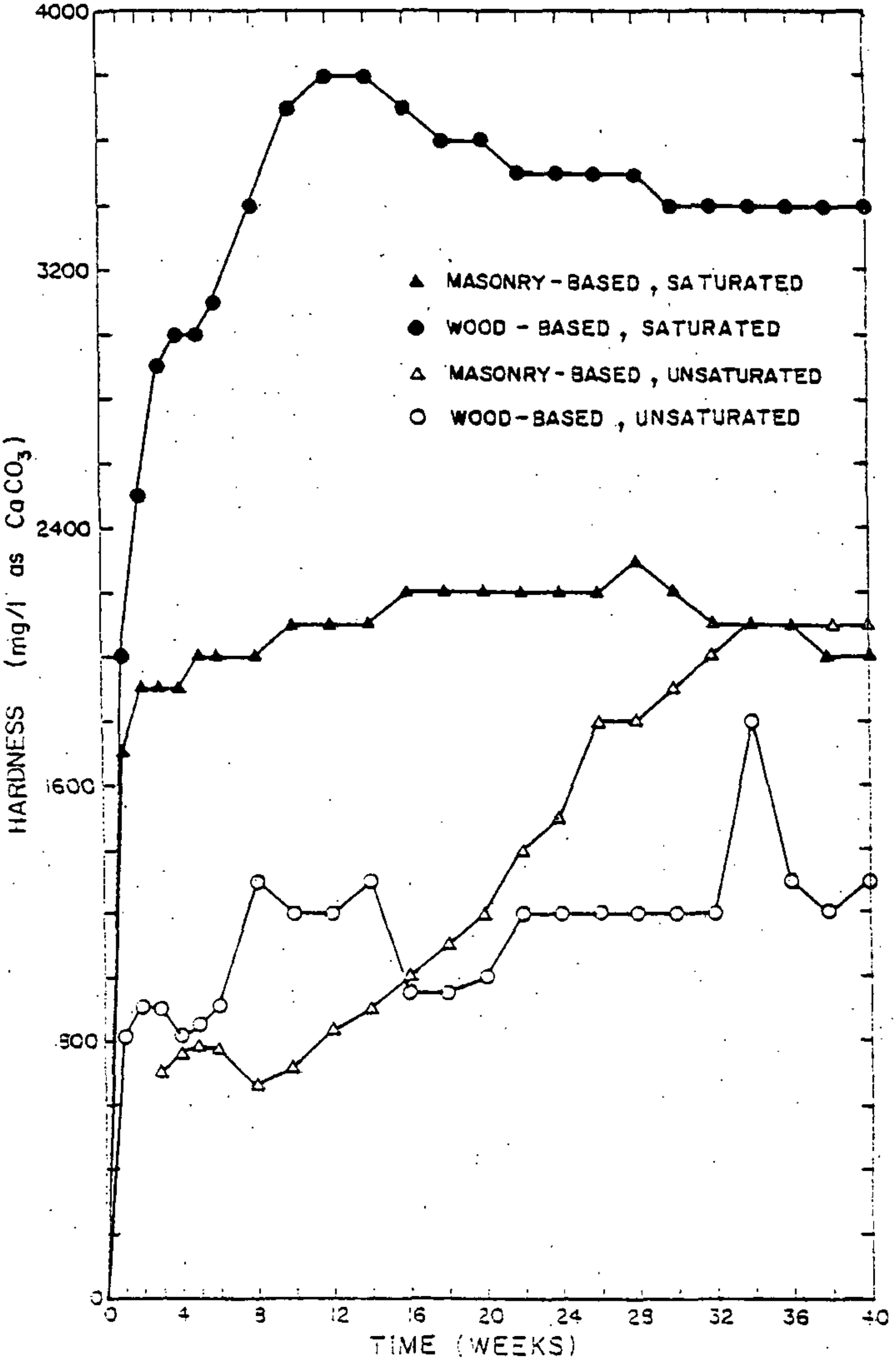

Figure 7. Hardness for Lysimeter Leachates

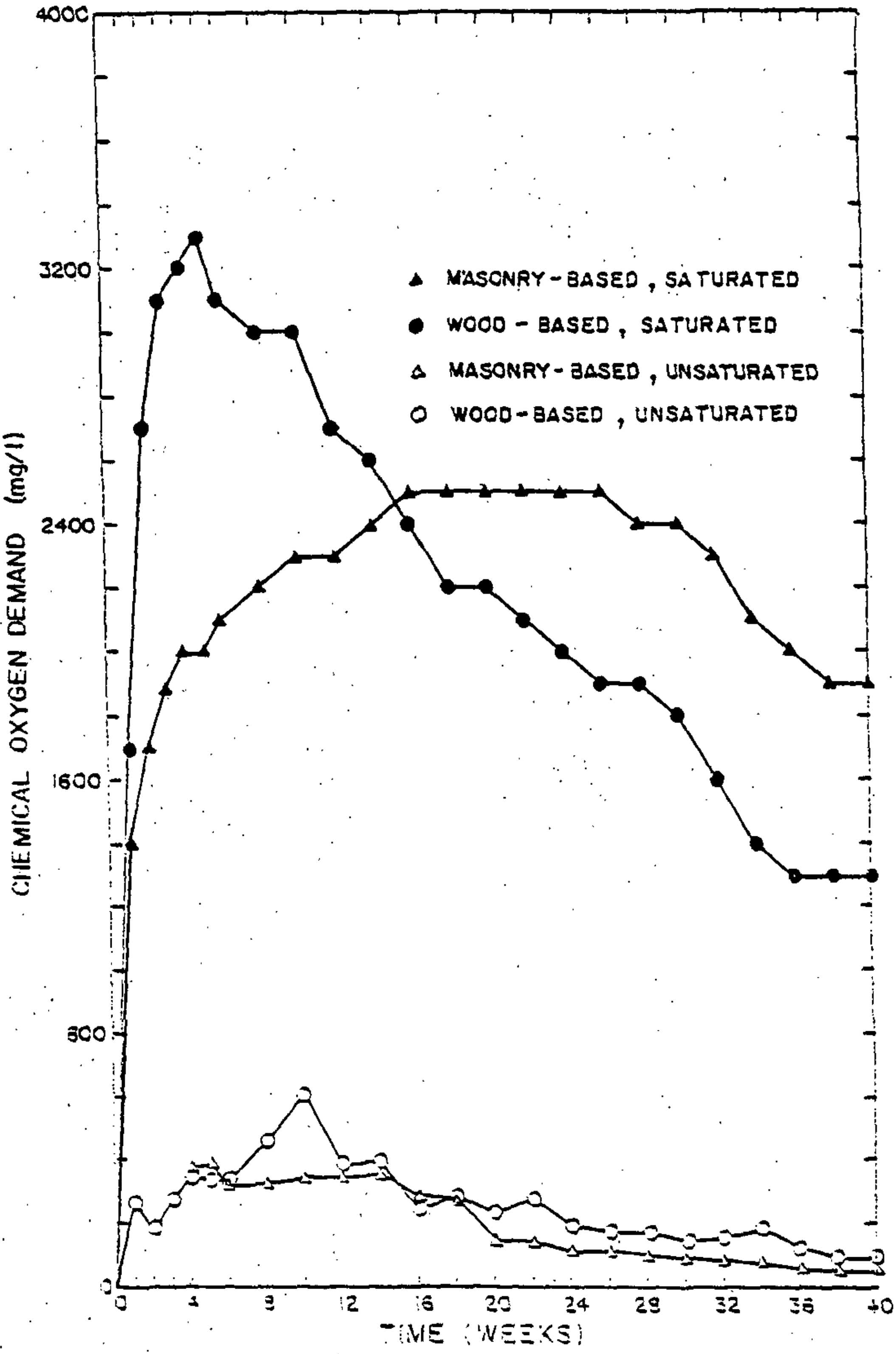

Figure 3. Chemical Oxygen Demand for Lysimeter Leachates

declines.

Total Carbon. The total carbon (TC) was measured to give a better measure of the organic matter than that available with the COD test. As performed, the determination included both the total organic and inorganic carbon and was therefore an over estimation of the true amount of organic material present.

Due to problems with the instrument the TC analysis was only performed for the first 24 weeks of the test period. The results for the saturated wastes generally followed the trends of the COD results. The TC of the saturated masonry-based waste was approximately 35 percent of the COD and the TC of the saturated wood-based waste was approximately 40 percent of the COD. For the unsaturated wastes the TC to COD ratio varied from 40 to 90 percent for the masonry-based waste and 50 to over 100 percent for the wood-based waste.

Iron. The total iron concentrations for three of the four leachates are reported in Figure 9. The fourth leachate, that of the saturated masonry-based waste, has been omitted due to its low concentration range of only 0.12 to 0.27 mg/ℓ. The remaining results were quite variable. The saturated wood-based leachate peaked at 84 mg/ℓ within four weeks and then rapidly declined to less than 1.0 mg/ℓ for the remainder of the test period. The unsaturated wood-based leachate peaked several times, but generally ranged

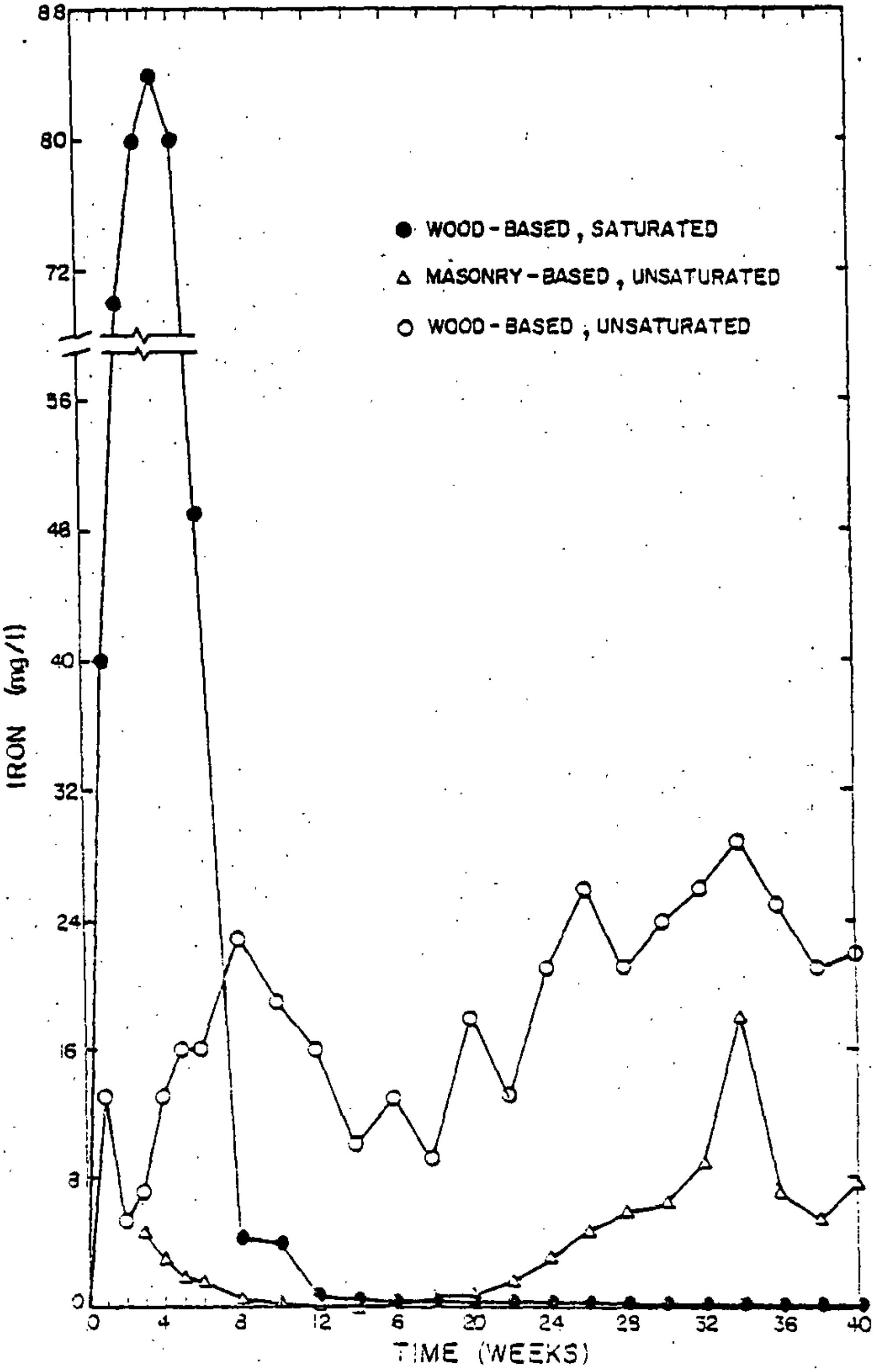

Figure 9. Iron Concentrations for Lysimeter Leachates

from 10 to 30 mg/ℓ. The unsaturated masonry-based leachate reached quite low concentrations due to the pH increase associated with weeks six through 14, but gradually increased to a peak of 18 mg/ℓ as the pH decreased.

Manganese. As for iron, the manganese data for three of the four leachates is illustrated. The leachate of the saturated masonry-based waste was omitted from the results in Figure 10 due to the extremely low manganese concentration of 0.01 to 0.09 mg/ℓ. Again the results were quite variable. The saturated wood-based waste peaked at 9.3 mg/ℓ within four weeks and then declined to less than 1.0 mg/ℓ for the rest of the test period. The unsaturated wood-based waste peaked twice and then declined steadily for the last 30 weeks. These peaks in the concentration of manganese for the unsaturated wood-based waste were about five times greater than the peak concentration for the saturated wood-based waste. This is the opposite of the results for iron in which the saturated wood-based waste was about three times the unsaturated value. The manganese concentration for the unsaturated masonry-based waste followed the same trend as that for iron. With the high pH during weeks 6 to 14 the manganese concentration dropped quite low and then increased as the pH decreased.

Zinc. The levels of zinc were quite low for all of the leachates with the exception of the saturated wood-based

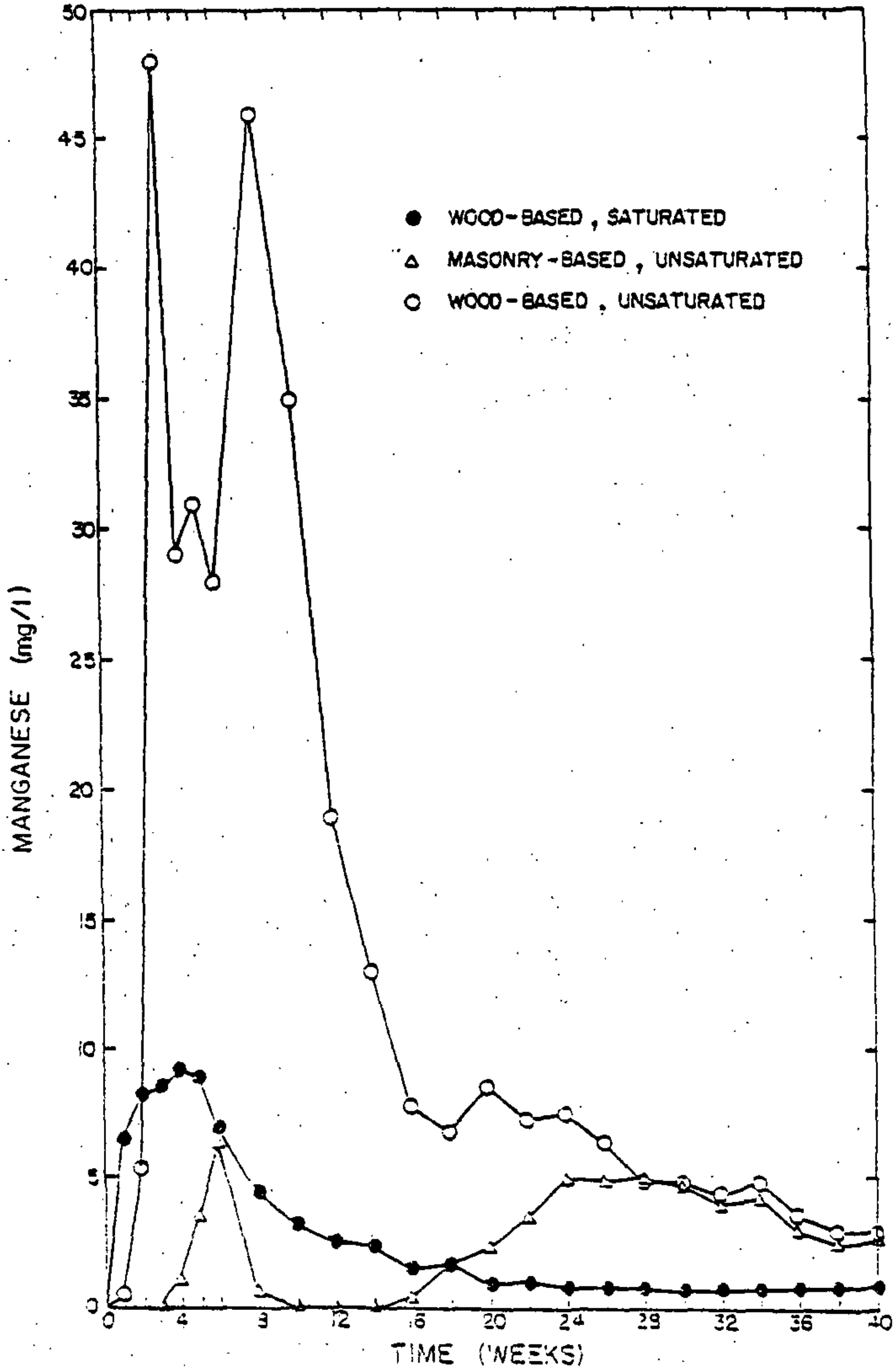

Figure 10. Manganese Concentrations for Lysimeter Leachates

waste. The concentration of zinc never exceeded 0.15 mg/ℓ for the saturated masonry-based waste, 0.40 mg/ℓ for the unsaturated masonry-based waste, and 0.80 mg/ℓ for the unsaturated wood-based waste.

For the saturated wood-based waste the results were much more significant. The concentration climbed to 50 mg/ℓ by the third week, peaked at 54 mg/ℓ on week four, declined to 0.83 mg/ℓ by week ten, and continued to decline to an average value of about 0.06 mg/ℓ during the last 20 weeks. The timing of this peak concentration for the saturated wood-based waste parallels that of iron and manganese.

Other Metals. Copper was analyzed for all the leachates on a regular basis during the first half of the test period and then abandoned due to the insignificant levels. The concentrations of copper never exceeded 0.22 mg/ℓ for either of the saturated waste leachates and 0.80 mg/ℓ for either of the unsaturated waste leachates.

On several occasions (week two, four, six, and 12) lead and cadmium were analyzed for all four leachates. The concentrations were found to be below normal detectable levels; lead less than 0.5 mg/ℓ and cadmium less than 0.25 mg/ℓ.

Total Amount Released. The total amount of material released in the lysimeter leachates for the entire test period was calculated and reported for a number of parameters. The amount of material released per unit weight is reported in Table 8 and the amount of material released per unit volume is reported in Table 9. To construct these tables the volume of leachate for each waste for every sampling interval had to be considered. The material released varied considerably between Table 8 and Table 9 because of the large difference in the density of the two wastes.

The results show an averaging effect over the entire test period and individual peaks are not recognizable. For the saturated wastes the values for alkalinity, hardness, and solids were approximately two to three times greater than the values for the unsaturated wastes. The COD values for the saturated wastes were much greater than those for the unsaturated wastes, and the results for metals were more variable. Although the peak concentration of iron was about three times greater for the saturated wood-based waste, the unsaturated wood-based waste actually released more iron over the test period. The unsaturated wood-based waste also released more manganese. For zinc, the saturated wood-based waste was the only waste that released significant amounts. Overall the two wood-based wastes released the most metals, the unsaturated masonry-based waste released only slightly lower levels of iron and manganese, and the saturated masonry-based waste released negligible amounts.

The total amounts of contaminants released can be used to predict field estimates of the leachate release for varying compositions of demolition waste under varying

TABLE 8

Total Amount of Material Released per Unit Weight in Lysimeter Leachate After 40 Weeks

Selected Parameter	Saturated Masonry-Based	 Wood-Based	Unsaturated Masonry-Based	 Wood-Based
Alkalinity	631	3670	346	1040
Hardness	2130	6950	1170	1840
COD	2270	4400	129	375
Total Solids	4810	11300	2230	2990
Iron	0.166	21.9	3.46	30.0
Manganese	0.028	5.14	2.27	17.0
Zinc	0.059	13.3	0.144	0.180

Values reported as grams released per metric ton of waste.

TABLE 9

Total Amount of Material Released per Unit Volume in Lysimeter Leachates After 40 Weeks

Selected Parameter	Saturated Masonry-Based	Saturated Wood-Based	Unsaturated Masonry-Based	Unsaturated Wood-Based
Alkalinity	419	1210	230	344
Hardness	1410	2300	774	607
COD	1510	1460	86.0	124
Total Solids	3190	3740	1480	990
Iron	0.110	7.23	2.30	9.94
Manganese	0.018	1.70	1.51	5.64
Zinc	0.039	4.39	0.096	0.060

Values reported as grams released per cubic meter of waste.

conditions. It should be understood that the laboratory study simulated improper disposal practices and that proper site selection and design may reduce or eliminate the problem.

Discussion

The intent of the laboratory investigation was to determine the parameters leached from demolition waste, the maximum concentrations of these parameters, and the total amount released over the test period. The laboratory design also allowed for the study of different compositions of demolition waste under different water percolation conditions. The principal objective was to predict the water pollution potential from the improper landfill disposal demolition waste.

Since there was a considerable difference between the saturated and unsaturated conditions, as well as between the two types of waste, the main points are outlined in Table 10.

Based on the laboratory results there are a number of generalizations and trends that appear. First, there was a substantial difference between the leachates generated under saturated versus unsaturated conditions as well as between the different compositions of waste. The leachates of the two unsaturated wastes were similar in composition and generally much weaker than the leachates of the two saturated wastes. The leachates from the saturated wastes

TABLE 10

Summary of the Relative Levels of Various Parameters for the Lysimeter Leachates

	Saturated		Unsaturated	
	Masonry-Based	Wood-Based	Masonry-Based	Wood-Based
pH	Very Alkaline	Slightly Acidic	Variable	Slightly Acidic
Conductivity	High	High	Moderate	Moderate
Solids	High	High	Moderate	Moderate
Alkalinity	Moderate	High	Moderate	Moderate
Hardness	High	Very High	Moderate-High	Moderate
COD	High	High	Low	Low
Iron	Very Low	High Peak	Moderate Peak	Moderate Peaks
Manganese	Very Low	Moderate Peak	Moderate Peaks	High Peaks

showed more variation between the two types of waste. This was due largely to the pH difference.

The second generalization is the effect of pH as a master variable. It was quite evident that high pH values limited the concentration of metals in the leachates. The results of the unsaturated masonry-based waste are illustrated in Figure 11 and exemplify the effect of pH on the solubility of iron and manganese.

A third generation deals with peak concentrations. For a number of parameters there was a plateau effect after a sharp initial peak. This was evident for conductivity, hardness, and solids, and to some extent, for COD and alkalinity. Iron and manganese, on the other hand, typically peaked over short periods of time and their values were much less predictable. In general all of the peaks for the saturated wastes occurred within the first ten weeks while the peaks for the unsaturated wastes were more scattered.

Another generalization can be made about the relationship between the trends of a number of the parameters. It appears that conductivity may be a good indicator of a number of parameters. In general conductivity paralleled the trends of hardness and total solids, and to a lesser extent, COD and alkalinity. For the concentration of metals pH has already been established as a good indicator, but for pH values below eight or nine it is important to analyzed for both iron and manganese due to the independence

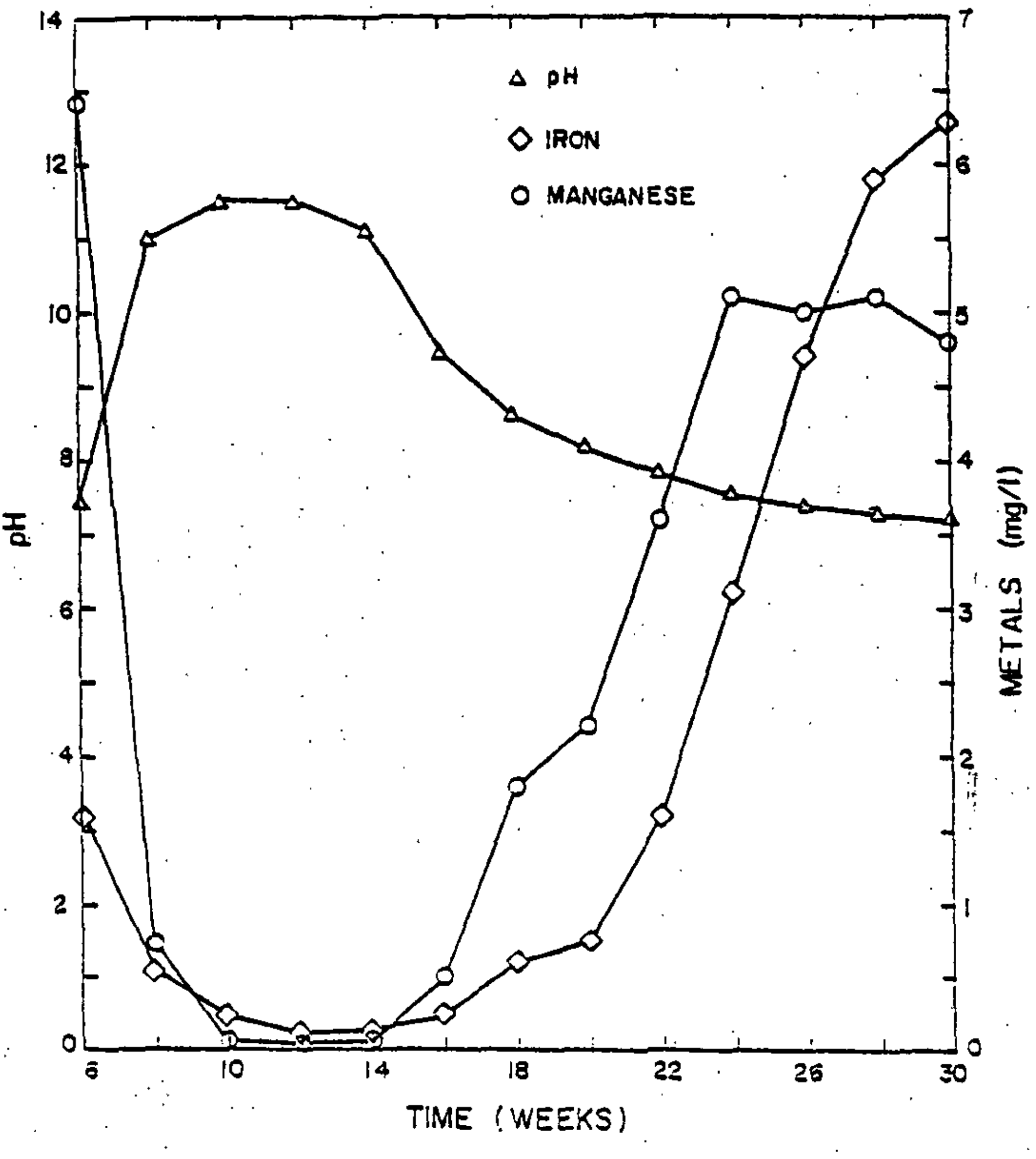

Figure 11. Relationship Between pH, Iron, and Manganese for the Unsaturated Masonry-Based Waste

CHAPTER V

CONTROL STRATEGIES FOR LEACHATE RELEASE

Methodology

The final phase of this research involved a study of the capacity of concrete rubble and crushed limestone substrates to remove metals from demolition waste leachate. The application of the control strategy would include the installation of a permeable substrate base beneath a demolition waste landfill to reduce the strength of the leachate released. This idea has the appeal of simplicity and a low materials cost since one of the major components of demolition waste is concrete rubble.

The theory for controlling leachate release from landfills was explained in the literature review and includes, in combination, the importance of pH, the ability of lime to increase the pH, the prolonged ability of concrete to leach lime, and the possibility of an "in situ" control of the release of various leachate elements. The majority of concrete rubble is Portland cement concrete and the major component of Portland cement is lime. Saturated limewater at $25^{\circ}C$ has a pH of 12.42 (17). An initial indication of this phenomenon was observed in the laboratory study where the saturated masonry-based waste generated a leachate that reached pH values greater than 12.0.

In an attempt to test the control strategy anaerobic and aerobic batch simulations were performed with different sizes of concrete rubble and crushed limestone. All analyses were performed with leachate spiked with high concentrations of iron and manganese. The saturated wood-based leachate from the lysimeter study was used because it had the greatest strength of the lysimeter leachates and was readily available at the end of the laboratory test period.

For simplicity batch simulations were performed in lieu of dynamic column simulations. With the batch simulations anaerobic conditions are difficult to maintain, although it is important to at least compare anaerobic results to those results obtained with the leachate exposed to the atmosphere. When anaerobic leachate is exposed to the atmosphere, metal precipitates form rapidly (33). Therefore, a simple batch study was performed under anaerobic conditions with a number of more detailed batch studies under aerobic conditions.

Anaerobic Study. The first portion of this phase of the research involved batch studies on small and large grain sizes of concrete and limestone substrates under anaerobic conditions. The concrete substrate was obtained by crushing Portland cement concrete blocks and sieving the rubble to 0.635-0.953 cm (0.25-0.375 inch) and 1.91-2.54 cm

(0.75-1.00 inch) grain sizes. The limestone was a crushed Dolomite limestone analyzed by the commercial distributor to be 21.4 percent elemental calcium and 12.0 percent elemental magnesium. The limestone was sieved to the same grain sizes as the concrete material.

The sample containers consisted of rectangular, plexiglass-columns 12.7 cm by 12.7 cm in cross section and 45.7 cm in height (5 x 5 x 18 inches). The total volume of each sample container was 7.374 liters (1.95 gallons). Each sample container was fitted with a drain tap in center of the top and the bottom. Four of the containers were completely filled with the various substrates and a fifth was used as a control. Each container was purged with nitrogen gas and filled from the bottom with spiked leachate. Anaerobic conditions were maintained in the leachate reservoir and sample containers by using a water trap to prevent the entrance of atmospheric oxygen. Each container was completely filled, tightly sealed, and left unagitated at room temperature.

The study was run for four weeks with samples extracted and analyzed after two weeks and again at the end of four weeks. At the end of two weeks approximately 100 ml was drained from each column and analyzed for pH, iron, and manganese. After four weeks the columns were completely drained and analyzed for pH, conductivity, alkalinity, hardness, COD, total solids, volatile solids, residual solids, iron, and manganese.

Aerobic Study. The aerobic portion of the study was designed as a simple technique to determine the capacity for removal of iron and manganese from leachate by the concrete and limestone substrates. Of concern was the effects of varying the grain size of the substrate, the amount of substrate, and the time period.

To determine the effect of time as well as grain size, batch studies were run in parallel for two hours, 48 hours, and two weeks. The substrates used were the same small and large grain size of concrete and limestone as used in the anaerobic study with the addition of a small and large grain size gravel substrate. Washed gravel was used as an inert control to determine the physical effect of the substrate in the removal process. All substrates were placed in 500 ml sample bottles and filled with 200 ml of spiked leachate. Due to the different densities and porosities of the substrates, the weight required for 200 ml of liquid to be level with the surface of the substrate needed to be experimentally determined. The amount of substrate used for both grain sizes was 350 grams for concrete, 600 grams for limestone, and 450 grams for gravel. All sample bottles were tightly sealed and left at room temperature for the pre-determined durations. All samples were analyzed for pH, iron, and manganese.

In order to better analyze the removal capacity of the concrete and limestone, an attempt was made to construct removal isotherms. The overall removal includes ion exchange, adsorption, and precipitation. For this reason, "removal isotherms" were constructed without specifying the mechanism, and the common adsorption isotherm equations and methodology was applied.

Adsorption, in general, is the process of collecting soluble substances that are in solution on a suitable interface (21). Of concern in this study is the case of adsorption at the liquid-solid interface. The process can be pictured as one in which molecules leave solution and are held on the solid surface by chemical and physical bonding. Chemical bonding involves very strong bonds and is considered irreversible. Physical bonding involves very weak bonds and the molecules are easily removed or desorbed by a change in concentration of the solution. The molecules adsorbed from solution are referred to as the "absorbate" and the solid is referred to as the "adsorbent". The amount of material adsorbed per unit weight of adsorbent can be determined experimentally as a function of the adsorbate concentration. The resulting function is called an adsorption isotherm and can be used as a measure of adsorptive capacity for a given temperature. The most common equations used to describe adsorption isotherms were developed by Freundlich and Langmuir. The

Langmuir equation was derived from rational considerations and assumes single layer adsorption and uniform adsorption energies (37). The Freundlich equation is a special case for heterogeneous surface energies and is basically empirical (37). The isotherm equations relate the amount of material in solution to that adsorbed as follows:

$$\text{Freundlich} \qquad \frac{X}{M} = kC^{1/n}$$

$$\text{Langmuir} \qquad \frac{X}{M} = \frac{abC}{1 + bC}$$

where: X/M = amount adsorbed per unit weight of adsorbent

C = residual concentration of adsorbate in solution after adsorption

k,n,a,b = empirical constants

To determine the removal isotherms and approximate the removal capacity of the concrete and limestone substrates a series of substrate additions was used. Three different substrate grain sizes were used: 0.635 to 0.953, 0.953 to 1.91, and 1.91 to 2.54 centimeter diameters (0.25-0.375, 0.375-0.75, 0.75-1.00 inches). The three concrete substrates were used in seven equal increments from 50 to 350 grams and the three limestone substrates were used in eight equal increments from 75 to 600 grams. All substrates were placed in 500 ml plastic sample bottles to which 200 ml of spiked leachate was added. The sample bottles were tightly covered and left at room temperature. The batch study was run for two weeks in an attempt to reach equilibrium. All samples were analyzed for pH, iron, and maganese.

Results

Anaerobic Study. For the four substrates used, the total weight, the effective volume of solids, and the resulting porosities are given in Table 43, Appendix C. Due to the different densities and porosities the total weight of each concrete substrate was approximately two-thirds that of the limestone substrates. The volume of leachate added also varied from 2.76 to 2.98 liters for the small and large limestone substrates and 3.15 to 3.49 liters for the small and large concrete substrates.

The results of the pH, iron, and manganese analysis after two and four weeks are reported in Table 11. For the end of the four week study a more detailed chemical analysis was performed on the five leachate samples and the results are reported in Table 12. The initial spiked leachate had a pH of 6.80, iron concentration of 106 mg/ℓ, and manganese concentration of 84 mg/ℓ. The other chemical parameters were assumed to be equal to the values obtained on week 40 of the test period since the leachate was extracted shortly afterward. After four weeks the control showed a decrease in the iron and manganese concentration to 20 and 39 mg/ℓ respectively, and no change in the pH.

TABLE 11

pH, Iron, and Manganese Values for Anaerobic Batch Studies After Two and Four Weeks

	Two Weeks			Four Weeks		
	pH	Fe	Mn	pH	Fe	Mn
Control*	6.80	21	39	6.80	20	39
Small limestone	6.60	0.50	6.8	6.70	0.46	5.1
Large limestone	6.60	2.2	9.4	6.70	0.88	7.2
Small concrete	8.20	2.5	0.31	8.90	5.8	0.14
Large concrete	7.80	14	0.79	8.35	25	0.39

All values of Fe and Mn in mg/ℓ

*Initial spiked leachate: pH = 6.80; Fe = 106 mg/ℓ, Mn = 84 mg/ℓ

TABLE 12

Results for Anaerobic Batch Studies after Four Weeks

		Limestone		Concrete	
Parameter	Control	Small	Large	Small	Large
Conductivity (μmhos/cm)	5300	5000	5000	4000	4700
Alkalinity (mg/ℓ as $CaCO_3$)	1200	1100	1000	80	140
Hardness (mg/ℓ as $CaCO_3$)	3600	3200	3200	2400	2700
COD (mg/ℓ)	1000	1000	1000	780	850
Total Residue (mg/ℓ)	5650	5450	5400	4500	4800
Fixed Residue (mg/ℓ)	3900	3750	3700	3700	3850
Total Volatile (mg/ℓ)	1750	1700	1700	800	950

Any decrease in the concentration of iron and manganese below the level of the control for the other samples was attributed to removal by the substrates.

After four weeks the small limestone substrate removed 86.9 percent of the manganese and 97.7 percent of the iron, and the large limestone substrate removed 81.5 percent of the manganese and 95.6 percent of the iron. For the concrete substrates, the small substrate removed 99.6 percent of the manganese and 71.0 percent of the iron, while the large substrate removed 99.0 percent of the manganese, and none of the iron.

For the large concrete substrate the iron concentration was in fact slightly higher than that of the control. Over the period from the second to the fourth week, the concentration of iron for both the small and large concrete substrates actually increased by approximately a factor of two. This apparent reversible behavior raises a question about the permanency of the removal capacity. This effect may be explained by the sampling technique which incorporated a minimal extraction of leachate after two weeks, and a relatively rapid draining of the sample container at the end of the four week period. Therefore, the reversibility of the removal capacity may have been due to draining the substrate and disturbing the equilibrium balance between the substrate and the solute.

The overall removal results for iron and manganese demonstrate that under these conditions the limestone substrates have a low capacity to remove manganese and a good capacity to remove iron. The concrete substrates, on the other hand, have an excellent capacity to remove manganese and a questionable capacity for iron removal. In both cases the small substrates were more effective in removing the metals than the large substrates. This is due to the increased surface area of the smaller substrates. Assuming perfect spheres the large substrate would only have 36 percent of the average surface area of the small substrate per unit volume.

The results of the other chemical parameters showed very little removal by the limestone substrates and varying removal capacities for the different constituents by the concrete substrates. The removal by concrete of the alkalinity was the most notable result, while the conductivity, hardness, COD, and total solids were reduced by 11 to 33 percent. The volatile component of the total solids was reduced by about 50 percent and accounted for almost all of the reduction in the total solid concentration. This reduction in alkalinity, with a similar reduction in hardness, and somewhat less of a reduction in volatile solids, indicates that calcium carbonate must be precipitating.

Aerobic Study. In the first portion of the aerobic batch studies spiked leachate was added to the two different sizes of the three different substrates and sampled after two hours, 48 hours, and two weeks. The results for iron and manganese are reported in Table 13 and the results for pH are illustrated in Figure 12.

An attempt was made to spike the leachate to the same levels as in the anaerobic study. Due to the aerobic conditions a reduction in the solubility of iron and manganese in the spiked leachate was anticipated. This reduction in solubility was not evident for manganese but was quite dramatic for iron. The resulting initial concentration of iron was 23 mg/ℓ as compared to 106 mg/ℓ under anaerobic conditions, while the manganese concentration was 84 mg/ℓ under both conditions. The final concentration in the control after two weeks was 42 mg/ℓ for manganese and only 5.9 mg/ℓ for iron.

The major difference in the solubility of iron and manganese was due to the pH of the solution. The reaction rates of this oxidation and the removal of these ions from solution are strongly pH dependent. Stumm and Morgan (31) reported on the disappearance of iron and manganese from solution at different pH values and showed that Fe(II) is rapidly oxidized above pH 6.6 and Mn(II) is rapidly oxidized above pH 9.0. They found the rate of oxygenation of Fe(II) in solutions of pH ≥ 5.5 to be first

TABLE 13

Iron and Manganese Concentration in Aerobic Batch Studies with Time

	Iron (mg/ℓ)*			Manganese (mg/ℓ)**		
	2 hrs	48 hrs	2 wks	2 hrs	48 hrs	2 wks
Control	-	-	5.9	-	-	42
Large gravel	19	5.6	4.0	77	63	29
Small gravel	21	7.4	1.3	87	72	16
Large limestone	14	2.9	1.0	53	19	2.5
Small limestone	15	2.1	1.5	43	2.0	2.6
Large concrete	17	8.1	2.3	49	5.6	0.94
Small concrete	16	4.8	1.9	36	3.5	0.56

*The initial iron concentration was 23 mg/ℓ.

**The initial manganese concentration was 84 mg/ℓ.

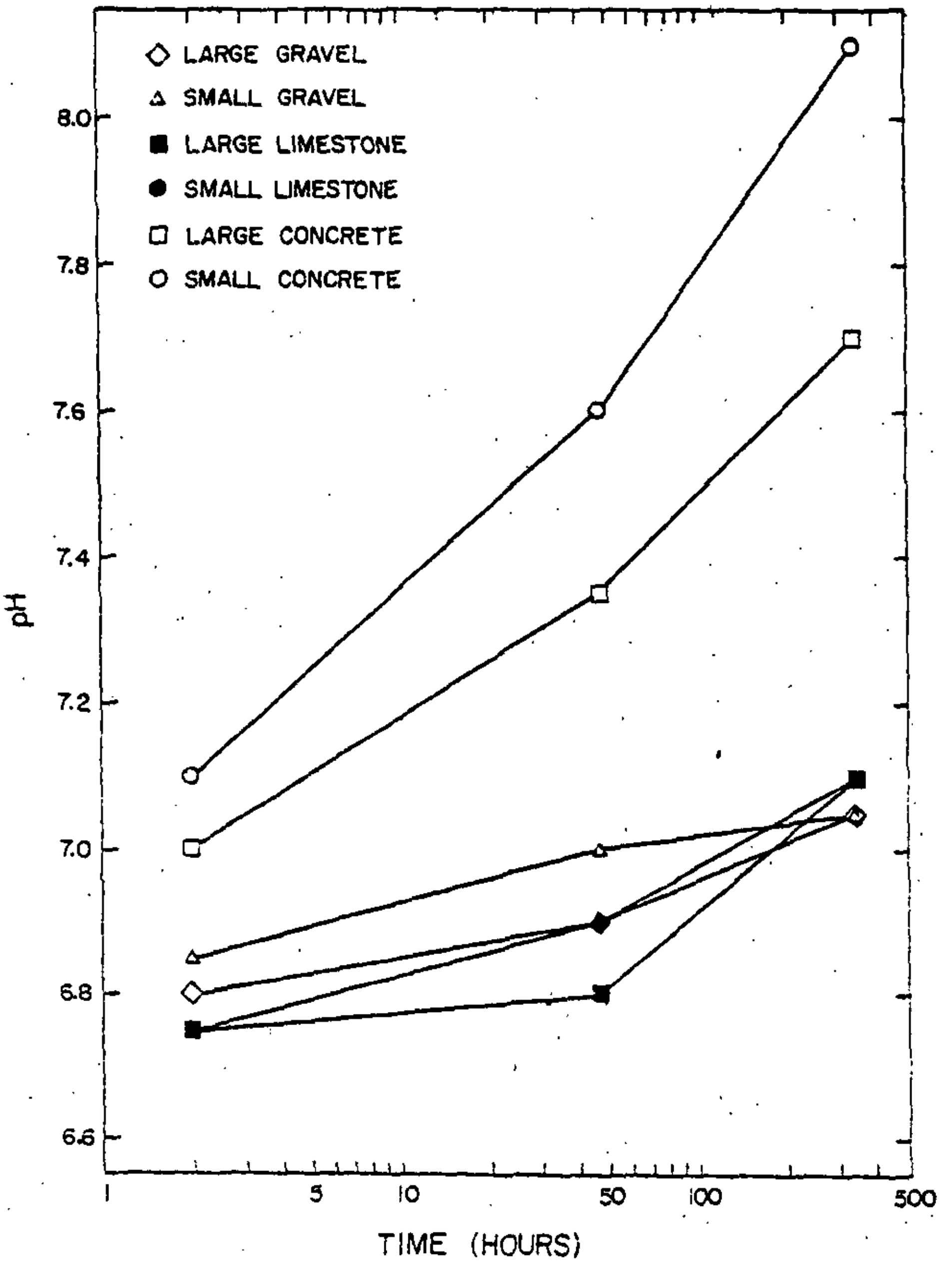

Figure 12. pH Variations with Time in Aerobic Batch Studies (Initial pH = 6.80).

order with respect to the concentrations of both Fe(II) and O_2, but that the decrease of the Mn(II) concentration with time suggested an autocatalytic reaction.

At the end of two weeks the pH for both the gravel substrates increased to 7.05, both limestone substrates increased to 7.10, the small and large concrete substrates increased to 7.70 and 8.10, respectively, and the control increased to 7.30. As in the anaerobic study, any decrease in the iron and manganese concentrations below the level of the control was considered to be removal by the substrates. For all the samples there was a continued decrease in the iron and manganese concentrations over time. After two weeks the manganese concentrations for the large and small gravel substrates were 29 and 16 mg/ℓ, respectively. The manganese concentrations for the limestone and concrete substrates were considerably lower than these values indicating good removal capacity. The iron concentrations for the large and small gravel were also lower than the control, but there was much less, if any, difference between the low values for the gravel substrates and the values for the limestone and concrete substrates. For this reason, only the removal capacity for manganese were considered valid in the aerobic batch studies.

By the end of two weeks the small limestone substrate removed 93.8 percent of the manganese and the large limestone substrate removed 94.0 percent. For the concrete substrates, the small substrate removed 98.7 percent of the manganese and the large substrate removed 97.8 percent. These removal capacities for the concrete substrates agree quite well with the anaerobic study after two weeks, but the limestone removal capacities were significantly greater under aerobic conditions.

Removal Isotherms. In the second portion of the aerobic batch studies the same spiked leachate was added to varying amounts of the three different size concrete and limestone substrates in order to construct removal isotherms. The pH results for varying amounts of substrate are shown in Figure 13 and agree reasonably well with the previous data. For iron and manganese, in general, there was a reduction in the concentrations with increasing amounts of the limestone and concrete substrates. Again, due to the aerobic conditions and the pH of the leachate, the results for iron were not considered valid and only the manganese results were reported in detail. The actual data for both iron and manganese is listed in Table 45, Appendix C.

For all the substrates the manganese removed per unit weight of substrate increased with increases in the residual concentration of manganese. This data fit the Freundlich equation reasonably well, but could not be fitted to the Langmuir equation.

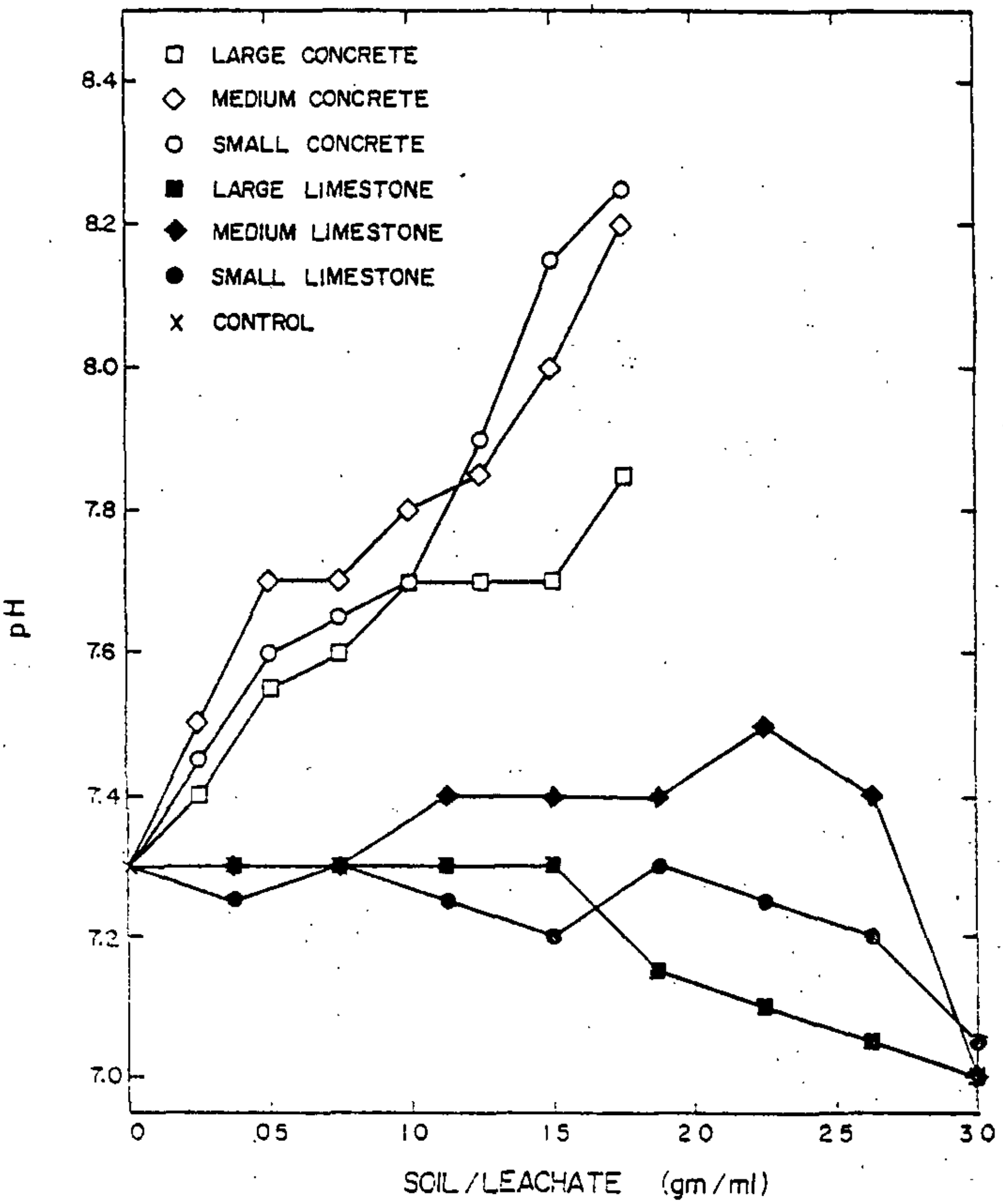

Figure 13. pH Variations with Varying Amounts of Substrate in Aerobic Batch Studies After Two Weeks

The constants for the Freundlich equation are evaluated by a straight line plot of the isotherms on double logarithmic paper (see Figures 14 and 15). To obtain the line of best fit a least-squares linear regression was performed on a programmable hand calculator. This procedure minimizes the sum of the squares of the deviations of the actual data points from the straight line of best fit. The Freundlich equations along with the correlation coefficients and the relative surface areas are reported in Table 14. The correlation coefficient (R) was calculated to determine how well the "best fit line" actually does approximate the data. The concrete substrates showed much better fits, although all the correlations were considered acceptable. The value of R would be ± 1 for a perfect correlation.

A comparison can be made between the empirical constant k, which roughly approximates the capacity, and the surface area of the different size substrates assuming spheres. Although the k values for the substrates were not directly proportional to the assumed surface areas there was a general increase in capacity with increased surface area.

In graphically comparing isotherms the slope is an indicator of the removal intensity and the relative position is proportional to the removal capacity (37). A comparison of the isotherms for the small and large

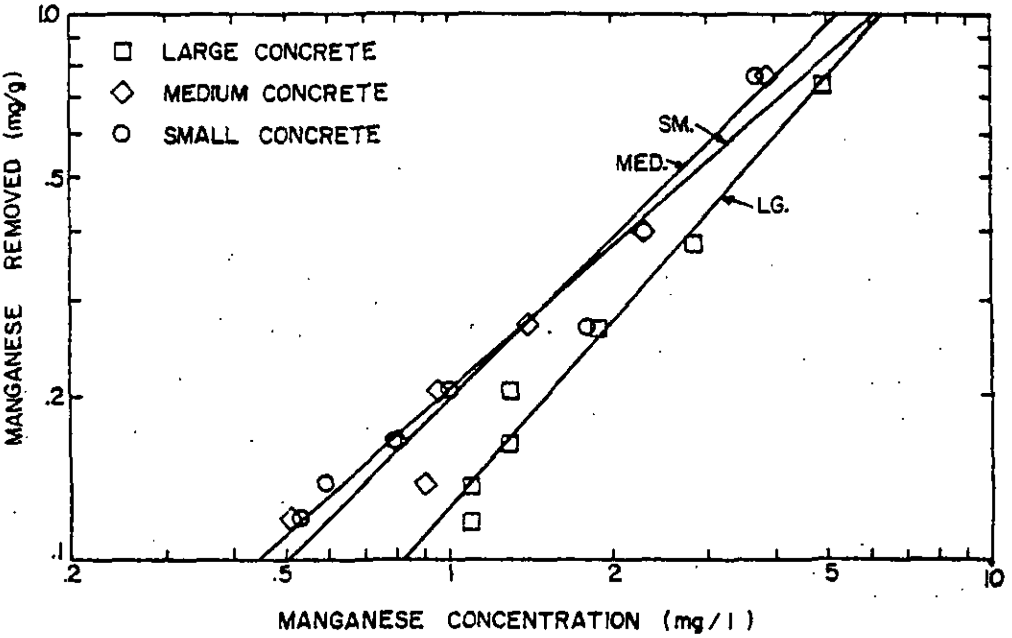

Figure 14. Manganese Isotherms for Varying Sizes of Concrete Substrate: Logarithmic Form

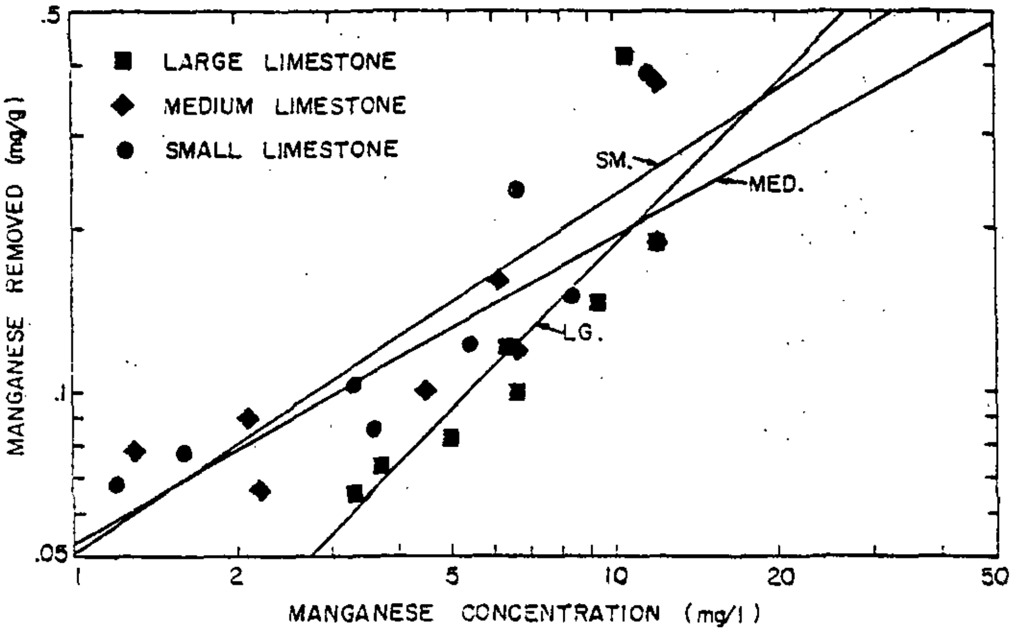

Figure 15. Manganese Isotherms for Varying Sizes of Limestone Substrate: Logarithmic Form

TABLE 14

Removal Isotherms, Correlation Coefficients, and Relative Surface Areas for Substrates

Substrate	Freundlich Equation	Correlation Coefficient	Relative Surface Area*
Small concrete	$\frac{X}{M} = 0.203C^{0.875}$	0.981	2.80
Medium concrete	$\frac{X}{M} = 0.196C^{0.980}$	0.936	1.56
Large concrete	$\frac{X}{M} = 0.124C^{1.13}$	0.985	1.00
Small limestone	$\frac{X}{M} = 0.051C^{0.655}$	0.890	2.80
Medium limestone	$\frac{X}{M} = 0.053C^{0.561}$	0.879	1.56
Large limestone	$\frac{X}{M} = 0.018C^{1.01}$	0.848	1.00

*Ratio of surface areas assuming average diameters and perfect spheres.

concrete and limestone substrates is shown in Figure 16. From the relative positions it is quite evident that concrete has a much better removal capacity for manganese than does limestone. From the relative slopes it appears that the large substrates have better removal capacities at high concentrations and less at low concentrations. This cross-over in the lines may be a result of the linear regression fit and may not be truly representative of the removal intensity.

Discussion

The objective of the batch studies was to investigate the use of concrete or limestone substrates beneath demolition waste landfills as a control strategy. The study was performed as a simple simulation to test the ability of the substrates to remove iron and manganese from leachate. Anaerobic conditions proved to be important to the solubility of iron, but not manganese, at the pH range encounter.

In the anaerobic study the limestone substrates demonstrated a low capacity to remove manganese and a good capacity to remove iron, while the concrete substrates demonstrated an excellent capacity to remove manganese and a questionable capacity for iron removal. In comparing the results after two weeks and four weeks the iron removal capacity of the concrete substrates was found

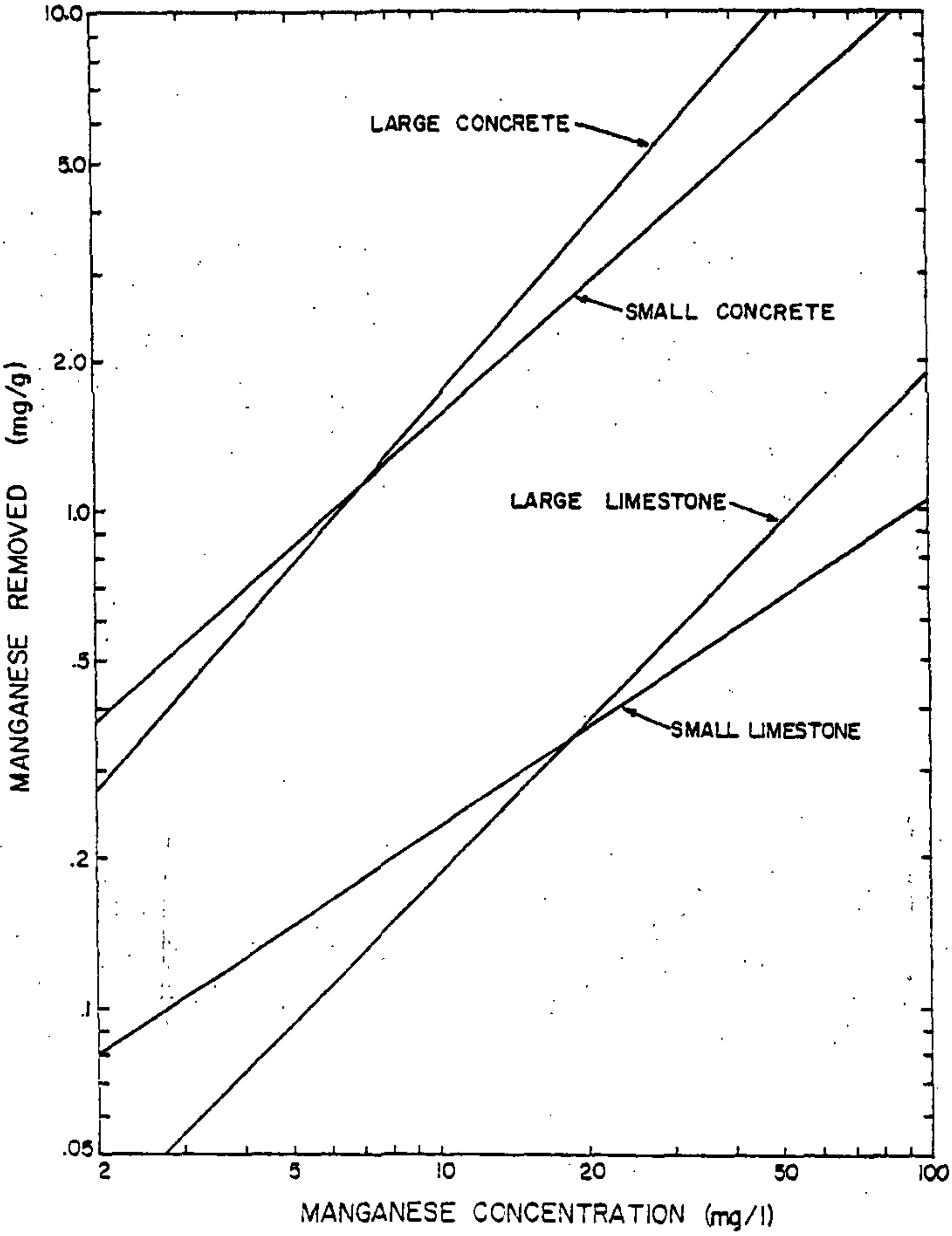

Figure 16. A Comparison of Manganese Isotherms for Concrete and Limestone Substrates: Logarithmic Form

to be reversible, and may have been due to the sampling technique. The concrete was also found to have a much better removal capacity for a number of other chemical constituents.

In the aerobic studies only the manganese results were considered valid. The pH range of the samples under aerobic conditions was directly responsible for the precipitation of iron and not manganese since iron is rapidly oxidized above pH 6.6 and manganese is not rapidly oxidized until pH 9.0. The limestone substrates showed minimal variations from the initial pH of 6.80, while the concrete substrates showed gradual increases. The reason for the increased pH for concrete, and not limestone, is the free lime which dissolves from the concrete. Lime consists of calcium oxide and reacts with water to form calcium hydroxide which is a strong base. Limestone, on the other hand, consists of calcium carbonate, is relatively insoluble, tends only to neutralize acidic solutions, and will not continue to raise the pH much above neutral.

The amount of manganese removal attributed to the gravel as an inert substrate was found to be 31 percent for the large substrate and 62 percent for the small substrate. Since the removals by the concrete and limestone substrates were measured as total removals without specifying the mechanism, the values for the gravel were not used in the removal capacity calculations. More specific removal mechanisms could be estimated by taking into account the proportions removed by the inert substrates.

From the results with varying time periods and varying amounts of substrate the concrete was found to have a much better capacity for manganese removal than the limestone. In general, the results indicate that concrete shows merit in its removal capacity and may indeed have application in a control strategy. Because the batch studies simulated saturated conditions further studies should be performed to determine the effects of possibly more realistic percolating flow. Additional studies should also examine the reversibility of the removal capacities.

CHAPTER VI

CONCLUSIONS

In summary, the main conclusions of this research are as follows:

1. The available information on demolition waste indicates that significant volumes of this waste are produced each year and that the predominant current practice is the disposal of this waste as landfill material.
2. From the field and laboratory results it was found that demolition waste can generate a leachate with high concentrations of conductivity, alkalinity, hardness, COD, iron, manganese, and zinc.
3. Leachates under saturated conditions can vary greatly with different compositions of demolition waste, while leachates under unsaturated flow are generally similar in composition. The saturated waste leachates are generally greater in strength than unsaturated leachates.
4. pH was found to be one of the most important variables and directly controls the solubility of a number of chemical constituents, with iron being the most predominant.

5. Concrete rubble has an excellent capacity for the removal of manganese and a moderate capacity to remove a number of other constituents. This may have application in control strategies as an alternative to the suggested use of limestone as a base beneath landfills.
6. The strength of demolition waste leachate resulting from improper disposal practices can be environmentally significant and has demonstrated the need, similar to municipal landfills, for proper site selection and design to reduce the possible degradation of water quality.

CHAPTER VII

RECOMMENDATIONS FOR FUTURE RESEARCH

Based on the results of this research the following recommendations are made:

1. Further laboratory studies are not expected to provide much additional information and the development of a predictive model using the present results may be more valuable.
2. Further field studies should be conducted which include measurements of the age, composition, and depth of the waste, as well as monitoring of ground and surface water and hydrologic conditions. Such an analysis could then be used to examine the applicability of the laboratory results as a predictive tool.
3. The use of limestone and concrete substrates as a control strategy should be studied further. Such research should be performed under anaerobic conditions and should also examine dynamic column flow under saturated and unsaturated conditions.

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