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CODISPOSAL OF DIMINIMUS LEVELS OF LOW-LEVEL RADIOACTIVE WASTE AND SANITARY WASTE

By

**E.S.K. Chian
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B. Kahn
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F.G. Pohland**

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**U.S. Department of Energy
Idaho Operations Office
Idaho Falls, Idaho 83407**

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A UNIT OF THE UNIVERSITY SYSTEM OF GEORGIA
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ATLANTA, GEORGIA 30332

1986



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<p>Codisposal of low-level radioactive waste (LLRW) with municipal refuse was investigated in two pilot-scale controlled landfills (concrete lysimeters; 3.05 m x 3.05 m x 4.28 m) which were lined with 30-mil HDA Gundline liner (elastomeric polyolefin alloy based high density polyethylene) and had provisions for leachate collection and leak detection. Shredded municipal refuse was placed within the landfills and spiked with radionuclides (Co-58, Sr-85, and Ce-141) at a level of 28 nCi/gm to simulate codisposal of LLRW with municipal refuse. Water was added to simulate normal rainfall events; the extent to which radionuclides and organics were leached from both landfills was recorded. To compare the effect of leachate</p> <p>(Continued)</p>					
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SUMMARY

Factors Affecting the Mobility of Selected Radionuclides
Codisposed with Municipal Refuse Within Landfills

Disposal of Low-Level Radioactive Waste (LLRW) has been accomplished by shallow land burial at licensed commercial sites since 1962. At some of the sites, however, subsurface migration of radionuclides, caused by leaching due to infiltration of groundwater, surface-water runoff, and incidental rainfall, have been reported. In addition, the gradual depletion of available volume for LLRW disposal at operational licensed commercial sites, due to site closure and increased generation of LLRW, has become a concern, since it may result in an increase in the volume requirements for storage of LLRW. Furthermore, the restrictions imposed on LLRW disposal, from January 1, 1986, by the Low-Level Radioactive Waste Policy Act of 1980 (PL96-573) may contribute to an additional increase in storage requirements for LLRW, since most of the states or the regional compacts (comprised of member states) currently do not have an operational site for LLRW disposal, and one of the provisions in this Act allows each state or a regional compact the authority to exclude LLRW generated outside its borders. In order to reduce the volume requirements for LLRW disposal, one of the suggested alternatives to shallow land burial of LLRW was the regulated release of low-activity wastes to a municipal sanitary

landfill. This alternative, namely, codisposal of selected radionuclides with municipal refuse within landfills, has been evaluated in this study.

The containment of the radionuclides, that are codisposed with municipal refuse, within a landfill during the landfill stabilization phase and the post-closure phase would allow the radionuclides to decay and minimize the possibility of impairment to the environmental setting around the landfill. In view of containment of LLRW codisposed with municipal refuse, leachate recycle would be the preferred mode of leachate management, since it would allow the LLRW to undergo their natural decay within the landfill. Also, the landfill should be lined and incorporate in its design the following structural elements: leachate and gas collection systems, a leak detection system, and provisions for leachate recycle.

The major objectives of this study were to evaluate the behavior of selected radionuclides (Co-58, Sr-85, and Ce-141) codisposed with municipal refuse within pilot-scale landfills during landfill stabilization, and to assess the potential for solubilization of Co-58, from cobalt sulfide precipitates, due to formation of complexes with humic substances present in leachate. A separate confirmation of such behavior was provided by the results of companion laboratory-scale studies on investigation of the potential of humic substances present in leachate to mobilize Co-58 from cobalt sulfide precipitate.

Two pilot-scale landfills, 3.05-m square and 4.28-m deep, were utilized to evaluate the factors affecting the mobility of selected radionuclides (Co-58, Sr-85, and Ce-141) codisposed with municipal

refuse during landfill stabilization. These landfills were reinforced concrete lysimeters, lined with a 0.76-mm (30-mil) high density polyethylene liner, and had provisions for leachate collection and recirculation. The municipal refuse placed within the cells was spiked with Co-58, Sr-85, and Ce-141 at levels of 0.91 mCi, 1.09 mCi, and 5.78 mCi, respectively. The resulting loading levels, based on dry weight of refuse, for Co-58, Sr-85, and Ce-141 were 1.6×10^{-7} Ci/kg, 2.0×10^{-7} Ci/kg, and 1.06×10^{-6} Ci/kg, respectively. Water was added to the landfills to simulate infiltration due to incidental rainfall in Atlanta, GA during the investigation, and leachate recycle was practiced in one of the landfills. During the study, the extent to which radionuclides and organics leached from both landfills was recorded. Analyses on leachate samples included detection of Co-58, Sr-85, and Ce-141 along with the indicator parameters, namely, pH, ORP, conductivity, total alkalinity, COD, BOD₅, TOC, TIC, volatile fatty acids (acetic, propionic, isobutyric, butyric, and valeric), sulfide, chloride, iron, manganese, nickel, zinc, cobalt, and strontium.

Based on evaluation of the data on radionuclides and indicator parameters, and the known progression of events that occur within a landfill undergoing stabilization, the following major findings complement the state-of-the-art knowledge on codisposal of LLRW with municipal refuse within landfills:

- i. The selected radionuclides, Co-58, Sr-85, and Ce-141, can be codisposed with municipal refuse at the loading levels utilized in this study.
- ii. The immobilization of Co-58 and Sr-85 present in leachate was

due to precipitation of these radionuclides as cobalt sulfide and strontium carbonate, while the immobilization of Ce-141 may be attributed to adsorption of the radionuclide onto the refuse.

- iii. The maximum levels of Co-58, Sr-85, and Ce-141, expressed as a percentage of initial loading levels, present in the leachate were 12.7%, 15.7% and less than 0.03%, respectively. The activity levels corresponding to these maximum levels of Co-58, Sr-85, and Ce-141 in leachate were lower than the maximum permissible concentrations of these radionuclides in water above natural background.
- iv. Humic substances, which constitute the major fraction of TOC of leachate after the degradation of readily degradable organics (final maturation phase), solubilize Co-58 from cobalt sulfide precipitate due to the formation of soluble complexes.
- v. Leachate recycle was the preferred mode of leachate management, since it allowed for containment of radionuclides within the landfill due to minimization of removal of leachate from the landfill. This allowed the radionuclides to undergo their natural decay within the landfill.

Thus, the radionuclides, Co-58, Sr-85, and Ce-141 can be safely codisposed with municipal refuse within landfills at the loading levels utilized in this study.

Further studies on the behavior of other radionuclides, which constitute LLRW viz., P-32, Cr-51, Se-75, Mo-99, I-131, Pu-238, and

Pu-240, within landfills during landfill stabilization is recommended, since the data obtained from such studies along with the data obtained from this study will form a data base on the behavior of LLRW within landfills, which in turn could be used to predict the behavior of LLRW with landfills.

CHAPTER I

INTRODUCTION

1.1 Problem Description

Disposal of Low-Level Radioactive Waste (LLRW) has been accomplished by shallow land burial at licensed commercial sites since 1962. However, at some sites, the current practice of LLRW disposal has resulted in subsurface migration of radionuclides caused by leaching due to infiltration of groundwater, surface-water runoff, and incidental rainfall (1-15, 18). Also, the gradual depletion of the available volume for LLRW disposal at the licensed commercial sites due to site closure, increased generation of LLRW, and volume allotment programs at sites, has become a concern since it may lead to an increase in the storage requirements for LLRW (15-17).

The Low-Level Radioactive Waste Policy Act of 1980 (PL96-573) imposes new restrictions on LLRW disposal. According to the provisions in this Act, effective January 1, 1986, each state or regional compact (comprised of member states) is responsible for assuring adequate disposal of LLRW generated within its borders. Also, this Act allows the states forming a regional compact, to designate a state with a LLRW disposal site as the host state within the regional compact, and the authority to exclude LLRW from outside their region. Thus, these restrictions imposed upon LLRW disposal by the Act will lead to the development of LLRW disposal sites within a state or a regional

compact, which currently does not have an operating site. Furthermore, these conditions of the Act may lead to an increase in the storage requirements for LLRW, since most of the states or regional compacts at present do not have an operating site (17).

In order to reduce the volume requirements for LLRW disposal, the Nuclear Regulatory Commission (NRC) has identified a need to develop a policy on regulating de minimus levels of radioactivity. Suggested alternatives to shallow land burial of low-activity wastes, under an exception to 10 CFR Part 20 included the regulated release of low-activity waste to uncontrolled disposal at a municipal sanitary landfill (19, 20).

The efficacy of codisposal of LLRW and municipal refuse hinges on the successful containment of LLRW within the landfill, during both the landfill stabilization phase and the post-closure phase. In view of total containment of LLRW, the design of landfills that receive municipal refuse and LLRW should incorporate provisions for leachate containment and recycle, since leachate recycle would contain LLRW within the leachate and the landfill environment until precipitants (e.g., sulfide, carbonates) are formed during landfill stabilization; these precipitants would precipitate LLRW within the landfills (21, 22, 23, 59, 64, 109). The precipitation of LLRW by potential-precipitant species, viz., sulfides, carbonates, and hydroxides, is counteracted by potential complexation of LLRW by humic substances present in leachate; this may lead to mobilization of precipitated LLRW. Thus, termination of leachate recycle during the final stages of landfill stabilization, namely, final maturation phase, would reduce the

potential mobilization of precipitated LLRW.

Codisposal of LLRW and municipal refuse within laboratory-scale lysimeters was investigated by Chang (21). Co-58, which was one of the radionuclides utilized to simulate LLRW, exhibited an increase in activity from 0.045 nCi/L to 0.1 nCi/L in leachate during methane fermentation phase of landfill stabilization. Chang (21) speculated that the increase in Co-58 in leachate was probably due to formation of soluble complexes of cobalt with humic substances present in leachate.

1.2 Objectives of the Research

Codisposal of selected radionuclides and municipal refuse within two pilot-scale landfills was investigated and the major objectives of the investigation were:

- i) to evaluate the behavior of Co-58, Sr-85, and Ce-141 codisposed with municipal refuse within the landfills during landfill stabilization; and,
- ii) to assess the potential of solubilization of Co-58 from precipitated cobalt sulfide by humic substances present in landfill leachates.

Since the containment of LLRW was a major determinant in the probable success of codisposal of LLRW with municipal refuse, both pilot-scale landfills were lined with 0.76-mm polyethylene liners (HDA Gundline, Gundline Lining Systems, Inc., Houston, TX) and designed for leachate collection and leak detection. One of the landfills was equipped with provisions for leachate recycle, since leachate recycle would accelerate landfill stabilization (22) and contain LLRW within the

leachate and the landfill environment until precipitants (e.g., sulfide, carbonates) are formed during landfill stabilization (23, 59).

1.3 Scope of the Research

An assessment of the technical feasibility of codisposal of selected radionuclides with municipal refuse within landfills would aid in evaluating this disposal technique as an alternative to shallow land burial for disposal of very low-activity wastes. The selected radionuclides, Co-58, Sr-85, and Ce-141, were codisposed with municipal refuse within two pilot-scale landfills and the extent to which these radionuclides along with the organics leached from the landfills was recorded. In addition, the potential for migration of these radionuclides through remobilization due to complexation with ligands, viz., humic substances, present in leachate was examined.

CHAPTER II

LITERATURE REVIEW

2.1 Low-Level Radioactive Waste (LLRW)

Low-Level Radioactive Waste (LLRW) is defined as waste that contains less than 10 nCi (nanocurie) of long-lived α -radiation per gram and has γ -radiation low enough to require minimal biological shielding and remote handling, or that average less than 35.7 curies of activity per m^3 (one curie of activity per cubic foot) of material or less than 200 mrem (millirem) per hour (26, 27). LLRW contains disposable filter cartridges, filter sludges, ion exchange resins, evaporator concentrates, contaminated clothing, and discarded tools (28).

2.1.1 Sources and Quantities of LLRW

Low-Level Radioactive Wastes are produced from fuel cycle, institutional research and development activities, medical and biological applications, and defense programs (27, 29). The fuel cycle accounted for 43% of LLRW produced annually from non-defense sources; the other sources (institutional, medical, and biological) for 47% to 49% of LLRW generated annually from non-defense sources (17, 29).

Commercial nuclear power plants are the largest generators of liquid LLRW from the nuclear fuel cycle. These wastes are classified as either boiling water reactor (BWR) wastes or pressurized water reactor (PWR) wastes. The important difference between these wastes is the absence of borates in the BWR wastes. The BWR and PWR wastes are

collected and segregated into one of the eight categories, namely, high purity waste, medium purity waste, chemical waste, detergent waste, spent resin, filter precoat, secondary system wastes, and miscellaneous waste. The sources for generation of these types of wastes are summarized in Table 1.

Recent estimates have indicated that total volume of LLRW generated from nuclear fuel cycle was $2.7 \times 10^5 \text{ m}^3$ (30, 31); of this volume, 80% was produced at light water reactor (LWR) power plants and 20% resulted from fabrication of fuel. While the volume of LLRW produced varied from plant to plant, $4.5 \times 10^{-5} \text{ m}^3$ of LLRW were produced per megawatt of power generated at LWR plants (32).

The total volume of LLRW generated from other sources was $1.8 \times 10^5 \text{ m}^3$. Anderson et al. (30) reported in 1978 that medical and biological sources constituted 80% of the LLRW ($2.2 \times 10^4 \text{ m}^3$) produced annually. Industry and medical facilities produced 47% of the total LLRW generated annually in Pennsylvania (17). The total LLRW produced from fuel cycle and other sources is referred to as commercial LLRW.

2.1.2 Characteristics of LLRW

The LLRW generated from the fuel cycle are classified into two categories, namely, wet wastes and dry wastes. The physical and chemical characteristics and the radiological properties of wet wastes generated from BWR, PWR, and Decontamination/Decommissioning Operations are presented in Tables 2 to 7. Since the precise composition of most of these wastes varied from plant to plant, the values listed in Tables 2 to 7 are typical values.

Table 1. Sources for Generation of Various Categories of Boiling Water Reactor (BWR) and Pressurized Water Reactor (PWR) Wastes (29)

Waste Category	Source
High purity waste	Equipment drainage (BWR)
Medium purity waste	Miscellaneous leakage and floor drainage (BWR)
Chemical waste	Ion exchanger regenerations, solutions and laboratory drains (BWR, PWR)
Detergent waste	Drainage from laundry machines and personnel decontamination stations (BWR, PWR)
Spent resin	Water slurry of depleted bead resin from ion exchangers (BWR, PWR)
Filter precoat	Water slurry of powdered resin, diatomaceous earth, etc., from precoat-type filters (BWR, PWR)
Secondary system wastes	Turbine building floor and equipment drainage, steam generator blowdown, condensate polisher regeneration solutions, etc. These wastes normally contain little or no radioactivity (PWR).
Miscellaneous waste	Equipment and floor drainage and miscellaneous leakage

Table 2. Typical Characteristics of Boiling Water Reactor (BWR) Wet Wastes (29)

Characteristic		Waste Category						
		High Purity Waste	Medium Purity Waste	Chemical Waste ^a	Detergent Waste	Bead Waste	Powdered Resin Filter Precoat	Diatomaceous Earth Filter Precoat
Temperature	(°C)	21 to 65	21	77	21 to 65	21	21	21
pH		6 to 8	6 to 8	6 to 8	7 to 9	-	-	-
Conductivity	(μmho/cm)	<10	10 to 1,000	>10,000	-	-	-	-
Density	(kg/m ³)	1,000	1,000	1,180	1,000	-	-	-
Activity ^b	(μCi/mL)	(0.88)(A+M)	(0.001)(A+M)	(40)(A+M)	See Table 4	1 to 10	0.1 to 500	0.05
Constituents:								
Water ^c	(wt %)	>99	>99	75	99	50	50	40
Oil	(ppm)	trace amts.	<1	-	-	-	-	-
Detergent	(wt %)	-	-	-	1	-	-	-
Suspended Solids	(ppm)	<10	10 - 100	1,000	1,000	-	5	10
Sodium Sulfate	(wt %)	-	-	22	-	-	-	-
Sodium Chloride	(wt %)	-	-	2	-	-	5	-
Bead Resin ^d	(wt %)	-	-	-	-	50	-	-
Powdered Resin ^e	(wt %)	-	-	-	-	-	40	-
Diatomaceous Earth	(wt %)	-	-	-	-	-	-	40

^aCharacteristics of chemical waste are after concentration by evaporation to solubility-limit of solution.

^bFor values of A and M, see Table 3.

^cFor bead resin and filter precoat type waste, water content listed is interstitial water only.

^dBead resin is a variable mix of anion and cation resin beads.

^ePowdered resin is a variable mix of powdered anion and cation resin beads.

Table 3. Typical Concentrations of Expected Radionuclides
in Reactor Coolant for Boiling Water Reactors
(BWRs) (29)

Nuclide	Reactor Coolant Fission Products A ($\mu\text{Ci/mL}$)	Nuclide	Reactor Coolant Activation Products M ($\mu\text{Ci/mL}$)
Sr-89	2.7×10^{-3}	P-32	2.0×10^{-5}
Sr-90	2.0×10^{-4}	Cr-51	5.0×10^{-4}
Zr-95	3.5×10^{-5}	Mn-54	4.0×10^{-5}
Nb-95	3.6×10^{-5}	Co-58	5.0×10^{-3}
Mo-99	2.0×10^{-2}	Co-60	5.0×10^{-4}
Ru-103	1.7×10^{-5}	Fe-59	8.0×10^{-5}
Ru-106	2.2×10^{-6}	Zn-65	2.0×10^{-6}
Ag-110m	6.0×10^{-5}		
Te-129m	3.4×10^{-5}		
Te-132	4.3×10^{-2}		
I-131	1.2×10^{-2}		
Cs-134	1.4×10^{-4}		
Cs-136	9.2×10^{-5}		
Cs-137	2.1×10^{-4}		
Ba-140	7.8×10^{-3}		
Ce-141	3.4×10^{-5}		
Ce-144	3.0×10^{-5}		
Pr-143	3.3×10^{-5}		
Np-239	2.1×10^{-1}		

Notes:

1. Only isotopes with half lives > 24 hr are listed.
2. Activity in main steam is 10^{-3} times activity in reactor coolant, except that, for halogen, activity in main steam is 10^{-2} times activity in reactor coolant.

Table 4. Typical Concentrations of Expected Radionuclides
in Laundry Wastes from Boiling Water Reactor (BWR)
and Pressurized Water Reactor (PWR) (29)

Nuclide	Activity ($\mu\text{Ci/mL}$)
Mn-58	1.6×10^{-6}
Co-58	6.4×10^{-6}
Co-60	1.4×10^{-5}
Zr-95	2.3×10^{-6}
Ru-103	2.3×10^{-7}
Ru-106	3.9×10^{-6}
Ag-110m	7.1×10^{-7}
I-131	9.7×10^{-7}
Cs-134	2.1×10^{-5}
Cs-137	3.9×10^{-5}
Ce-144	8.0×10^{-6}

Note: Only isotopes with half-lives > 24 hr are listed.

Table 5. Typical Characteristics of Pressurized Water Reactor (PWR) Wet Wastes (29)

Characteristic		Waste Category	
		Miscellaneous Chemical Waste ^a	Secondary Side Condensate Polisher Regenerative Waste ^a
Temperature	(°C)	77	77
pH		2.5 to 4.0	2.5 to 4.0
Boric Acid	(wt %)	12	-
Activity ^b	(μ Ci/mL)	(0.2)(R)	(4×10^4)(S)
Sodium Sulfate	(wt %)	-	14.9
Ammonium Sulfate	(wt %)	-	9.6
Sodium Chloride	(wt %)	-	2.0

^aCharacteristics of miscellaneous chemical waste and secondary side condensate polisher regenerative waste are after concentration by evaporation to solubility limit of solution.

^bSee Table 6 for R and S.

Note:

Characteristics of Detergent Wastes, Bead Resin, and Filter Precoat Backwash are the same as those of BWR (See Table 2).

Table 6. Typical Concentrations of Expected Radionuclides
in Reactor Coolant and Secondary Side for
Pressurized Water Reactors (PWRs) (29)

Nuclide	Reactor Coolant Fission and Activation R ($\mu\text{Ci/mL}$)	Secondary Side Fission and Activation S ($\mu\text{Ci/mL}$)
Cr-51	1.7×10^{-3}	2.2×10^{-7}
Mn-54	2.7×10^{-4}	5.8×10^{-8}
Fe-55	1.4×10^{-3}	1.8×10^{-7}
Co-58	1.4×10^{-2}	1.9×10^{-6}
Fe-59	8.8×10^{-4}	1.4×10^{-7}
Co-60	1.8×10^{-3}	2.4×10^{-7}
Sr-89	3.1×10^{-4}	5.4×10^{-8}
Sr-90	8.8×10^{-6}	1.3×10^{-9}
Y-91	5.6×10^{-5}	8.1×10^{-9}
I-131	2.4×10^{-1}	3.6×10^{-5}
Rb-86	7.7×10^{-5}	1.2×10^{-8}
Cs-134	2.2×10^{-2}	3.5×10^{-6}
Cs-136	1.2×10^{-2}	1.8×10^{-6}
Cs-137	1.6×10^{-2}	2.4×10^{-6}

Note: Only isotopes with half-lives > 24 hr are listed.

Table 7. Characteristics of Typical Decontamination Waste Solutions (29)

Characteristic		Component Decontamination Waste ^a	System Decontamination Waste ^b	Building Decontamination Waste ^c
Temperature	(°C)	21	21	21
pH		4	4	5
Conductivity	(µmho/cm)	>10 ⁴	>10 ⁴	-
Dow NS-1 ^d	(wt %)	7	7	-
Water	(wt %)	93	93	88
Iron Oxide	(ppm)	600 to 900	1200	-
Co-60	(ppm)	1.4 to 1.6	-	-
Radiac Wash ^e	(wt %)	-	-	5
Crud	(wt %)	-	-	5
Oil	(wt %)	-	-	2
Activity:				
I-131	(µCi/mL)	-	-	0.6
Ce-134	(µCi/mL)	-	-	0.4
Ce-136	(µCi/mL)	-	-	0.2
Ce-137	(µCi/mL)	-	-	1.4
La-140	(µCi/mL)	-	-	5.2
Ba-140	(µCi/mL)	-	-	3.6
Total Activity (µCi/mL)		1.2 to 2.2 ^f	0.48	-

^aData for component decontamination is based on decontamination of regeneration heat exchangers at Peach Bottom Units 2 and 3.

^bData for system decontamination is based on test loop set-up for decontamination of Dresden Unit 1 reactor coolant system.

^cData for building decontamination is based on decontamination of auxiliary building at Three Mile Island Unit 2.

^dDow NS-1 is a proprietary mixture of EDTA, inorganic salts, and organic inhibitors.

^eRadiac Wash is a mixture of EDTA, mild detergent and other proprietary additives.

^fTotal activity for component decontamination is based on 50/50 volume % mixture of decontamination solution and rinse water.

^gTotal activity for component decontamination is based on 30/70 volume % mixture of decontamination solution and rinse water.

2.1.3 Disposal of LLRW

Disposal of LLRW, by tradition, has been accomplished by shallow land burial. While other disposal methods including intermediate depth burial, disposal in engineered facilities, disposal in drilled holes, deep-well injection, disposal in hydrofractured strata, disposal in cavities, and sealed disposal have been both proposed and reviewed, shallow land burial has been the most utilized method for disposal of LLRW (33).

The total volume of commercial LLRW buried within the United States has increased rapidly since 1962; currently, more than 5.66×10^4 m³ of commercial LLRW are buried annually at licensed commercial sites (34). Defense operations have generated three times more LLRW than commercial operations. The volume of LLRW generated annually due to defense operations has leveled at 2.8×10^3 m³. The LLRW generated from defense operations are buried at 17 government-owned sites which are managed by the U.S. Department of Energy (DOE).

The shallow land burial of LLRW generated from commercial operations has been accomplished at six shallow-land burial sites, namely, Beatty, Nevada; Maxey Flats, Kentucky; West Valley, New York; Richland, Washington; Sheffield, Illinois; and Barnwell, South Carolina. Of these sites, Maxey Flats and West Valley sites have been closed due to migration of radionuclides caused by water management problems. The site at Sheffield, Illinois experienced lengthy delay with license renewal. Thus, only three sites at Beatty, Nevada; Richland, Washington; and Barnwell, South Carolina have been receiving LLRW for disposal (18). In addition, the site at Barnwell, South Carolina has

instituted a volume allotment program for receiving LLRW for disposal; this has caused a three-fold increase in transportation cost incurred by some Midwestern utilities due to their LLRW being shipped to Richland, Washington and Beatty, Nevada.

2.1.4 Problems Associated with Shallow-Land Burial of LLRW

The major problems identified at licensed-commercial sites for shallow-land burial are the migration of radionuclides at some sites and the gradual depletion of available volume for LLRW disposal due to site closure and increased generation of LLRW (1, 5, 6-16, 18, 35-51).

The problems associated with shallow-land burial are further compounded by the restrictions that are imposed on LLRW disposal by the Low-Level Radioactive Waste Act of 1980 (PL 96-573) from January 1986. This Act requires each state to be responsible for assuring adequate disposal for LLRW generated within its borders. Also, this Act allows states that have agreed to form a regional compact, the authority to exclude LLRW from outside their region. Six regional compacts of states, namely, Northwest, Central, Midwest, Southeast, Northeast, and Rocky Mountain have been formed. The status of low-level regional compacts is presented in Figure 1. These compacts have won legislative approval from their represented states. However, Congress has not approved any of the compacts (18).

Although the Nuclear Regulatory Commission (NRC) does not view storage as an alternative to disposal, one of the options for compacts without an operating site would be to provide interim storage capacity for their LLRW while developing disposal capacity. NRC has estimated that in the final environmental impact statement (FEIS) for 10 CFR 61

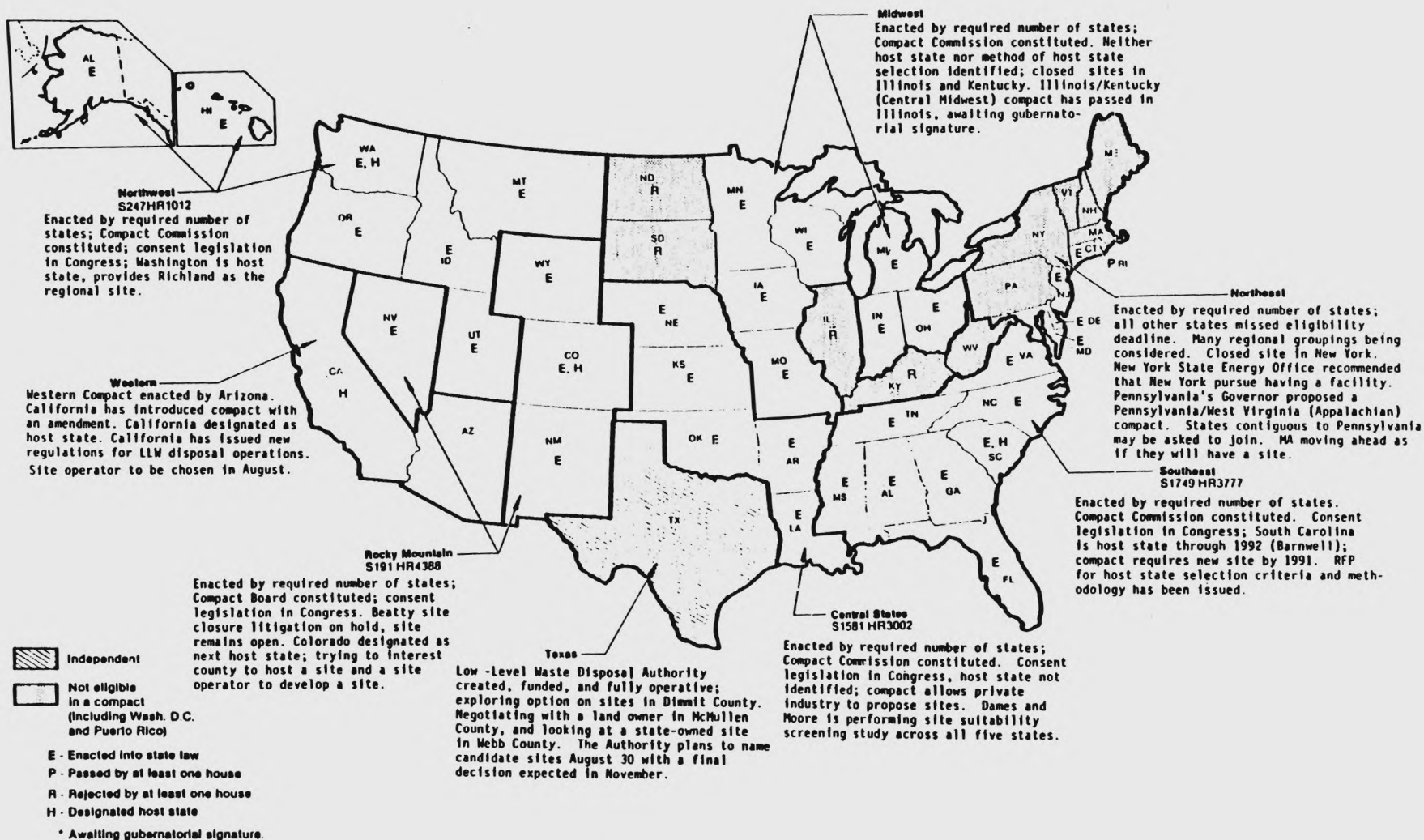


Figure 1. The Status of Regional Low-Level Waste Compacts (18).

licensing rule for land disposal of LLRW, site selection and characterization would take up to two years, and the licensing process would take one to two years. Thus, depending on the characteristics of the site and the complexity of the NRC review, the process could take two to four years which would obviously be extended in the event of a protracted hearing (18). Consequently, the interim storage of LLRW within regional compacts without an operating site may be inevitable; this would lead to an increase in the volume of LLRW to be disposed at a later date.

2.1.4.1 Suggested Remedial Actions for Reduction in Volume of LLRW. In order to reduce the volume requirement for LLRW disposal, the NRC has identified a need to develop a policy on regulating de minimus levels of radioactivity. Among the suggested alternatives to shallow land burial, under an exclusion in 10 CFR 20, was the regulated release of low-activity (de minimus levels) waste to uncontrolled disposal at a municipal sanitary landfill. The disposal of dry active waste (DAW) below de minimus levels within a landfill is one of the strategies being considered by utilities in regional compacts without an operating site (19, 20). Thus, codisposal of de minimus levels of LLRW and municipal refuse within landfills is being regarded as a viable strategy for reduction in the volume requirement for LLRW disposal.

2.2.1 Controlled Landfill

Sanitary landfills have become the most common method of municipal refuse disposal in the United States, not only because of tradition, but also because of their economic advantage over other methods of solid waste management. Based on the type of solid waste it

receives, a landfill can be identified as one of the three types of landfills: "Sanitary Landfill" intended for municipal refuse disposal, "Secure Landfill" intended for hazardous waste disposal, and "Controlled Landfill" intended for codisposal of municipal refuse and hazardous waste. A leachate and gas collection/treatment strategy forms the basis of a "Controlled Landfill" since the potential migration of either leachate or gas from the landfill would result in environmental impairment (52).

2.2.2 Municipal Refuse

The sources of municipal refuse are largely dependent on location and may be classified as residential, commercial, industrial, and agricultural. The type of municipal refuse produced from these sources usually include food wastes, rubbish, ash and residues, demolition and construction debris, treatment plant sludges, agricultural wastes, and hazardous wastes. Wastes originating from residential and commercial activities constitute municipal refuse. The composition of municipal refuse is presented in Table 8; of the various constituents, food and garden wastes represent the readily biodegradable fraction. The projected municipal refuse production in 1990, at 3.5 percent annual compound growth, will be 230 million metric tons (53).

2.2.3 Landfill Characterization

A "controlled landfill" is usually adjoined by a natural environmental setting and the potential impairment of this setting is inextricably linked with the potential migration of the transport phases, namely, leachate and gas, from the landfill. A proper

Table 8. Composition of Municipal Refuse Produced in the United States (Percent Dry Weight)

Component	Tchobanoglous, <u>et al.</u> , (1977)	Myer, <u>et al.</u> , (1979)	Chain, <u>et al.</u> , (1977)	U.S. EPA (1968)	Walsh and Kinman (1981)	Solid Waste Management (1972)
	(53)*	(54)*	(55)*	(56)*	(57)*	(58)*
Food Wastes	5.76	0.94	14.4	11.1	3.6	6.05
Garden Wastes	6.14	0.41	3.1	6.9	10.7	6.76
Paper	48.14	44.79	36.5	48.6	42.6	47.5
Cardboard	4.86	-	-	-	-	-
Plastics	3.71	-	2.8	-	-	0.86
Rubber	0.64	9.03	-	-	8.7	0.75
Leather	0.51	-	-	-	-	0.34
Textiles	2.30	3.08	0.7	0.7	-	0.68
Wood	2.04	0.49	1.8	2.1	2.6	2.42
Glass	9.98	7.61	6.8	8.3	12.2	7.41
Tin Cans	7.42	-	-	-	-	-
Non-Ferrous Metals	1.28	10.82	2.7	11.1	12.0	9.78
Ferrous Metals	2.43	-	11.8	11.1	-	-
Dirt, Ashes, and Brick	4.82	-	14.9	8.3	3.2	16.20

*Indicates reference number

understanding of the processes contributing to landfill stabilization is important, especially when the landfill is not "secured", since the variations in leachate and gas during landfill stabilization indicate the setting's vulnerability to change. A better understanding of landfill behavior and development of more efficient design, operation, and control procedures for "controlled landfills" have resulted from investigations on codisposal of municipal refuse and hazardous wastes within "controlled landfills" (23, 54, 59, 64).

All candidate landfill sites should be evaluated initially in terms of climatological events and their impacts on local hydrogeology. The factors affecting landfill stabilization are presented in Figures 2 and 3. As shown in Figure 2, the rainfall-derived infiltration constitutes the principal source of moisture for generation of leachate during the life of a landfill; thus, a reliable determination of areal rainfall and its opportunity to enter, persist within, and exit the landfill environment are of prime importance. Moreover, rainfall-derived infiltration will be affected by the landfill management strategies employed at the site, i.e., cover, lining, codisposal, irrigation/leachate recycle, and gas extraction procedures (60). The interrelationships between different parameters, within a landfill environment are shown in Figure 3.

2.2.4 Landfill Stabilization

Landfills receiving municipal refuse along with varying quantities of hazardous wastes proceed through a series of stabilization phases; the importance and the longevity of each phase are determined by the existing climatological conditions, operational

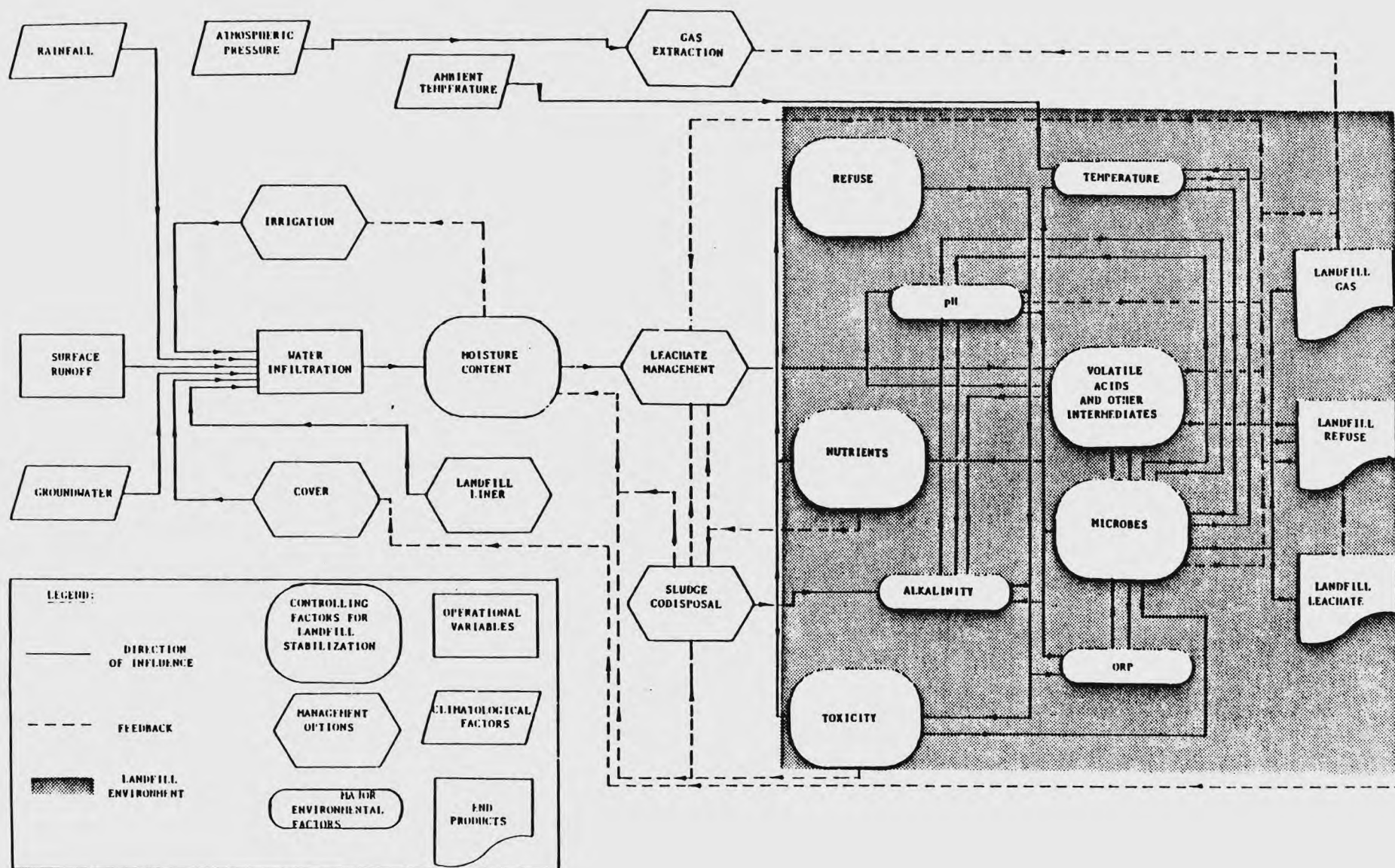


Figure 2. Factors and Management Techniques Influencing Leachate and Gas Production During Landfill Stabilization (60).

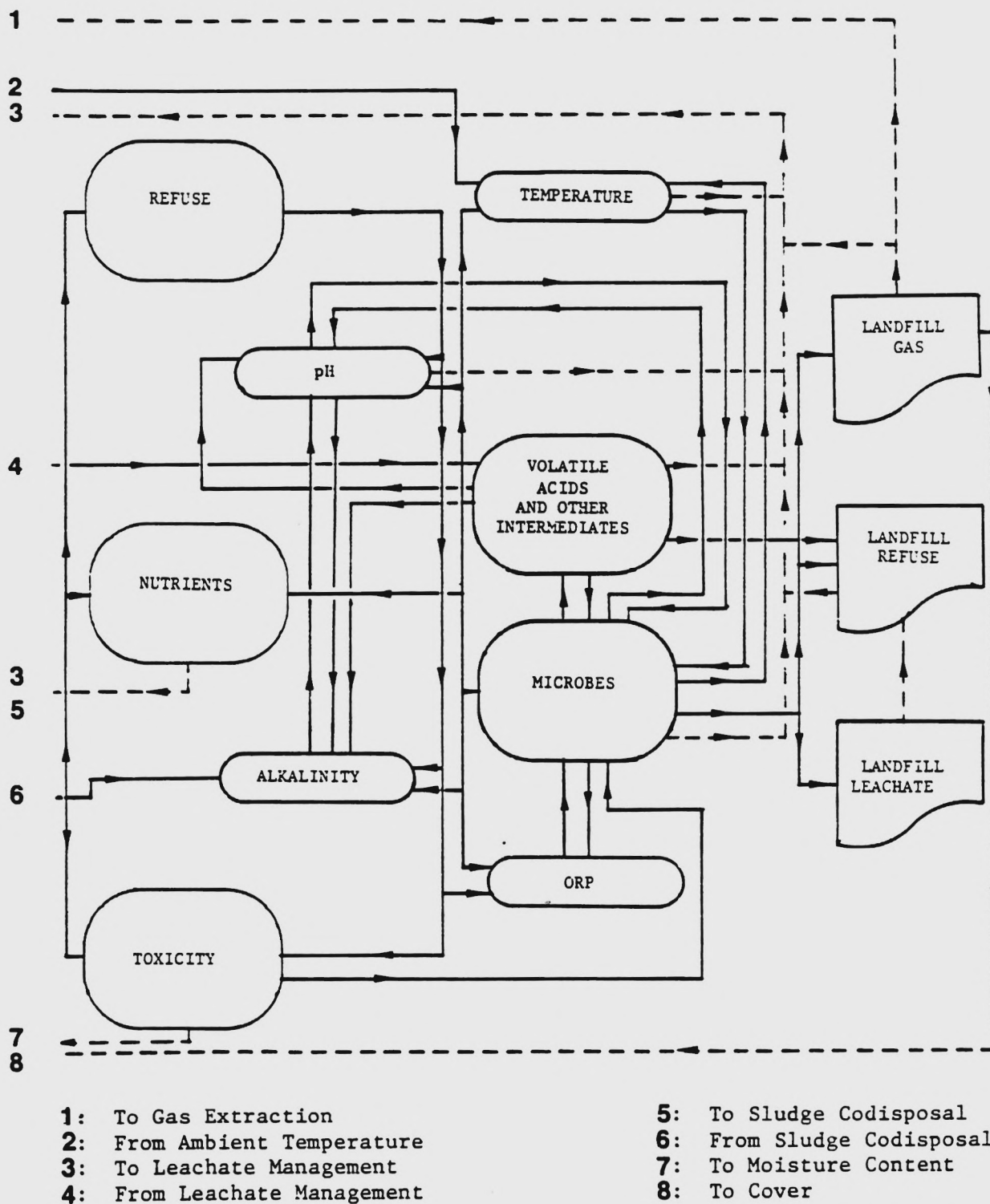


Figure 3. Interrelationships Between Different Parameters Within a Landfill Environment (60).

variables, management options and control factors operative or being applied externally or internally (52, 60). The concept of sequential behavior can be ascertained from results of studies on landfill stabilization (21, 23, 52, 59, 60). The choice of analysis of certain environmental parameters is governed by the fact that a landfill during its active life is analogous to a batch anaerobic digester with limited inputs or outputs and has a functional retention time in years instead of days. Thus, the existence of discrete and sequential phases of refuse conversion and stabilization within a landfill receiving municipal refuse can be illustrated in terms of principal events and parametric changes by the following five stabilization phases: Initial Adjustment Phase, Transition Phase, Acid Formation Phase, Methane Fermentation Phase, and Final Maturation Phase (52, 60).

Phase I: Initial Adjustment--

- Initial placement of waste and preliminary accumulation of moisture occur.
- Initial subsidence and closure of each landfill area.
- Changes in environmental parameters are first detected to reflect the onset of stabilization processes, these processes trend in a logical fashion.

Phase II: Transition--

- "Field capacity" is exceeded and leachate is formed.
- A transition from initial aerobic to anaerobic microbial stabilization occurs.
- The primary electron acceptor shifts from oxygen to nitrates and sulfates with the displacement of oxygen by

carbon dioxide in the gas.

- A trend toward reducing conditions is established.
- Measurable intermediates such as the volatile fatty acids first appear in the leachate.

Phase III: Acid Formation--

- Intermediary volatile fatty acids become predominant with the continuing hydrolysis and fermentation of waste and leachate constituents.
- A precipitous decrease in pH occurs with a concomitant mobilization and possible complexation of metal species.
- Nutrients such as nitrogen and phosphorus are released and utilized in support of the growth of biomass commensurate with the prevailing substrate conversion rates.
- Hydrogen may be detected and affect the nature and type of intermediary products being formed.

Phase IV: Methane Formation--

- Intermediary products appearing during the acid formation phases are converted to methane and excess carbon dioxide.
- The pH returns from a buffer level controlled by the volatile fatty acids to one characteristic of the bicarbonate buffering system.
- Oxidation-reduction potentials are at their lowest values.
- Nutrients continue to be consumed.
- Complexation and precipitation of metal species proceed.
- Leachate organic strength is dramatically decreased in

correspondence with increases in gas production.

Phase V: Final Maturation--

- Relative dormancy following active biological stabilization of the readily available organic constituents in the waste and leachate.
- Nutrients may be limiting.
- Measurable gas production all but ceases.
- Natural environmental conditions become reinstated.
- Oxygen and oxidized species may slowly reappear with a corresponding increase in oxidation-reduction potential.
- More microbially resistant organic materials may slowly be converted with the possible production of humic-like substances capable of complexing with and re-mobilizing heavy metals.

All of the major events selected to describe and separate these landfill stabilization phases are encountered at one time or another in landfills containing municipal refuse, provided that the associated microbially mediated processes have been augmented by a sufficiency of moisture and nutrients, and are not being exposed to the inhibited influences of toxic materials (52, 60).

Since the landfill stabilization phases usually overlap during the active life of a landfill, they are often viewed in a collective fashion that trends to obscure their discrete and sequential existence; this limits the recognition and the understanding of the progress of events that result in maturation of stabilization processes. The time required for the maturation of stabilization processes may be described

in terms of landfill "age". In fact, a landfill, during its active life, does not have a single "age" but rather a collection of landfill "ages" with different "ages" being associated with microenvironments within the landfill complex due to their respective progress towards stabilization (60). Moreover, the rate of progress towards stabilization depends upon the physical, chemical, and microbial conditions within the microenvironments. For example, high compaction may restrict the movement of moisture and nutrients throughout the mass of municipal refuse, acidic conditions established during acid formation phase may prevent the onset of active methane fermentation, and the presence of toxic substances may induce microbial inhibition within microenvironments of the landfill complex (52, 60).

2.2.5 Indicator Parameters Descriptive of Landfill Stabilization Phases

There are certain indicator parameters or indices capable of being used to detect and describe the presence, the intensity, and the longevity of each phase of landfill stabilization (60). Many of these parameters apply to the analysis of leachate; thus, they are important after leachate production has commenced. In addition, their applications and interrelationships within an overall landfill perspective are reflected by the nature of analyses, namely, physical, chemical, or biological. For example, pH and ORP are physical parameters that indicate acid-base and oxidation-reduction conditions, respectively. COD and BOD₅ are chemical and biological parameters, respectively, but are indicative of relative biodegradability of the leachate produced from the landfill. Other critical parameters, that are used to identify and assess the intensity and the longevity of each solubilization

phase, include alkalinity (buffer capacity), conductivity (ionic strength/activity of ions), heavy metals (potential inhibition of microbial-mediated processes), individual and total volatile acids (breakdown of polymers to monomers, availability of substrate to methane formers), and chloride (dilution, mixing, and washout of organic and inorganic constituents present in leachate).

Ranges in intensity or concentration of indicator parameters will vary throughout the stabilization phases due to the principal function of each stabilization phase and dilution resulting from continuous infiltration of moisture into the landfill. Table 9 provides a compilation of data on the general ranges of intensity and concentration of indicator parameters throughout the landfill stabilization phases (25). An examination of these data indicates some overlap in the intensity or concentration of indicator parameters, particularly during the acid formation and the methane fermentation phases.

The processes contributing to landfill stabilization are most pronounced during the acid formation and the methane fermentation phases. The combined duration of these phases is reduced with leachate recycle due to accelerated stabilization (21-23, 59, 64, 109). During the contracted time interval, however, the intensity and the concentration of indicator parameters are magnified. Thus, the time span during which environmental impairment of the natural setting encompassing the landfill can occur is reduced with leachate recycle.

2.2.6 Metal Solubility in Landfill Leachates

The solubility of heavy metals in landfill leachates is an important determinant in the performance of an anaerobic biological

Table 9. Leachate Concentrations and General Significance During Landfill Stabilization (25).

PARAMETER	PHASES OF STABILIZATION				REPORTED RANGE
	TRANSITION	ACID FORMATION	METHANE FERMENTATION	FINAL MATURATION	
Biochemical Oxygen Demand (BOD ₅), mg/l	Low to high (100-10,900); Low to high biodegradability	Moderate to high (1000-57,700); Moderate to high biodegradability	High to low (3400-600); High to low biodegradability	Low (4-120); Low biodegradability	4-57,700
Chemical Oxygen Demand (COD), mg/l	Low to high (480-18,600); Low to high oxidation potential	Moderate to high (1500-71,700); Moderate to high oxidation potential	High to low (9760-580); High to low oxidation potential	Low (31-900); Low oxidation potential	31-71,700
Total Organic Carbon (TOC), mg/l	Low to moderate (100-3000); Low to moderate organic carbon content	Low to high (500-27,700); Low to high organic carbon content	Moderate to low (2230-300); Moderate to low organic carbon content	Low (70-260); Low organic carbon content	70-27,700
Total Volatile Acids (TVA), mg/l Acetic	Absent to low (0-500)	Low to high (3000-18,800); Volatile acid formation	High to low (4000-250); Volatile acid conversion to methane	Absent to low (0-50)	0-18,800
BOD ₅ /COD	Low to high (0.23-0.87); Low to high substrate biodegradability	Moderate to high (0.4-0.8); Moderate to high substrate biodegradability	Moderate to low (0.64-0.17); Moderate to low substrate biodegradability	Low (0.02-0.13); Low substrate biodegradability	0.02-0.87
COD/TOC	High (4.3-4.8); Low oxidation state of organics	Moderate to high (2.1-3.4); Moderate to low oxidation state of organics	Moderate to low (3.0-2.0); Moderate to high oxidation state of organics	Low (0.4-2.0); High oxidation state of organics	0.44-4.8
Total Kjeldahl Nitrogen (TKN), mg/l	Moderate to high (180-860)	Low to high (14-1970); Leaching and assimilation of nitrogenous compounds	Low (25-82); Assimilation of nitrogenous compounds	Low to moderate (7-490)	7-1970
Ammonia Nitrogen (NH ₃ -N), mg/l	Moderate (120-225)	Low to high (2-1030); Production and biological uptake	High to low (430-6); Biological uptake	High to low (430-6); Biological uptake and release	2-1030

Table 9. Continued...

PARAMETER	PHASES OF STABILIZATION				REPORTED RANGE
	TRANSITION	ACID FORMATION	METHANE FERMENTATION	FINAL MATURATION	
Nitrate Nitrogen (NO ₃ ⁻ -N), mg/l	Low to high (0.1-51); Oxidation of ammonia	Moderate to low (19-<0.05); Nitrate reduction	Absent	Low (0.5-0.6)	<0.05-51
NH ₃ -N/TKN	Low to high (0.1-0.91)	Low to high (0.02-0.98); Production and utilization	Low to high (0.1-0.84); Production and utilization	Moderate to high (0.54-0.97); Bio- logical uptake and release	0.02-0.98
Total Phosphate (PO ₄ ⁼ -P), mg/l	Low (0.6-1.7)	High to low (120-0.16); Bio- logical assimilation and complexation	Moderate to low (14- 0.7); Biological assimilation and com- plexation	Low to moderate (0.16-14)	0.16-120
Total Coliform, $\frac{\text{CFU}}{100\text{ml}}$	Moderate to low (10 ⁵ -10 ⁰)	Moderate to low (10 ⁵ -10 ⁰)	Absent	Absent	10 ⁰ -10 ⁵
Fecal Coliform, $\frac{\text{CFU}}{100\text{ml}}$	Moderate to low (10 ⁵ -10 ⁰)	Moderate to low (10 ⁵ -10 ⁰)	Absent	Absent	10 ⁰ -10 ⁵
Fecal Streptococci, $\frac{\text{CFU}}{100\text{ml}}$	Moderate to low (10 ⁶ -10 ⁰)	Moderate to low (10 ⁶ -10 ⁰)	Absent	Absent	10 ⁰ -10 ⁶
Viruses, PFU/100ml	-	Absent	Absent	Absent	-
pH	Neutral (6-7)	Neutral to low (7.7-4.7); Volatile acid formation and inhibition	Neutral to high (6.3- 8.8); Volatile acid conversion to methane	Neutral to high (7.1-8.8); Absence of volatile acids	4.7-8.8
Oxidation-Reduction Potential (ORP), mV E _c	High (+40); Incipient aerobic conditions	High to low (+80 to -240); Facultative and anaerobic conditions	Low (-73 to -240); Anaerobic methane fermentation	High (+97 to +163); High oxidation state of organics	-240 to +163

Table 9. Continued...

PARAMETER	PHASES OF STABILIZATION				REPORTED RANGE
	TRANSITION	ACID FORMATION	METHANE FERMENTATION	FINAL MATURATION	
Conductivity, $\mu\text{mhos/cm}$	Low (2450-3310); Anion and cation leaching	Low to high (1600-17,100); Volatile acid formation, anion and cation leaching and precipitation	Moderate to low (7700-2900); Conversion of volatile acids to methane, metal precipitation	Low (1400-4500)	1400-17,100
Solids (TS), mg/l	Low (2450-2960); Anion and cation leaching	Low to high (4120-55,300); Volatile acid formation, anion and cation leaching	Moderate (6410-2090); Conversion of volatile acids to methane, metal precipitation and filtration	Low (1460-4640)	1460-55,300
Alkalinity, mg/l as CaCO_3	Low (200-2050)	High to low (9650-140); Volatile acid formation and neutralization	Low to moderate (760-5050); Volatile acid conversion to methane	Low to moderate (200-3520)	140-9650
Chloride (Cl^-), mg/l	Low to high (30-5000); Leaching and washout	Low to high (30-5000); Leaching and washout	Low to high (30-5000); Leaching and washout	Low to high (30-5000)	30-5000
Sulfate ($\text{SO}_4^{=}$), mg/l	Low to moderate (10-458); Leaching and oxidation of sulfur compounds	High to low (3240-10); reduction of sulfate	Absent	Low (5-40)	5-3240
Sulfide ($\text{S}^{=}$), mg/l	Absent	High to low (818-1.6); Reduction of sulfate and metal precipitation	Low (0.9); Metal precipitation	Absent to low (0-0.9)	0.9-818
Cadmium (Cd), mg/l	Low (<0.005-0.01)	Low to moderate (<0.005-0.39); Inhibition and precipitation	Low (<0.005-0.1); Precipitation	Low (0.004)	<0.005-0.39
Calcium (Ca), mg/l	Low (190-490)	Low to high (70-3900); Leaching and precipitation	Moderate to low (490-76); Precipitation	Low (76-254)	70-3900

Table 9. Continued...

PARAMETER	PHASES OF STABILIZATION				REPORTED RANGE
	TRANSITION	ACID FORMATION	METHANE FERMENTATION	FINAL MATURATION	
Chromium (Cr), mg/l	Low (0.023-0.28)	High to low (18-0.06); Inhibition and precipitation	Low (0.05); Precipitation	Low (0.05)	0.02-18
Copper (Cu), mg/l	Low (0.085-0.39)	High to low (2.2-0.005); Leaching and precipitation	Low (0.03-0.18); Precipitation	Low (0.02-0.56)	0.005-2.2
Iron (Fe), mg/l	Low to moderate (68-312)	High to low (2200-90); Leaching and precipitation	Moderate to low (336-115); Precipitation	Low (4-20)	4-2200
Lead (Pb), mg/l	Low (0.001-0.004)	High to low (1.44-0.01); Inhibition and precipitation	Low (0.01-0.1); Precipitation	Low (0.01-0.1)	0.001-1.44
Magnesium (Mg), mg/l	Low (66-96)	Low to high (3-1140); Leaching	Moderate to low (505-81)	Low (81-190)	3-1140
Manganese (Mn), mg/l	Low (0.6)	High to low (41-0.6); Leaching and precipitation	Low (0.6); Precipitation	Low (0.6)	0.6-41
Nickel (Ni), mg/l	Low to moderate (0.02-1.55)	High to low (79-0.03); Inhibition and precipitation	Low (0.1-1.0); Precipitation	Low (0.07)	0.02-79
Potassium (K), mg/l	Low to high (35-2300)	Low to high (35-2300); Leaching and washout	Low to high (35-2300); Leaching and washout	Low to high (35-2300)	35-2300
Sodium (Na), mg/l	Low to high (20-7600)	Low to high (20-7600); Leaching and washout	Low to high (20-7600); Leaching and washout	Low to high (20-7600)	20-7600
Zinc (Zn), mg/l	Low to moderate (0.06-21)	High to Low (220-0.65); Inhibition and precipitation	Low (0.4-6); Precipitation	Low (0.4)	0.06-220

system since the degree of toxicity of a metal to microorganisms, that contributes to microbial inhibition, is dependent on the amount of metal available to penetrate its cell wall (56). From the point of view of codisposal of LLRW and municipal refuse within controlled landfills, the solubility of LLRW in leachate is important since the presence of LLRW in the liquid transport phase (leachate) within a landfill increases the potential of LLRW migration in the event of a liner failure. The factors controlling metal solubility in leachates include the presence of potential precipitant species such as hydroxide, carbonate, and sulfide, the presence of complexing ligands (humic substances) that will tend to enhance metal solubility, and pH and ORP, due to their impact on the dissociation of species.

Sulfide, even at low concentrations, is a powerful precipitant and will precipitate most heavy metals, with the exception of chromium (59), as metal sulfides. Sulfide is produced under reducing conditions by microbially-mediated reductions of sulfates and other sulfur containing inorganic compounds (62). Sulfide in solution dissociates forming three species: H_2S , HS^- , and S^{2-} and the distribution of these species is pH dependent. Equilibrium concentrations of soluble sulfide up to 200 mg/L as S^{2-} did not exert any significant toxic effects on anaerobic treatment (62, 63).

The distribution of carbonate species (H_2CO_3 , HCO_3^- , and CO_3^{2-}) in leachate is pH dependent. Carbonates and hydroxy carbonates have the potential to control the solubility of metals that are incapable of forming metal sulfides such as strontium (64). Also, carbonates and hydroxy carbonates may control the solubility of heavy metals in

absence of sulfide. Hydroxide acts as another precipitant for heavy metals in leachate. When hydroxide and sulfide are simultaneously present in leachate, the solubility is usually controlled by sulfide, except for trivalent chromium (59).

In complicated matrices such as leachate produced from a landfill, complexing agents both inorganic and organic in nature are present along with the potential precipitant species like sulfides, carbonates, and hydroxides. The occurrence of humic substances in landfill leachate has been reported by Chian and DeWalle (78), Raveh and Arnimelech (79) and Fuller and Artiole (80). Humic substances are complex polymers that possess both carboxyl and phenolic hydroxyl acidic functional groups. These functional groups are involved in metal complexation reactions (73).

The precipitation of heavy metals present in leachate by the potential-precipitant species, viz., sulfides, carbonates, and hydroxides, is counteracted by the formation of soluble metal-humate complexes; this leads to an increased solubilization of heavy metals in leachate. Pohland et al. (59, 64) reported the presence of aromatic hydroxyl as a mobilizing ligand for heavy metals in leachate from landfills receiving municipal refuse and varying quantities of hazardous wastes (metal sludges); the solubility of zinc and nickel was enhanced by aromatic hydroxyl even in the presence of sulfides, carbonates, and hydroxides. Heavy metals (copper, cadmium, nickel, zinc, manganese, lead, and iron) are known to form complexes with humic acid (67-77). Batch experiments on the extent to which heavy metal ions, namely, copper, lead, zinc, and cadmium are removed from aqueous solutions by humic acid revealed that

metal-humate complexes are usually formed within five days (67, 70); at equilibrium, humic acids enhanced the solubility of heavy metals even in presence of sulfide (67, 74).

2.2.7 Metal Speciation by Chemical Modeling

If the medium under study is assumed to be in equilibrium, the concentration of each species of a given component (metal or ligand) can be calculated on the basis of thermodynamics, provided that the following are known: the total concentration of all the components; and the equilibrium constants of all the reactions that occur between the various species under the conditions (pressure, temperature, and ionic strength) of the medium being studied.

This approach has the considerable advantage of requiring only the determination of total concentrations of components. The model, however, suffers from limitations mentioned below:

- i) The equilibrium constants of all possible reactions should be known. Since little information is available on mixed complexes, their contribution to formation of complexes cannot be accurately assessed. Likewise, little is known about the adsorption of free ions, ligands and complexes on solids (65).
- ii) Since, the approach is based on thermodynamic considerations, the medium is assumed to be in equilibrium. Kinetic aspects, however, are often important for reactions occurring with polymeric species, many of which are metastable compounds. Kinetics also play a major role in precipitation reactions and for processes involving mass

transfer at interfaces.

Even with these limitations, this approach can be used to describe the potential formation, at equilibrium, of metal-ligand precipitates and soluble complexes within the medium. Thus, the forms in which heavy metals may exist in the leachate and landfill environment during different phases of landfill stabilization, can be estimated.

A computer program for the calculation of chemical equilibrium composition of aqueous systems (MINEQL) developed by Westall et al. (66) was modified, then used to assess the potential formation of metal-ligand precipitates and soluble complexes for the set of metals and ligands that are summarized in Table 10.

The equilibrium composition of metal complexes in leachates (ionic strength = 0.03 M) at pH values of 5 and 7, are presented in Figures 4-7. As shown, at both pH values sulfide was the most powerful precipitant for heavy metals (Fe^{2+} , Zn^{2+} , Ni^{2+} , and Co^{2+}) in the data set with the exception of trivalent chromium and manganese. The solubility of trivalent chromium was controlled by hydroxide, whereas the solubility of managanese was controlled by acetate at a pH value of 5.00 and by carbonates at a pH value of 7.00 (in absence of acetate). The solubility of cerium was controlled by acetate at a pH value of 5.00. In absence of acetate, however, at a pH value of 7.00 cerium was present as Ce^{3+} . Sodium and potassium were present as Na^+ and K^+ , respectively. Calcium and magnesium formed complexes with acetate at a pH value of 5.00, whereas at a pH value of 7.00 in the absence of acetate these metals formed complexes with carbonates.

Table 10. Data Set Used to Study Metal Speciation
by Chemical Modelling (MINEQL)

Component	Concentration* M
Metal:	
Ca ²⁺	1.0 x 10 ⁻²
Mg ²⁺	1.0 x 10 ⁻²
Sr ²⁺	1.0 x 10 ⁻⁵
K ⁺	2.0 x 10 ⁻²
Na ⁺	2.0 x 10 ⁻²
Fe ²⁺	1.0 x 10 ⁻²
Mn ²⁺	5.0 x 10 ⁻⁴
Zn ²⁺	1.0 x 10 ⁻³
Ni ²⁺	5.0 x 10 ⁻⁴
Co ²⁺	2.0 x 10 ⁻⁴
Cr ³⁺	1.0 x 10 ⁻⁴
Ce ³⁺	7.0 x 10 ⁻⁶
Ligand:	
CO ₃ ²⁻	2.0 x 10 ⁻² , 1.0 x 10 ⁻²
S ²⁻	1.0 x 10 ⁻²
Acetate	5.0 x 10 ⁻¹ , 1.0 x 10 ⁻¹ , None

*Electroneutrality was ignored.

Note: The stability constants for formation of complexes and solids from metals and ligands are presented in Morel, Francois M. M., "Principles of Aquatic Chemistry," A Wiley-Interscience Publication New-York, 1983.

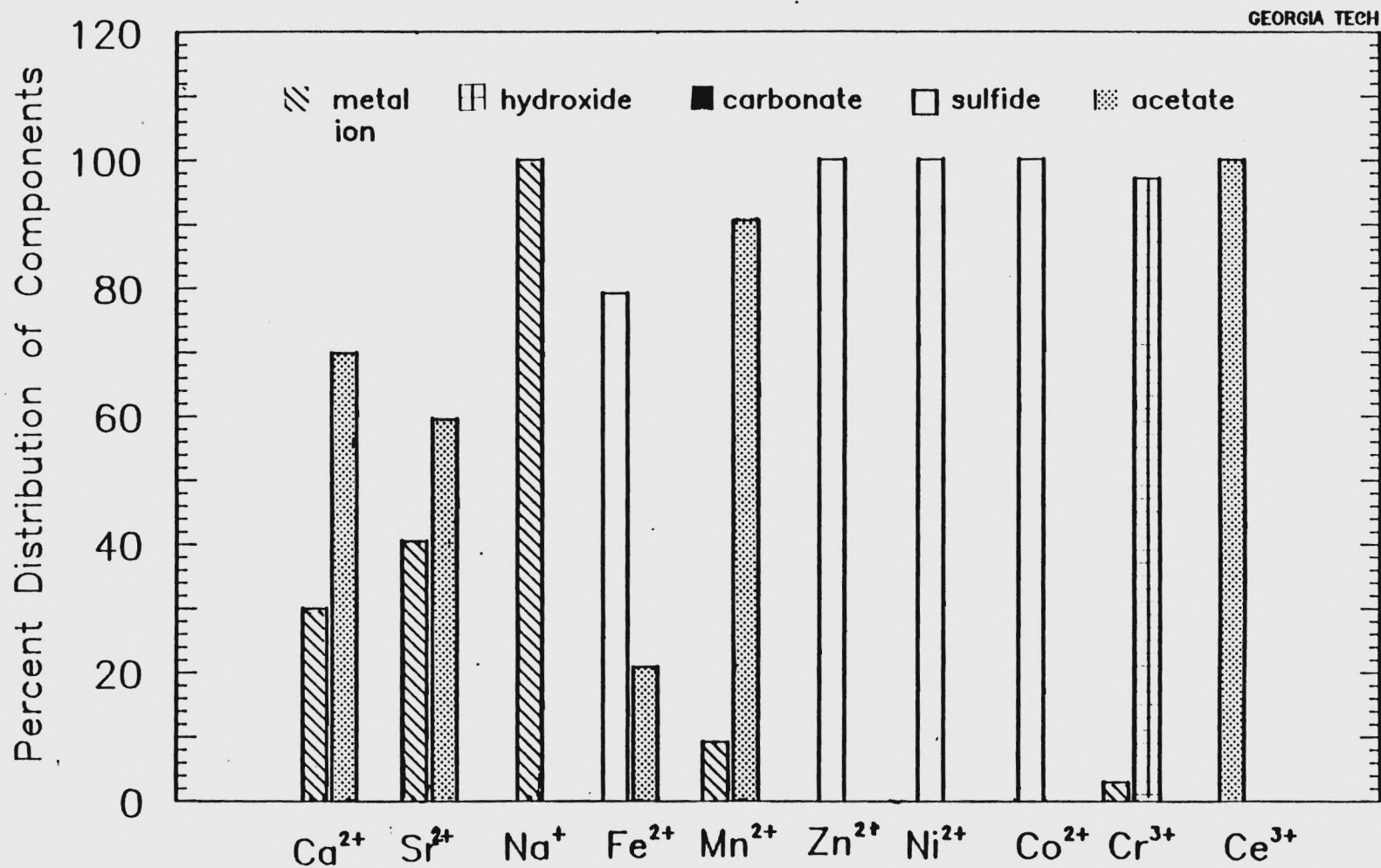


Figure 4. Speciation of Metals by Chemical Modeling (MINEQL)
(Carbonate=.01M,Acetate=.5M,Sulfide=.01M, pH=5.00).

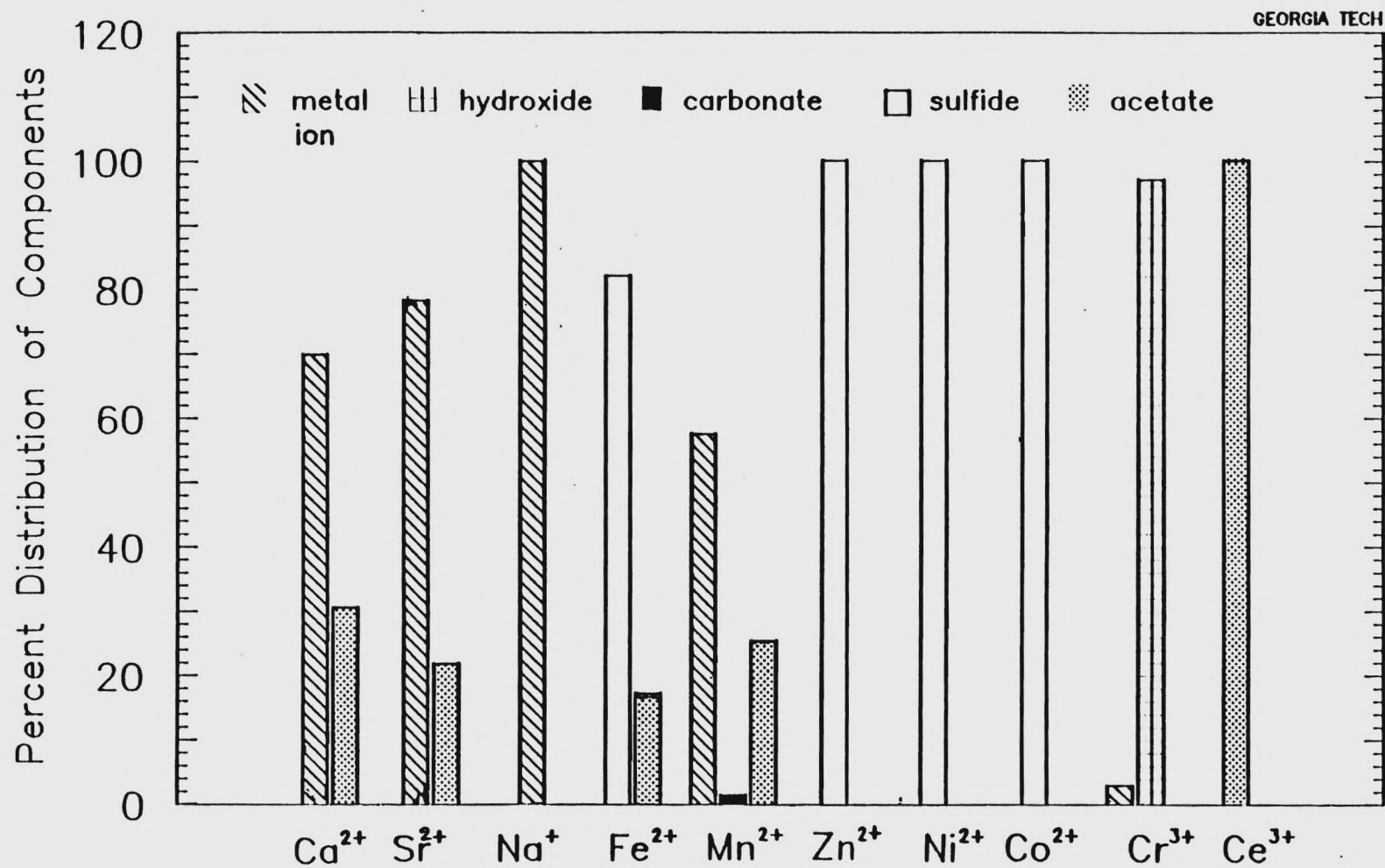


Figure 5. Speciation of Metals by Chemical Modeling (MINEQL)
 (Carbonate=.01M,Acetate=.1M,Sulfide=.01M, pH=5.00).

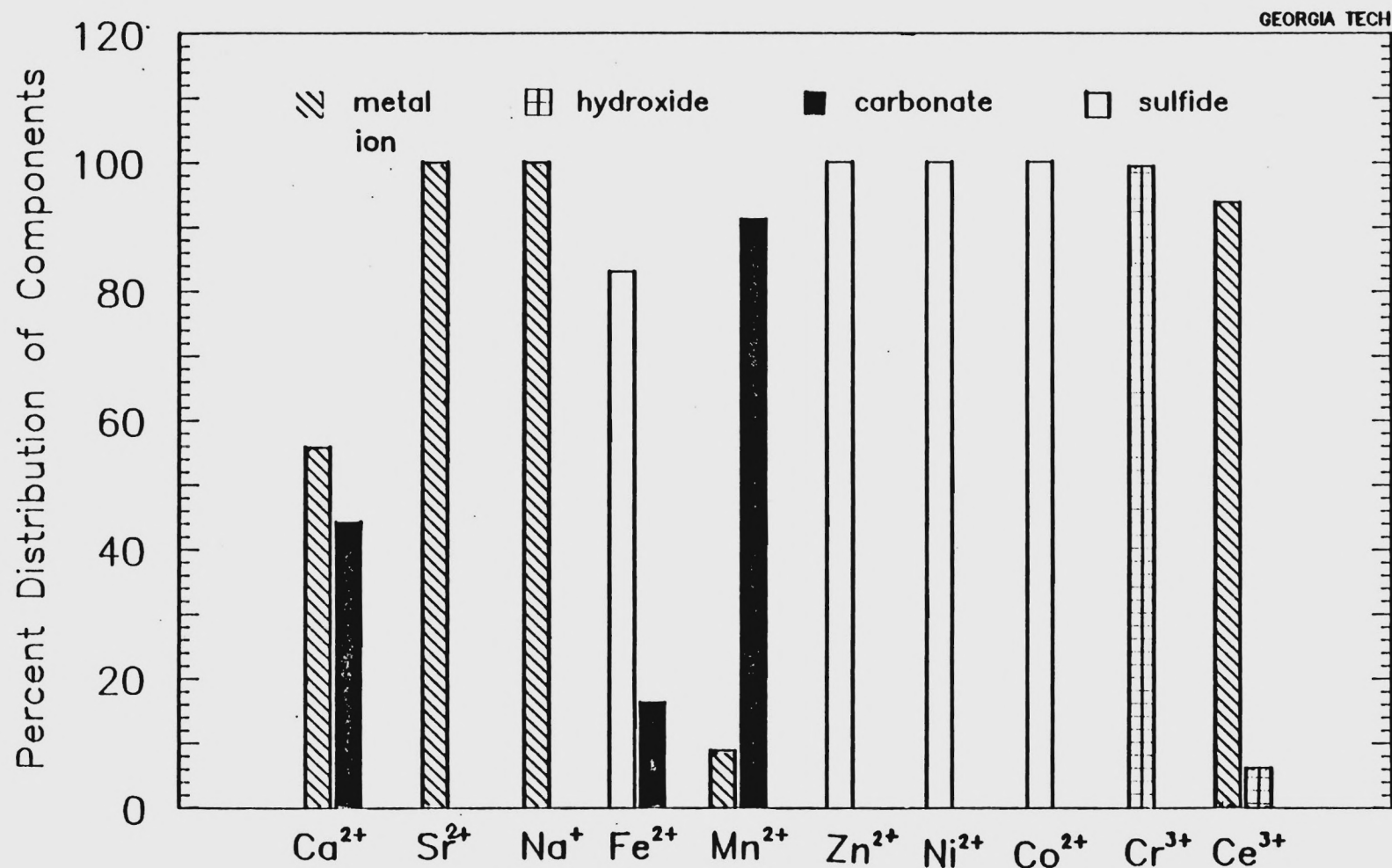


Figure 6. Speciation of Metals by Chemical Modeling (MINEQL)
(Carbonate=.01M,Sulfide=.01M, pH=7.00).

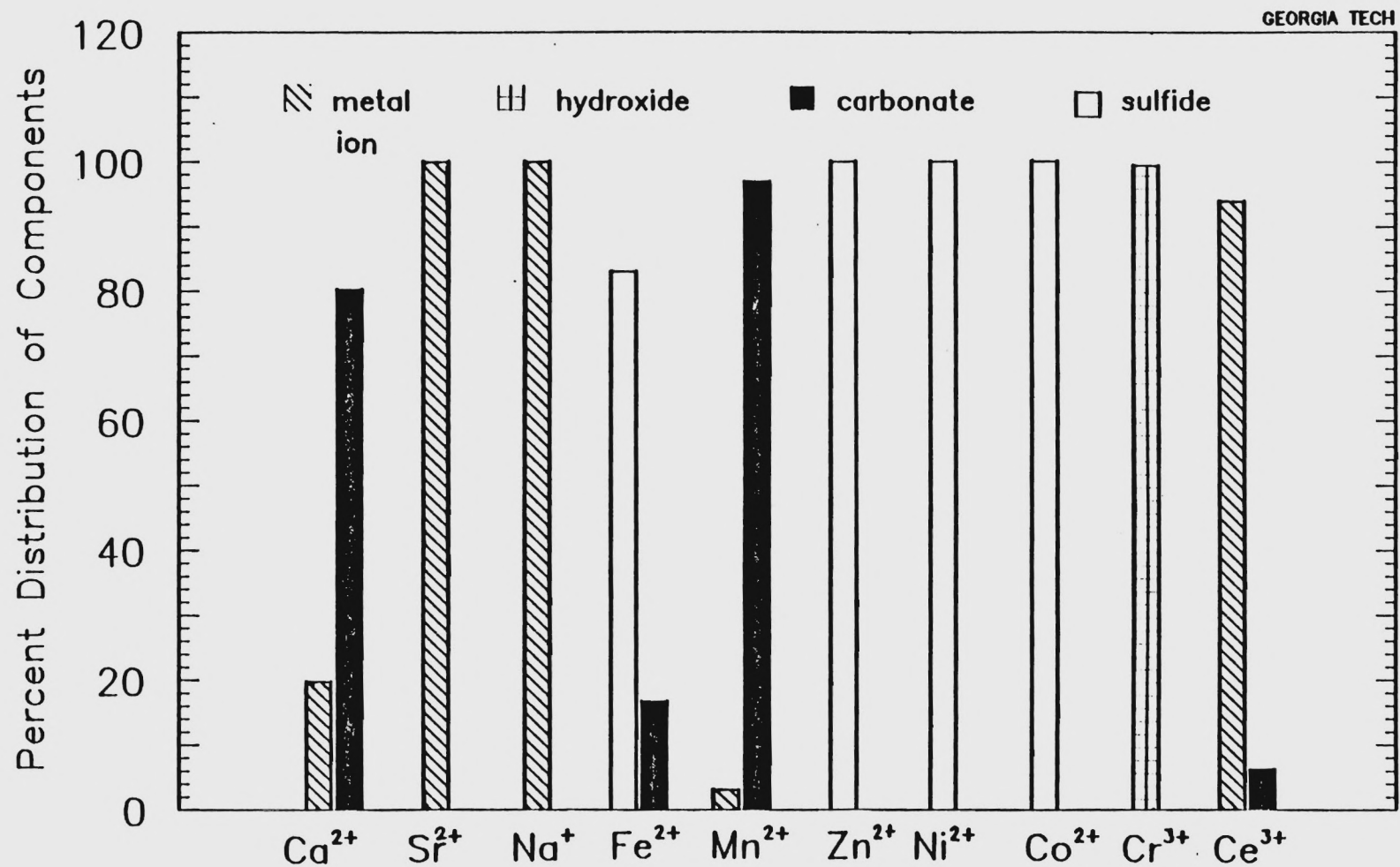


Figure 7. Speciation of Metals by Chemical Modeling (MINEQL)
(Carbonate=.02M,Sulfide=.01M, pH=7.00).

2.2.8 Complexation of Radionuclides with Humic Acid

The codisposal of LLRW and municipal refuse within laboratory-scale lysimeters, with and without leachate recycle, was investigated by Chang (21). One of the radionuclides (Co-58) utilized to simulate LLRW, exhibited a two-fold increase in leachate during the later phases of landfill stabilization. Chang (21) speculated that the increase in levels of Co-58 in leachate was probably due to the formation of soluble complexes of cobalt with humic substances present in leachate.

Swanson (81) reported enhanced mobility of Ni-63 and Pu-239 due to formation of complexes with organic ligands (EDTA, NTA, picolinate, citrate, and oxalate). Based on their studies on complexation of radionuclides with organic chelating agents present in nuclear waste, Toste et al., (5) have identified a need for studies on complexation of radionuclides, that constitute LLRW, with naturally-occurring organics, such as humic acids.

CHAPTER III

MATERIALS AND METHODS

3.1 Construction of Pilot-Scale Landfill Cells

The pilot-scale landfills were simulated in two cells 3.05-m (10 ft) square and 4.28-m (14-ft) deep. The cell walls were constructed with reinforced concrete blocks and installed on sloping reinforced concrete slabs. These cells were originally constructed and utilized for studies on "Controlled Landfill Stabilization by Leachate Recycle" (23). To prevent moisture infiltration from the adjoining soils, the inner faces of the cell walls were coated with successive coats of a foundation sealant and an epoxy based paint. Since containment of the radionuclides which were codisposed with shredded municipal refuse within the pilot-scale landfills was necessary, both cells were lined with 0.76-mm (30-mil) high density polyethylene liners (HDA Gundline Liner, Gundline Lining Systems, Inc., Houston, TX). The liner for each cell was assembled on-site, then placed above the underdrain network installed in a 15.2-cm (6-in) sand layer to provide continuous monitoring for liner integrity. Each underdrain network consisted of a 5.08-cm (2-in) PVC main with three 5.08-cm (2-in) laterals extending on either sides; these pipes were perforated with 0.95-cm (3/8-in) holes drilled 15.24 cm (6 in) on center. The mains extended through the front wall of each cell to a control sump. One of the cells (Cell A) was designed for single pass operation, while the other (Cell B) was

designed for leachate recycle.

Each cell was equipped with appurtenances for collecting leachate. The collection networks, similar in arrangement to the underdrain networks, were placed just above the liner at the bottom of each cell in a 15.24-cm (6-in) layer of sand. The collection network main extended through the liner and the front wall of each cell to two collection sumps. These collection sumps along with the collection sump for underdrain network were 208-L (55-gal) polyethylene containers; they were placed in a fiber glass coated plywood tank 2.43 m (8 ft) x 0.91 m (3 ft) x 1.52 m (5ft) (length x width x height). A submersible pump with epoxy coated housing and polyethylene impeller was utilized to recycle the leachate collected from Cell B. The leachate recycle network was constructed with a 5.08-cm (2-in) diameter Schedule 40 PVC pipe and compatible PVC fittings. A leachate distribution network, similar in arrangement to the underdrain network, was installed in a 0.3-m (1-ft) bed of gravel above the shredded municipal refuse placed in each cell. Figure 8 presents a sectional view of Cell B.

The cells were tested for leaks by filling them with water and noting the change in water levels after 24 hours. An insignificant change in the water level in each cell was interpreted as an assurance of sealing of all the joints in the liner. After draining the water, a layer of Geocloth (Fabric, Gundle Lining Systems, Inc., Houston, TX) was placed above the liner in each cell, to separate the liner from any sharp edged material contained in the shredded municipal refuse. The cells were then ready for placement of shredded municipal refuse.

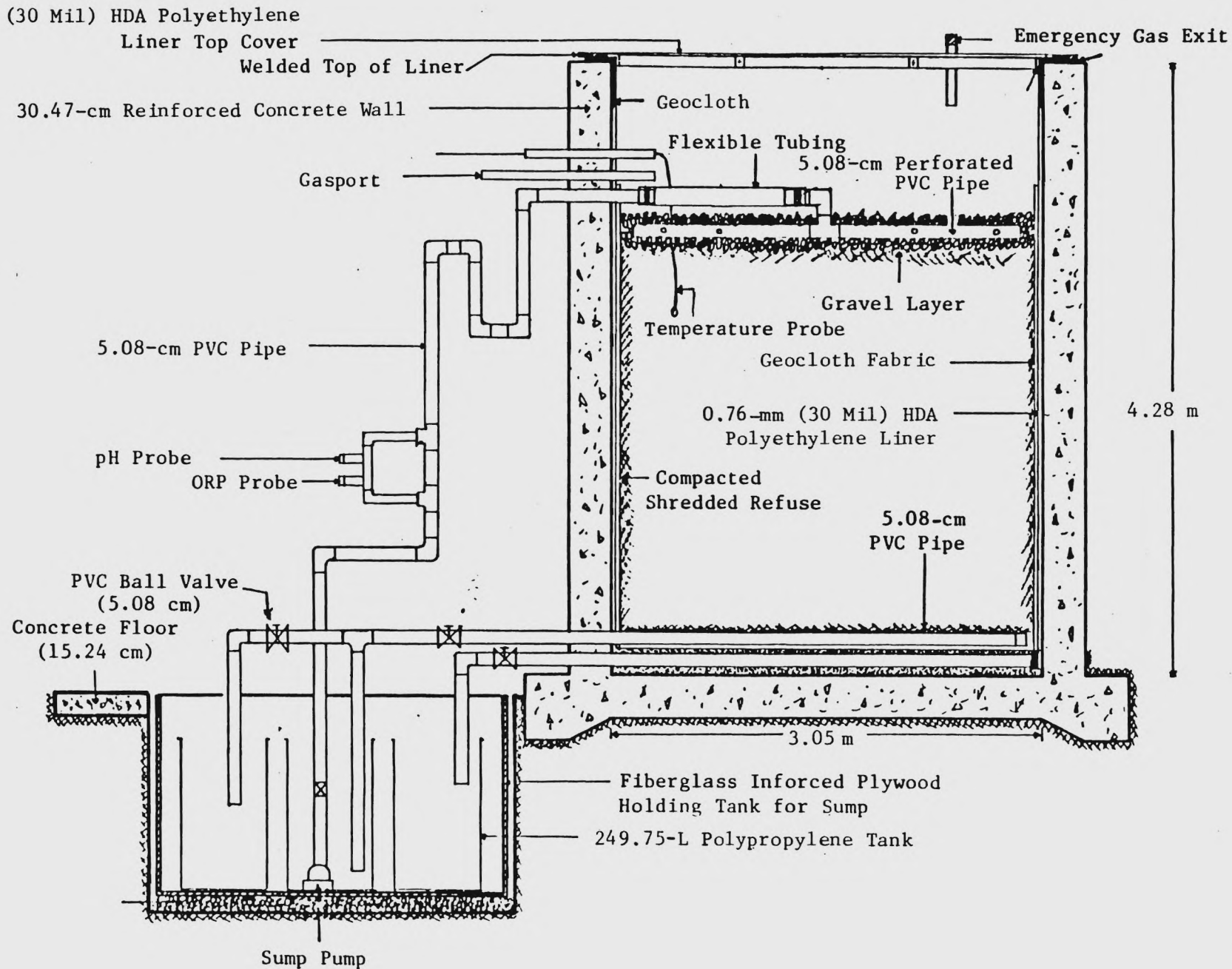


Figure 8. Schematic of simulated landfill with recycle.

3.1.2 Placement of Shredded Municipal Refuse

Shredded municipal refuse was loaded in a transfer truck at the Shredding Facility in Dekalb County, Georgia on 1 July 1982 and delivered onsite the following morning. The refuse was placed within the lined cells with a front-end loader and a conveyor system. During placement, the refuse was compacted by two hand-held gasoline powered tampers. The density of the compacted refuse within the landfills was estimated by compacting representative samples of refuse in 208-L (55-gal) drums by hand-held tampers concomitantly with the loading operation, then determining the weight of the compacted refuse. Also, representative samples of refuse were collected in 3.78-L (1-gal) glass containers at the beginning and the end of each work day; these samples were analyzed for moisture content, carbon, hydrogen, nitrogen, and calorific values. Refuse placement was interrupted when height of compacted refuse reached 1.22 m (4 ft) in each cell and a 0.03-m (1.5-in) layer of SaniFoam (urea-formaldehyde based plastic foam) was sprayed as a cover on the surface of the refuse in both cells.

Placement of refuse was resumed on 3 July 1982. Prior to loading of refuse in each cell the cured foam layer was crushed manually. A solution containing Co-58 (half life = 70.8 d), Sr-85 (half life = 64.5 d), and Ce-141 (half life = 32.4 d) was diluted to 50 mL with 0.5 M HCl. A 0.05-mL (50 λ) aliquot was withdrawn from the 50-mL radionuclide solution, diluted to 400 mL with 0.5 M HCl and placed in a cottage cheese container for counting. Three such solutions were prepared for counting. The volume in the 50-mL volumetric flask was brought to 50 mL by adding 0.5 M HCl and 25 mL of this solution were spiked on the

refuse placed in each cell. The resulting amounts of Co-58, Sr-85, and Ce-141 in each cell were 0.91 mCi, 1.09 mCi, and 5.78 mCi, respectively. Compaction of refuse was resumed after a 0.52-m (2-ft) layer of refuse was placed above the crushed foam layer. Upon completion of refuse placement, the average heights of compacted refuse were 2.54 m (8 ft 4 in) and 2.72 m (8 ft 10 in) in Cell A and Cell B, respectively.

Final closure of the cells was initiated on 4 July 1982. In each cell a distribution network was embedded in a 0.3-m (1-ft) layer of gravel. On 6 July 1982, the top of each cell was sealed with the liner material and the creases along the joints were vacuum tested for leaks. The entire structure was covered by a plywood roof. An instrumentation shed was constructed on a reinforced concrete slab; it housed leachate collection sumps, pH/ORP meter, flow meter for gas measurement, temperature recorders, and 208-L (55-gal) drums for storing leachate from Cell A (single pass cell). The entire structure was fenced off and access to the shed was controlled. The project site was inspected and approved by the Radiological Safety Officer of Georgia Tech Nuclear Safeguards Committee, Georgia Tech, Atlanta, GA.

3.2 Moisture Addition

Leachate generation from each cell was initiated by raising the moisture content of the refuse to field capacity (Cell A, 101.8%; Cell B, 100.5%; moisture content was based on the dry weight of the refuse). Since rainfall occurred between the conclusion of refuse placement and the initiation of final closure, a rainfall equivalent of 492 L (130 gal) was included in the calculation for the volume of water needed to

raise the moisture content of the refuse to field capacity. The total volumes of water (tap water + rainfall equivalent) added to Cell A and Cell B were 1893 L (500 gal) and 1949 L (575 gal), respectively. Thereafter, rainfall was simulated by adding rainfall equivalent volumes to both cells through the distribution networks. Addition of rainfall equivalent was terminated on Day 56, since accumulation of leachate at the bottom of Cell B exceeded 0.3-m (1 ft) head. Rainfall data for Atlanta, GA were obtained from the Georgia Tech Research Institute (GTRI), Atlanta, GA.

3.3 Analytical Methods

Leachate collection from each cell was initiated on 27 July 1982 (Day 0). Samples were collected at weekly intervals up to Day 150, at 2-week intervals from Day 150 to Day 570, and at monthly intervals from Day 570 to Day 970. From Day 290 to Day 350 collection of leachate samples was suspended and, the cells were tested for gas leaks during that period. After repeated testing, it was concluded that the cells were not leak proof for gas. Leachate collection was resumed on Day 350. Leachate recycle, however, was maintained at a frequency of once a week during that period (Day 290 to Day 350). Leachate samples were analyzed for pH, oxidation-reduction potential (ORP), five-day biochemical oxygen demand (BOD₅), chemical oxygen demand (COD), total carbon (TC), total inorganic carbon (TIC), total organic carbon (TOC), conductivity, total alkalinity, individual and total volatile fatty acids (acetic, propionic, isobutyric, butyric, and valeric), sulfide, chloride, metals (Fe, Zn, Ni, Mn, and Co), and radionuclides (Co-58,

Sr-85, and Ce-141). In addition to these parameters, nutrients such as nitrogen and phosphorus present in leachate are important indicator parameters which describe landfill stabilization. In this study, however, leachate samples were not analyzed for nutrients.

3.3.1 Gross Parameters

Total alkalinity, BOD₅, and COD were measured as per methods included in "Standard Methods" (1975). TC and TIC were measured with a Beckman Model 915 Total Carbon Analyzer (Fullerton, CA). A Fisher Model 144 pH/mV meter was used to measure pH and ORP. Conductivity was measured with a Yellow Springs Instrument Company Model 31 Conductivity Bridge (Yellow Springs, OH).

3.3.2 Volatile Fatty Acids

The volatile fatty acids present in leachate samples were measured using an Internal Standard Method on a Hewlett Packard Model 5710A gas chromatograph (GC) equipped with a flame ionization detector (FID; Avondale, PA). Leachate samples were filtered through a 0.45- μ m filter (Gelman, Ann Arbor, MI), then acidified to a pH value less than 2 with 85% phosphoric acid (3.5 mL acid/100 mL sample). In addition, 1-butanol (20- L 1-butanol/100 mL acidified sample) was added as the internal standard. Table 11 lists the GC program conditions utilized to measure volatile fatty acids.

3.3.3 Heavy Metals

Leachate samples were analyzed for the following heavy metals: iron (Fe), zinc (Zn), nickel (Ni), manganese (Mn), and cobalt (Co). These heavy metals were chosen as they were expected to behave similar to the behavior exhibited by cobalt. In addition to these heavy metals

Table 11. Gas Chromatographic Conditions Used for
Measuring Volatile Fatty Acids (83)

Gas Chromatograph	Hewlett Packard 5710A/ 3380A Integrator
Injection Temperature	250°C
Detector Temperature	250°C
Oven Temperature Program	
Initial Temperature and Time	100°C for 4 min
Rate of Increase in Temperature	4°C/min
Final Temperature and Time	180°C for 0 min
Column Details	
Material and Dimensions	Glass; 2m long by 2mm I.D.
Stationary Phase	1% Carbowax 20 M; 15% phosphoric acid on Carbopack B, #60H#80 mesh.
Carrier Gas	Nitrogen @ 40 mL/min

the leachate samples were also monitored for chromium (Cr) and copper (Cu). The analyses for these metals were discontinued after Day 50, since their concentrations in leachate were always less than 1 mg/L. The samples were preserved as per procedures included in "Methods of Chemical Analysis of Water and Wastes" (1979) (113). Two atomic absorption spectrophotometers, i.e., Perkin Elmer Model 303 and Perkin Elmer Model 703 (Norwalk, CT) were utilized to determine the concentration of each heavy metal present in digested leachate samples. Model 703 spectrophotometer was used to measure metal concentration in $\mu\text{g/L}$ range, and Model 303 spectrophotometer was used to measure metal concentration in mg/L range.

Leachate samples, prior to analysis, were digested as per the following procedure (84): to 100 mL of sample contained in a 250-mL beaker, a 5-mL aliquot of 1:1 diluted nitric acid (HNO_3 ; Ultrex) was added, covered with a watch glass, then evaporated (without boiling) to near dryness on a hot plate. After the sample was cooled, a 4-mL aliquot of concentrated HNO_3 (Ultrex) was added, and again evaporated to near dryness. After cooling, a 1-mL aliquot of 1:1 HNO_3 (Ultrex) and a 3-mL aliquot of 30% hydrogen peroxide (H_2O_2) were added to the sample which was then heated until the effervescence from the addition of H_2O_2 subsided. After cooling, the addition of 30% H_2O_2 in 1-mL aliquots and evaporation of the sample were repeated until the effervescence was minimal or the general sample appearance was unchanged. No more than 10 mL of 30% H_2O_2 was added to the sample. After cooling, a 1-mL aliquot of 1:1 HNO_3 (Ultrex) was added and the sample was then heated for ten minutes. The sample was then cooled, diluted to 100 mL

with deionized water, and stored in a polyethylene container.

Digestion blanks of deionized water were prepared along with digestion of leachates.

3.3.4 Sulfide

Leachate samples were analyzed for sulfide with a silver/sulfide electrode (Orion Model 94-16; Orion Research, Inc., Cambridge, MA) and a pH/mV meter (Fisher Accumet Model 610; Fisher Scientific, Atlanta, GA). The method, included in the manufacturer's instructions, is presented in subsequent sub-sections.

3.3.4.1 Preparation of Deoxygenated Deionized Water (DDW).

Approximately 3 L of deionized water was deoxygenated by purging the water with nitrogen for approximately 30 minutes. This water, hereafter referred to as DDW, was used for preparing all working solutions.

3.3.4.2 Sulfide Anti-Oxidant Buffer (SAOB 11). To 600 mL of DDW placed in a 1-L volumetric flask, 200 mL of 10 M NaOH, 35 g ascorbic acid, and 67 g disodium ethylenediaminetetraacetate were added. The mixture was swirled, then diluted to 1 L with DDW. Fresh SAOB 11 ranged in color from clear to pale yellow-brown and was stored in a tightly stoppered bottle. Upon oxidation, SAOB 11 turned dark brown and was discarded.

3.3.4.3 Sulfide Standards. A stock solution of saturated sodium sulfide was prepared by dissolving approximately 100 g of reagent grade $\text{Na}_2\text{S} \cdot 9\text{H}_2\text{O}$ in 100 mL of DDW. The mixture was well shaken, allowed to stand overnight, and stored in a tightly stoppered bottle.

A Weekly Sulfide Standard was prepared by pipetting 1 mL of the

stock solution into a 100-mL volumetric flask, then adding 50 mL of SAOB 11. The mixture was diluted to 100 mL with DDW and the exact concentration C was determined by titrating 40 mL of Weekly Sulfide Standard with 0.1 M lead perchlorate $Pb(ClO_4)_2$ and noting the change in potential (mV). At the end point of titration the change in potential (ΔmV) was the largest.

$$C = 3200 (V_t/V_s) \quad (1)$$

where;

C = concentration of sulfide, mg/L

V_t = volume of titrant at end point, mL

V_s = volume of standard, mL

To prepare a calibration curve, sulfide standards were prepared by making ten-fold serial dilutions of the Weekly Sulfide Standard with equal volumes of SAOB 11 and DDW.

3.3.4.4 Sulfide Measurement. Three sulfide standards were prepared by serial dilution of the Weekly Sulfide Standard (Section 3.3.4.3). The silver/sulfide electrode was placed in the beaker containing the least concentrated standard and the potential (mV) was noted while the standard was stirred gently. The electrodes were then taken out, rinsed, blot dried, and the procedure was repeated for the mid-range standard and the most concentrated standard. A calibration curve was prepared by plotting sulfide concentrations (log axis) versus millivolt readings (linear axis) on a semilogarithmic paper.

To measure sulfide present in a leachate sample, 50 mL of SAOB 11 was added to 50 mL of sample in a 150-mL beaker. The solution was

stirred gently and the potential (mV) was noted. The concentration was read directly from the calibration curve which was prepared within two hours.

3.3.5 Chloride

Leachate samples were analyzed for chloride using a potentiometric method similar to the method used for sulfide (Orion Research, Inc., Cambridge, MA). Pretreated leachate samples were titrated with 0.0142 N silver nitrate (AgNO_3).

Leachate samples, prior to titration, were pretreated as per the following method: a 10-mL aliquot of sample was diluted to 100 mL with deionized water, acidified with a 2-mL aliquot of concentrated sulfuric acid (H_2SO_4), and boiled for five minutes. After cooling, a 5-mL aliquot of 50% hydrogen peroxide (H_2O_2) was added, and the sample was boiled again for 15 minutes. After cooling, a 10-mL aliquot of 10 M sodium hydroxide (NaOH) was added, and the sample was diluted to 100 mL with deionized water. The sample was boiled again for five minutes, then filtered while hot with a glass fiber filter (Gelman, Ann Arbor, MI). After cooling, a 4-mL aliquot of nitric acid (HNO_3 ; Ultrex) was added and the sample was diluted to 100 mL with deionized water. This pretreated sample was then titrated 0.0142 N solution of silver nitrate.

3.3.6 Temperature

Temperature within the refuse in each cell was recorded on a temperature recorder (Series 9354; Cole-Parmer Instrument Company, Chicago, IL) with a temperature probe (Series 703, YSI; Cole-Parmer Instrument Company, Chicago, IL) which was embedded within the refuse

at 0.6 m from the refuse surface.

3.3.7 Radioactivity

Leachate samples in 400-mL aliquots were filtered through 0.45- μ m filters (Gelman, Ann Arbor, MI), placed in cottage cheese containers, and analyzed for Co-58, Sr-85, and Ce-141 on a Ge(Li) detector. Table 12 lists the equipment used to detect Co-58, Sr-85, and Ce-141.

3.4 Complexation Studies

During landfill stabilization, the precipitation of heavy metals, that are present in the leachate, by potential-precipitant species, such as sulfides, carbonates, and hydroxides is counteracted by the potential formation of soluble metal-humate complexes. During the course of investigation, however, the radionuclides, Co-58, Sr-85, and Ce-141, present in leachate, decayed to below detectable levels. Thus, their potential to form soluble metal-humate complexes could not be evaluated. Since Chang (21), based on a study on codisposal of municipal refuse, Co-58, Sr-85, Cs-137, and H-3, reported an increase only in the Co-58 levels in leachate during the methane fermentation phase of landfill stabilization, a companion laboratory-scale investigation was embarked upon to evaluate the potential of humic substances in leachate to mobilize Co-58 from previously precipitated cobalt sulfide by forming soluble cobalt-humate complexes. Leachate samples from the pilot-scale landfill with leachate recycle (Cell B) were utilized to assess the potential of the humic substances present in leachate to mobilize Co-58 from cobalt sulfide.

The analytical methodology utilized for the investigation of

Table 12. Equipment Used to Measure Co-58, Sr-85, and Ce-141
Present in Leachate Samples Collected from the
Pilot-Scale Landfills

Detector	Ge(Li)
Company	Canberra
Shield	1.75 mm lead wall with 3.17 mm copper liner (ADT Co.)
Liquid Nitrogen	Type LR-31 Serial Number 80RF-76 (Union Carbide Co.)
Preamplifier	
Company	Canberra
Model	#2001
High Voltage Power Supply	
Company	Canberra
Model	#3105
Operating Voltage	+2500 V
Spectroscopy Amplifier	
Company	Canberra
Model	#141
Coarse Gain	10
Fine Gain	4.34
Multichannel Analyzer	
Company	Canberra
Model	#8100; One channel equivalent to 1 KeV
Counting Time	2000 sec
Printer	
Company	Hewlett Packard
Model	#9830A
Calculator/Plotter	
Company	Hewlett Packard
Model	#9862A

potential complexation of Co-58 with leachate was developed using model compounds, since reproducibility and reliability of the results produced from the methodology utilized were key to the success of this investigation. A swamp-based humic acid (A-2HA) was selected as the model humic acid, since it was well-characterized (85) and was of aquatic origin. Cobalt was selected as the model metal since the shredded municipal refuse was spiked with Co-58 to simulate codisposal of municipal refuse and LLRW within the pilot-scale landfills. Analytical grade cobalt sulfide (CERAC, Milwaukee, WI) was utilized to simulate precipitated cobalt within landfills. The precipitated cobalt sulfide within a landfill, however, will be associated with the precipitates of other heavy metals and inert solids. The use of analytical grade cobalt sulfide to examine the potential of humic substances present in leachate to mobilize cobalt from cobalt sulfide precipitated within a landfill will yield higher levels of cobalt in leachate, since the humic substances in leachate will only complex cobalt, as opposed to the case in an actual landfill, wherein cobalt along with other heavy metals will be complexed by the humic substances. Thus, the values generated, at equilibrium, for the mobilization of Co-58 from cobalt sulfide by humic substance in leachate can be regarded as "worst-case" results. Stable cobalt was replaced by a mixture of Co-58 and stable cobalt in the confirmatory study since Co-58 is measured more accurately than stable cobalt in dilute solutions of cobalt and the results of this study ascertained the complexation of Co-58 by humic substances present in leachate.

Development of the analytical methodology was implemented in

five phases:

- i) development of the molecular-weight fractionation scheme;
- ii) development of the analytical methods for equilibrium studies on complexation of cobalt with model humic acid (A-2HA);
- iii) development of the analytical methods for kinetic studies on complexation of cobalt with model humic acid (A-2HA);
- iv) equilibrium and kinetic studies on complexation of cobalt with leachate from the pilot-scale landfill with leachate recycle (Cell B); and,
- v) equilibrium studies on complexation of Co-58 with model humic acid (A-2HA) and leachate from Cell B.

3.4.1 Molecular-Weight Fractionation Scheme

A molecular-weight fractionation scheme, which included ultrafiltration (UF) and gel permeation chromatography (GPC) was developed for this study. Chian and DeWalle (78) reported that parallel ultrafiltration of leachate samples resulted in a more representative separation than serial ultrafiltration; thus, parallel ultrafiltration was incorporated in the molecular-weight fractionation scheme. Aqueous solutions of the model humic acid (A-2HA) at pH values of 5.5, 7.0, and 8.5 were prepared using the procedure outlined in Figure 9. These solutions and leachate samples were fractionated into different molecular-weight fractions by UF and GPC. Figure 10 presents the molecular-weight fractionation scheme.

3.4.1.1 Ultrafiltration (UF). Ultrafiltration (UF) of model humic acid (A-2HA) and leachate samples was performed in a stirred UF

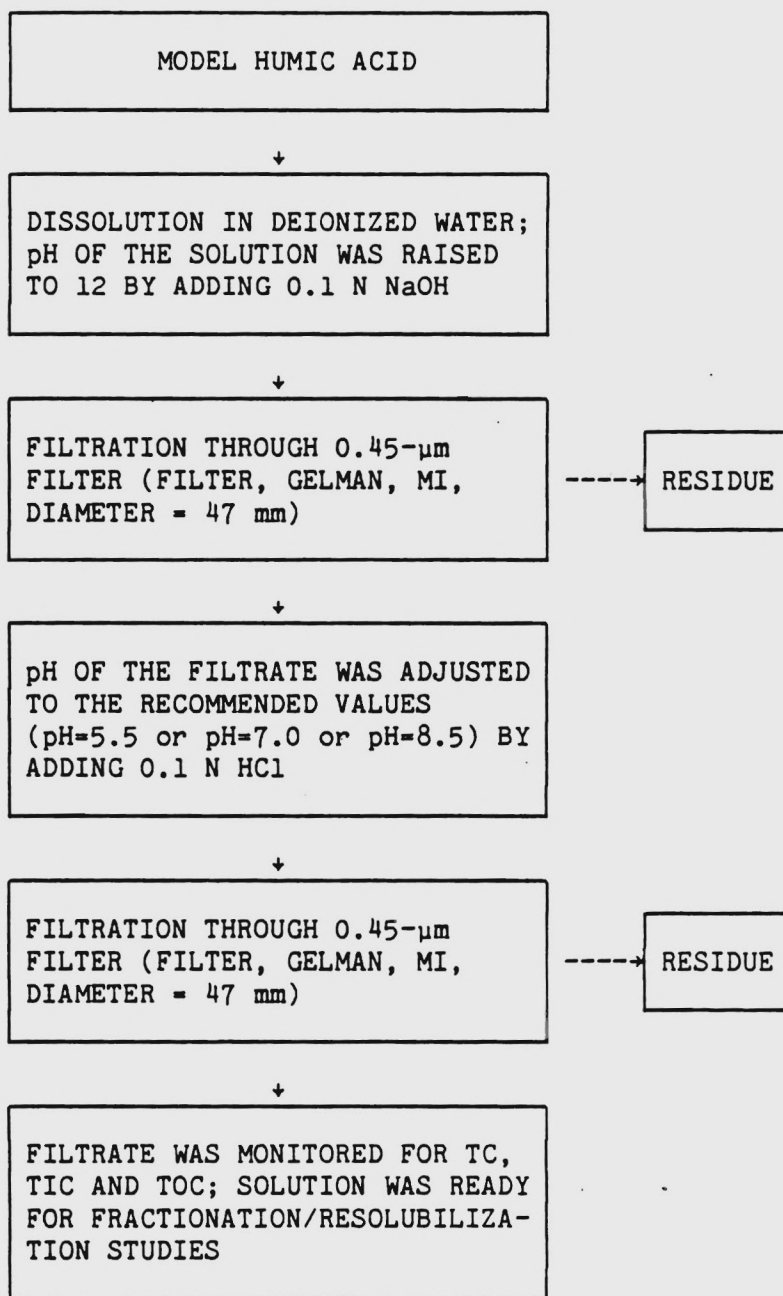


Figure 9. Preparative Procedure of Aqueous Solution of Model Humic Acid

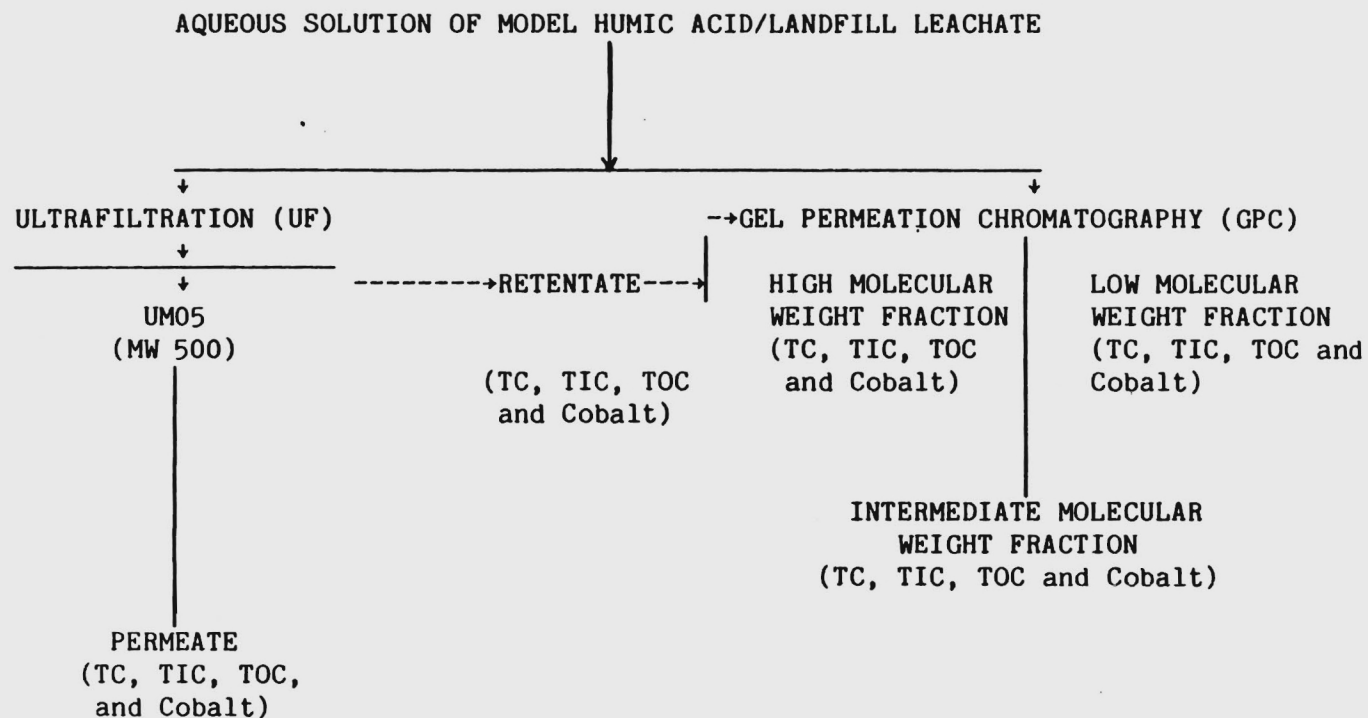


Figure 10. Molecular Weight Fractionation of Model Humic Acid and Landfill Leachates.

cell (Amicon, Bedford, MA) with UM05 UF membrane (Diaflo Ultrafilters, Amicon, Bedford, MA). This membrane had a nominal weight cutoffs of 500 daltons. Prior to ultrafiltration, the UF membranes were checked for leaks by testing the retention of Blue Dextran ($MW > 2 \times 10^6$ daltons). Nitrogen was used to pressurize the sample within the UF cell; the operating pressure was maintained at 2.41×10^5 Pa (35 psi). Table 13 presents the experimental details of ultrafiltration.

Ultrafiltration was used to determine the extent of cobalt bound to the high-molecular-weight fraction of both the model humic acid (A-2HA) and the humic substances present in the leachate. This technique is based on the premise that a UM05 UF membrane (MW cutoff: 500 daltons) would retain the high-molecular-weight organic fraction and any cobalt associated with it. The original sample and its ensuing fractions were monitored for cobalt and TOC.

3.4.1.2 Gel Permeation Chromatography (GPC). The determination of molecular-weight distribution profiles of humic substances found in soils, sediments, and natural waters by gel permeation chromatography (GPC) especially, with Sephadex gels (Pharmacia Fine Chemicals, NJ), has been reported by many researchers (87-108). Gel permeation chromatography is influenced by several factors that include column size, pH, volume, concentration, and composition of sample, and pH, type, and concentration of eluant. Furthermore, the validity of molecular-weight distribution profiles generated by GPC is enhanced by minimizing gel-solute interactions. There are two main kinds of gel-solute interactions: coulombic interaction caused by charged sites of gel and solute, and adsorption effects. The former, especially for

Table 13. Experimental Protocol Utilized for Ultrafiltration of Leachate

Ultrafiltration Cell			Ultrafiltration Membranes		
Cell Type	Operating Pressure Pa(psi)	Gas Used to Pressurize the Sample	Type	Nominal Molecular Weight Cutoff (daltons)	Nominal* Pore Diameter (μm)
Stirred UF Cell; Max Sample Volume = 500 mL	2.41×10^5 (35)	Nitrogen	UM05	500	0.0021
Max pressure =	3.44×10^5 (50)				

*Perry, R. H., and Chilton, C. H., "Chemical Engineering Handbook", 5th Ed., McGraw Hill Book Company, pp 19-84.

polyelectrolytes impose an additional charge-exclusion resulting in an apparent increase in the nominal molecular weight, while the latter results in an apparent decrease. Thus, molecular-weight distribution profiles generated under controlled conditions can be compared, while the practice of assigning nominal molecular weights to the excluded fractions of humic acids is questionable. In this study, GPC under controlled conditions was used to compare molecular-weight distribution profiles of the model humic acid (A-2HA) and the leachate samples with/without soluble cobalt-humate complexes present. Sephadex gels G-50 and G-75 (Pharmacia Fine Chemicals, Piscataway, NJ) were used as the stationary phase. Deionized water was used as the mobile phase and pumped through the entire system by a constant flowrate pump (Milton Roy Co., St. Petersburg, FL). The valve system used for sample application consisted of two SRV-4 four-way valves (Pharmacia Fine Chemicals, Piscataway, NJ) and a 1-mL sample loop. The valve system was operated in three configurations: column standby; filling the sample loop; and, sample application and normal running. A detailed block diagram of the apparatus and the operating sequence of the valve system are presented in Figure 11.

Blue Dextran ($MW > 2 \times 10^6$ daltons) and phenol ($MW=94$ daltons) were used as standards for the high-molecular-weight fraction and the low-molecular-weight fraction, respectively. The high-molecular-weight cut and the low-molecular-weight cut were established by measuring the absorbance of the ensuing fractions at $\lambda = 254$ nm. Standards, model humic acid (A-2HA), and leachate samples were injected into the sample loop in 0.5-mL aliquots; the mobile phase, pumped through the sample

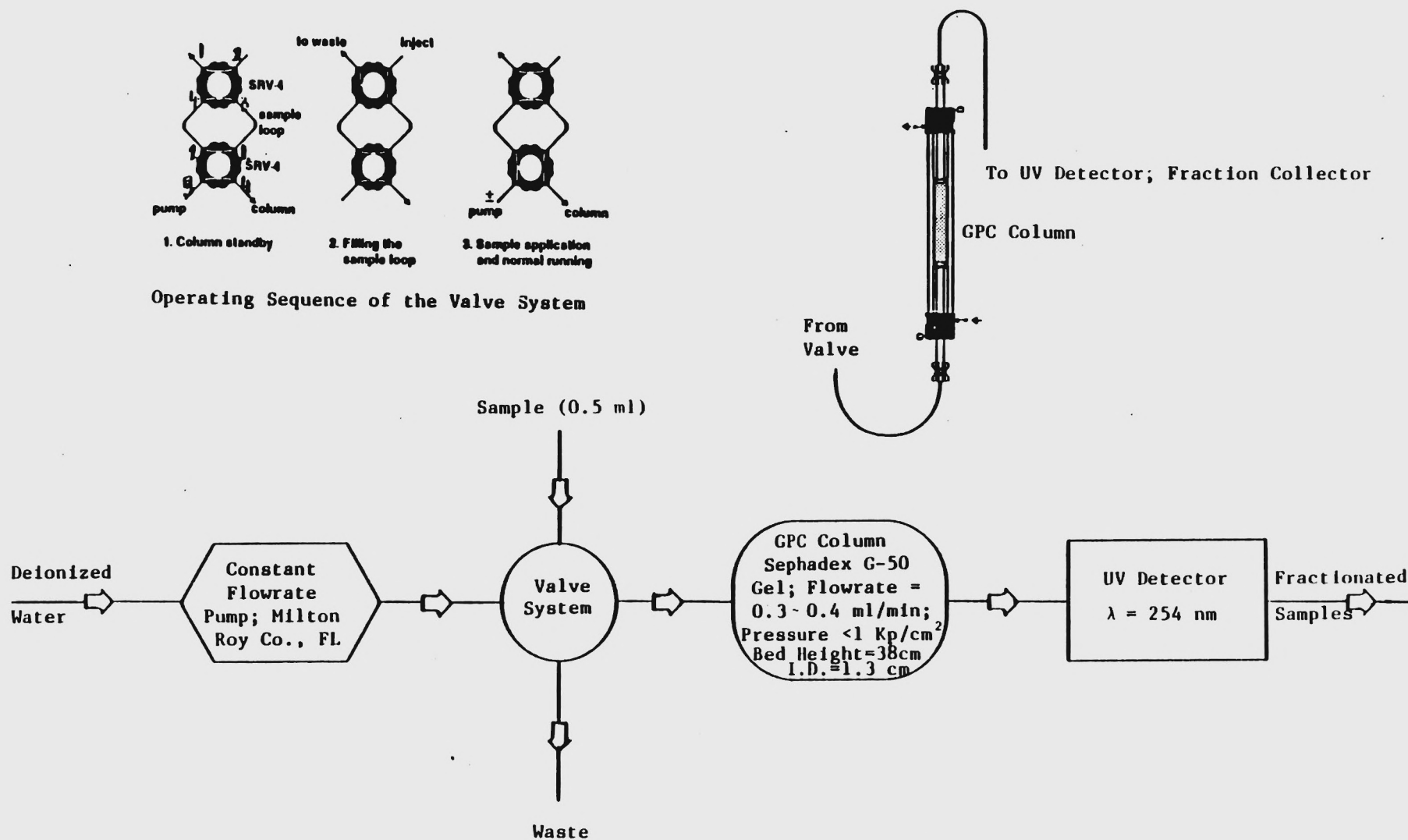


Figure 11. A Block-Diagram of the Apparatus Used for Fractionating A-2HA and Landfill Leachates by Gel Permeation Chromatography (GPC).

loop, introduced the sample into the column, and the ensuing fractions of the model humic acid (A-2HA) and the leachate samples were analyzed for TOC and cobalt. The experimental details of GPC are presented in Table 14.

3.4.2 Equilibrium Studies

A quality assurance and quality control (QA/QC) study was performed to determine the significance of the difference in the equilibrium levels of cobalt solubilized from cobalt sulfide by deionized water and the model humic acid (A-2HA), respectively. Measured amounts of cobalt sulfide were added to 20 replicates of deionized water ($\text{CoS} = 0.065 \text{ g} \pm 0.011 \text{ g}$) and 17 replicates of the model humic acid (A-2HA; 0.5 g/L ; $\text{pH} = 7.00$, $\text{CoS} = 0.061 \text{ g} \pm 0.008 \text{ g}$), each contained in a 125-mL French Square bottle without any headspace. These bottles were loaded in a rotary shaker (1 rpm) and allowed to equilibrate for three weeks. The shaker was housed in an incubator; the temperature within the incubator was maintained at $20^\circ\text{C} \pm 1^\circ\text{C}$.

After the equilibration period, samples were withdrawn from the rotary shaker and centrifuged in a ultracentrifuge (Model J2-21, Rotor: JA-20.1, Beckman Instruments, Inc., Palo Alto, CA) at $20.0^\circ\text{C} \pm 0.5^\circ\text{C}$. The centrates from the centrifuged samples were digested as per the digestion procedure included in Section 3.3.3; the digested samples were analyzed for cobalt with an atomic absorption spectrophotometer (Model 703, Graphite Furnace, Perkin Elmer, Norwalk, CT).

After completion of the QA/QC study, equilibrium studies on complexation of cobalt (from cobalt sulfide), by model humic acid at pH values of 5.5, 7.0, and 8.5 were performed. These pH values were

Table 14. Experimental Protocol Utilized for Gel Permeation Chromatography (GPC) of Leachate

Stationary Phase		Mobile Phase	Standards	Samples Fractionated
Gel	Column Details			
<p>Sephadex: a modified dextran obtained by crosslinking linear macromolecules, consisting of a three-dimensional network of polysaccharide chains.</p> <p>G-50 (Fractionation range: 500 MW-10,000 MW) G-75 (Fractionation range: 1,000 MW-50,000 MW)</p>	<p>Bed Height = 38.5 cm</p> <p>Inner Diameter (I.D.) = 1.3 cm</p>	<p>Deionized Water</p> <p>(Flowrate: 0.4-0.5 mL/min)</p>	<p>Blue Dextran (MW$>2 \times 10^6$)</p> <p>Phenol (MW=94)</p>	<p>Standard, A-2HA and filtered landfill leachate (0.45-μm filter; Gelman, MI)</p>

selected since the pH of landfill leachate may vary from 4.5 during acid formation phase to 9.0 as may be the case due to generation of ammonia by microbially-mediated reduction of nitrates or presence of caustic metal sludges in the landfill. Moreover, microenvironments with different pH values are known to coexist within a landfill. Thus, equilibrium studies on complexation of cobalt with model humic acid (A-2HA) were performed at different pH values.

The procedure utilized for these studies was similar to the procedure developed for the QA/QC study. In addition, the centrates from centrifuged samples were fractionated as per the molecular-weight fractionation scheme included in Section 3.4.1 (Figure 3). In the UF fractionation scheme, however, only a UMO5 UF membrane (MW cutoff: 500 daltons) was used, thereby fractionating the sample into the permeate (low-molecular-weight) and the retentate (high-molecular-weight) fractions. The sample, the permeate, and the retentate were digested as per the procedure included in Section 3.3.3, then analyzed for cobalt by atomic absorption spectroscopy (Perkin Elmer Model 703, Graphite Furnace Atomic Absorption Spectrophotometer, Norwalk, CT). The procedure utilized for equilibrium studies on complexation of cobalt (from cobalt sulfide) by filtered leachate samples (0.45 μ m filter, Gelman, Ann Arbor, MI) was similar to the aforementioned procedure, and is presented in Figure 12.

3.4.3 Kinetic Studies

To determine the time required by the model humic acid (A-2HA) for solubilizing cobalt to equilibrium levels from cobalt sulfide, a kinetic study was performed. Known amounts of cobalt sulfide were

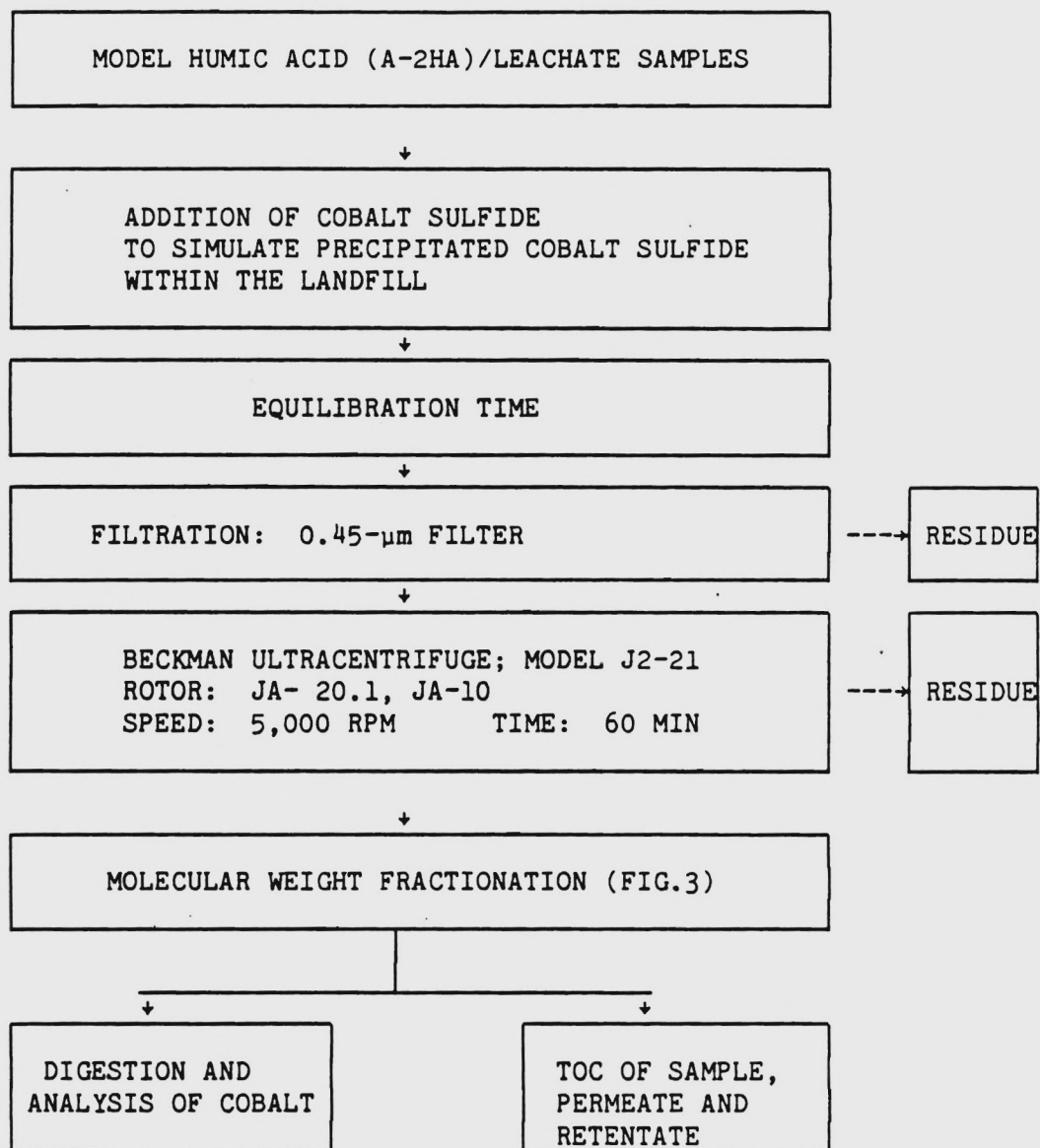


Figure 12. Methods Used for Equilibrium Studies on Complexation of Cobalt by Model Humic Acid (A-2HA) and Leachate Samples.

added to the model humic acid (A-2HA; 0.5 mg/L; pH=7.04; and TOC = 276.8 mg/L) in 125-mL French Square bottles; these bottles were loaded in a rotary shaker. Samples were withdrawn at predetermined intervals filtered, centrifuged, digested, and analyzed for cobalt by atomic absorption spectroscopy (Perkin Elmer Model 703, Graphite Furnace, Atomic Absorption Spectrophotometer, Norwalk, CT). The procedure utilized for leachate samples was similar to the procedure developed for the model humic acid (A-2HA) and is presented in Figure 13.

3.5 Equilibrium Studies with Co-58

The solubilization of Co-58 from cobalt sulfide (containing Co-58) by model humic acid (A-2HA) and leachate samples was investigated. Upon reaching equilibrium, the samples containing resolubilized Co-58 were fractionated with gel permeation chromatography (GPC); thus, molecular-weight distribution profiles of radioactive cobalt were generated.

The radioactive cobalt (Co-58, 1 mCi) was purchased from Amersham (Arlington Heights, IL), and received by the Radiological Safety Officer, Office of Radiological Safety, Georgia Institute of Technology, Atlanta, GA. The procedure utilized in this study was approved by the Georgia Tech Nuclear Safeguards Committee and the experiments were performed in the Radiochemistry Laboratory of the Nuclear Research Center at Georgia Tech.

The detailed procedure of the equilibrium experiment is presented in four sections: preparation of the tracer solution, precipitation of cobalt sulfide, vacuum filtration step, and the equilibrium experiments.

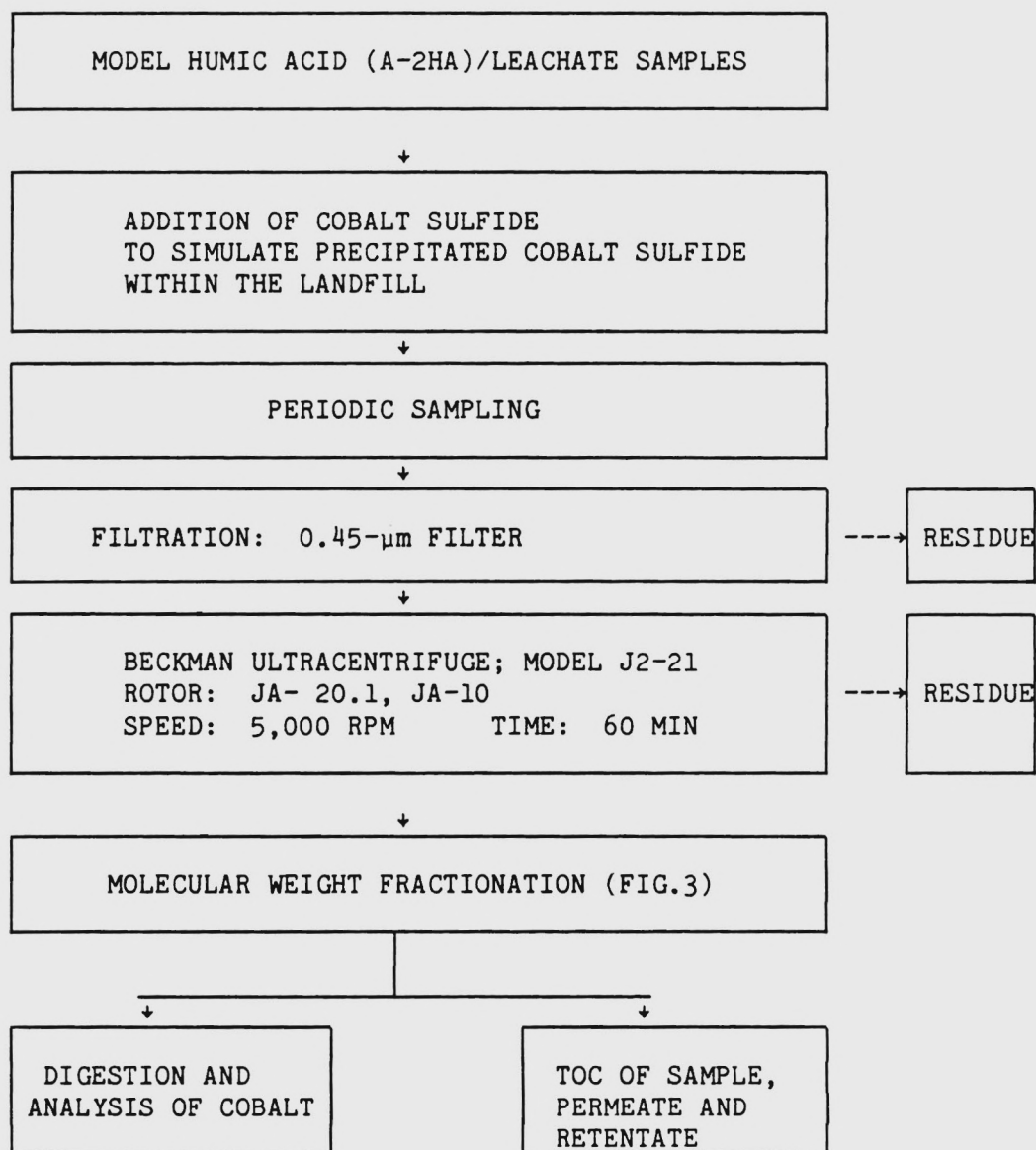


Figure 13. Methods Used for Kinetic Studies on Complexation of Cobalt by Model Humic Acid (A-2HA) and Leachate Samples.

3.5.1 Preparation of Tracer Solution

The tracer solution (20 $\mu\text{Ci/mL}$) was prepared with the addition of a 2-mL aliquot of Co-58 tracer containing 1 mCi to approximately 20 mL of 0.01 M hydrochloric acid (HCl) in a 50-mL volumetric flask; the final volume was adjusted to 50 mL with 0.01 M HCl. This tracer solution was used in the subsequent cobalt sulfide preparation step.

3.5.2 Precipitation of Cobalt Sulfide

To approximately 10 mL of deionized water in a 25-mL volumetric flask, 8 mL of cobalt solution (Atomic Absorption Standard, 1 mL = 1 mg; Fisher Chemicals, Atlanta, GA) and 0.5 mL of Co-58 tracer solution were added. The final volume was adjusted to 25 mL and a 0.2-mL aliquot was withdrawn for radioactivity counting. The standard cobalt solution, containing both stable and radioactive cobalt, was transferred into a 125-mL Erlenmeyer flask, then a 5-mL aliquot of sodium sulfide (NaS ; 4.3 gm/L) was added. The formation of a black cobalt sulfide suspension was observed.

3.5.3 Vacuum Filtration Step

The cobalt sulfide suspension generated in the previous step (3.5.2) was filtered through a 0.45- μm filter (Micro Filtration Systems, Dublin, CA). The Erlenmeyer flask and the filter holder were rinsed with deionized water and the rinse water was filtered through the same paper. The volume of the filtrate was recorded and a 5-mL aliquot was withdrawn for radioactivity counting.

3.5.4 Equilibrium Experiment

The filter papers, together with the residues, were placed within correspondingly labelled vials (volume: 24 mL); the vials were

then filled (no headspace) with the model humic acid (A-2HA) at different pH values (5.5, 7.0, and 8.5) or leachate samples, and capped tightly with screw tops. These vials, along with packing material, were placed within wide-mouthed square polyethylene (PE) reagent bottles (180 mL Fisher Chemicals, Atlanta, GA). This minimized movement of the vials during the equilibration period. Furthermore, it reduced the possibility of contamination which could result from an accidental breakage of the vials. The reagent bottles were loaded in a rotary shaker (1 rpm) and the samples were allowed to equilibrate.

The solutions in the vials were monitored for radioactivity at intervals of 12, 28, and 90 days. These intervals were chosen, since equilibrium experiments on mobilization of stable cobalt by humic acid (A-2HA) and landfill leachate indicated that equilibrium was reached within 14 days. After 12 days, the vials were withdrawn from the PE bottles. One-mL aliquots were withdrawn for radioactivity counting and 1-mL aliquots of corresponding ligand solutions were added to the vials. The vials were placed within the PE bottles and allowed to equilibrate in the rotary shaker for 16 more days. After a total of 28 days, 1-mL aliquots were withdrawn and fractionated into different molecular-weight fractions with gel permeation chromatography (GPC). The samples were further equilibrated for a total of 90 days and 1-mL aliquots were withdrawn for radioactivity counting. The samples and the ensuing fractions were analyzed for radioactivity using a sodium iodide detector with a thallium activator [NaI(Tl)]; the other equipments used for radioactivity counting are mentioned in Table 12.

CHAPTER IV

RESULTS AND DISCUSSION

Results of analyses performed on the shredded municipal refuse samples which were collected concomitantly with the loading operation and the leachate samples from the two pilot-scale landfill cells along with the environmental data are presented in this chapter. In the time scale used for data presentation, Day 0 corresponds to the initiation of leachate collection on July 27, 1982. Also included in this chapter are the results of the studies performed on the solubilization of Co-58 from cobalt sulfide (containing both Co-58 and stable cobalt) by model humic acid (A-2HA), and leachate samples from the landfill cell equipped with leachate recycle (Cell B). Since the only radionuclide present in the samples was Co-58, the radioactivity in samples has been expressed in counts per minute normalized to Day 0 (March 7, 1985) of the solubilization study.

4.1 Characterization of Shredded Municipal Refuse

The characteristics of shredded municipal refuse placed in the pilot-scale landfills (Cell A and Cell B) with respect to as placed refuse density, moisture content, carbon, hydrogen, and nitrogen (CHN), and heat value are presented in Table 15. The as placed refuse density, 379.9 kg/m^3 , was computed by simulating compaction of municipal refuse

Table 15. Original Characteristics of Shredded Municipal Refuse Placed within Pilot-Scale Landfills

Parameter	Characteristics of Shredded Municipal Refuse*
Density** of shredded municipal refuse as placed, kg/m ³	379.9
Carbon, % dry basis	43.8
Hydrogen, % dry basis	5.8
Nitrogen, % dry basis	6.2
Moisture content as placed, %	40
Heat value as placed, kJ/kg	15,210

*Average of four samples

**The as placed density was computed by simulating compaction of municipal refuse with hand-held tampers within 208-L drums concomitantly with the loading operation.

with hand-held tampers within 208-L drums concomitantly with the loading operation; this value, albeit within the range of refuse density reported by Tchobanoglous et al. (53), is higher than value (318.6 kg/m^3) reported by Pohland (23). The mass of refuse, on an as placed basis, in Cell A and Cell B was 8,976 kg (19,789 lb) and 9,584 kg (21,130 lb), respectively. The mass of refuse placed in each cell was calculated by multiplying the volume of compacted refuse in each cell with the as placed refuse density (379.9 kg/m^3).

The carbon, hydrogen, and nitrogen contents of the shredded municipal refuse, on dry weight basis, were 43.8%, 5.8%, and 6.2%, respectively. These values are similar to values reported in the literature (23,53), albeit the nitrogen content of 6.2% was higher than values reported by Pohland (23) and Chang (21) for shredded municipal refuse from the same shredding facility. Since the chemical composition of shredded municipal refuse at the same shredding facility may vary seasonally, the higher nitrogen content may be attributed to higher content of food wastes, garden trimmings, textile wastes, and leather wastes in the shredded municipal refuse used in this study as compared to the municipal refuse samples used by Chang (21) and Pohland (23). The heat value of refuse, on a dry basis, was 15,210 kJ/kg; this value was within the range of values reported by Tchobanoglous et al. (53).

4.2 Moisture Balance

A periodical evaluation of the moisture present within a landfill is important since it is one of the factors which enhances landfill stabilization. The volume of moisture (m^3) retained within pilot-scale landfill cells at any time, t , can be expressed as:

$$R/t=t = MA/t=0 + M/t=0 + WR/t=t - L/t=t - LS/t=t - LF/t=t \quad (1)$$

where:

R = the volume of moisture retained within the landfill cell (m^3).

MA = the volume of water added to bring the landfill cell to field capacity (100% moisture content based on dry weight of refuse placed within the cell) (m^3).

M = the volume of water corresponding to the initial moisture content of the refuse placed within the landfill cell (m^3).

WR = the cumulative volume of water added to the landfill cell to simulate incidental rainfall (m^3).

L = the cumulative volume of leachate drained from the landfill cell (m^3).

LS = the cumulative volume of leachate collected from the landfill cell for analysis (m^3).

LF = the cumulative volume of leachate lost from the landfill cell due to liner failure (m^3).

Since the cells were sealed, the entire structure was covered

with plywood roofing, and the leachate collection systems were housed in a shed, the volume of moisture lost due to evaporation could be considered negligible. Moreover, the term LF assumes significance only in an event of a liner failure. Since the absence of leachate in the underdrain system was noted throughout the period of this investigation, the term LF has been dropped from the expression for evaluating R; thus, the modified expression for the volume of moisture retained with the landfill cell at any time, t, is:

$$R/t=t = MA/t=0 + M/t=0 + WR/t=t - L/t=t - LS/t=t \quad (2)$$

The moisture balances for Cells A and B are presented in Figure 14. Water additions to the cells simulating incidental rainfall were terminated on Day 56 (September 22, 1982) for two reasons: to simulate the prevention of infiltration after sealing off the cells with covers (constructed from the liner material) and to limit the accumulation of leachate to 0.3 m (1 ft) over the liner at the bottom of Cell B. The moisture content (based on as placed weight of refuse) in Cell A varied between a low of 59.8% (Day 970) and a high of 69.2% (Day 56). The moisture content (based on dry weight of refuse) in Cell B varied between a low of 60.3% (Day 0) and a high of 73.9% (Day 56).

The maintenance of a minimal volume of leachate within landfills by limiting the volume to a 0.3-m head over the bottom of the landfill has two operational advantages: a reduction in size of the appurtenances required for recirculation of leachate and a reduction in volume

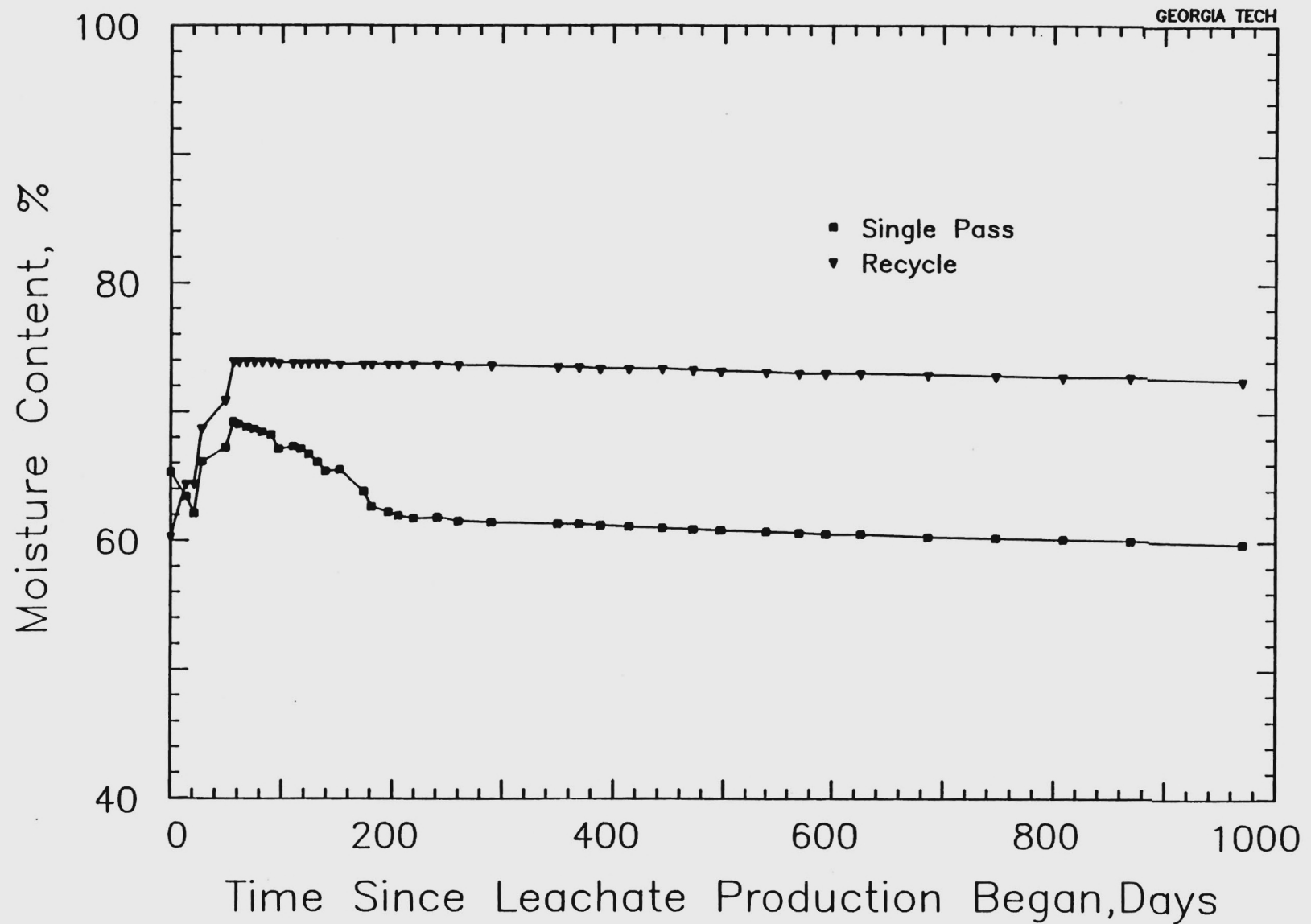


Figure 14. Variations in the Moisture Content of the Municipal Refuse Within Both Cells.

of leachate that may require external treatment. Also, the maintenance of a minimal volume of leachate within landfills will reduce the volume that can potentially migrate into the adjoining environment in the event of a liner failure. Since the presence of moisture content at levels greater than field capacity is essential for the maturation of stabilization processes, the volume of leachate in excess of field capacity can be minimized without disturbing the stabilization processes by recirculation of leachate as reported by Pohland (22).

4.3 Environmental Conditions

The data on temperature, including maximum and minimum ambient temperatures and maximum temperatures within the cells, and incidental rainfall are presented in Figures 15 and 16, respectively. The maximum temperatures within the cells followed the variations in the ambient temperatures during the study.

4.4 Leachate Analysis

The causes of changes in the various constituents present in leachates from the landfill cells were different; in Cell A (single pass), the leachate generated was collected and stored, whereas in Cell B (leachate recycle), the leachate produced was recycled on a daily basis (Day 0 to Day 152 and Day 350 to Day 556), weekly basis (Day 152 to Day 350 and Day 556 to Day 970). The recirculation of leachate provided a continuous exposure of the complex constituents in the leachate and the refuse to the microbial population within the landfill cell. This prolonged exposure enabled the microbial population to

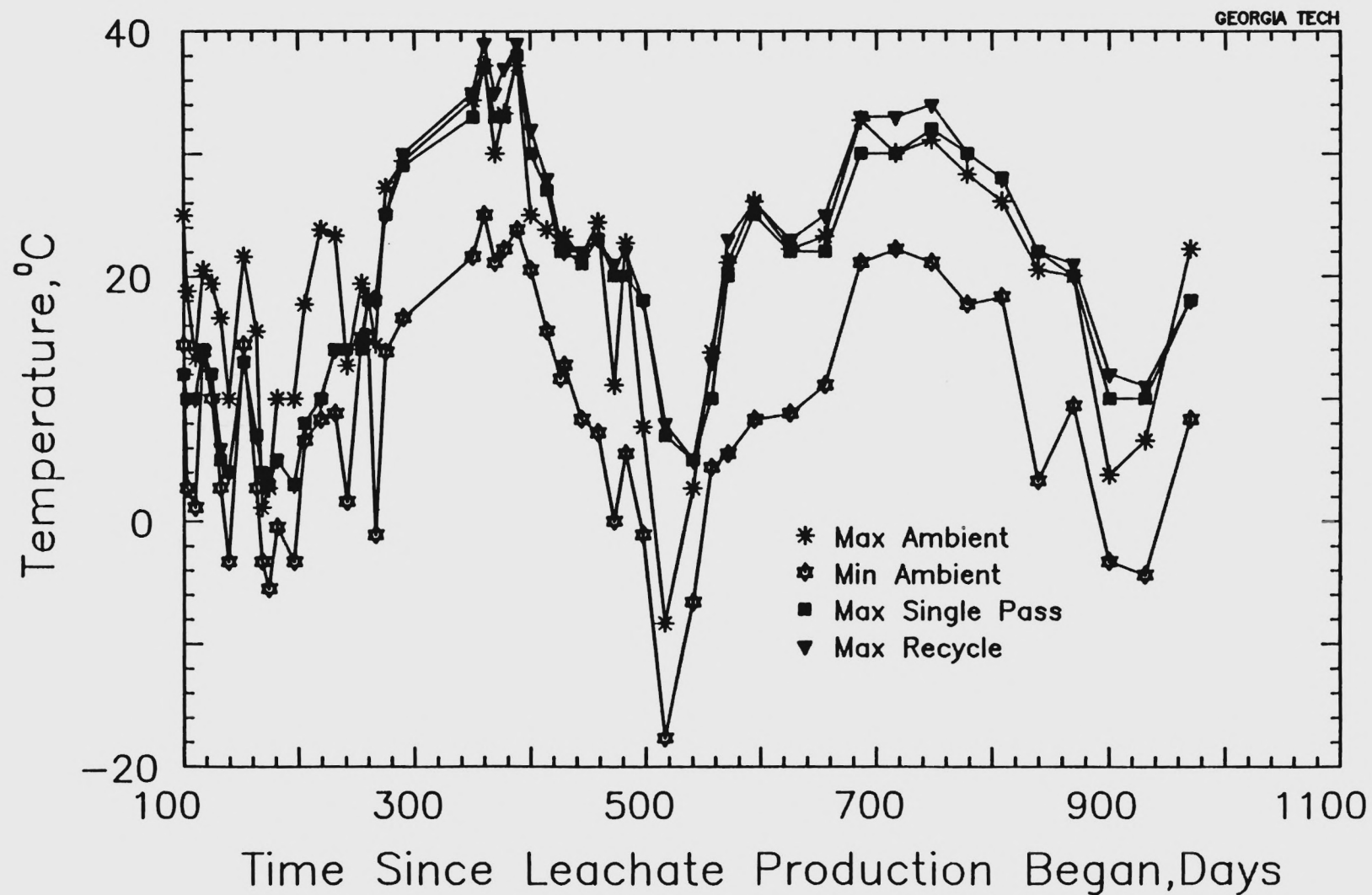


Figure 15. Maximum and Minimum Ambient Temperatures and Temperatures Within Cell A and Cell B.

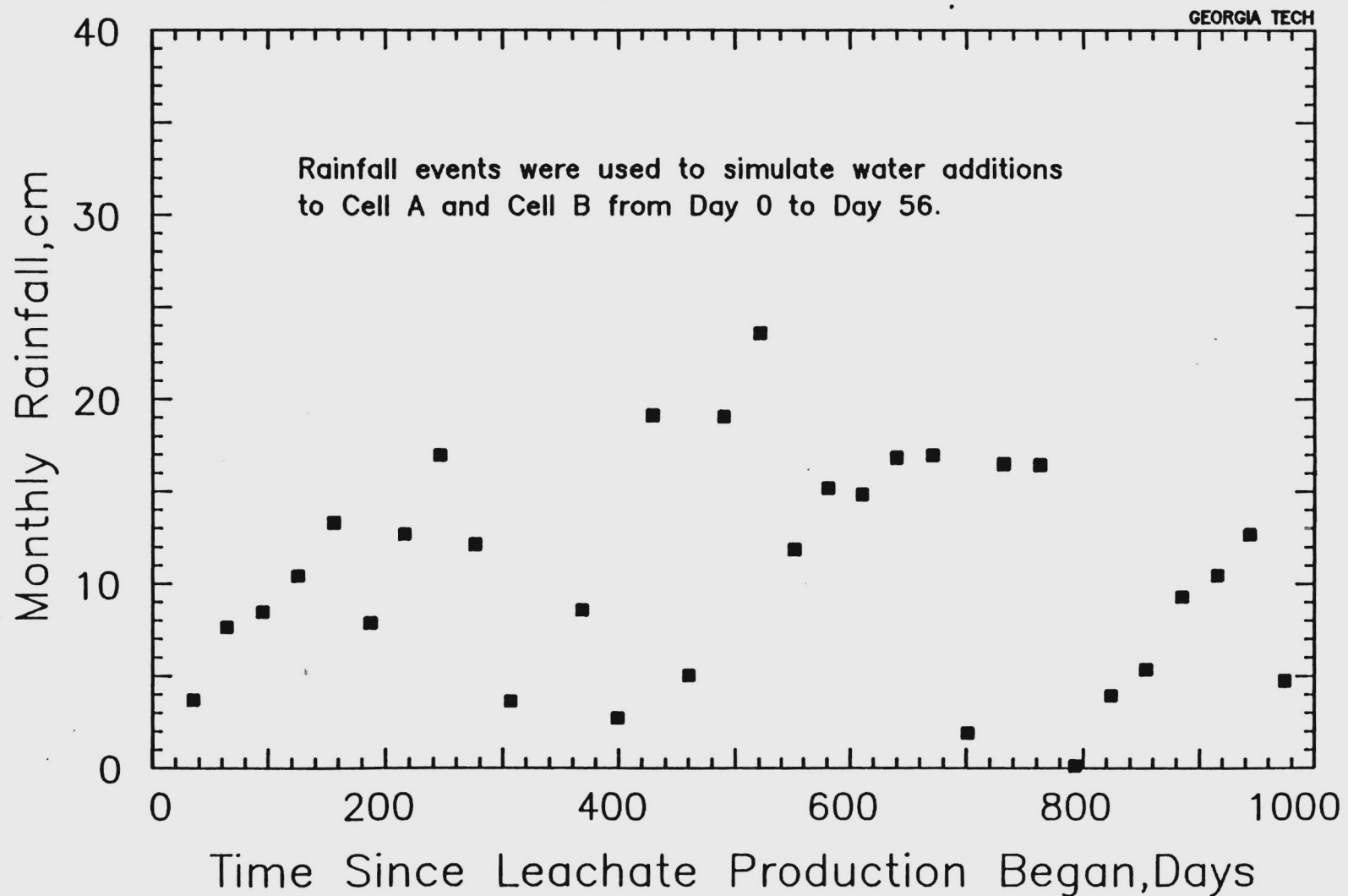


Figure 16. Rainfall (Cumulative Monthly) in Atlanta During the Study (Source NOAA)

degrade the complex constituents in the leachate and the refuse, thus providing an on-site treatment of the leachate generated from the landfill cell.

The causes for changes in constituents present in leachate from Cell A, in contrast to Cell B, were a combination of additions of rainfall-equivalent volumes of water, washout due to removal of leachate from the cell, and a result of microbially-mediated stabilization processes that occur in the landfill. Thus, the leachate collected from Cell A during the early stages of landfill stabilization, particularly, the Initial Adjustment Phase, the Transition Phase, and the Acid Formation Phase would require external treatment. In addition, since the addition of rainfall-equivalent volumes to both cells was terminated on Day 56, the changes in constituents present in leachate from both cells due to dilution were minimized beyond Day 56.

4.4.1 Organic Pollutant Parameters

Organic pollutant parameters (COD, BOD₅, and TOC) were used to reflect the polluttional impact of the leachate produced from the landfill cells. The variations in COD, BOD₅, and TOC along with the changes in ratios of BOD₅/COD and COD/TOC are presented in Figures 17-21.

As indicated in Figures 17-19, COD, BOD₅, and TOC exhibit a similar trend, wherein an increase is followed by a decrease in the concentrations. During the period COD, BOD₅, and TOC concentrations decreased, the pH of leachate (Figure 25) increased, and the ORP (Figure 28) and the total volatile fatty acids (Figures 22 and 23) decreased. The decrease in COD, BOD₅, and TOC in leachate from Cell B

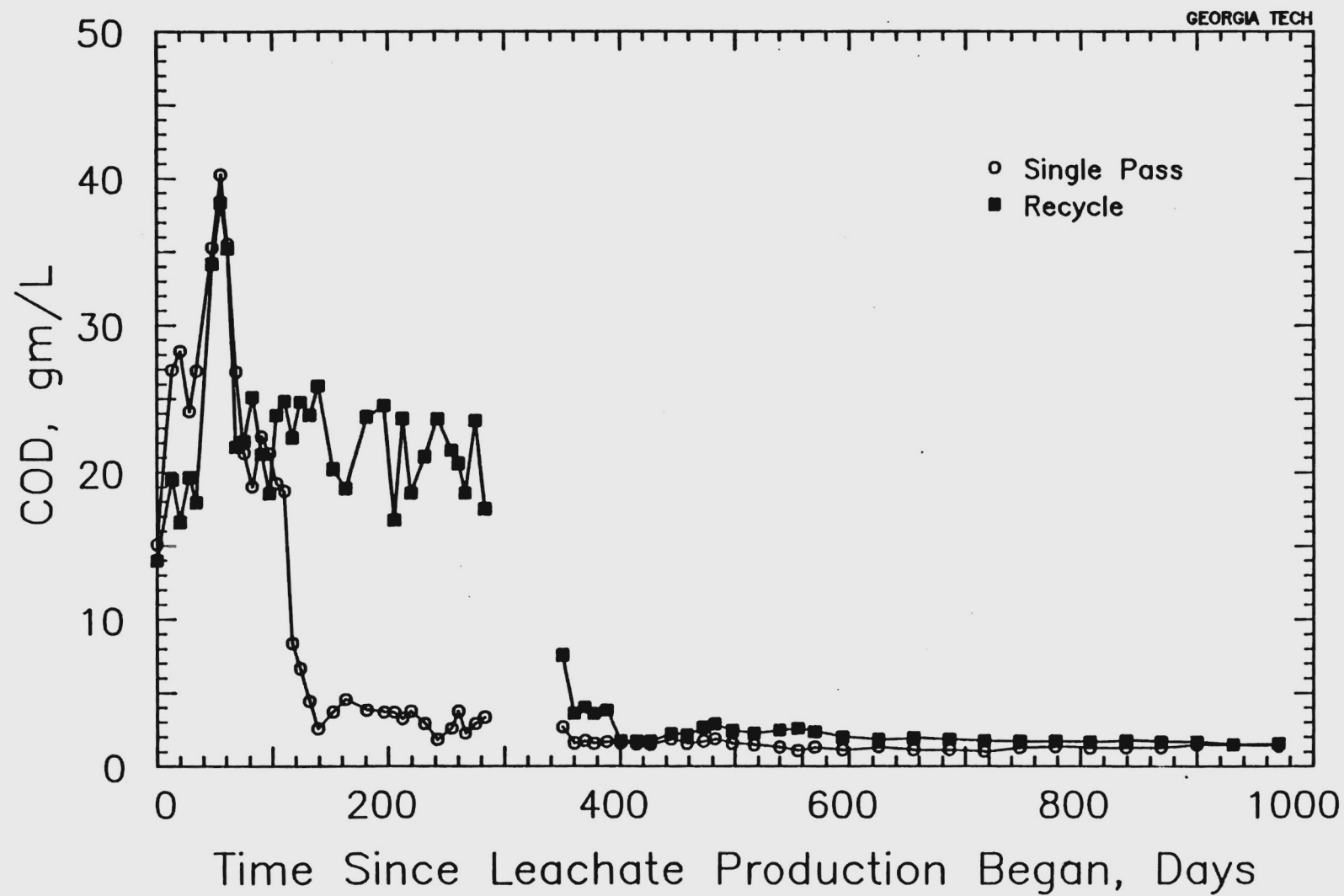


Figure 17. COD of Leachate from Both Cells.

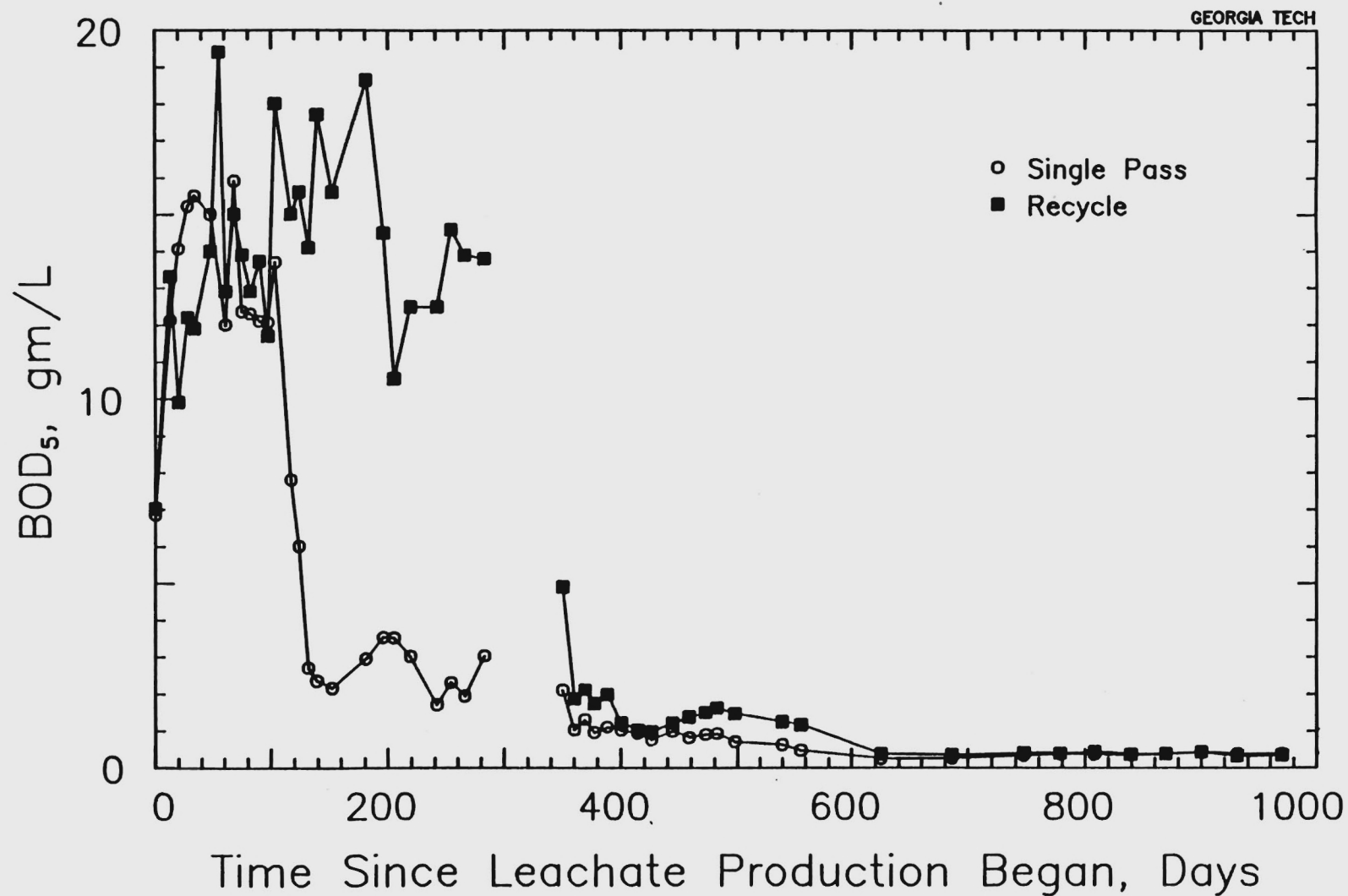


Figure 18. BOD₅ of Leachate from Both Cells.

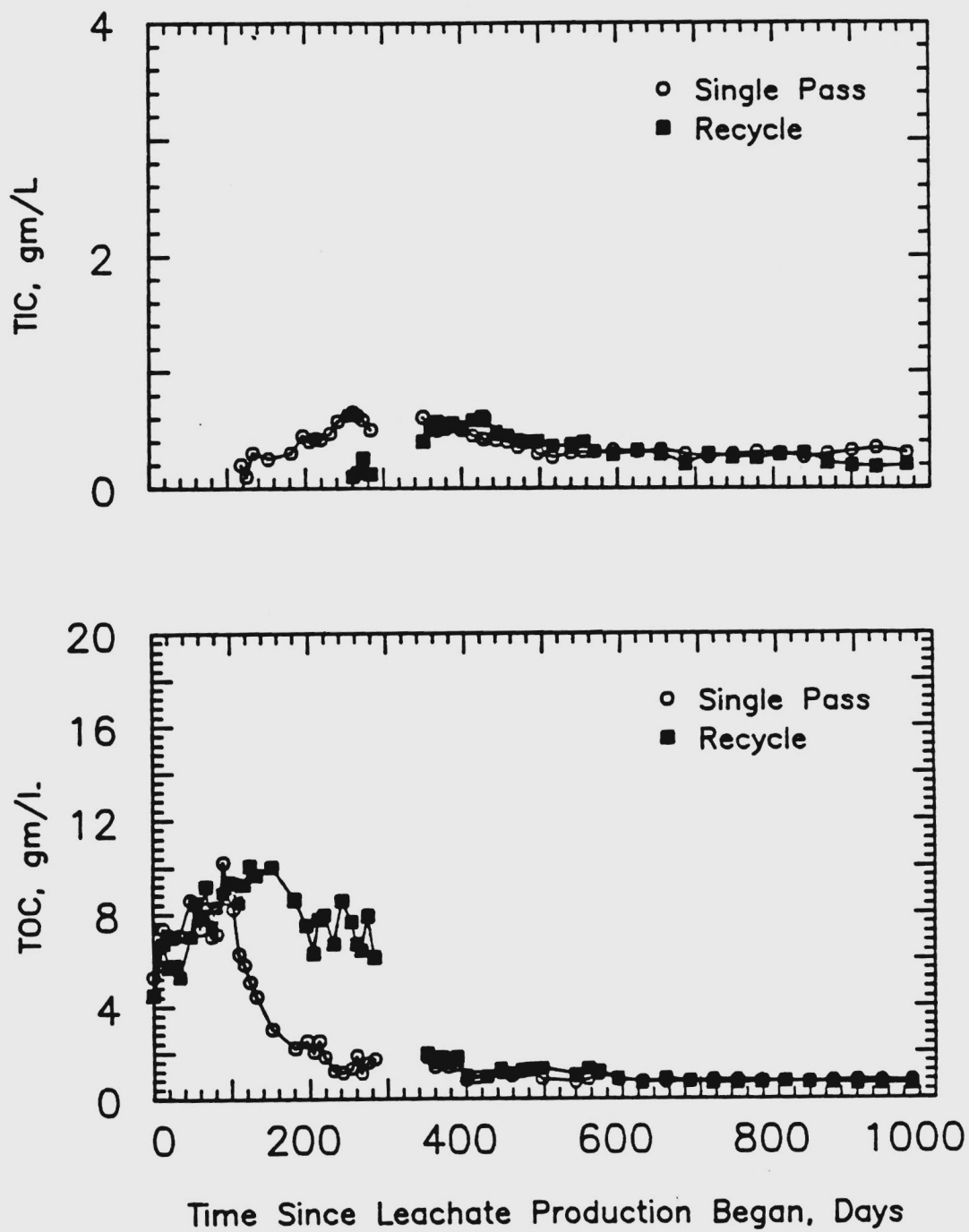


Figure 19. TOC and TIC of Leachate from Both Cells.

occurred during the period (Day 290 to Day 350) when leachate collection was suspended. The cells, however, were tested for gas leaks during this period. Had leachate samples been collected and analyzed during this period (Day 290 to Day 350), the analyses would have reflected the decrease in COD, BOD₅, and TOC in leachate and expected changes in other indicator parameters. The rate of decrease in these parameters in leachate from Cell B, however, can be envisioned as similar to the rate of decrease in corresponding parameters in the leachate from Cell A, since similar microbially-mediated stabilization processes occurred in both cells.

The initial concentrations of COD, BOD₅, and TOC in leachate collected from Cell A were 15,100 mg/L, 6,849 mg/L, and 5,300 mg/L, respectively, whereas the final ones were 1,349 mg/L, 370 mg/L, and 767 mg/L. In comparison, the initial concentrations of COD, BOD₅, and TOC in leachate collected from Cell B were 13,975 mg/L, 7,025 mg/L, and 4,500 mg/L, respectively, whereas the final ones were 1,522 mg/L, 342 mg/L, and 609 mg/L. The maxima of COD, BOD₅, and TOC concentrations in leachate from Cell A were 40,256 mg/L (Day 55), 22,200 mg/L (Day 55) and 10,200 mg/L (Day 90), respectively. In contrast, the maxima of COD, BOD₅, and TOC concentrations in leachate from Cell B were 38,327 mg/L (Day 55), 19,400 mg/L (Day 55), and 10,050 mg/L (Day 124), respectively. The changes in the maxima and the final concentrations of COD, BOD₅, and TOC in leachate from Cell A represent reductions of 96%, 98%, and 92%, respectively. In comparison, the changes in the maxima and the final concentrations of COD, BOD₅, and TOC in leachate from Cell B represent reductions of 96%, 98%, and 94%, respectively.

The delay in the occurrence of maxima of COD, BOD₅, and TOC in leachate from Cell B as compared to Cell A can be attributed to the absence of washout of these parameters from Cell B due to recirculation of leachate.

The decline in COD, BOD₅, and TOC concentrations in the leachate from Cell A between Days 55 and 139 was due to a combination of washout of the organic constituents and in situ treatment. During the same period, COD, BOD₅, and TOC concentrations in the leachate from Cell B did not exhibit a similar decline since washout of the organic constituents was prevented by recycling leachate to the top of the refuse. In either case, concentrations were influenced by the relative dilution effects of water additions. Although gas production wasn't monitored because of the presence of gas leaks in the cells, the production of gas concomitantly with the decrease in the organic pollutant parameters is envisioned.

The TIC concentrations in the leachate samples from both cells are presented in Figure 19. As shown, the initiation of TIC concentrations in leachate in measurable levels occurred with an increase in the pH of leachate from acidic levels to neutral levels (Figure 25), a decrease in the volatile fatty acids in leachate (Figures 23 and 24) and a shift in the primary buffer system in the leachate from that of acetate to carbonate.

Ratios of BOD₅/COD and COD/TOC are presented in Figures 20 and 21, respectively. The BOD₅/COD ratio is indicative of the biodegradability of the leachate; a decrease in this ratio indicates a decrease in biodegradability and consequently, a maturation of stabilization processes. The COD/TOC ratio is indicative of the oxidized state of the organic carbon that is available for biodegradability; a decrease in this

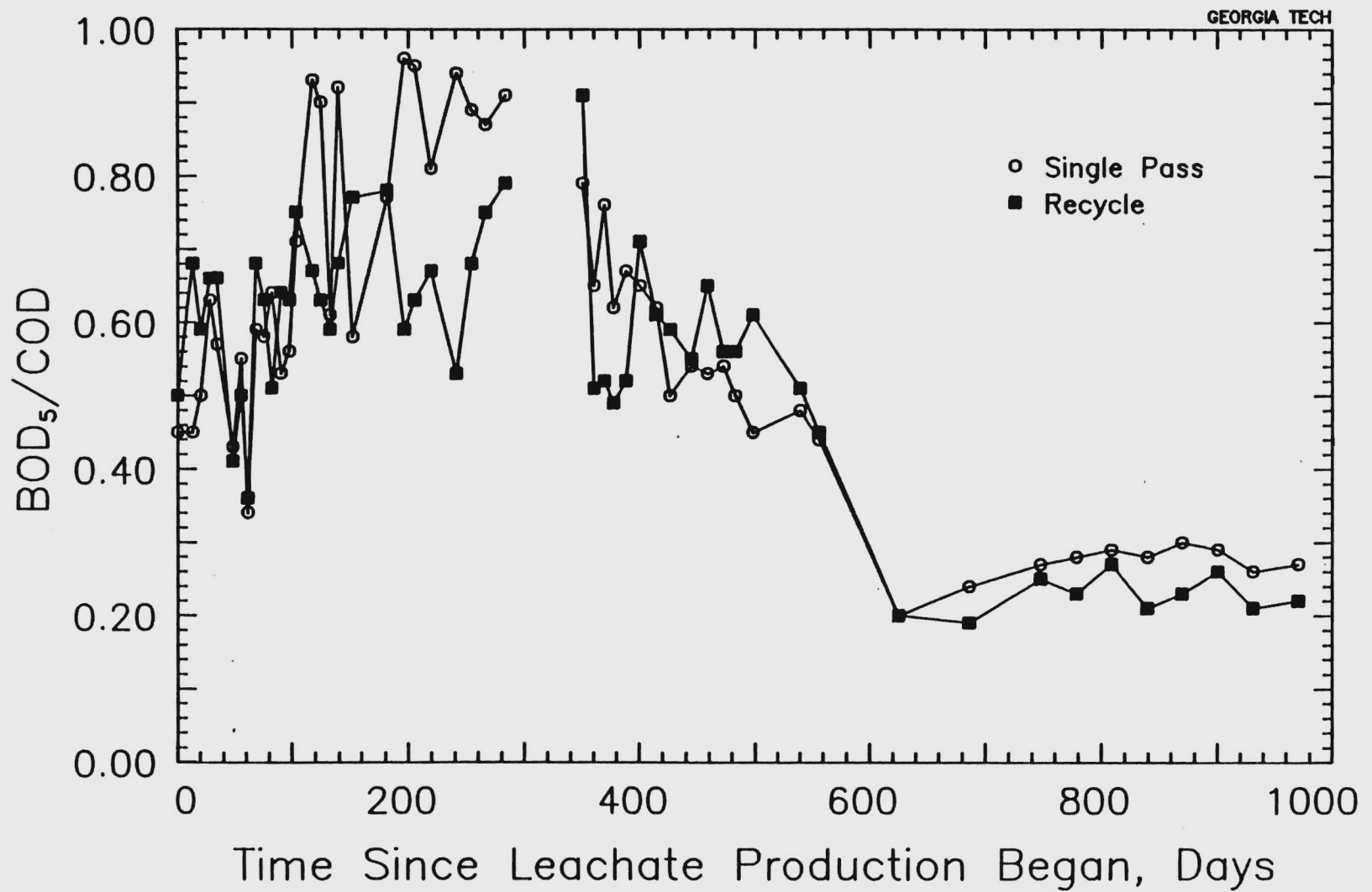


Figure 20. BOD_5/COD Ratio of Leachate from Both Cells.

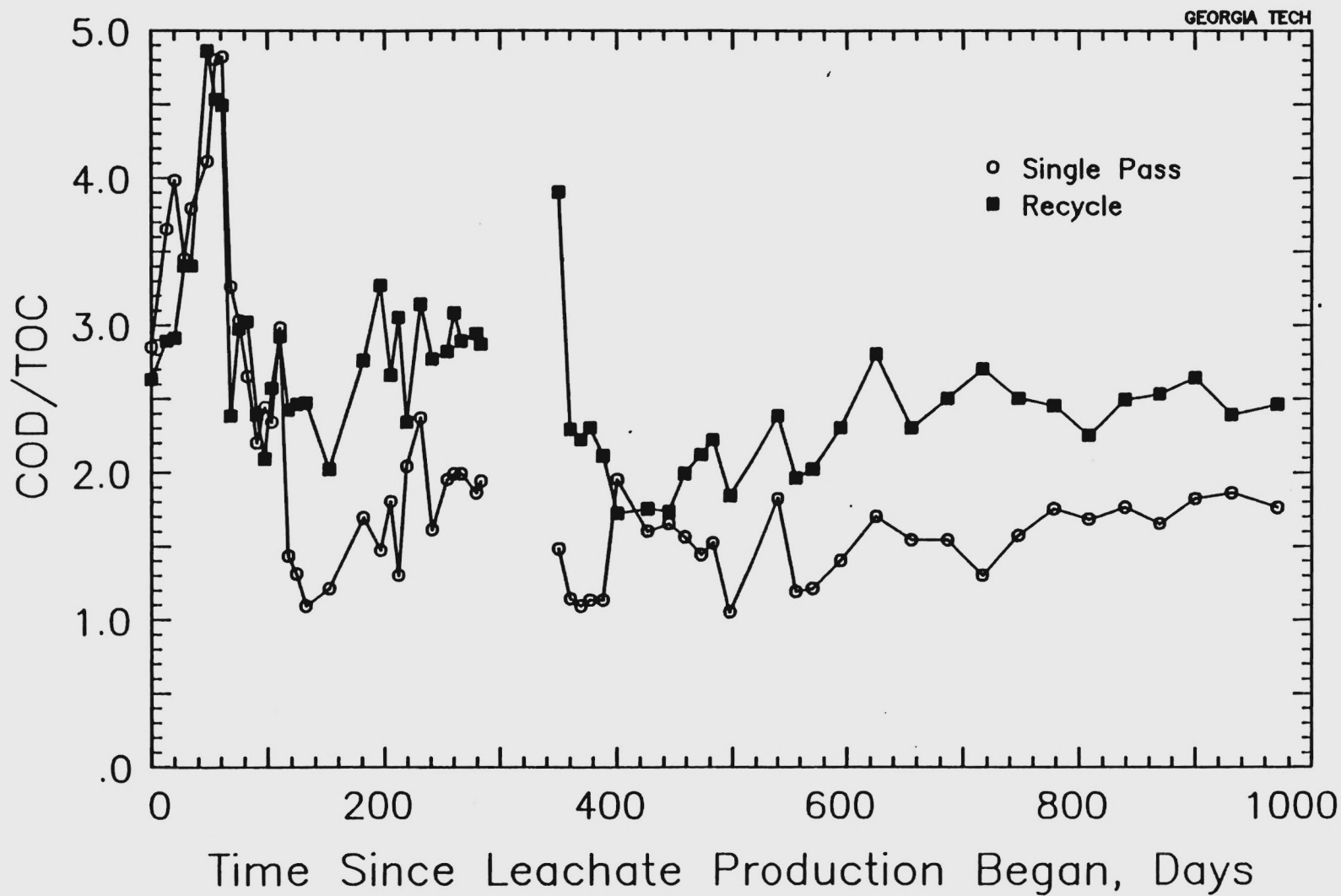


Figure 21. COD/TOC Ratio of Leachate from Both Cells.

ratio represents a more oxidized state of organic carbon due to an increase in the presence of products of microbially-mediated processes which lead to the maturation of stabilization processes. As expected, these ratios decreased with the progress of landfill stabilization except from Day 205 to Day 290 when the BOD₅/COD ratio increased for the leachate produced from Cell B; this increase was due to an increase in the total volatile acid concentration which resulted from increase in propionic acid concentration in the leachate during the same period (Figure 23). The accumulation of propionic acid is attributable to the prevention of microbially-mediated degradation of propionic acid due to acidic conditions of the leachate.

4.4.2 Volatile Fatty Acids

In a landfill undergoing anaerobic stabilization, the concentration of volatile fatty acids in the leachate produced from the fill is one of the most important indicator parameters. Indeed, the stage and the progress of anaerobic conversion may be reflected by the concentration of these low molecular weight volatile fatty acids, which include acetic, propionic, isobutyric, butyric, and valeric acids.

The concentrations of individual and total volatile fatty acids in leachate from Cell A and Cell B are presented in Figures 22 and 23, respectively. As expected, the most abundant acid was acetic acid followed by butyric, propionic, valeric, and isobutyric acids. The maxima of total volatile fatty acid concentrations in leachate samples collected from Cell A and Cell B were 20,892 mg/L as acetic acid (Day 41) and 12,857 mg/L as acetic acid (Day 68), respectively. The total volatile fatty acid concentrations in leachate from Cell A and Cell B were undetectable beyond Day 426.

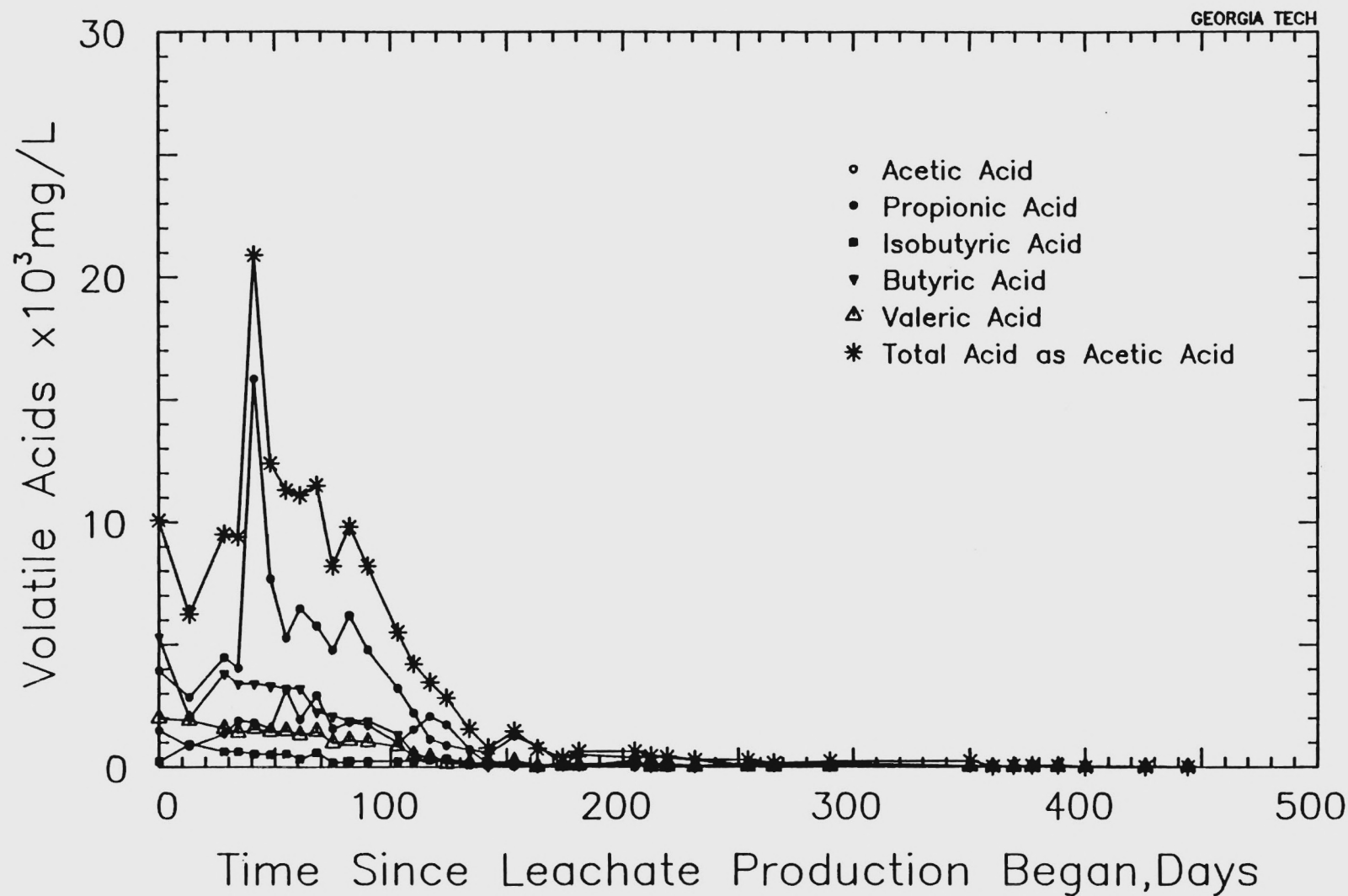


Figure 22. Individual and Total Volatile Fatty Acid Concentrations in Leachate from Cell A (Single Pass).

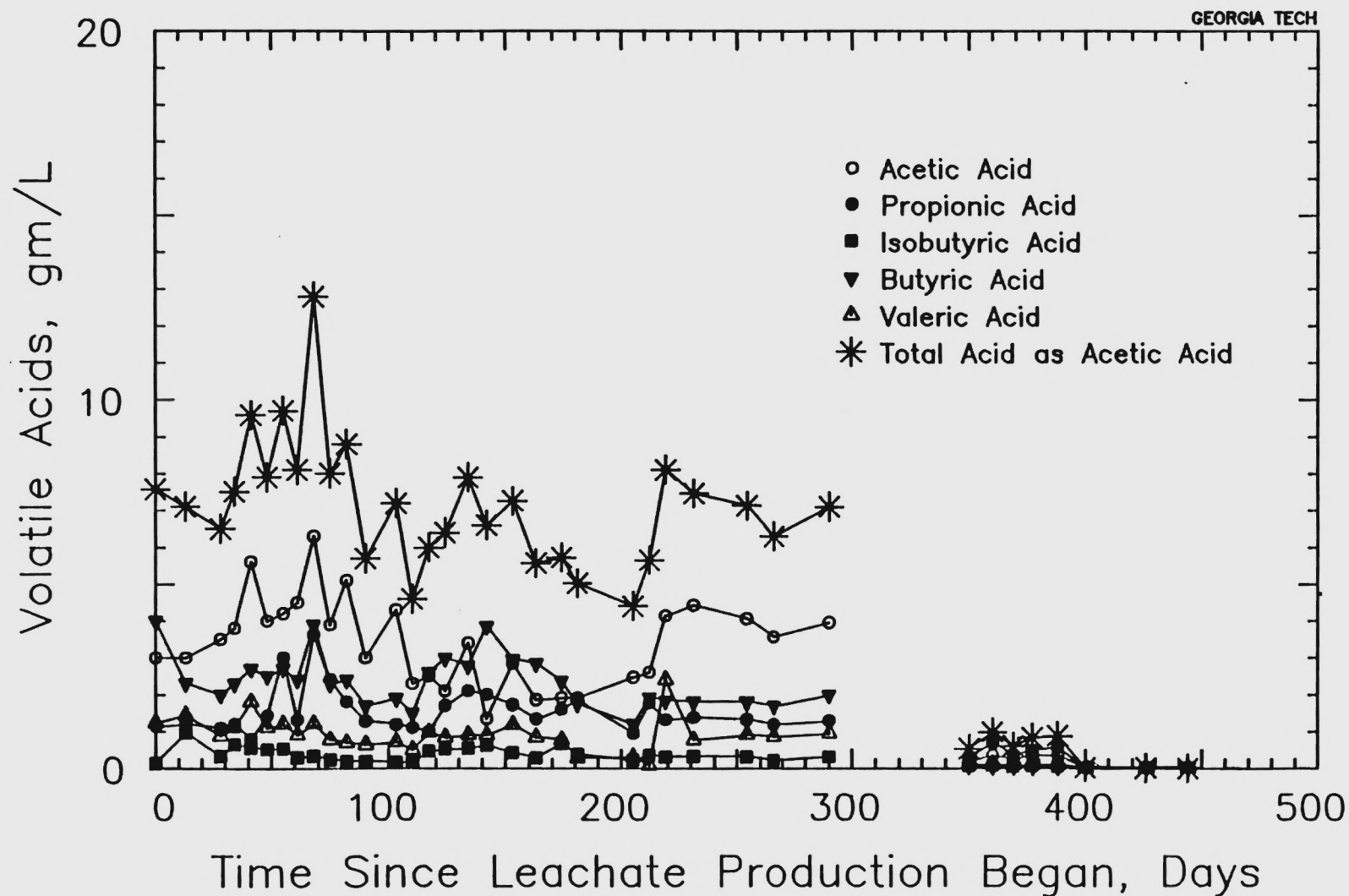


Figure 23. Individual and Total Volatile Fatty Acid Concentrations in Leachate from Cell B (Recycle).

As mentioned earlier (Section 4.4), the causes of changes in the volatile fatty acid concentrations in leachate from Cell A and Cell B were different. After reaching a maximum of 20,892 mg/L as acetic acid on Day 41, the concentration of total volatile acid in leachate from Cell A steadily decreased due to washout and biodegradation; this decrease was accompanied with an increase in the pH (Figure 24). In comparison, during the same period (from Day 41 to Day 152), the leachate from Cell B did not exhibit a similar decrease in the pH and the total volatile acid concentration. Consequently, the frequency of leachate recycle was changed from once a day to once a week on Day 152 in an attempt to encourage the growth of methane formers by reducing its exposure to the high total volatile fatty acid concentrations and the low pH conditions prevailing in Cell B. This frequency of leachate recycle was maintained until Day 350. The frequency of leachate recycle was reduced to encourage the onset of methane formation phase since the acidic condition of the leachate was preventing the onset of the methane fermentation phase. During this period (Day 152 to Day 350), the pH of leachate increased to neutral pH levels and the total volatile acid concentration decreased concomitantly, thereby indicating a successful recovery of Cell B from acidic conditions. From Day 350 to Day 570 leachate was recycled once a day; thereafter, the frequency of leachate recycle was reduced to once a week, since most of the indicatory parameters (COD, BOD, TOC, pH, ORP, alkalinity, and conductivity) did not exhibit significant changes.

During the initial stages of landfill stabilization the organic carbon present in the total volatile acid constitutes a significant

fraction of the TOC of leachate produced from the landfill. As stabilization progresses, the magnitude of this fraction decreases, due to the decrease in the total volatile fatty acid concentration. This decrease in the total volatile fatty acid concentration in the leachate is attributable to the landfill being in methane forming phase or final maturation phase, and a paucity of readily biodegradable organic compounds in the leachate. During the period when decreases in the volatile acid concentrations occurred, gas production couldn't be measured due to the presence of gas leaks in the liner. The production of gas, however, is expected with a decrease in the volatile acid concentrations.

The ratios of TOC equivalent of total fatty acid to the TOC of leachate from Cell A and Cell B, respectively, are presented in Figure 24. As shown, the decrease in this ratio corresponds with the decrease in the total volatile fatty acids (Figure 24) and an increase in the pH.

4.4.3 pH

The pH of the leachate produced from a landfill undergoing stabilization is a critical indicator parameter since it is an important determinant of the presence of volatile fatty acids and type and capacity of the predominant buffer system (viz., acetate or carbonate). Increases in the pH of leachate are usually accompanied by decreases in the volatile fatty acid concentrations.

The pH values of the leachate samples collected from Cell A and Cell B during this study are presented in Figure 25. The initial and the final pH values of the leachate from Cell A were 5.2 and 7.1, respectively. The pH values of the leachate from Cell A were within a

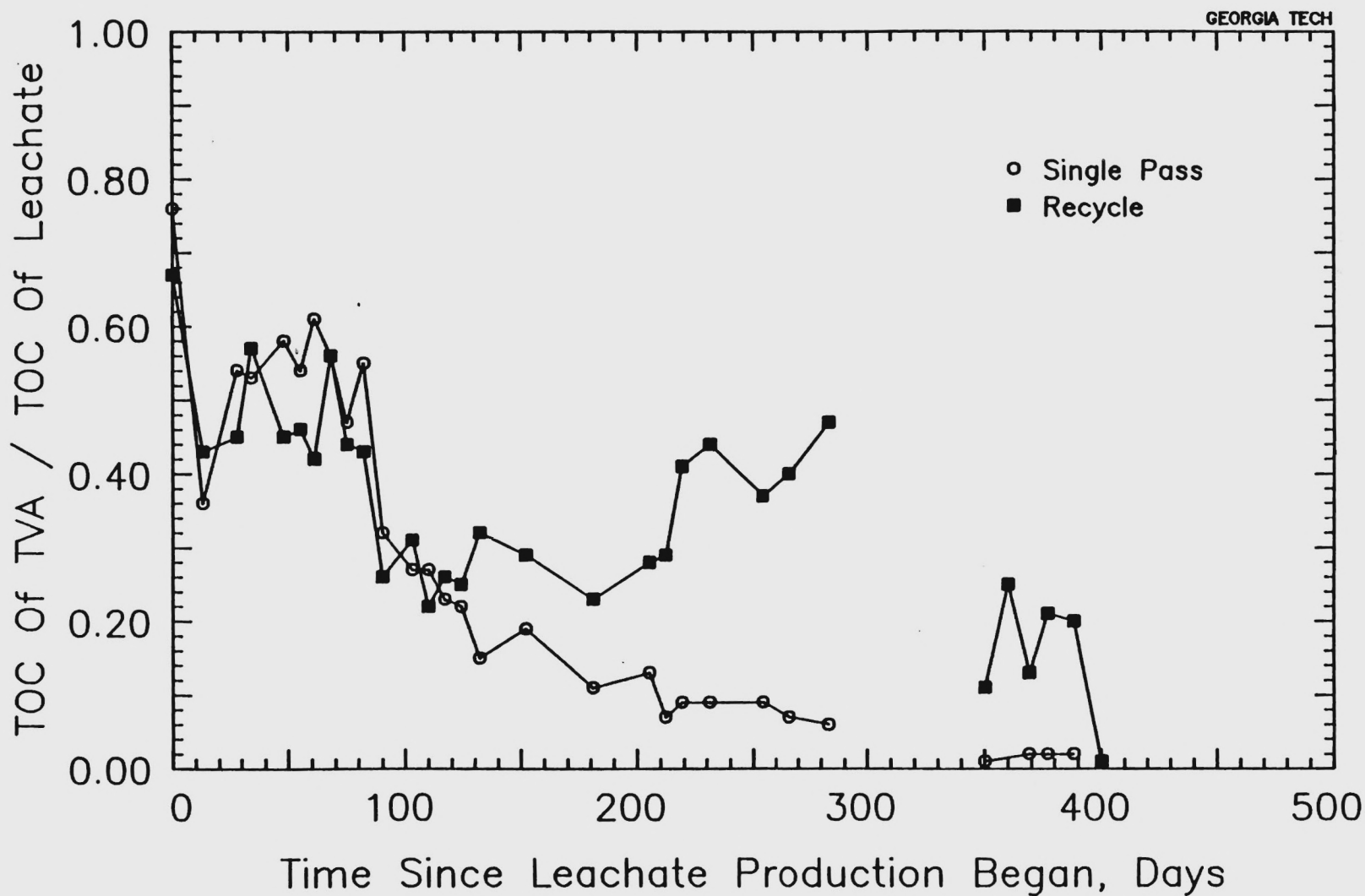


Figure 24. Ratio of TOC Equivalent of Total Volatile Fatty Acid to TOC of Leachate from Both Cells.

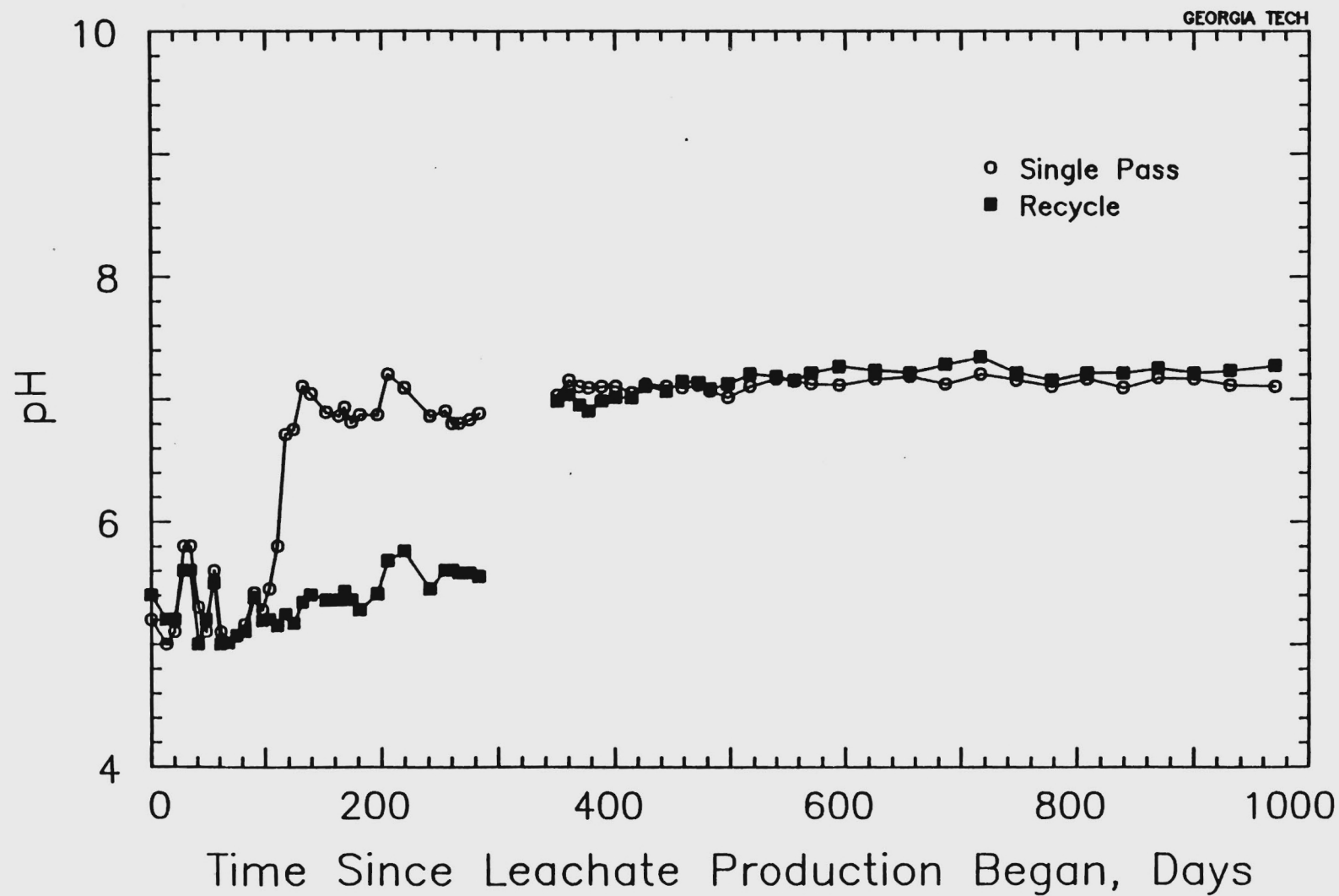


Figure 25. pH of Leachate from Both Cells.

range 5.00 to 5.60 from Day 0 to Day 103, then increased to 7.14 in 36 days. This increase in pH was concomitant with a decrease in the total volatile fatty acid concentration and a shift in the buffer system from acetate to carbonate. Thereafter, the pH values of the leachate samples from Cell A were within a range 6.80 to 7.21.

The initial and the final pH values of the leachate from Cell B were 5.40 and 7.21, respectively. During the period of daily recycling of leachate (Day 0 to Day 152), the pH values were within a range 5.00 to 5.40. From Day 152 to Day 350, when weekly recycle of leachate was practised, the pH values were within a range 5.28 to 6.98; the increase in pH presumably due to the growth of methane formers within microenvironments of the landfill. This cause for an increase in the pH was further substantiated by a concomitant decrease in the total volatile acid concentration. From Day 350 to Day 970, the pH values were within a range 6.98 to 7.27. Similar to the behavior exhibited by the leachate from Cell A, the buffer system present in the leachate from Cell B shifted from acetate to carbonate as the pH increased from 5.00 to 7.27.

4.4.4 Alkalinity

The alkalinity of leachate is indicative of its buffer capacity. Also, alkalinity together with pH and total volatile fatty acid concentration indicate the predominant type of buffer system present in the leachate produced from a landfill.

The total alkalinity present in leachate samples collected from Cell A and Cell B during this investigation is presented in Figure 26. The total alkalinity present in leachate from Cell A increased from

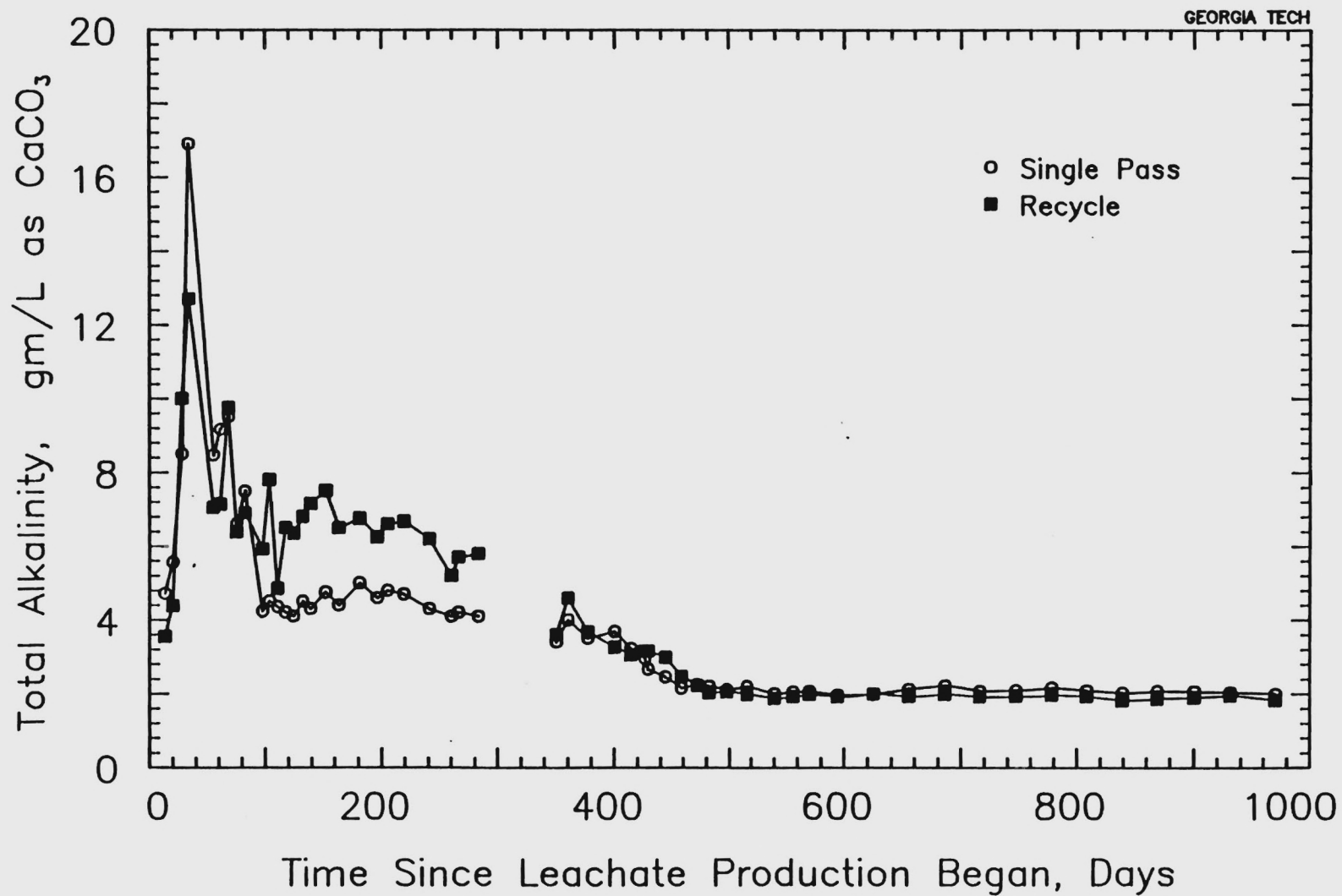


Figure 26. Total Alkalinity of Leachate from Both Cells.

4,717 mg/L as CaCO_3 (Day 13) to 16,900 mg/L as CaCO_3 (Day 34), then decreased to 1,990 mg/L as CaCO_3 (Day 970). In comparison, the total alkalinity present in leachate from Cell B increased from 3,547 mg/L as CaCO_3 (Day 13) to 12,700 mg/L as CaCO_3 (Day 34), then decreased to 1,825 mg/L as CaCO_3 (Day 970).

The increase in total alkalinity of leachate was due to increase in total volatile acid concentration since the volatile acids would add to the alkalinity pool as volatile acid salts. The decrease in volatile acid concentration would reduce the contribution of the volatile acids to the alkalinity pool. Thus, the trend exhibited by total alkalinity (Figure 25) of leachate closely follows the trend exhibited by volatile acids (Figures 22 and 23). An increase in pH (Figure 24) along with a decrease in volatile acid concentration and total alkalinity indicated a shift in the predominant buffer system from acetate to carbonate.

4.4.5 Conductivity

Conductivity of leachate is a measure of the presence of ionic species and along with total volatile fatty acid and alkalinity present in the leachate produced from a landfill undergoing stabilization, tends to have an inverse relationship with pH. Variations in conductivity result from microbially mediated processes, physical-chemical removal processes and dilution effects which occur within the leachate and the landfill environment.

The conductivity of a landfill leachate is indicative of its ionic strength. Since ionic strength of leachate inversely affects the activity coefficient of ions present in leachate, an increase in

conductivity of leachate results in a decrease in the activity coefficient, thereby leading to an increase in the concentration of ions in leachate.

The conductivity of the leachate samples collected from Cell A and Cell B are shown in Figure 27. As shown, the conductivity of leachate from Cell A increased from 6,400 μmhos to 11,500 μmhos , then decreased to 2,150 μmhos , whereas the conductivity of leachate from Cell B increased from 5,400 μmhos to 12,600 μmhos , then decreased to 1,855 μmhos .

The increases in the conductivity of leachate from Cell A and Cell B were due to leaching of organics and inorganics from the refuse and microbially mediated production of volatile acids. After reaching a maximum, the conductivity of leachate decreased due to a decrease in total volatile fatty acid concentrations, organic and inorganic salts, and metals, which were precipitated by sulfides. In addition, washout contributed to the decrease in conductivity of leachate from Cell A.

4.4.6 Oxidation-Reduction Potential (ORP)

The ORP of the leachate from a landfill is an indicator parameter which indicates the presence of oxidizing or reducing conditions within the leachate and the landfill environment. Except for the early phases of stabilization (Initial Adjustment Phase and Transition Phase), the ORP should reflect the presence of reducing conditions, particularly when methane fermentation is effective in converting the intermediate organic leachate constituents to more stabilized forms.

The ORP of leachate samples from Cell A and Cell B are presented in Figure 28. During the early phase of landfill stabilization (Day 0

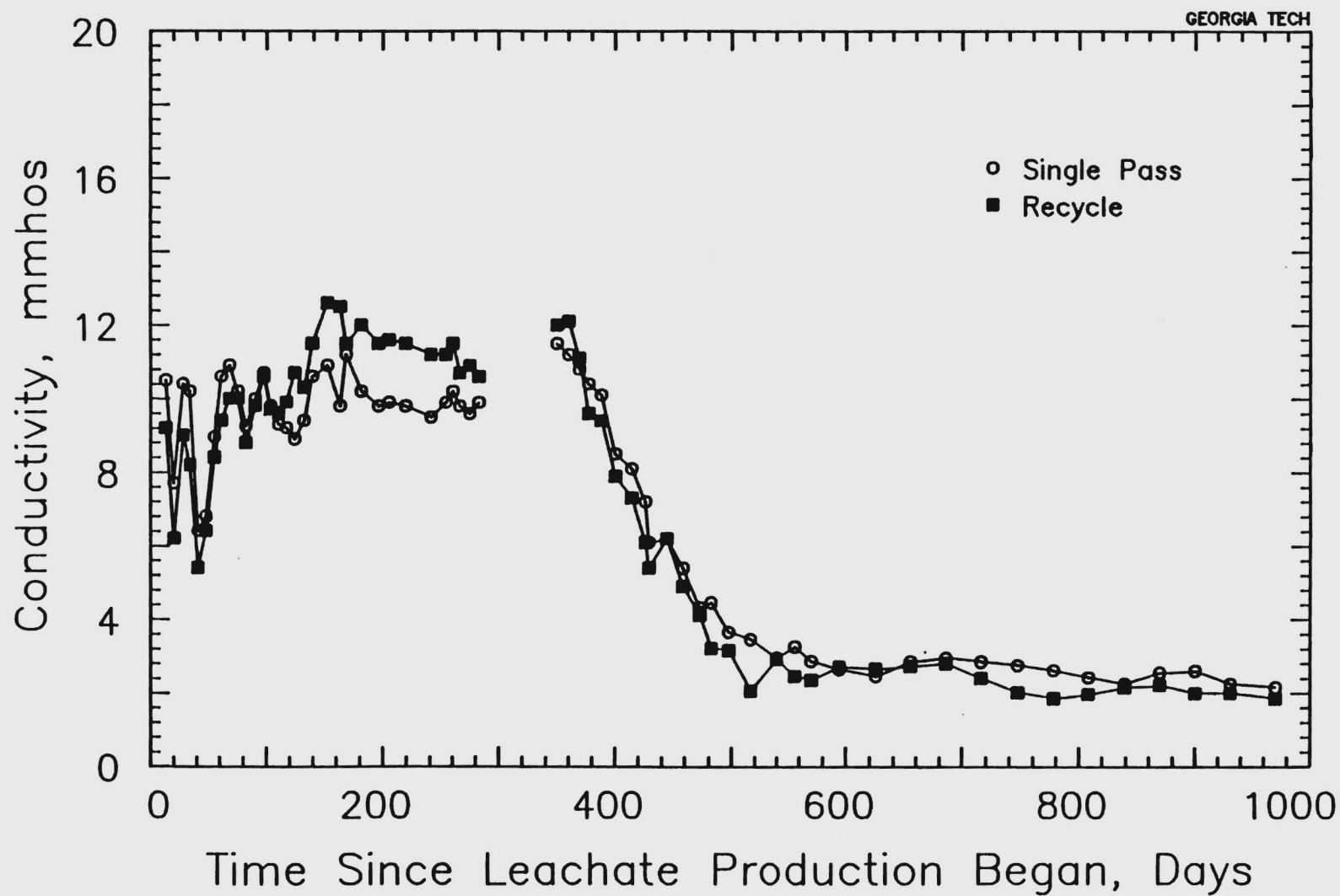


Figure 27. Conductivity of Leachate from Both Cells.

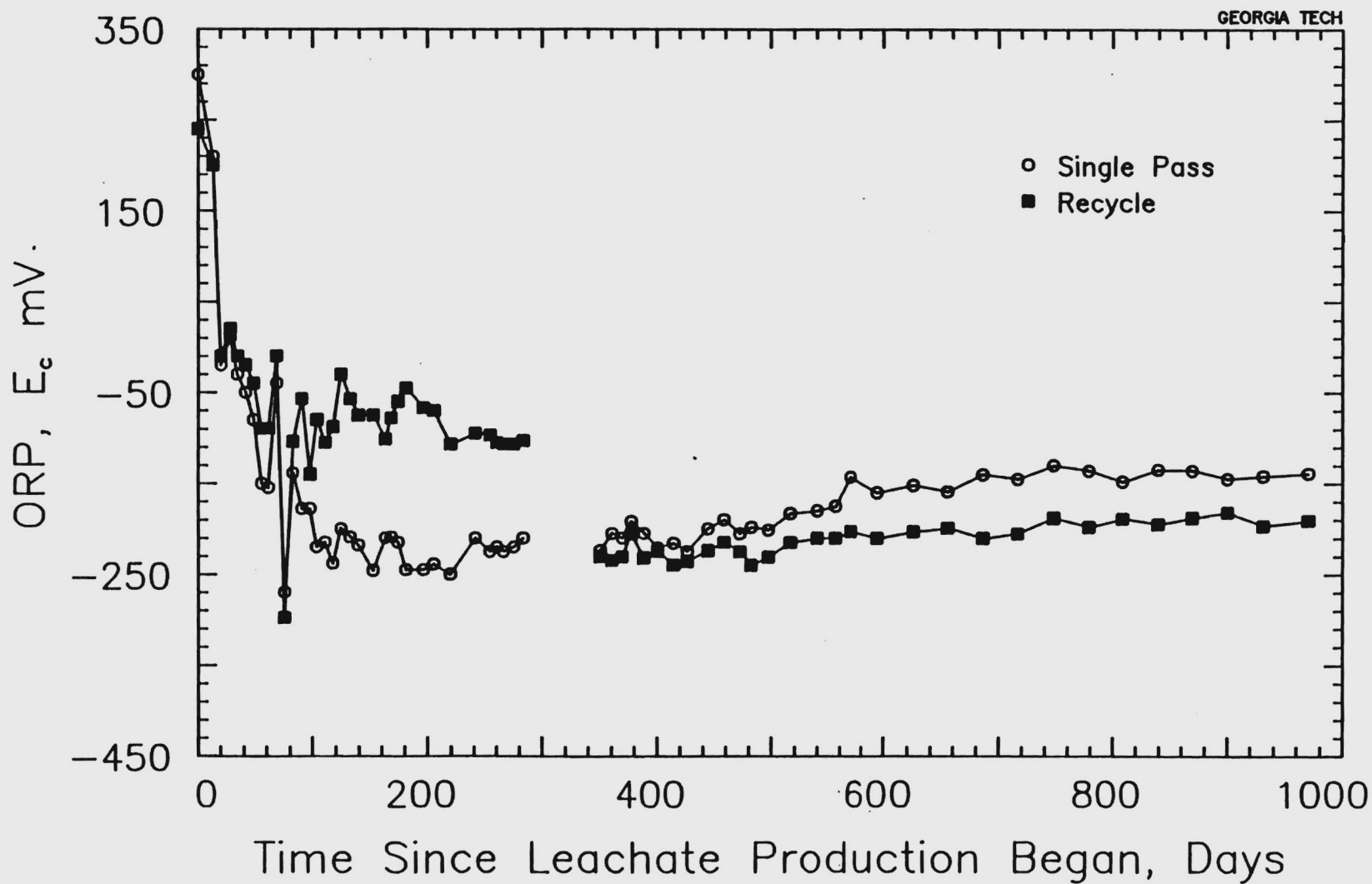


Figure 28. ORP of Leachate from Both Cells.

to Day 34) the ORP of leachate from both cells decreased from positive to negative values (Cell A: 300 mV E_C to -30 mV E_C ; Cell B: 240 mV E_C to -10 mV E_C) which indicated the onset of reducing conditions within the leachates and the landfill environments. Thereafter, the ORP of the leachate from both cells remained negative till Day 970. ORP values of -400 mV to -500 mV that are optimum for methane fermentation were never recorded during the period of this investigation; this is probably due to inherent problems with the instrumental measurement of this parameter since the exposures of leachate, collected from a highly reducing environment, to atmosphere during measurement will record ORP values that are more positive than the actual ORP values.

Without new sources of oxidizing potential and entry of oxygen, the reducing environment established by microbially-mediated processes within a landfill will likely persist for an indefinite period of time. From Day 34 to Day 970 the ORP values reflected the existence of reducing conditions within both cells; these conditions along with increases in pH values and decreases in volatile acids induced sulfide formation and precipitation of heavy metals as subsequently discussed.

4.4.7 Sulfide

Sulfide is produced within a landfill environment during the Acid Formation and the Methane Fermentation Phases of stabilization as a result of microbial reduction of sulfate under reducing conditions by microorganisms which are strict anaerobes and use sulfate as an electron acceptor. The sulfide so produced precipitates most heavy metals present in the leachate as metal sulfides.

The total sulfide concentration in the leachate samples from

Cell A and Cell B are presented in Figure 29. The total sulfide concentration in the leachate from Cell A increased from 0.23 mg/L (Day 43) to 0.85 mg/L (Day 283), then decreased to 0.1 mg/L (Day 778), whereas the total sulfide concentration in the leachate from Cell B increased from 0.45 mg/L (Day 350) to 4.50 mg/L (day 444), then decreased to 0.13 mg/L (Day 839). The maximum sulfide concentrations in the leachate samples from Cell A and Cell B corresponded to ORP values of -210 mV E_C and -224 mV E_C , respectively.

In a sulfate-sulfide system, the reduction of sulfate to sulfide usually occurs around -500 mV E_C . Since ORP of leachate is a very sensitive parameter, it usually increases sharply as soon as the leachate is exposed to atmosphere (for measurement); thus, the "as-measured" value will probably be higher than the actual value. The trends reported for leachate samples from both cells, however, would remain unchanged. Presence of sulfide in leachate samples was accompanied by precipitation of heavy metals (shown later in Figures 36-40). The maximum sulfide concentration (4.5 mg/L) in leachate from Cell B was higher than the maximum sulfide concentration (0.85 mg/L) in leachate from Cell A; this was attributed to the contrasting modes of management of leachate from Cell A (single pass) and Cell B (leachate recycle).

Sulfide exists in solution in three forms: H_2S , HS^- , and S^{2-} . The presence of these forms is pH dependent with H_2S and HS^- being the dominant forms at acidic pH values (pH=5.0). However, with an increase in the pH of leachate from 5 to 7 as is evidenced in landfills, sulfide is also present in S^{2-} form along with H_2S and HS^- ; this is important,

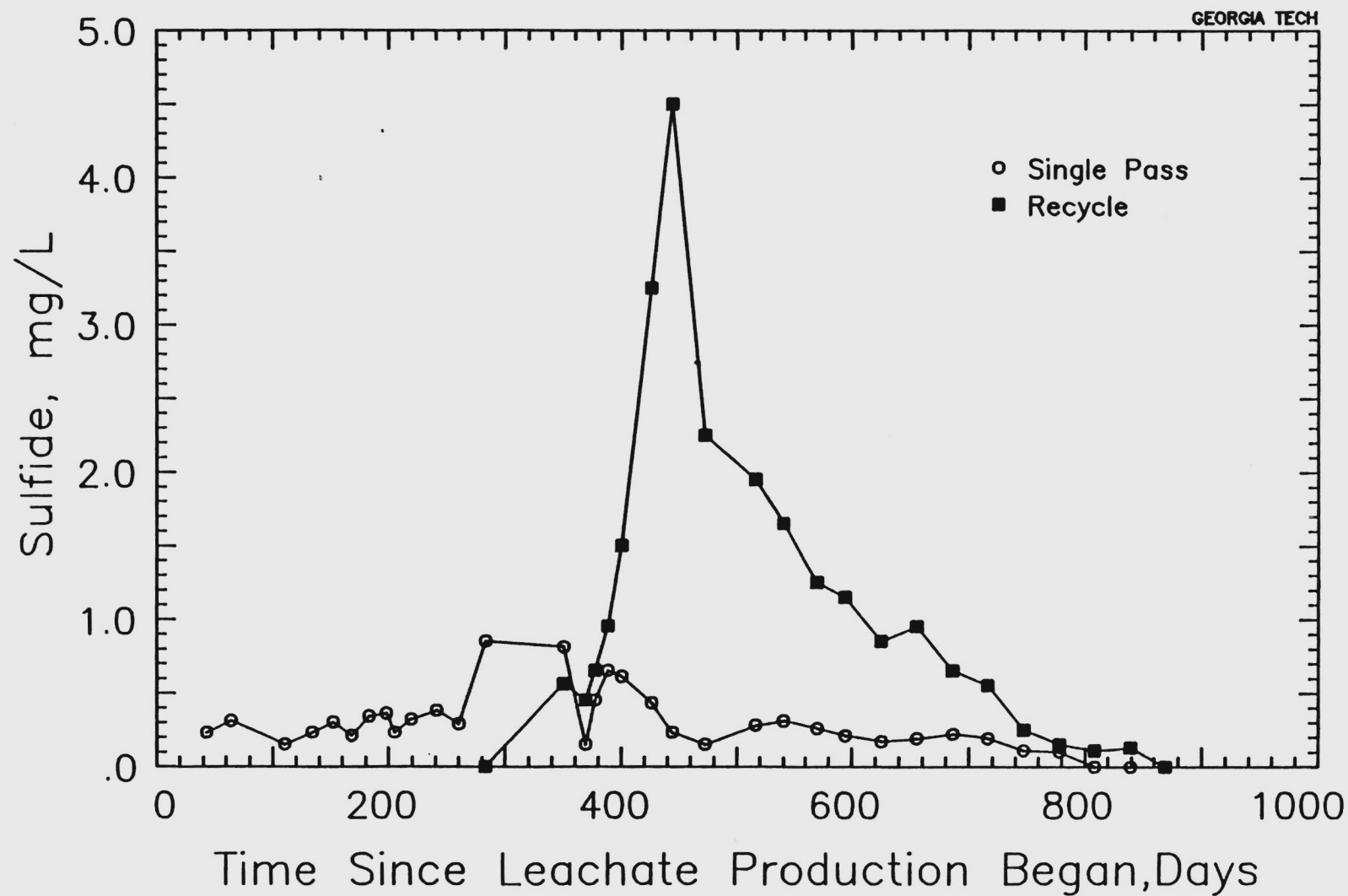


Figure 29. Sulfide Concentrations in Leachate from Both Cells.

since S^{2-} is responsible for precipitation of heavy metals.

4.4.8 Chloride

In an anaerobic system such as a landfill undergoing stabilization, a conservative constituent is often utilized as a tracer to assess dilution, mixing, and washout effects. Chloride (Cl^{-}) has been selected as a tracer, since it is essentially a non-reactive component with a high solubility in water. The chloride concentrations in the leachate samples from Cell A and Cell B are shown in Figure 30.

Chloride concentration in leachate from Cell A decreased from an initial value of 667 mg/L (Day 13) to 507 mg/L (Day 68), then increased to a maximum of 719 mg/L (Day 96); thereafter, the concentration decreased steadily to a final value of 165 mg/L (Day 970). In comparison, chloride concentration in leachate from Cell B decreased from an initial value of 470 mg/L (Day 13) to 372 mg/L (Days 55 and 68), then increased to a maximum of 713 mg/L (Day 110); thereafter, the concentration decreased steadily to a final value of 300 mg/L (Day 970).

The initial decline in the chloride concentration in leachate from both cells was due to dilution caused by continual water additions in rainfall-equivalent volumes to the cells through the distribution networks placed over the refuse in each cell to simulate incidental rainfall. The increase in concentrations can be ascribed to the leaching of chloride from the refuse. After reaching a maximum, the chloride concentration in leachate from both cells declined steadily, and as expected, chloride concentration in leachate from Cell A declined more than the chloride concentration in leachate from Cell B.

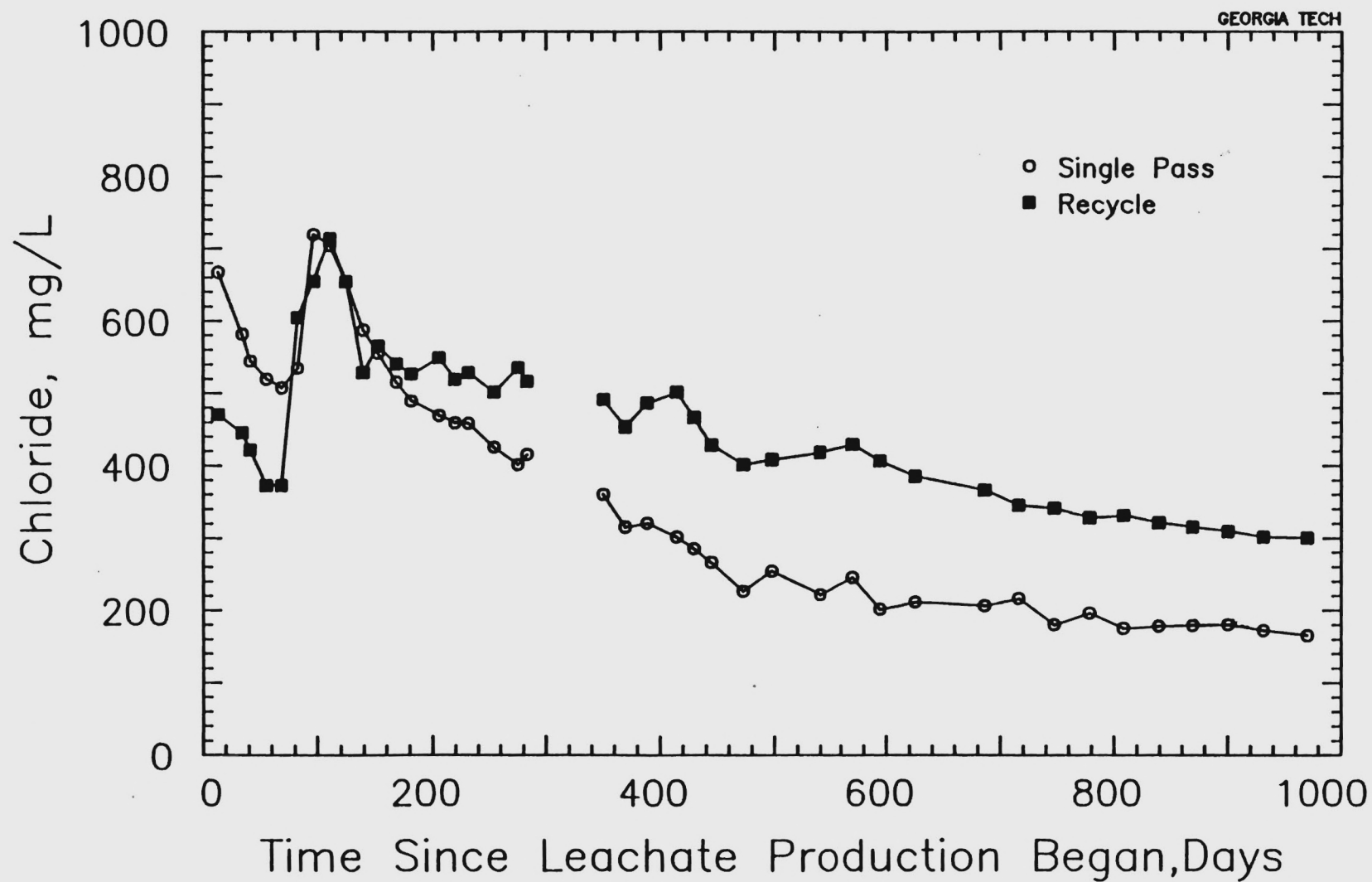


Figure 30. Chloride Concentration in Leachate from Both Cells.

This behavior can be ascribed to washout resulting from single pass operation.

A correction factor, based on chloride concentrations, was utilized to assess the changes in constituent levels in leachate samples from both cells. The correction factor (C.F.) is defined as follows:

$$C.F. = \frac{[Cl^-]_t}{[Cl^-]_{max}} \quad (3)$$

where;

C.F. = concentration factor

$[Cl^-]_t$ = chloride concentration at any time t, mg/L

$[Cl^-]_{max}$ = maximum chloride concentration, mg/L

The correction factors for leachate samples collected from both cells are shown in Figure 31.

The correction factor for leachate samples from Cell A decreased from 1.00 (Day 96) to 0.23 (Day 970), whereas that from Cell B decreased from 1.00 (Day 110) to 0.41 (Day 970). To further illustrate the washout effect, the "As-Measured" COD and the "Corrected" COD values of leachate samples from both cells are presented in Figure 32. Based on the maximum "Corrected" COD values of leachate samples from Cell A and Cell B, the reductions in COD by Day 970 were 89% and 95%, respectively. In comparison, the reduction based on maximum "As-Measured" COD values of leachate from both cells was 96%. Thus, the difference in reductions in COD values (89% vs. 96%) of leachate samples from Cell A can be attributed to washout.

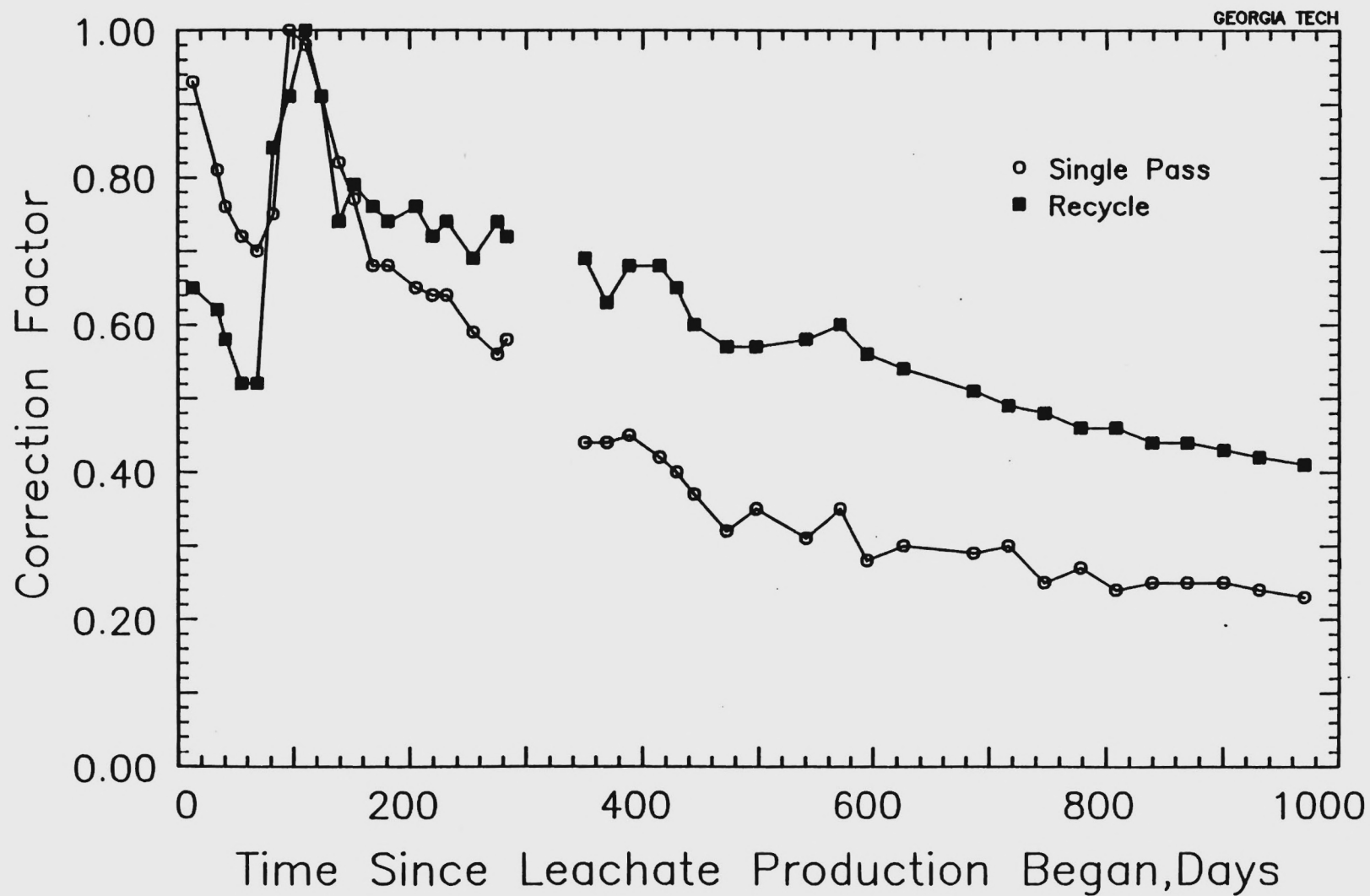


Figure 31. Chloride-Based Correction Factor for Washout of Leachate Constituents from Both Cells.

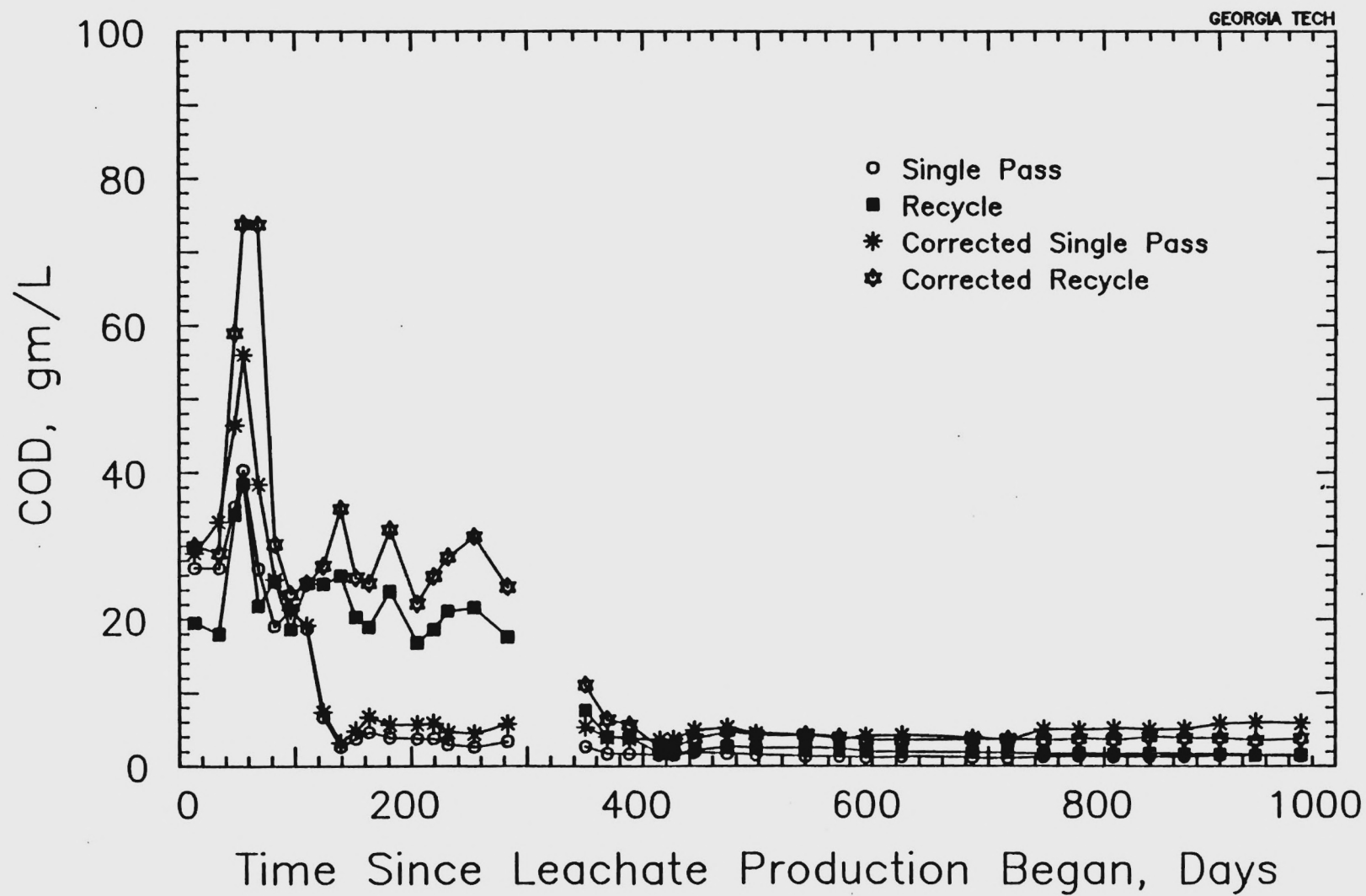


Figure 32. As-Measured COD and Dilution-Effect-Corrected COD of Leachate from Both Cells.

4.4.9 Ionic Strength and Activity

Ionic strength (I) indicates the intensity of the electric field in the leachate from a landfill. The ionic strength variations are caused by microbially-mediated processes, physical-chemical removal processes, and dilution effects within the leachate and the landfill environment.

The ionic strength (I) of leachate from Cell A and Cell B was computed using the following equation:

$$I = 1.6 \times 10^{-5} \times C \quad (4)$$

where:

I = Ionic Strength, M

C = Conductivity, μmhos

Equation 4 was reported by Lind (110) and Russell (111). The ionic strength of leachate from Cell A decreased from 0.168 M (Day 13) to 0.102 M (Day 41), then increased to 0.174 M (Day 68); between Days 68 and 350, the ionic strength fluctuated between 0.142 M and 0.184 M. After Day 350, the ionic strength declined steadily to 0.034 M (Day 970). In comparison, the ionic strength of leachate from Cell B decreased from 0.147 M (Day 13) to 0.086 M (Day 41), then increased to 0.201 M (Day 152). After Day 360, the ionic strength declined steadily to 0.029 M (Day 970; Figure 33).

The initial decline in the ionic strength of leachate from both cells can be ascribed to dilution. Reducing conditions coupled with enhanced biological conditions within the leachate and the landfill environment caused the production of ionic species such as sulfides and

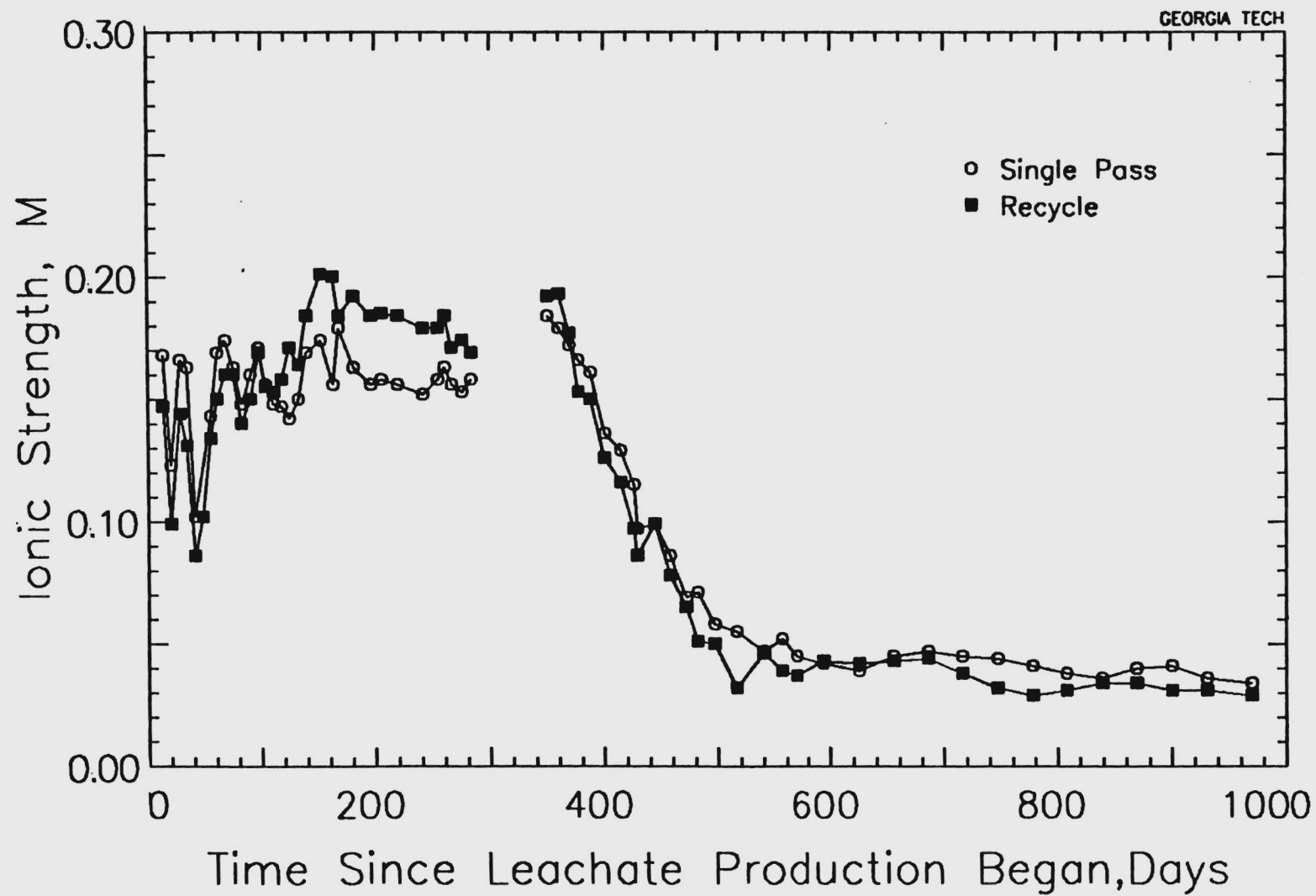


Figure 33. Ionic Strength of Leachate from Both Cells.

ionized volatile fatty acids as the pH of leachate increased to neutral levels from acidic levels. The reductions in ionic strength and conductivity were concomitant with removal of ionic species such as sulfides and heavy metals primarily by precipitation with heavy metals (Figures 29, 36-40).

The activity coefficient can be computed using any of the four following approximations: Debye-Huckel Approximation, Extended Debye-Huckel Approximation, Guntelberg Approximation, and Davies Approximation (112).

Debye-Huckel Approximation (Ionic Strength $< 0.5 \times 10^{-3}$ M)

$$\log \gamma_1 = - 0.51 Z_1^2 I^{1/2} \quad (5)$$

where:

γ_1 = activity coefficient of species 1

Z_1 = charge on species 1

I = ionic strength

Extended Debye-Huckel Approximation (Ionic Strength < 0.1 M)

$$\log \gamma_1 = - \frac{A Z_1^2 I^{1/2}}{1 + B a_1 I^{1/2}} \quad (6)$$

where;

γ_1 = activity coefficient of species 1

Z_1 = charge on species 1

I = ionic strength

A = $0.509 \text{ mol}^{-1/2} \text{ kg}^{1/2}$, at 25°C for water

B = $3.291 \times 10^9 \text{ m}^{-1} \text{ mol}^{-1/2} \text{ kg}^{1/2}$, at 25°C for water

a_1 = effective diameter of the ion, m

Guntelberg Approximation (Ionic Strength $\leq 0.1M$)

$$\log \gamma_i = - \frac{0.5 Z_i^2 I^{1/2}}{1 + I^{1/2}} \quad (7)$$

where:

γ_i = activity coefficient of species i

Z_i = charge on species i

I = ionic strength

Davies Approximation (Ionic Strength $\leq 0.5M$)

$$\log \gamma_i = - 0.51 Z_i^2 \left(\frac{I^{1/2}}{1 + I^{1/2}} - 0.3I \right) \quad (8)$$

where;

γ_i = activity coefficient for species i

Z_i = charge on species i

I = ionic strength

Davies Approximation was used to compute the activity coefficient of leachate samples collected from Cell A and Cell B, since the ionic strength values were greater than 0.1 M from Day 13 to Day 414, except on Days 20 and 41, when the ionic strength of leachate samples collected from Cell B was less than 0.1 M. The activity coefficients for monovalent, divalent, and trivalent ions present in leachate samples from Cell A and Cell B are shown in Figures 34 and 35, respectively.

An increase in the activity coefficients of monovalent, divalent, and trivalent ions was accompanied by a decrease in ionic strength, conductivity, and alkalinity. The activity coefficients of

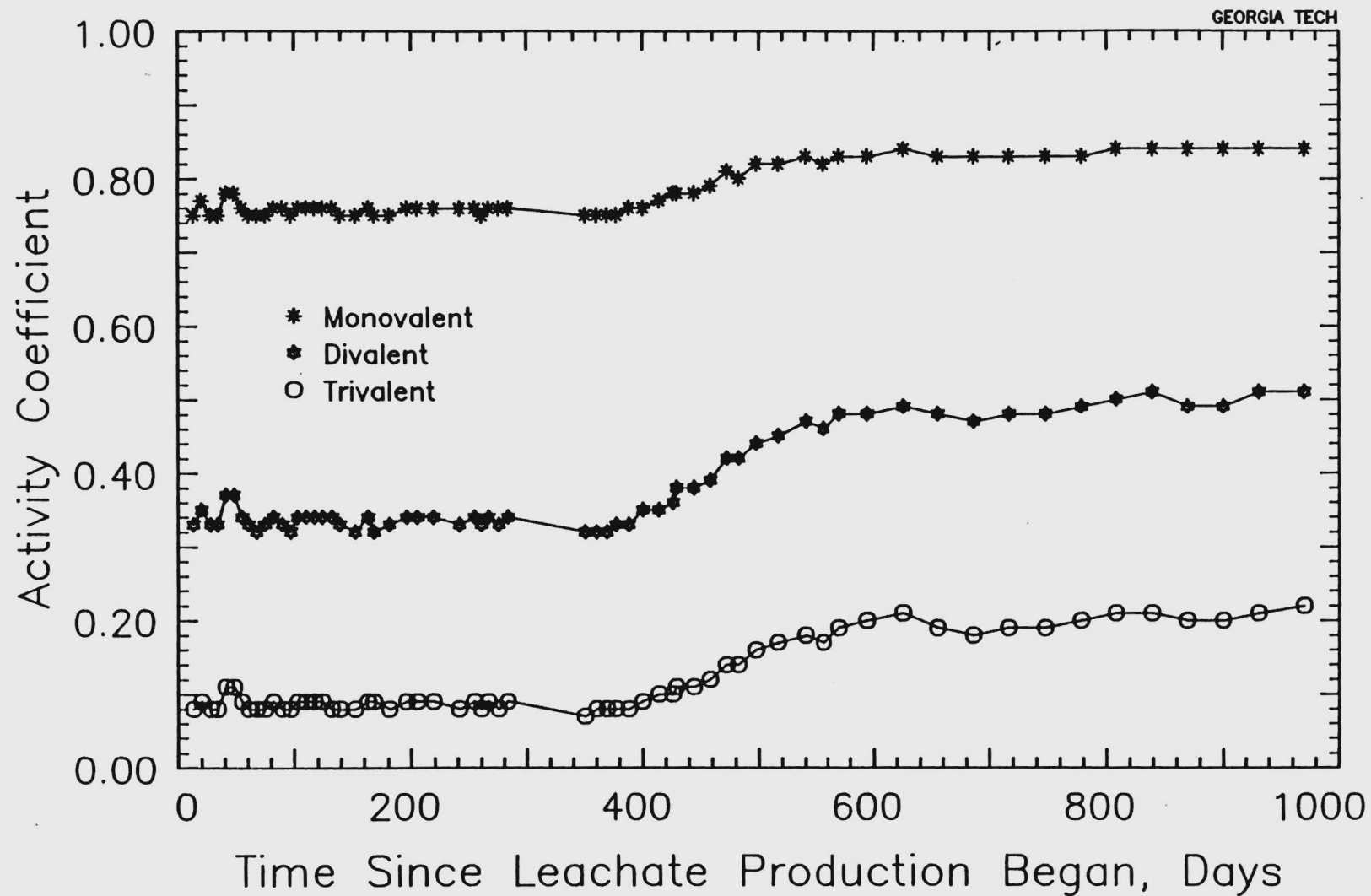


Figure 34. Activity Coefficients for Monovalent, Divalent, and Trivalent Ions Present in Leachate Samples from Cell A.

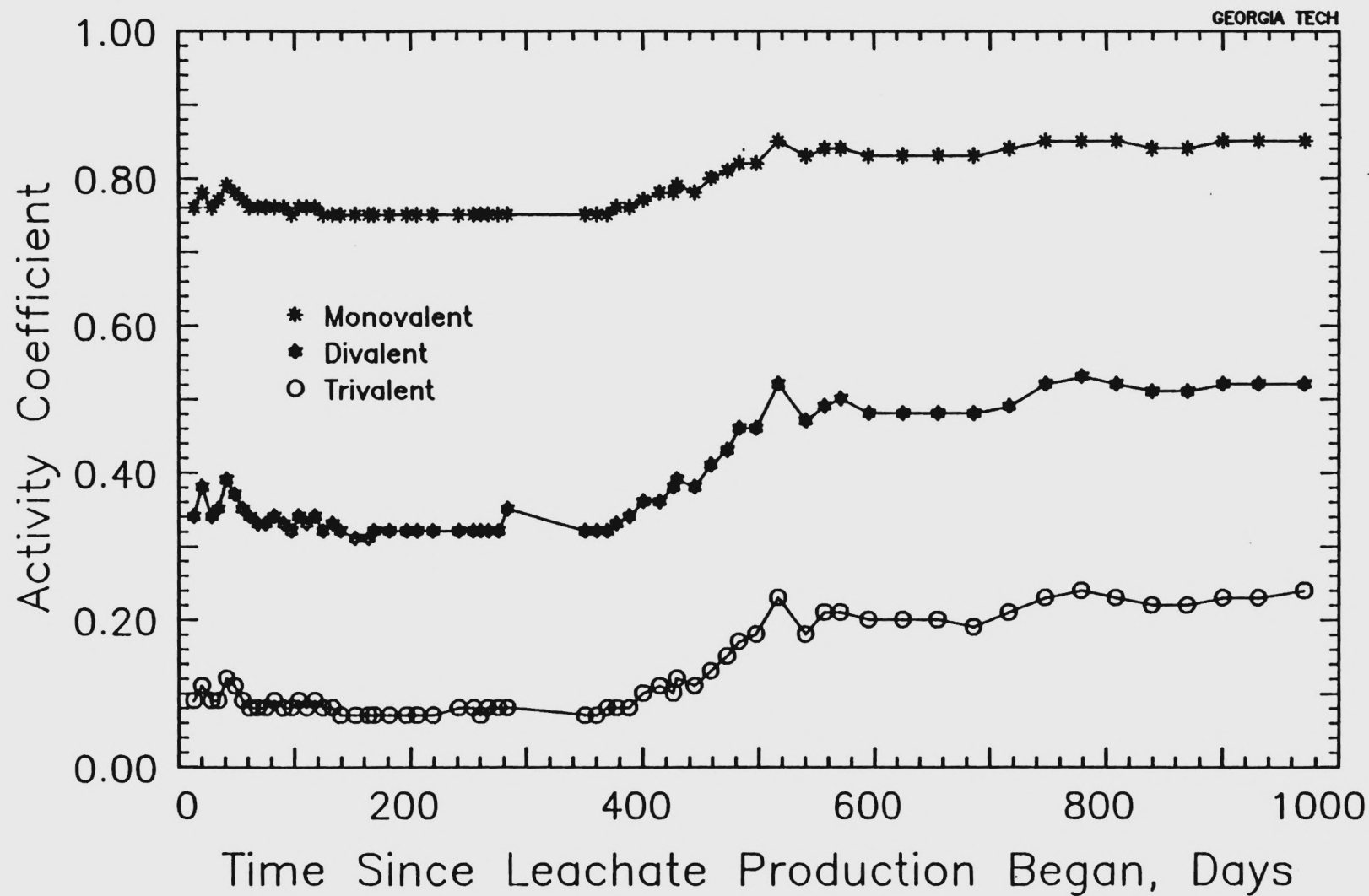


Figure 35. Activity Coefficients for Monovalent, Divalent, and Trivalent Ions Present in Leachate Samples from Cell B.

monovalent, divalent, and trivalent ions were between 0.75 and 0.85, 0.33 and 0.53, and 0.08 and 0.24, respectively. These values are in agreement with the values reported by Chang (21) and Pohland (23).

The activity coefficient of an ion affects the presence of that ion in the leachate produced from a landfill. In general, lower activity coefficients result in higher levels of ions in leachate. For example, the solubility of cobalt sulfide can be described as;

$$\{Co^{2+}\} \{S^{2-}\} = K_{SO} \quad (9)$$

assuming an activity coefficient of 0.5 for divalent ions Equation 9 will be modified to;

$$0.5 [Co^{2+}] 0.5 [S^{2-}] = K_{SO} \quad (10)$$

$$[Co^{2+}][S^{2-}] = 4K_{SO} \quad (11)$$

Since K_{SO} does not change, the product of cobalt and sulfide concentrations in leachate increases by a factor of four; this results in higher cobalt and sulfide concentrations in leachate at equilibrium than the concentrations predicted for dilute solutions.

4.4.10 Iron, Manganese, Zinc, Nickel, Cobalt, and Strontium

Iron, manganese, zinc, nickel, cobalt and strontium concentrations in leachate samples collected from Cell A and Cell B are presented in Figures 36 through 40. Iron concentration in leachate from Cell A increased from 140 mg/L (Day 13) to 395 mg/L (Day 55), then decreased to an ultimate value of 8.1 mg/L (Day 970), whereas iron concentration in leachate from Cell B increased from 83 mg/L (Day 13) to 625 mg/L

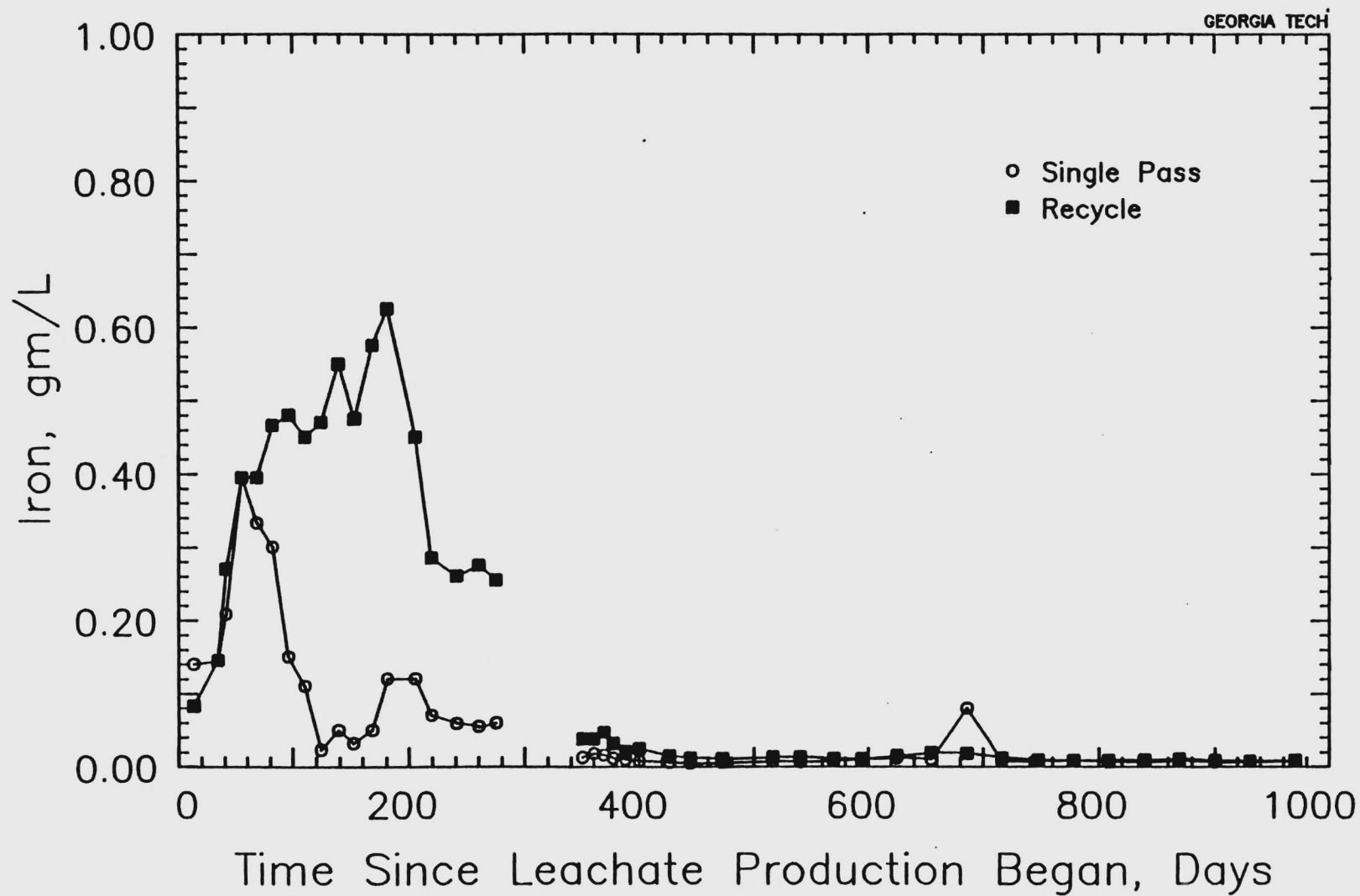


Figure 36. Iron Concentration in Leachate from Both Cells.

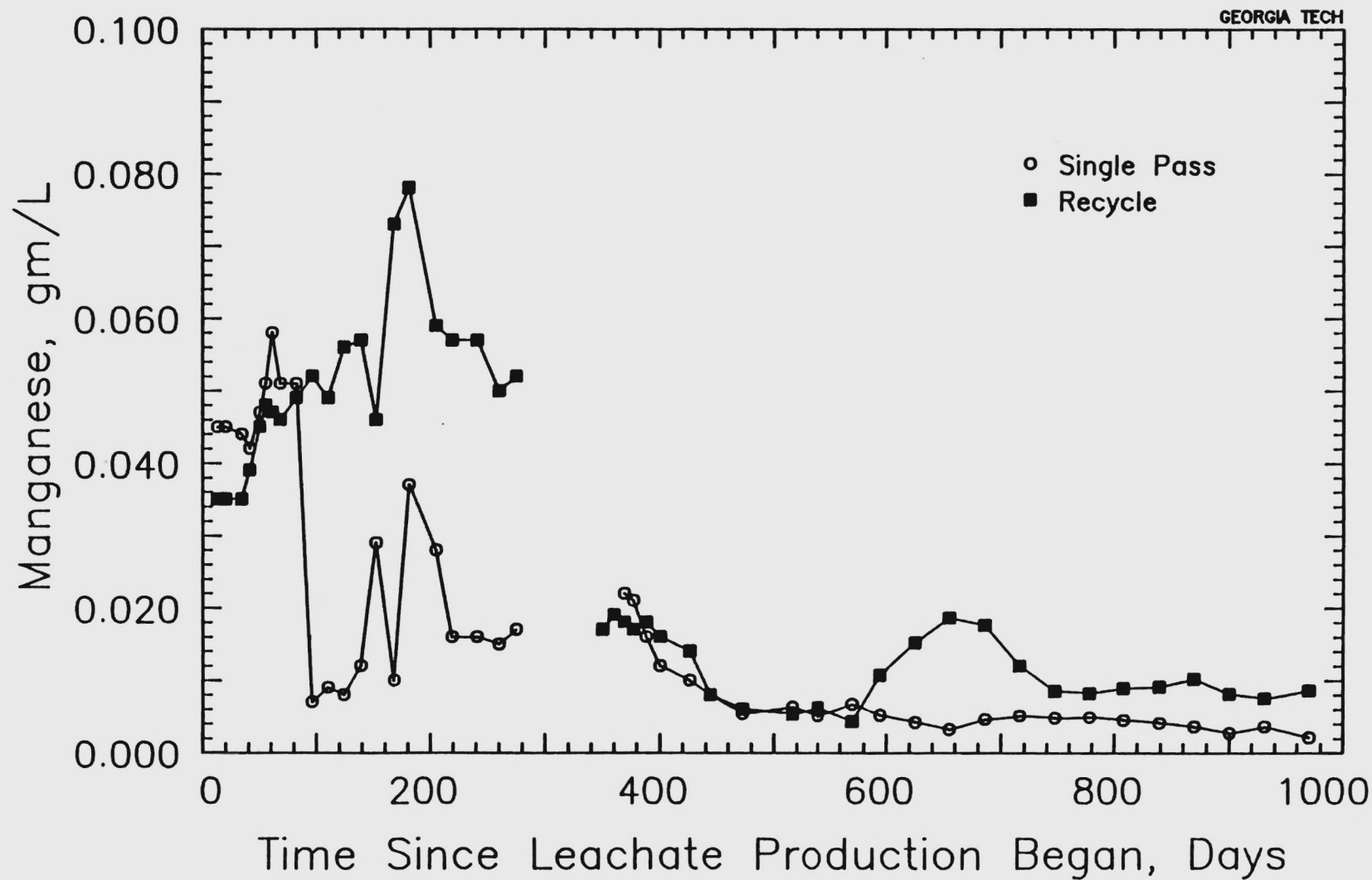


Figure 37. Manganese Concentration in Leachate from Both Cells.

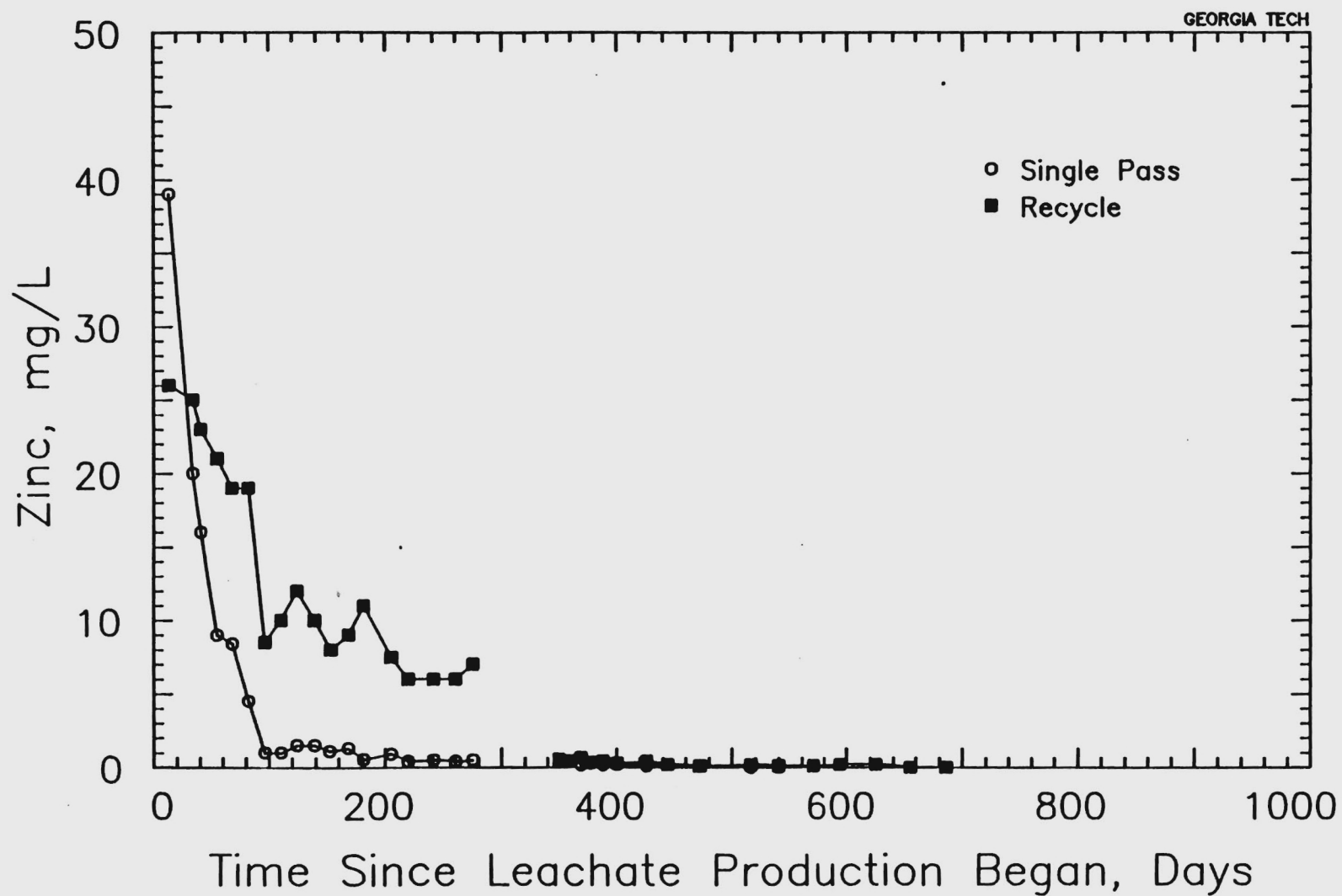


Figure 38. Zinc Concentration in Leachate from Both Cells.

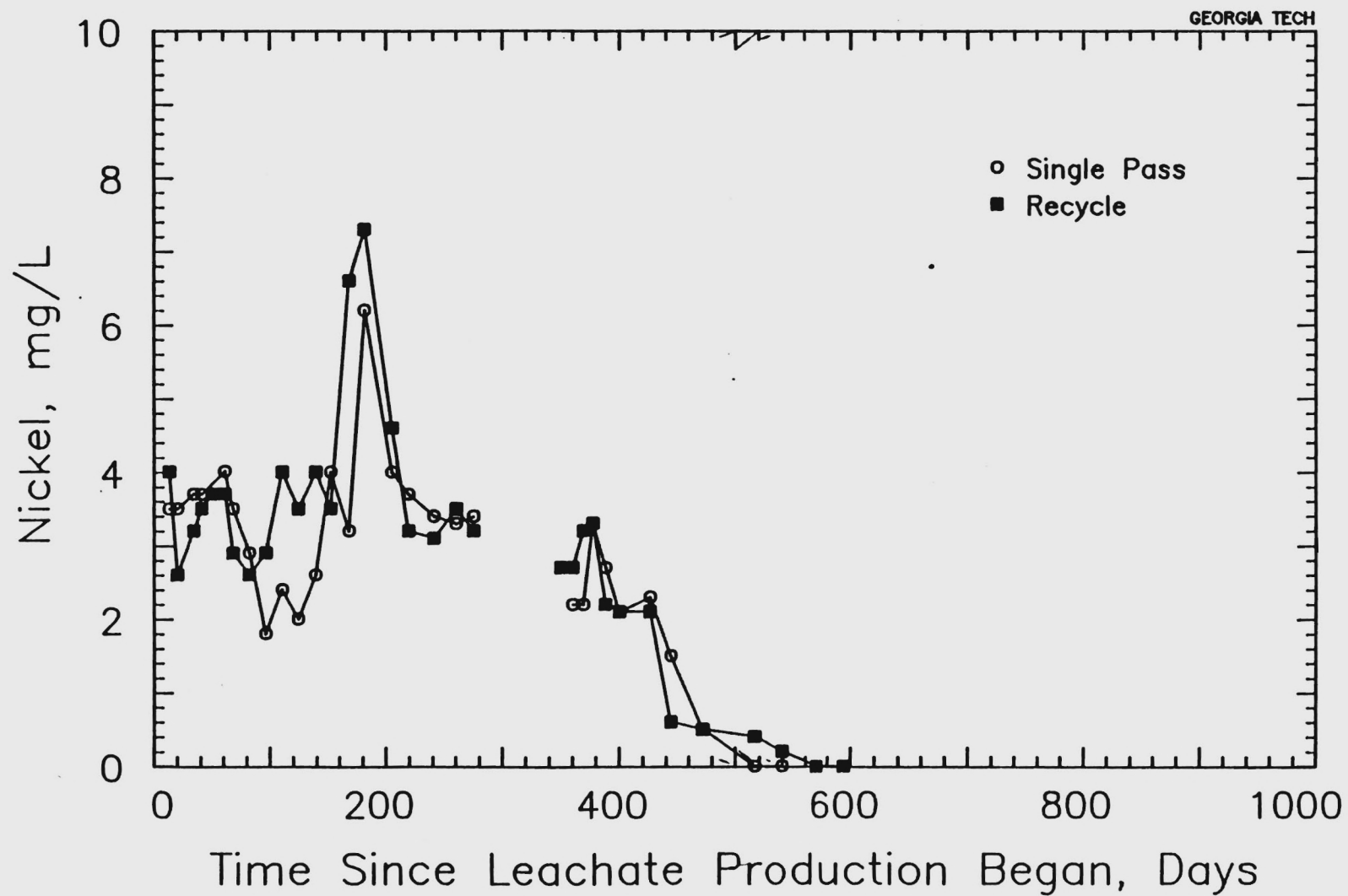


Figure 39. Nickel Concentration in Leachate from Both Cells.

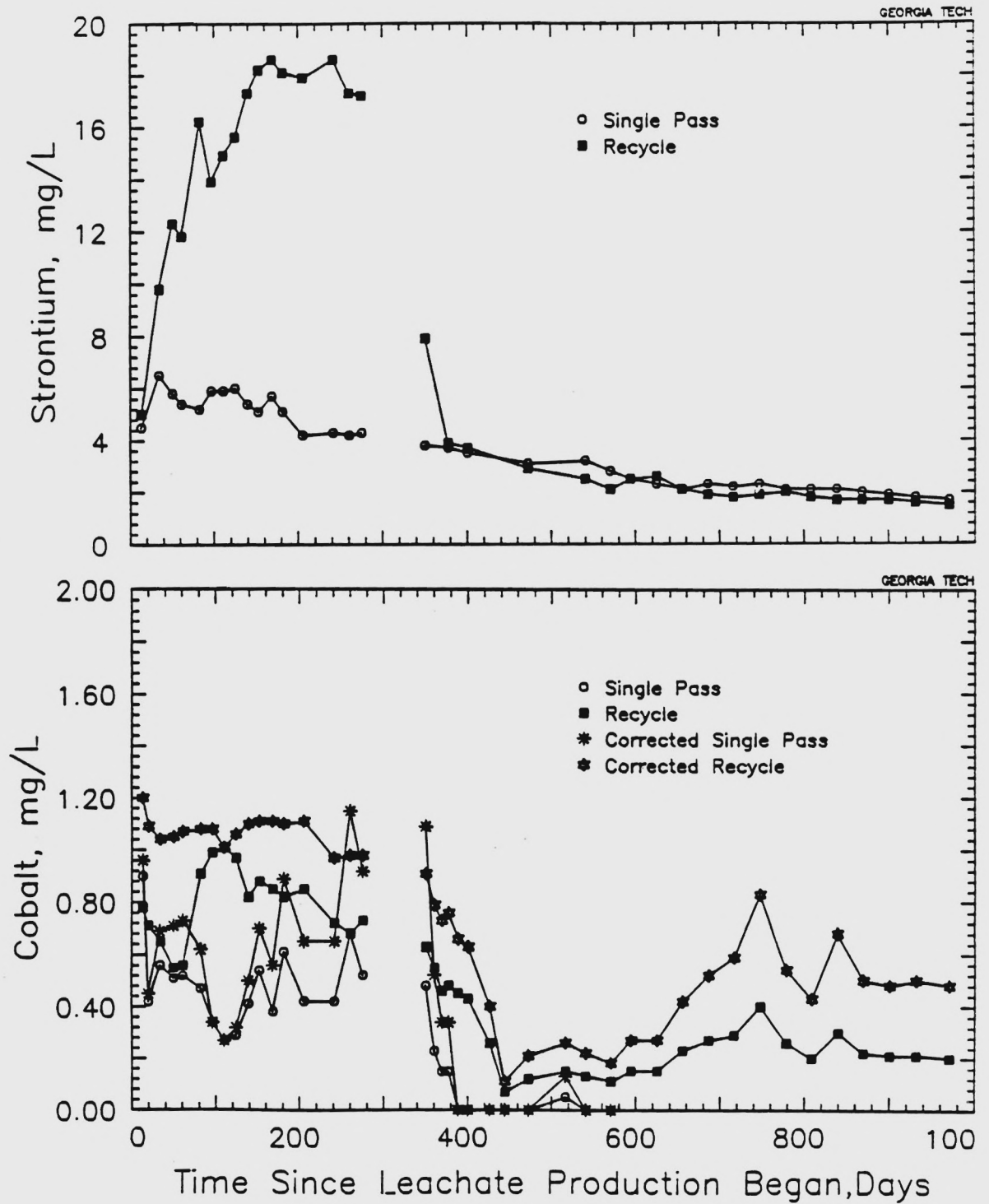


Figure 40. As-Measured and Dilution-Effect-Corrected Cobalt Concentrations and Strontium Concentrations in Leachate from Both Cells.

(Day 152), then decreased to an ultimate value of 8.6 mg/L (Day 970). Manganese concentration in leachate from Cell A increased from 45 mg/L (Day 13) to 51 mg/L (Days 55, 68, and 82), then decreased to an ultimate value of 2.1 mg/L (Day 970), whereas manganese concentration in leachate from Cell B increased from 35 mg/L (Day 13) to 79 mg/L (Day 181), then decreased to an ultimate value of 2 mg/L (Day 970). Zinc concentrations in leachate samples from both cells decreased steadily from 39 mg/L (Day 13; Cell A) to 0.1 mg/L (Day 472; Cell A) and 26 mg/L (Day 13, Cell B) to 0.2 mg/L (Day 625; Cell B). Nickel concentrations in leachate from Cell A increased from 3.5 mg/L (Day 13) to 6.2 mg/L (Day 181), then decreased to an ultimate value of 0.5 mg/L (Day 472), whereas nickel concentration in leachate from Cell B increased from 4.0 mg/L (Day 13) to 7.3 mg/L (Day 181), then decreased to an ultimate value of 0.2 mg/L (Day 540). Cobalt concentration in leachate from Cell A fluctuated between 0.27 mg/L (Day 110) and 0.90 mg/L (Day 13) till Day 350, then decreased sharply to 0.15 mg/L by Day 377. In contrast, cobalt concentration in leachate from Cell B increased from an initial value of 0.78 mg/L (Day 13) to 1.01 mg/L (Day 110), then decreased to 0.07 mg/L (Day 444), and finally increased to an ultimate value of 0.20 mg/L (Day 970). The strontium concentration in leachate from Cell A varied from 4.5 mg/L (Day 13) to 1.7 mg/L (Day 970), with a high of 6.0 mg/L (Day 124), whereas in leachate from Cell B, the strontium concentration varied from 5.0 mg/L (Day 13) to 1.5 mg/L (day 970), with a high of 18.6 mg/L (Days 168 and 241).

The increase in concentrations of iron and manganese in leachate from both cells during the transition and the acid formation phases of

landfill stabilization was primarily due to acidic pH of the leachate; the concentrations decreased with an increase in pH from acidic to neutral levels together with the appearance of sulfide (Figure 29) and a shift of buffer system from acetate to carbonate in leachate from both cells, thereby indicating precipitation of these metals as sulfides and carbonates. In addition to precipitation, washout also contributed to the decrease in concentrations of iron and manganese in leachate samples from Cell A.

Zinc and nickel behaved similarly to iron. Thus, the primary removal mechanism for heavy metals was precipitation; this is in agreement with the evidence reported by other researchers (21,25,59 and 109).

The "as-measured" and the "dilution-effect-corrected" concentrations of stable cobalt in leachate from both cells are presented in Figure 40. As expected, cobalt concentrations in leachate increased during the transition and the acid formation phases of landfill stabilization due to acidic pH of the leachate; the concentrations then decreased with an increase in pH from acidic to neutral levels together with the appearance of sulfide (Figure 29) and a shift in the primary buffer system from acetate to carbonate in leachate from both cells. While precipitation of cobalt by the potential-precipitant species such as, carbonates and sulfides was the major removal mechanism for removal of cobalt from leachate, the "as-measured" concentrations of cobalt in leachate were higher than those predicted by solubility.

As shown in Figure 40, the strontium concentrations in leachate from Cell A and Cell B increased during the transition and the acid

formation phases of landfill stabilization due to acidic pH of the leachate. The trend, however, was more pronounced in the strontium concentration in leachate from Cell B compared to the strontium concentration in leachate from Cell A. This behavior can be attributed to the prolonged acidic pH of leachate from Cell B as compared to leachate from Cell A, and prevention of washout of constituents present in leachate from Cell B due to recirculation of leachate. With the shift of the primary buffer system from acetate to carbonate along with an increase in pH of leachate from acidic levels to neutral levels, and appearance of TIC concentrations in leachate, strontium concentrations in leachate from both cells decreased. Based on the solubility product of strontium carbonate ($\log K = 9.0$), strontium concentration in a leachate (ionic strength = 0.03 M, Figure 33, activity coefficients for divalent ions = 0.5, Figure 35) will be 5.5 mg/L. This value is comparable to the residual strontium concentrations (2 mg/L) in leachate. Thus, the decrease in strontium concentration, albeit more pronounced in leachate from Cell B, is attributed to the probable precipitation of strontium as strontium carbonate.

The solubility of cobalt as a function of hydroxide, carbonate, and sulfide is presented in Figure 41; sulfide is the strongest precipitant for cobalt, followed by carbonate and hydroxide. The "as-measured" cobalt concentrations in leachate from Cell B are also shown in Figure 41, these concentrations are divided into two pH ranges: one group with pCo varying from 4.7 to 5.0 at pH values ranging from 5.00 to 5.60 and the other group with pCo varying from 5.0 to 5.9 at pH values ranging from 6.90 to 7.34.

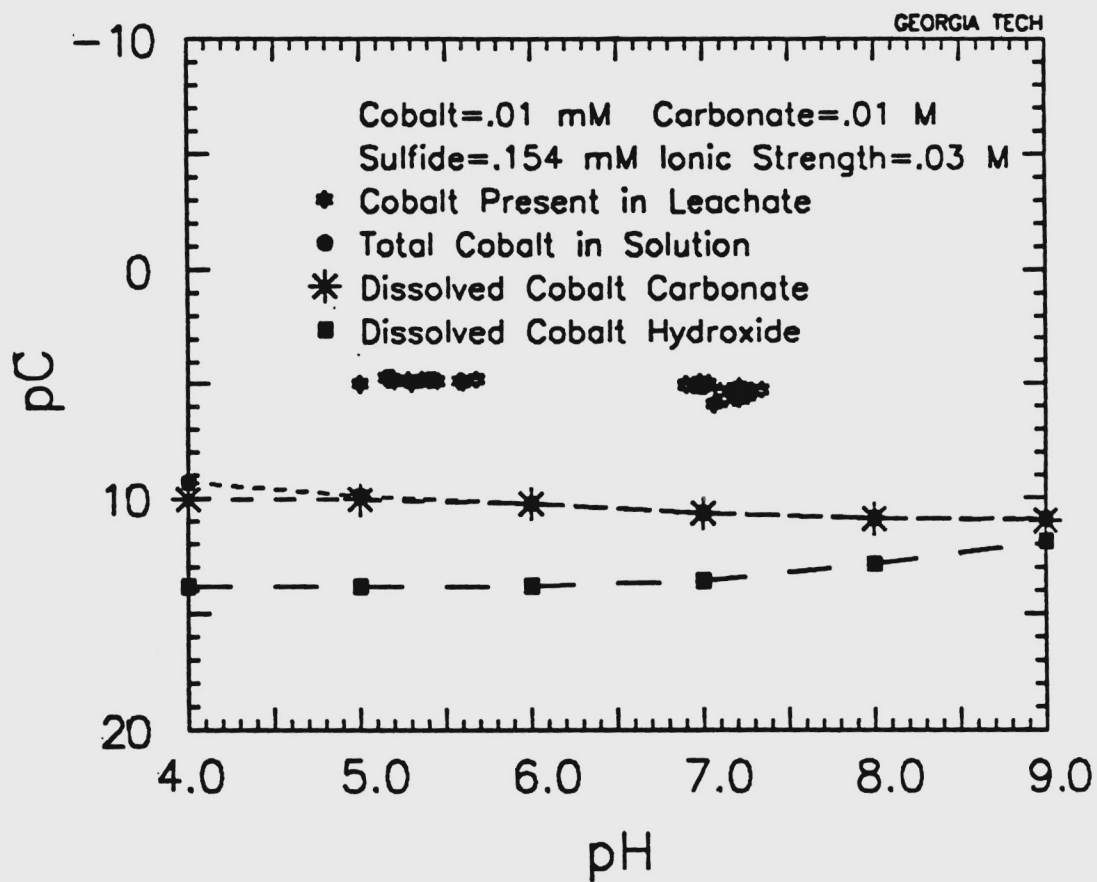


Figure 41. Cobalt Solubility in the Presence of hydroxide, Carbonate, and Sulfide.

Figure 41 indicates that while sulfide and carbonate (shift of the primary buffer system from acetate to carbonate due to increase in pH values of leachate from acidic levels to neutral levels and reduction in the concentrations of volatile acids to below detection limits) precipitated cobalt, cobalt concentrations were present in excess of the concentrations predicted by solubility. This can be attributed to the probable complexation of cobalt by humic substances present in leachate.

4.4.11 Radionuclides

The results of the radionuclide analyses performed on the leachate samples collected from Cell A and Cell B are summarized in Table 16. The causes for the decrease in the radionuclide levels in leachate from Cell A and Cell B were different; while washout contributed to the decrease in the radionuclide levels in Cell A, the radionuclides in leachate from Cell B were contained within the landfill system (lined landfill with cover, leachate collection, recirculation, distribution networks, and leachate collection sumps) and allowed to decay.

The maximum levels of Ce-141 present in leachate from both Cell A and Cell B, during the period of investigation, were less than 0.22 nCi/L. Based on Day 0 concentrations of Ce-141 in Cell A and Cell B, less than 0.03 percent of the Ce-141 spiked on the refuse were present in the leachate from both cells. Thus, greater than 99.97 percent of the Ce-141 spiked on the refuse were retained within the cells. The results of chemical modeling on speciation of metals (MINEQL) indicate

Table 16. Radionuclide Concentrations, nCi/L, (on July 2, 1982) in the Leachate Samples from Cell A (Single Pass) and Cell B (Recycle)

Time Since Leachate Production Began, Days	Radionuclide Concentration					
	Cell A			Cell B		
	Ce-141	Sr-85	Co-58	Ce-141	Sr-85	Co-58
	nCi/L	nCi/L	nCi/L	nCi/L	nCi/L	nCi/L
0	<0.22	4.35	4.00	<0.22	3.60	3.50
13	<0.21	4.37	3.68	<0.24	6.37	5.84
20	<0.22	6.90	5.79	<0.21	6.42	5.78
27	<0.21	4.94	3.83	<0.33	8.36	6.70
34	<0.34	6.10	4.56	<0.39	9.72	7.68
41	<0.27	5.41	3.77	<0.43	9.46	6.99
48	<0.31	5.02	2.75	<0.44	10.8	8.23
55	<0.32	5.36	2.70	<0.47	11.7	8.83
62	<0.37	5.22	1.98	<0.64	11.3	7.96
69	<0.60	5.48	2.10	<0.77	12.7	9.02
76	<0.45	5.20	1.70	<0.74	15.6	10.40
83	<0.53	5.64	1.67	<1.1	16.2	10.80
90	<0.62	5.77	1.73	<1.2	15.2	8.97
97	<1.1	6.58	2.14	<1.2	14.0	8.45
103	<1.3	6.54	1.66	<1.6	17.6	9.83
110	<1.6	6.43	1.29	<2.0	16.3	9.69
117	<1.9	6.23	0.40	<2.4	17.3	9.48
124	<2.2	5.90	0.22	<2.5	17.4	10.3
132	<1.3	5.52	0.32	<2.1	16.8	9.07
139	<1.5	5.99	0.30	<2.5	18.9	10.8
153	<4.1	1.73	0.30	<3.8	20.6	9.66
160	<4.4	5.07	0.25	<3.5	19.5	9.89
168	<5.1	5.35	0.15	<4.2	20.3	10.2
174	<6.3	4.71	0.15	<5.2	20.8	10.5
181	<7.1	5.31	0.16	<5.4	21.0	10.4
190	<7.0	4.72	0.50	<8.4	20.4	11.2
196	<8.6	2.88	0.16	<9.6	18.0	9.88
205	<4.9	4.34	0.30	<7.6	20.4	9.58
220	<8.1	5.29	0.25	<12.0	22.1	11.0
231	<26.0	2.00	0.20	<29.0	21.6	8.36
241	<26.0	4.70	0.20	<17.0	22.9	10.4
283	N.D. ^a	N.D.	N.D.	N.D.	N.D.	N.D.
350	N.D.	N.D.	N.D.	N.D.	N.D.	N.D.

^a None Detectable

Notes:

Volume of Water in Cell A = 5,475 L

Volume of Water in Cell B = 5,785 L

Radionuclide Concentration in Cells on 2 July 1982

- Cell A: Co-58 = 165 nCi/L, Sr-85 = 200 nCi/L, and Ce-141 = 1,055 nCi/L

- Cell B: Co-58 = 157 nCi/L, Sr-85 = 190 nCi/L, and Ce-141 = 1,000 nCi/L

that Ce-141 is expected to form soluble complexes with acetic acid in the presence of acetic acid, carbonate, sulfide, and hydroxide, at pH = 5.00, and exist as Ce^{+3} in the presence of carbonate, sulfide, and hydroxide, at pH = 7.00. This model (MINEQL), however, does not account for physical-chemical separation, viz., adsorption and ion-exchange. Since less than 0.03 percent of the Ce-141 spiked on the refuse was present in the leachate from both cells, the remaining (99.97%) Ce-141 was probably adsorbed on the refuse placed in the cells.

The as-measured and the dilution-effect-corrected concentrations of Co-58 and Sr-85 in leachate from both cells are presented in Figures 42 and 43, respectively. In comparison with Chang's findings on codisposal of low-level radioactive waste within landfills (21), the concentrations of Co-58 and Sr-85 in leachate from the cell with leachate recycle (Cell B), behaved similarly. The different behavior of Co-58 and Sr-85 in leachate from single-pass cell (Cell A), when compared to the results reported by Chang (21), is attributed to the variation in simulations of infiltration of moisture due to incidental rainfall; Chang (21) added one liter of water per week to the lysimeters, to simulate infiltration of moisture due to incidental rainfall (one liter per week was an average rainfall-equivalent volume over a three year period), throughout the period of investigation, whereas, in this study, actual rainfall-equivalent volumes, based on incidental rainfall in Atlanta, GA during the period of investigation, were added to the cells. Moreover, the additions of rainfall-equivalent volumes were terminated on Day 56 since a 0.3-m (1 ft) head

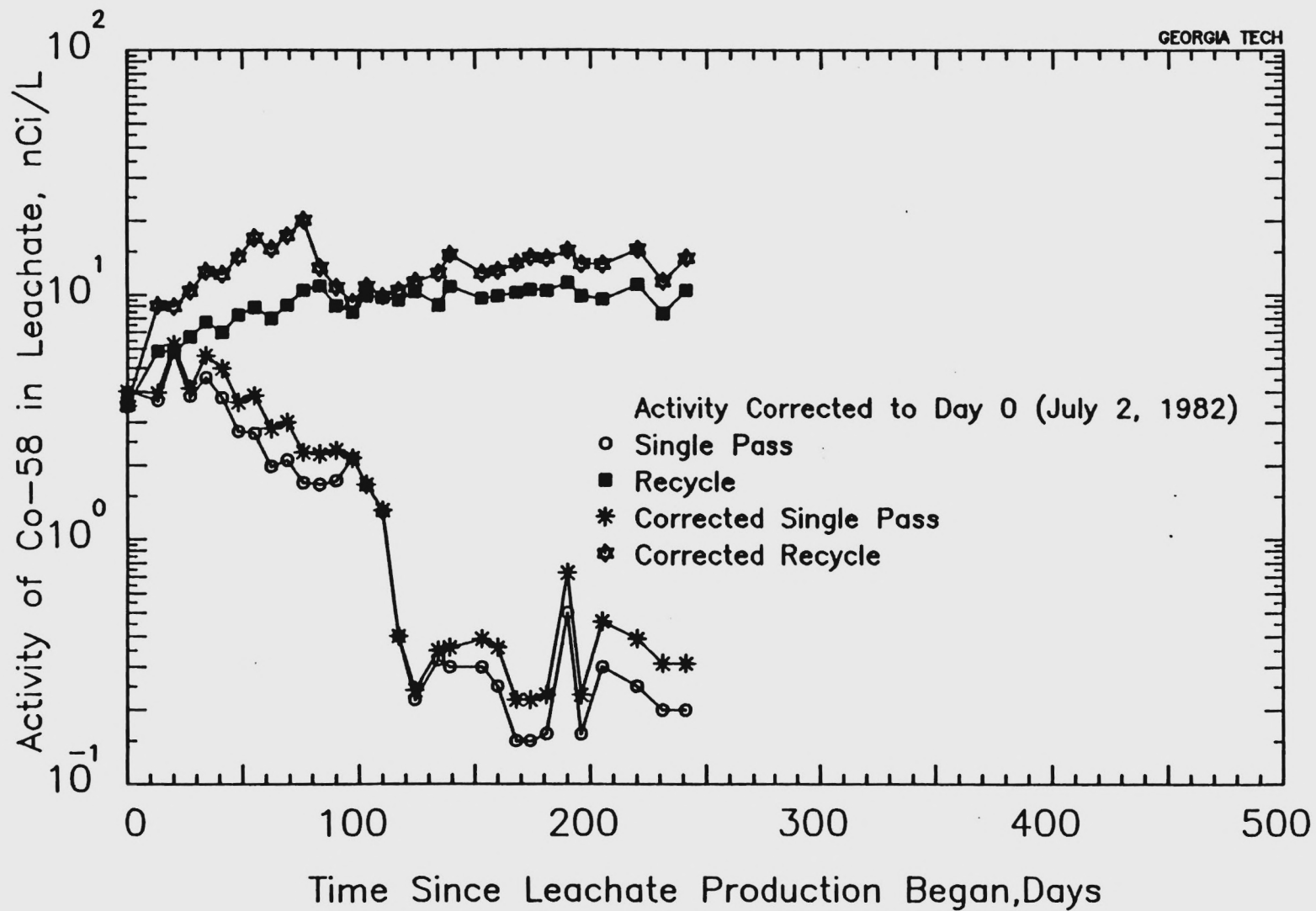


Figure 42. As-Measured and Dilution-Effect-Corrected Co-58 Present in Leachate from Both Cells.

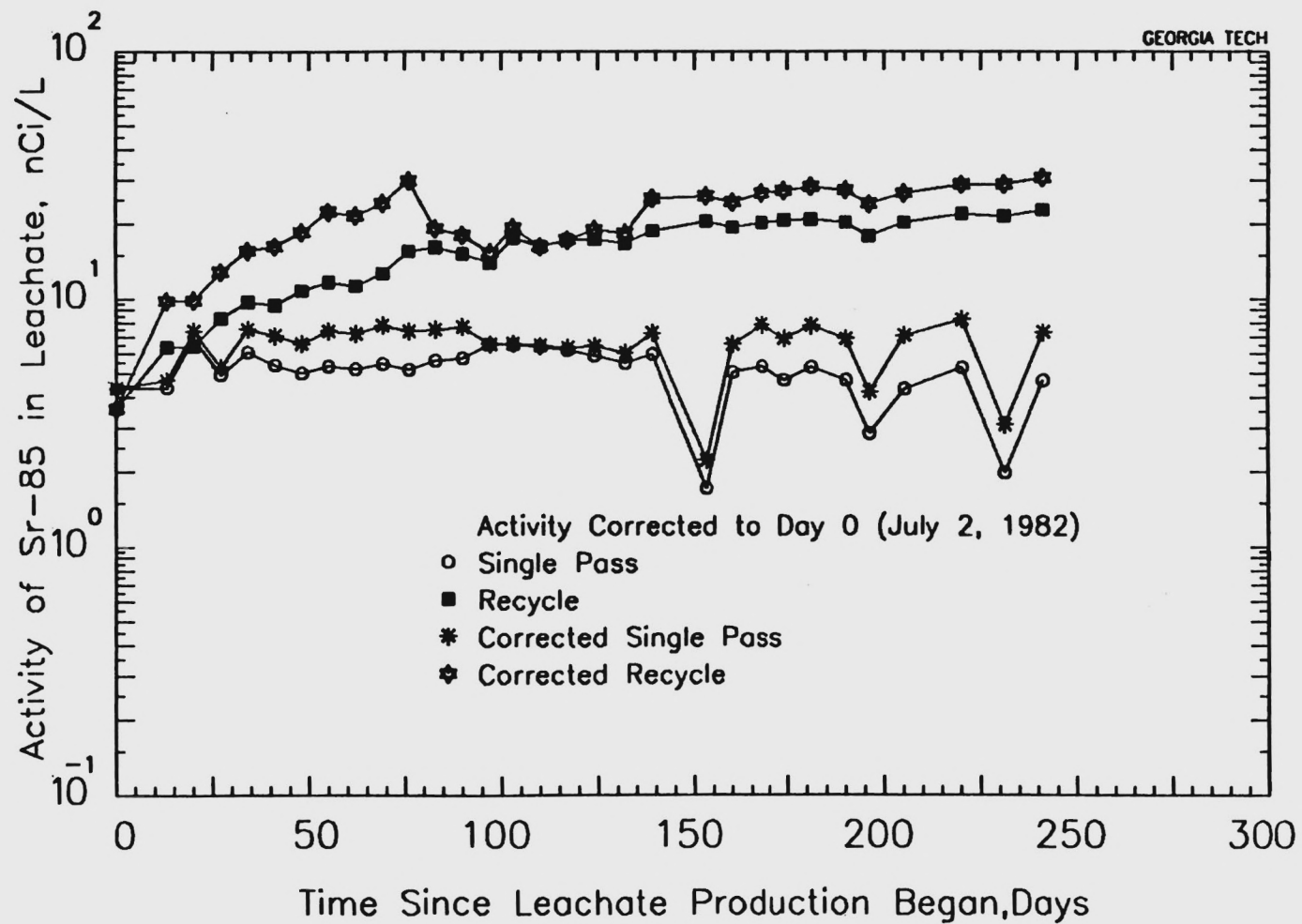


Figure 43. As-Measured and Dilution-Effect-Corrected Sr-85 Present in Leachate from Both Cells.

of leachate had accumulated on the bottom of the cell; this 0.3-m value was computed by assuming the porosity of refuse as 0.4. In Chang's study, the constituents present in the refuse placed in the single-pass cell leached continually due to continual addition of water to the cell, whereas, in this study, the leaching of constituents from the refuse, due to addition of rainfall equivalent volumes, ceased after Day 56.

Based on the initial spiked levels of Co-58 and Sr-85, the maximum levels of Co-58 and Sr-85 in leachate from Cell B were 12.7 percent and 15.7 percent, respectively. The radionuclides in leachate from both Cell A and Cell B decayed to below detectable limits by Day 350.

The specific activities of Co-58 and Sr-85 in leachate from both cells are presented in Table 17. The decrease in specific activity of Co-58 in leachate from Cell A is attributable to the inconsistent leaching patterns of Co-58 and stable cobalt due to termination of addition of rainfall-equivalent volumes to the cell. In leachate from Cell B, the specific activities of Co-58 and Sr-85 were $11.21 \text{ nCi/mg} \pm 2.14 \text{ nCi/mg}$ and $1.07 \text{ nCi/mg} \pm 0.009 \text{ nCi/mg}$, respectively. The coefficient of variation for the specific activities of Co-58 and Sr-85 in leachate from Cell B were 19.1% and 8.4%, respectively. The coefficient of variations for Co-58 and Sr-85 indicate that trends exhibited by cobalt and strontium in leachate from Cell B can be used to simulate the behavior of Co-58 and Sr-85 within Cell B.

One of the objectives of this study was to investigate the potential solubilization of precipitated LLRW by humic substances

Table 17. Specific Activities of Co-58 and Sr-85 in Leachate from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Specific Activity			
	Co-58		Sr-85	
	Cell A	Cell B	Cell A	Cell B
	nCi/mg	nCi/mg	nCi/mg	nCi/mg
13	4.0	7.4	1.0	1.3
20	13.7	8.1	-	-
34	8.1	11.8	0.9	1.0
50	-	-	0.9	-
61	3.8	14.2	1.0	0.8
82	3.6	11.8	1.1	1.0
96	6.3	8.5	1.1	1.1
110	4.8	9.7	1.0	1.1
124	0.8	10.6	0.9	1.1
139	0.8	13.2	1.2	1.1
152	0.6	11.0	0.3	1.2
168	0.4	12.0	1.0	1.0
181	0.2	12.6	1.1	1.1
205	0.7	11.3	1.0	1.2
241	0.5	14.4	1.0	1.3

present in the landfill leachate during the later phases of landfill stabilization (methane formation phase and final maturation phase). Since the radionuclides chosen to simulate LLRW had decayed to levels below the detectable limits by Day 350, a solubilization study was simulated, wherein Co-58 was chosen as the model radionuclide and its solubilization from cobalt sulfide (previously precipitated to simulate precipitation of LLRW by sulfide) by leachate from Cell B (during the later phases of stabilization) was investigated. Co-58 was chosen, since Chang (21) had reported an increase in Co-58 in the leachate from a lysimeter (equipped with leachate recycle) during the later phases of landfill stabilization. The results of the solubilization study are presented in the following section.

4.5 Solubilization Study

The results of the study on the solubilization of Co-58, from cobalt sulfide (containing both Co-58 and stable cobalt) by model humic acid (A-2HA) and leachate samples from Cell B are presented in the following sections. Also presented in the following sections are the results of the Quality Assurance/Quality Control (QA/QC) study, wherein cobalt sulfide and a swamp based humic acid (A-2HA) were utilized as model compounds.

4.5.1 Fractionation of Model Humic Acid and Leachate

Ultrafiltration (UF) and gel permeation chromatography (GPC) were utilized to fractionate model humic acid (A-2HA) and leachate samples into different molecular-weight fractions. The results of UF and GPC fractionations are summarized in Tables 18 and 19, respectively.

As shown in Table 18, model humic acid (A-2HA) at three different pH values (5.5, 7.0, and 8.5) and three leachate samples were fractionated into permeate (low-molecular-weight) and retentate (high-molecular-weight) fractions. The percent local membrane rejection, R , reported for each sample was computed using the following expression;

$$R = 1 - \frac{\ln \left(1 - \frac{V_p C_{pc}}{V_f C_f} \right)}{\ln \left(1 - \frac{V_p}{V_f} \right)} \quad (9)$$

where;

- R = local membrane rejection, %
- V_p = permeate volume, mL
- V_f = feed volume, mL
- C_{pc} = permeate concentration (composite), mg/L
- C_f = feed concentration, mg/L

The local membrane rejections indicate the percentage of TOC associated with the high-molecular-weight fraction which will be retained on an UF membrane at any time during ultrafiltration.

The local membrane rejection of the UM05 ultrafiltration membrane (nominal cutoff = 500 daltons; Amicon, Bedford, MA) for the model humic acid (A-2HA) samples at pH values of 5.5, 7.0, and 8.5 were $97.5\% \pm 0.2\%$, 95% and 96% , respectively. In comparison, the local membrane rejections for the leachate samples were 46.9% (Day 749, pH=7.15), $57.2\% \pm 2.1\%$ (Day 810, pH=7.21), and $63.1\% \pm 1.2\%$ (Day 841, pH=7.21). The significant difference in the local membrane rejections for the

Table 18. Total Organic Carbon (TOC) Distribution in Different Molecular Weight Fractions of Model Humic Acid (A-2HA) and Leachate Samples

Sample	pH	TOC	Ultrafiltration		
			UM05 (500 MW)		Local Membrane Rejection, R
			Permeate	Retentate	
		mg/L	mg/L	mg/L	%
A-2HA (1.0 g/L)	7.0	525 (200 ml)*	22.8 \pm 1.3 (148.5 ml \pm 1.3 ml)	1981.6 \pm 34.2 (51.5 ml \pm 1.3 ml)	97.5 \pm 0.2
A-2HA** (0.5 g/L)	5.5	282 (100 ml)	20.2 (49 ml)	592 (51 ml)	95
A-2HA** (0.5 g/l)	8.5	284 (100 ml)	16.5 (51 ml)	591 (49 ml)	96
Leachate, ** Day 749	7.15	665 (200 ml)	445.9 (146 ml)	1312.6 (54 ml)	46.9
Leachate, Day 810	7.21	720 \pm 25.4 (100 ml)	371 \pm 9.6 (49 ml \pm 1.8 ml)	1040 \pm 37.6 (51 ml \pm 1.8 ml)	57.2 \pm 2.1
Leachate, Day 841	7.21	698 \pm 32.4 (100 ml)	309 \pm 13.3 (47.3 ml \pm 3.7 ml)	1024 \pm 59.1 (52.7 ml \pm 3.7 ml)	63.1 \pm 1.2

*Volume of sample.

**Results are average of two samples.

model humic acid (A-2HA) samples and the leachate samples can be attributed to absence of the fulvic acid (low-molecular-weight) fraction in the model humic acid (A-2HA), whereas the leachate samples contained both fractions.

Model humic acid at three pH values (5.5, 7.0, and 8.5) and two leachate samples were fractionated into high-molecular-weight, intermediate-molecular-weight, and low-molecular-weight fractions using gel permeation chromatography. The gel used (Sephadex G-50; Pharmacia Fine Chemicals, Piscataway, NJ) had exclusion limits of 10,000 MW and 500 MW. As shown in Table 19, the TOC distributions obtained by gel permeation chromatography were similar to the local membrane rejections obtained by ultrafiltration (Table 18). In comparison with the model humic acid (A-2HA), the leachate samples exhibited a higher TOC content in the low-molecular-weight fraction. This can be attributed to the absence of the fulvic acid (low-molecular-weight) fraction in the model humic acid (A-2HA), since the fulvic acid fraction was separated from the model humic acid (A-2HA) during its isolation by Reuter (85). The percent recovery based on TOC ranged from 82.8% to 91.2% for model humic acid (A-2HA) and leachate samples, respectively.

4.5.2 Quality Assurance/Quality Control (QA/QC) Study

In order to test the significance of the difference in the solubilization levels of cobalt sulfide in deionized water and model humic acid (A-2HA) a quality assurance/quality control (QA/QC) study was conducted. The results obtained from this study are summarized in Table 20. The cobalt concentrations in model humic acid and deionized water were $8.3 \text{ mg/L} \pm 1.2 \text{ mg/L}$ and $3.7 \text{ mg/L} \pm 0.7 \text{ mg/L}$, respectively.

Table 19. Total Organic Carbon (TOC) Distribution in Different Molecular Weight Fractions (Generated by GPC) of Model Humic Acid (A-2HA) and Leachate

Sample	pH	TOC mg/L	Volume of Sample Injected Into The GPC Column mL	TOC			
				High Molecular Weight mg/L	Intermediate Molecular Weight mg/L	Low Molecular Weight mg/L	Recovery $\bar{x} \pm \sigma_x$ %
A-2HA (4)*	7.0	525	0.5	6.5 ± 0.5	5.5 ± 0.2	1.6 ± 0.3	88.1 ± 6.1
A-2HA (2)	5.5	520	0.5	5.4	5.9	1.8	86.9
A-2HA (2)	8.5	560	0.5	6.7	6.6	1.5	91.2
Leachate, Day 810 (4)	7.21	720	0.5	2.6 ± 0.4	6.6 ± 0.4	9.1 ± 0.7	85.4 ± 6.0
Leachate, Day 841 (4)	7.21	698	0.5	2.9 ± 0.3	6.3 ± 0.4	8.0 ± 0.8	82.9 ± 6.3

*Number of replicates.

Table 20. Cobalt Solubilized from Cobalt Sulfide by Model Humic Acid (A-2HA) and Deionized Water in the Quality Assurance/Quality Control (QA/QC) Experiment

Sample	Cobalt Sulfide Added ($\bar{x} \pm \sigma$), g/L	Cobalt Detected ($\bar{x} \pm \sigma$), mg/L	t-Test
A-2HA (17)* (0.5 g/L; pH=7.0)	0.49 \pm 0.07	8.3 \pm 1.2	$t_{\text{exp}} = 15.3$ $t_{\text{v}}=35, \alpha=0.05$
Deionized Water (20)*	0.52 \pm 0.09	3.7 \pm 0.7	$t_{\text{v}}=30, \alpha=0.05$ (Significant)

*Number of replicates.

The Student t-Test on the means revealed that the difference between the amounts of cobalt solubilized by model humic acid (A-2HA) and deionized water is statistically significant; this indicated probable formation of cobalt-humic complexes. The calculations of the Student t-Test are presented in Appendix A.

4.5.3 Equilibrium Study

The results of the equilibrium study on solubilization of cobalt, from cobalt sulfide, by model humic acid (A-2HA), leachate samples, and deionized water (blank) are presented in Table 21. Deionized water solubilized 6.1 mg/L of cobalt, whereas model humic acid (A-2HA) at pH values of 5.5, 7.0, and 8.5 solubilized 10.8 mg/L, 8.3 mg/L, and 26.1 mg/L of cobalt, respectively. In comparison, the leachate samples, which were collected on Days 749, 810, and 941, solubilized 40.5 mg/L, 45.6 mg/L, and 42.3 mg/L of cobalt, respectively. The local membrane rejection for blank was 15%, whereas for model humic acid (A-2HA) at pH values of 5.5, 7.0, and 8.5 the local membrane rejections were 74.8%, 91.5%, and 94.9%, respectively. In comparison, the local membrane rejections of the leachate samples collected on Days 749, 810, and 841 were 64.1%, 43.3%, and 42.5%, respectively.

The local membrane rejection of cobalt indicates the percentage of solubilized cobalt retained along with the high-molecular-weight fraction on an ultrafiltration membrane. Thus, higher rejections of cobalt (i.e., 64.1%, 43.3%, and 42.5%) by the UM05 ultrafiltration membrane (nominal cutoff = 500 daltons; Amicon, Bedford, MA) for the leachate samples than for the blank (15%) indicate that significant amounts of solubilized cobalt are associated with the molecular-weight

Table 21. Cobalt Solubilized from Cobalt Sulfide by Model Humic Acid (A-2HA), Leachate Samples, and Deionized Water

Sample	pH	Cobalt Detected				
		Background	Solubilized	Permeate (UM05)	Retentate (UM05)	Local Membrane Rejection %
		mg/L	mg/L	mg/L	mg/L	
Blank (CoS + Deionized Water)	6.8	N.D.*	6.1	5.4 (45 mL)**	6.7 (55 mL)**	15.0
A-2HA (0.5 g/L)	5.5	N.D.*	10.8	3.5 (52 mL)	18.0 (48 mL)	74.8
A-2HA (0.5 g/L)	7.0	N.D.*	8.3	1.2 (148 mL)	27.9 (52 mL)	91.5
A-2HA (0.5 g/L)	8.5	N.D.*	26.1	1.8 (50 mL)	47.8 (50 mL)	94.9
Leachate, Day 749	7.15	0.4	40.5	17.1 (42 mL)	59.1 (58 mL)	64.1
Leachate, Day 810	7.21	0.2	45.6	29.3 (47 mL)	61.2 (53 mL)	43.3
Leachate, Day 841	7.21	0.3	42.3	27.5 (47 mL)	54.3 (53 mL)	42.5

*None Detectable

** () Volume of fractionated sample.

fractions greater than 500 daltons. This can be attributed to the complexation of cobalt by humic substances present in the leachate samples. The increase in local membrane rejection of cobalt with pH for model humic acid (A-2HA) can be attributed to the increase in the solubility of model humic acid (A-2HA) with pH.

4.5.4 Kinetic Study

Results of the kinetic study (Figure 44) indicate that 80% of the final levels of solubilized cobalt were attained within three days. Thus, solubilization of cobalt from cobalt sulfide by leachate samples can be considered as a fairly rapid reaction since a period of three days can be envisioned as a short period when compared to the time period (measured in years) a landfill usually resides in the methane formation phase and the final maturation phase.

4.5.5 Confirmatory Study Using Cobalt-58

The solubilization of Co-58 from cobalt sulfide (containing both Co-58 and stable cobalt) by model humic acid (A-2HA; at pH values of 5.5, 7.0, and 8.5), leachate and deionized water (blank) was investigated and the results are presented in this section. The characteristics of the leachate used in this study are summarized in Table 22. Prior to utilization, the leachate was filtered through a 0.45 μ m filter. Since Co-58 was the only radionuclide present in the samples, radioactivity in each sample has been reported in counts per minute normalized to Day 0 (March 7, 1985) of the confirmatory study.

As shown in Table 23, the radioactivity in the residue (CoS) was the difference between the average radioactivity in the initial cobalt solutions and the radioactivity in the filtrate. The average radio-

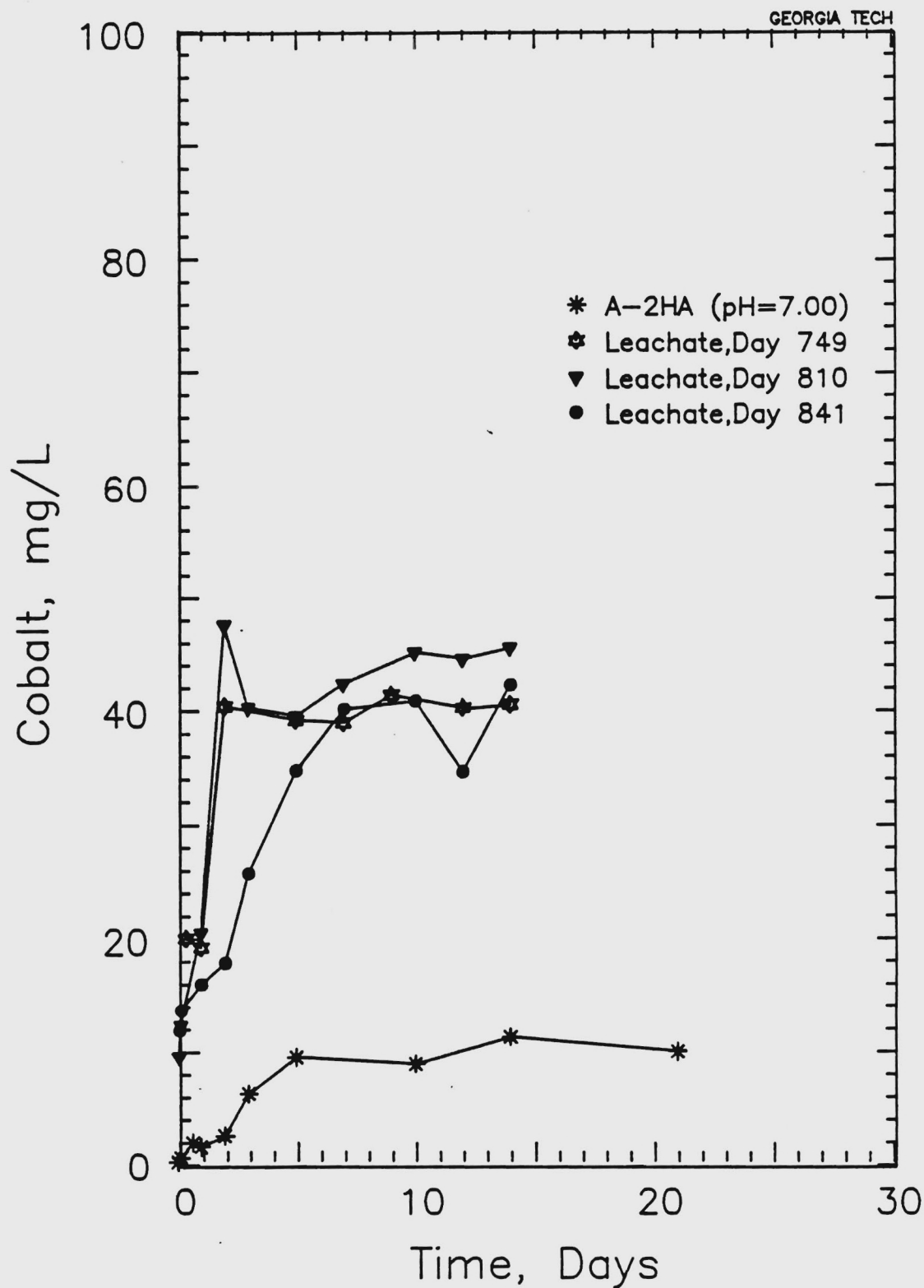


Figure 44.. Solubilization of Cobalt by A-2HA and Landfill Leachate.

Table 22. Characteristics of Leachate (Day 970) That Was Utilized to Investigate the Solubilization of Co-58 from Cobalt Sulfide (Containing both Co-58 and Stable Cobalt)

Parameter	Value
pH	7.27
ORP	-191 mV E _C
Total alkalinity as CaCO ₃	1,825 mg/L
Conductivity	1.85 mmhos
Ionic strength	0.04 M
COD	1,522 mg/L
BOD ₅	342 mg/L
TOC	619 mg/L
Total volatile acid as acetic acid	N.D.*
Cobalt	N.D.
Iron	8.6 mg/L
Manganese	2.0 mg/L
Sulfide	N.D.

*None Detectable

Table 23. Radioactivity of the Cobalt Solution and the Filtrate

Sample	pH	Counts Per Minute		
		Cobalt Solution	Filtrate	Cobalt in Cobalt Sulfide Residue on Filter Paper
		($\times 10^{-6}$)	($\times 10^{-6}$)	($\times 10^{-6}$)
I	II	III	IV	V=III _{avg} *-IV
A-2HA, #1	7.00	20.79	4.73	12.95
A-2HA, #2	7.00	19.50	4.87	12.81
A-2HA, #3	7.00	19.67	4.99	12.69
A-2HA, #4	7.00	19.78	4.69	12.99
A-2HA, #5	5.50	19.23	5.01	12.67
A-2HA, #6	5.50	17.95	5.71	11.97
A-2HA, #7	8.50	14.83	4.24	13.44
A-2HA, #8	8.50	16.13	5.22	12.46
Leachate**, #1	7.27	16.49	5.10	12.58
Leachate, #2	7.27	17.17	5.23	12.45
Leachate, #3	7.27	16.84	5.71	11.97
Leachate, #4	7.27	17.26	5.11	12.57
Blank***, #1	6.80	16.07	5.27	12.41
Blank, #2	6.80	16.15	5.20	12.48
Blank, #3	6.80	17.47	4.88	12.80

*III_{avg} = 17.66×10^6 counts per minute

**Leachate Sample from Cell B; Day 970

***Deionized Water

activity in the initial cobalt solutions was 17.68×10^6 counts per minute and the average radioactivity retained on the filter papers (0.45 μm , MSI, Fisher Scientific, Atlanta, GA) was 11.94×10^6 counts per minute. Thus, the radioactivity retained on a filter paper was $71.3\% \pm 0.07\%$.

The levels of Co-58 solubilized from cobalt sulfide on Days 12 and 28 are presented in Table 24. As shown, the levels of Co-58 solubilized increased significantly from Day 12 to Day 28. Also, the percent cobalt solubilized increased for each ligand from Day 12 to Day 28. These increases indicate that the solubilized levels of Co-58 were not at equilibrium on Day 12. No significant increases in Co-58 levels were observed on Day 90. The data presented in Table 24 correspond to equilibrium solubilized cobalt levels (Day 28) of 6.2 mg/L, 15.6 mg/L, 17.7 mg/L, 20.9 mg/L and 24.9 mg/L, respectively, for blank, leachate, A-2HA (pH=8.5), A-2HA (pH=5.5), and A-2HA (pH=7.0). As expected, these values are similar to the solubilized cobalt levels reported for stable cobalt (Table 21).

The different fractions obtained by fractionating Day 28 samples with gel permeation chromatography were analyzed for radioactivity. The relative radioactivity in each fraction expressed as a percent of the total radioactivity in the sample, is shown in Figures 45 through 48. Blue dextran ($\text{MW} > 2 \times 10^6$ daltons) and phenol ($\text{MW} = 94$ daltons) were used as standards and for a given set of conditions (column details, flow rate of mobile phase, and type of mobile phase) the elution pattern indicated the volumes within which blue dextran (high-molecular-weight fraction) and phenol (low-molecular-weight fraction)

Table 24. Co-58 Solubilized from Cobalt Sulfide by Model Humic Acid (A-2HA),
Leachate Sample from Cell B (Day 970), and Deionized Water (Blank)

Sample	pH	Counts Per Minute		Co-58 Solubilized from Cobalt Sulfide			
		Day 12	Day 28	Day 12		Day 28	
		($\times 10^{-5}$)	($\times 10^{-5}$)	% x_1	% $\bar{x}_1 \pm \sigma_{x_1}$	% x_2	% $\bar{x}_2 \pm \sigma_{x_2}$
A-2HA, #1	7.00	6.39	10.05	4.9		7.7	
A-2HA, #2	7.00	6.46	9.75	5.0		7.6	
					5.0 ± 0.2		7.5 ± 0.1
A-2HA, #3	7.00	6.47	9.57	5.0		7.5	
A-2HA, #4	7.00	6.98	9.66	5.3		7.4	
A-2HA, #5	5.5	6.54	9.07	5.1		7.1	
					4.6^*		6.3^*
A-2HA, #6	5.5	4.95	6.64	4.1		5.5	
A-2HA, #7	8.5	5.05	7.33	3.7		5.4	
					3.3^*		5.35^*
A-2HA, #8	8.5	3.68	6.65	2.9		5.3	
Leachate, #1	7.27	4.89	-	3.8			
Leachate, #2	7.27	4.13	4.35	3.3		3.4	
					3.5 ± 0.2		4.7 ± 1.1
Leachate, #3	7.27	4.26	6.50	3.5		5.4	
Leachate, #4	7.27	4.29	6.89	3.4		5.5	
Blank, #1	6.80	2.42	2.89	1.9		2.3	
Blank, #2	6.80	2.07	2.38	1.6		1.9	
Blank, #3	6.80	1.50	1.92	1.2	1.6 ± 0.3	1.5	1.9 ± 0.4

*Average of two samples.

were eluted. These volumes were $15 \text{ mL} \pm 0.16 \text{ mL}$ and $35 \text{ mL} \pm 0.52 \text{ mL}$ for blue dextran and phenol, respectively.

As shown in Figures 45-48, the Co-58 solubilized by blank (deionized water) coeluted with phenol. However, the elution patterns for the solubilized Co-58 in model humic acid (A-2HA) and leachate sample exhibited a shift towards left relative to the profile for the blank; this indicates that a fraction of the solubilized Co-58 is associated with high-molecular-weight and intermediate-molecular-weight organics. As expected, the largest shift to the left occurred for model humic acid (A-2HA) at pH=8.5 and the smallest shift occurred for model humic acid (A-2HA) at pH=5.5. This can be attributed to a decrease in the presence of the low-molecular-weight fraction of the model humic acid (A-2HA) with an increase in pH from 5.5 to 8.5 since the solubility of the model humic acid (A-2HA) increased with the increase in pH. Thus, the molecular-weight profiles generated by GPC confirm the complexation of Co-58 by humic substances present in landfill leachate.

4.6 Migratory Activity Levels of Co-58, Sr-85, and Ce-141 from the Landfills in the Event of a Liner Failure

The loading levels, based on dry weight or refuse, of Co-58, Sr-85, and Ce-141 utilized in this study were $1.6 \times 10^{-7} \text{ Ci/kg}$, $2.0 \times 10^{-7} \text{ Ci/kg}$, and $1.06 \times 10^{-6} \text{ Ci/kg}$, respectively. During the period of investigation, the maximum levels, expressed as a percent of initial loading levels, of Co-58, Sr-85, and Ce-141 in leachate from the pilot-scale landfill with provisions for leachate recycle (Cell B) were 12.7%, 15.7%, and 0.03%, respectively. The maximum levels of Co-58,

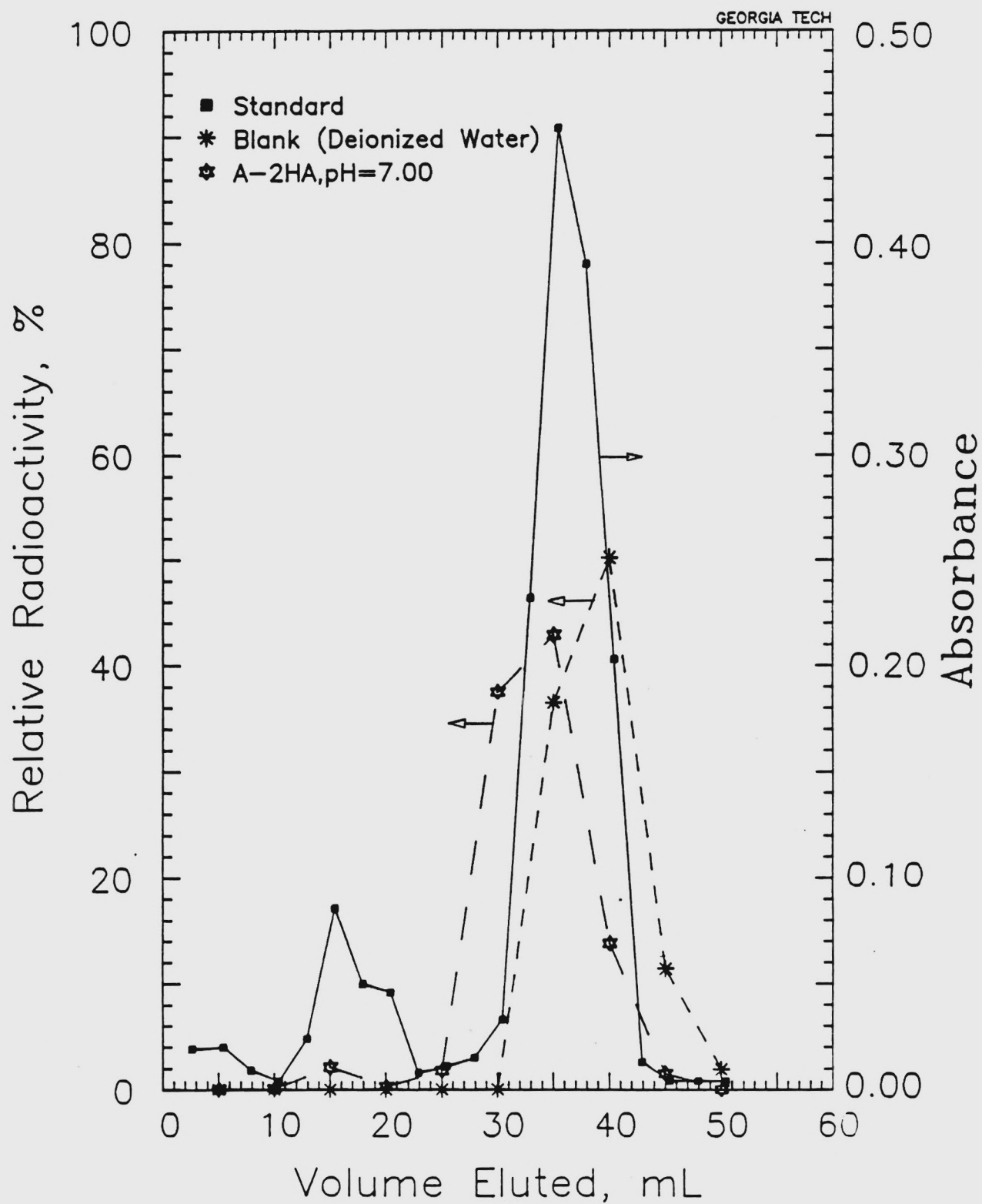


Figure 45. Radioactivity Present in Different GPC Fractions of Blank and A-2HA (pH=7.00).

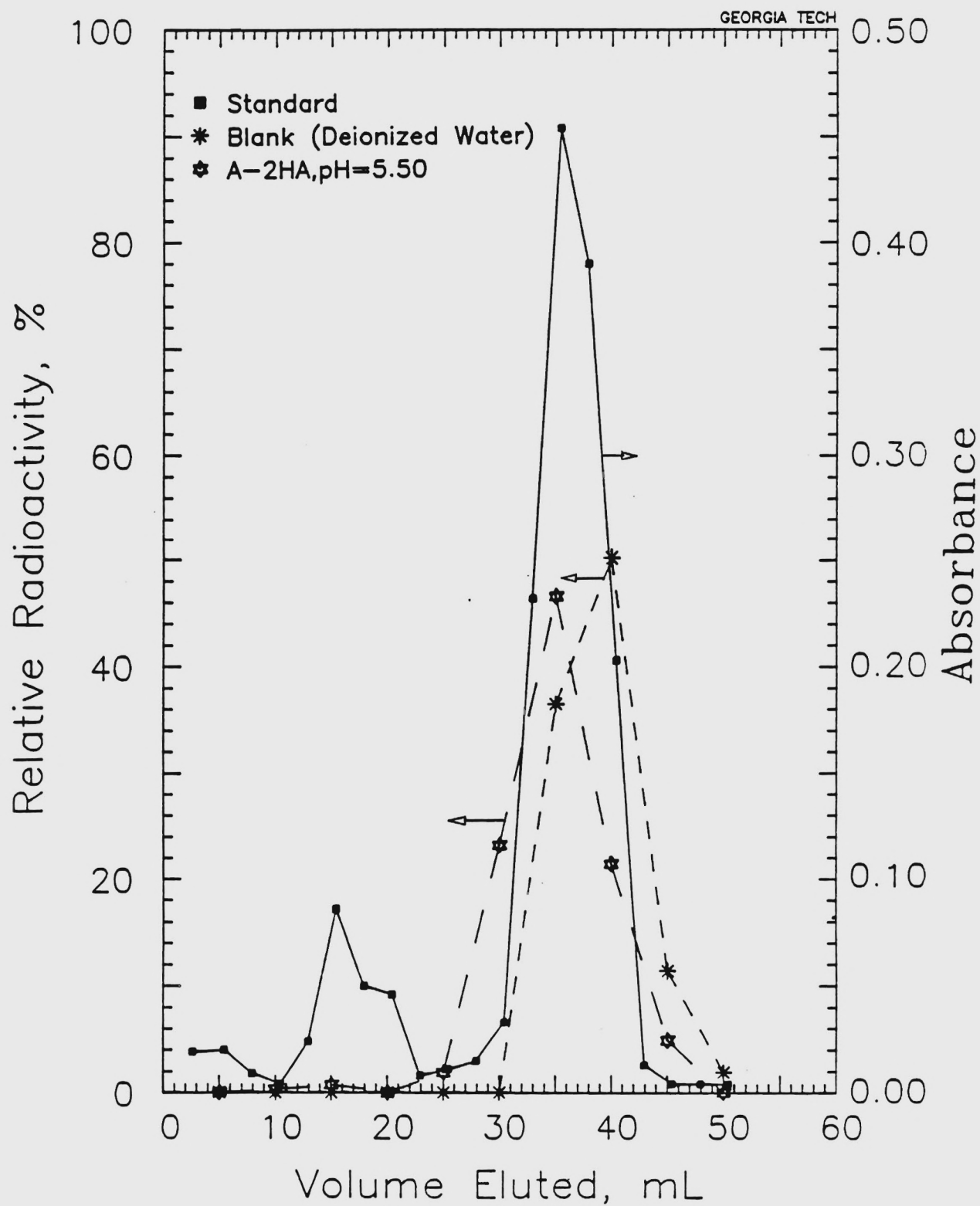


Figure 46. Radioactivity Present in Different GPC Fractions of Blank and A-2HA (pH=5.50).

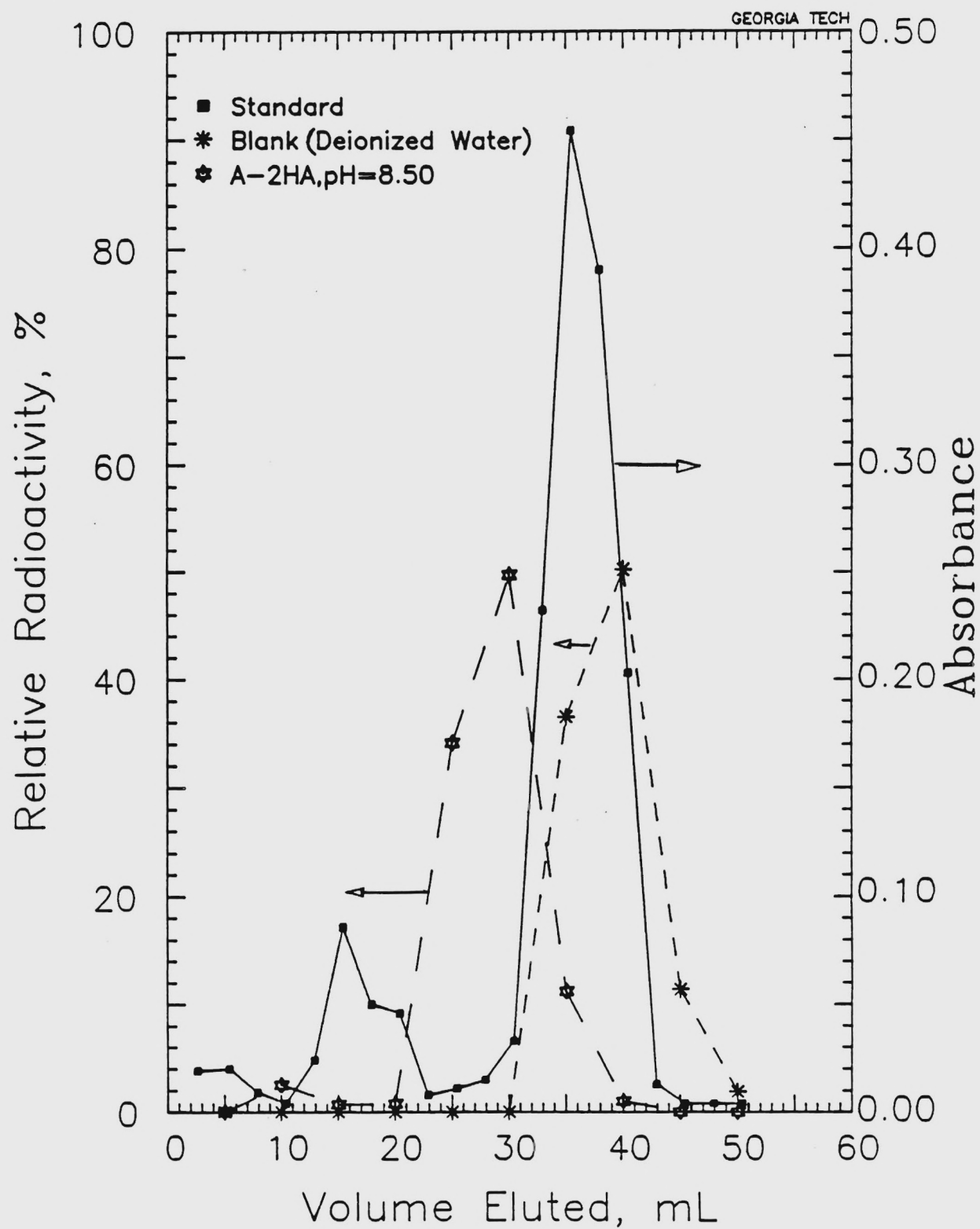


Figure 47. Radioactivity Present in Different GPC Fractions of Blank and A-2HA (pH=8.50).

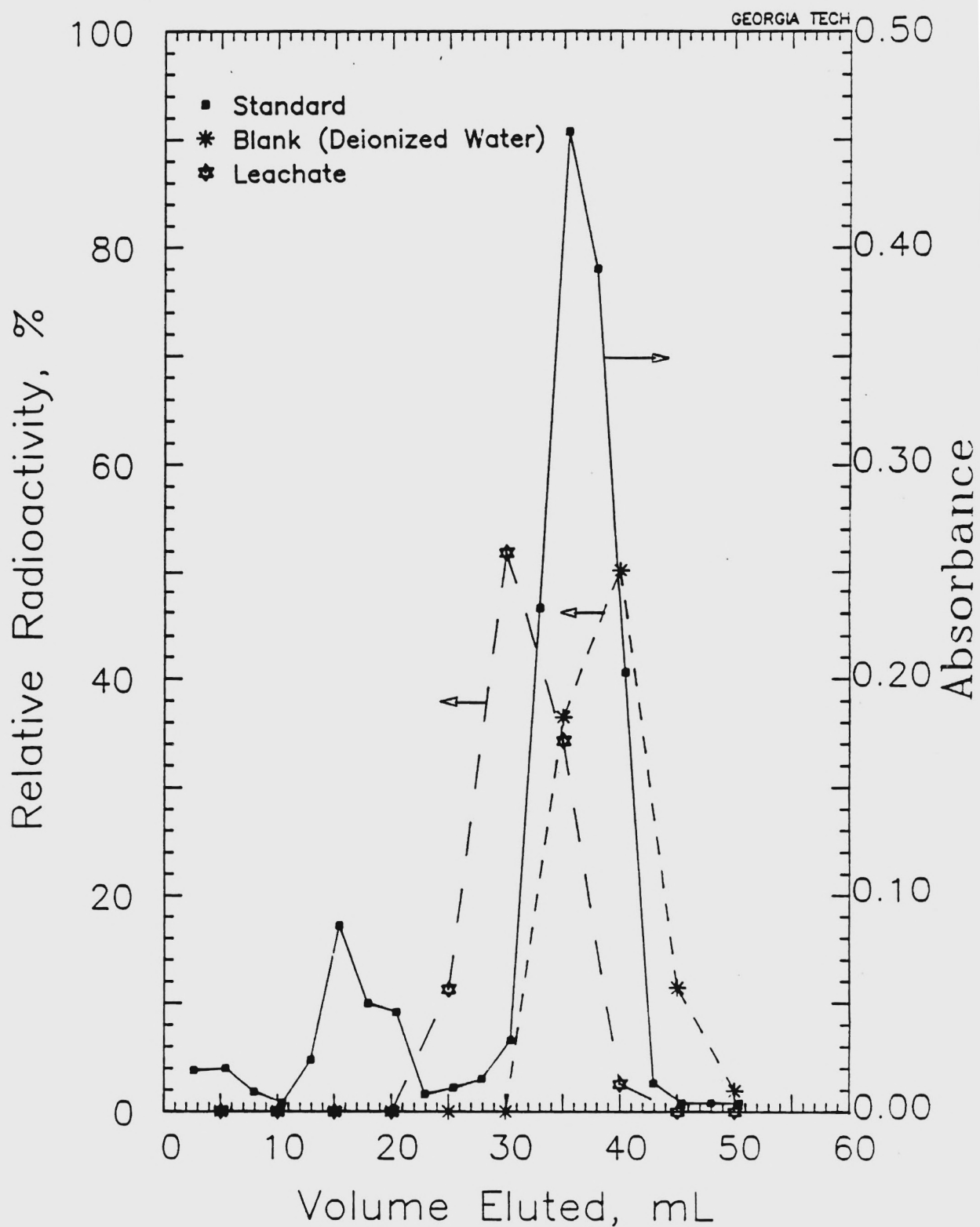


Figure 48. Radioactivity Present in Different GPC Fractions of Blank and Leachate.

Sr-85, and Ce-141 in leachate from the pilot-scale landfill without provisions for leachate recycle (Cell A) were lower than the levels present in leachate from Cell B. Based on these maximum levels present in leachate (Co-58 = 12.7%, Sr-85 = 15.7%, and Ce-141 = 0.03%), and neglecting any dilution, the migratory levels of Co-58, Sr-85, and Ce-141, in the event of a liner failure were 0.09 mCi/m³, 0.13 mCi/m³, and 0.0015 mCi/m³, respectively. Detailed computations are presented in Appendix C. These levels are below the maximum permissible concentrations of these radionuclides in water (Co-58 = 1 mCi/m³, Sr-85 = 3 mCi/m³, and Ce-141 = 3 mCi/m³; 10CFR, Part 20, Appendix B). Thus, the codisposal of the selected radionuclides, Co-58, Sr-85, and Ce-141, with municipal refuse is feasible at the loading levels (Co-58 = 1.6×10^{-7} Ci/kg, Sr-85 = 2.0×10^{-7} Ci/kg, and Ce-141 = 1.06×10^{-6} Ci/kg) utilized in this study.

CHAPTER V

CONCLUSIONS

Based on the results obtained from this study, the following conclusions can be drawn:

1. Co-58, Sr-85, and Ce-141 can be safely codisposed with municipal refuse within landfills at the loading levels utilized in this study, since the migratory activity levels of these radionuclides, in the event of a liner failure, will be less than the maximum permissible concentrations of these radionuclides in water above natural background. The loading levels, based on dry weight of refuse, for Co-58, Sr-85, and Ce-141 were 1.6×10^{-7} Ci/kg, 2.0×10^{-7} Ci/kg, and 1.06×10^{-6} Ci/kg, respectively.
2. The containment mechanism for Co-58 within the landfills was the formation of insoluble cobalt sulfide. The actual cobalt concentrations in leachate, however, were greater than equilibrium concentrations of cobalt in the presence of sulfide, carbonate, and hydroxide. This increase in the concentrations of cobalt in leachate is attributed to the formation of soluble complexes with other ligands (viz., humic substances) present in leachate during landfill stabilization. A separate confirmation of complexation of cobalt with humic substances present in leachate was provided

by companion laboratory-scale studies, wherein, leachate samples from the landfill operated as a landfill with leachate recycle (Cell B) solubilized Co-58 from cobalt sulfide precipitate.

3. The containment of Sr-85 within the landfill may be attributed to the formation of insoluble strontium carbonate complexes. This behavior was particularly reflected by strontium in leachate from the landfill operated as a landfill with leachate recycle (Cell B), as the strontium concentrations in leachate decreased with the change in the predominant buffer system from acetate to carbonate and increases in pH and TIC of leachate.
4. During landfill stabilization, the maximum levels of Co-58 and Sr-85 in leachate from Cell B, expressed as a percent of the initial loading levels, were 12.7% and 15.7%, respectively. The activity of Co-58 and Sr-85 in leachate corresponding to these levels, however, were lower than the maximum permissible concentration of these radionuclides in water above natural background.
5. Greater than 99.97% of the initial loading level of Ce-141 was retained within the landfills. The containment of Ce-141 within the landfills may be attributed to its adsorption onto the refuse.
6. During landfill stabilization, the behavior exhibited by Co-58 and Sr-85 in leachate from the landfill operated as a landfill with leachate recycle (Cell B) was similar to the

behavior exhibited by their respective stable forms. Thus, there was no preferential partitioning of the radionuclides as compared to their stable forms.

7. Since the total volatile fatty acid concentration in leachate from the landfill operated as landfill with leachate recycle (Cell B) persisted at a high level (7-10 g/L) up to Day 152, an attempt was made to enable the stabilization processes to proceed from the acid formation phase to the subsequent phases, namely, the methane fermentation phase and the final maturation phase, by reducing the frequency of leachate recycle from once a day to once a week between Days 252-350. This led to the successful recovery of Cell B with respect to progress of stabilization processes from the acid formation phase to the subsequent phases.
8. In a landfill receiving both municipal refuse and Co-58, the mobilization of Co-58 by humic substances present in leachate can be minimized by terminating leachate recycle after the degradation of readily degradable organics has occurred (final maturation phase), since the termination of leachate recycle will reduce both the formation of humic substances, and the contact opportunities provided to the humic substances in leachate to complex Co-58.

CHAPTER VI

RECOMMENDATIONS

Based on the results of this study, the codisposal of Co-58, Sr-85, and Ce-141 with municipal refuse, at the loading levels of 1.6×10^{-7} Ci/kg, 2.0×10^{-7} Ci/kg, and 1.06×10^{-6} Ci/kg, respectively, within landfills is recommended. In addition, further studies at higher loading levels of these radionuclides are recommended. Furthermore the technical feasibility of codisposal of radionuclides, namely, P-32, Cr-51, Se-75, Mo-99, I-131, Pu-238, Pu-239, and Pu-240, with municipal refuse within landfills should be evaluated. These radionuclides constitute LLRW and are generated from two sources: nuclear fuel cycle and nuclear medicine. The codisposal of LLRW, that are short-lived (half-life less than 300 days), with municipal refuse within landfills is an attractive alternative to the disposal of these radionuclides by shallow land burial, since it may alleviate some of the storage problems for such radionuclides. Dominant scenarios and potential pathways with respect to LLRW codisposed with municipal refuse within landfills should be evaluated. The results obtained from such studies would aid in assessment of the behavior of LLRW, within landfills, during landfill stabilization.

APPENDIX A

Appendix A

Sample Calculation of Testing Means by Student t-test.

A-2HA

D.I. Water

$$n_1 = 17$$

$$n_2 = 20$$

$$v = (n_2 - 1) = 16$$

$$v_2 = (n_2 - 1) = 19$$

$$\bar{x}_1 = 8.3 \text{ mg/L}$$

$$\bar{x}_2 = 3.71 \text{ mg/L}$$

$$\sigma_1 = 1.2 \text{ mg/L}$$

$$\sigma_2 = 0.7 \text{ mg/L}$$

$$SS_1 = 22.9$$

$$SS_2 = 8.8$$

$$s^2 = \frac{22.9 + 8.8}{16 + 19}$$

$$s_D = s_x - s_x = \sqrt{s^2 \left(\frac{1}{n_1} + \frac{1}{n_2} \right)} = 0.3$$

$$t = \frac{8.3 - 3.7}{0.3} = 15.33$$

From Tables:

$$t_{v=40, \alpha=0.05} = 2.021$$

$$t_{v=30, \alpha=0.05} = 2.042$$

$$\therefore \bar{x}_1 \neq \bar{x}_2$$

APPENDIX B

```

C *****
C
C MINEQL WAS MODIFIED FOR CDC APPLICATION BY S.B.GHOSH
C
C *****
PROGRAM MAIN(INPUT, INPUT2, INPUT3, OUTPUT, TAPE5=INPUT,
1TAPE10=INPUT2, TAPE11=INPUT3, TAPE6=OUTPUT)
COMMON/PAARM/NXDIM, NYDIM, ITMAX, ITER, EPS, NN(6), NNN
COMMON/VAR/GX(30), X(30), T(30), Y(30), Z(30,30),
1C(400), GC(400), GK(400), A(400,30), IDX(30), IDY(400)
C
NYDIM=400
NXDIM=30
ITMAX=100
EPS=1.E-4
ITER=0
THRSH=.01
XMU=0.01
C
CALL INPUT
CALL INION
CALL IONCOR(XMU)
CALL OINCMP
CALL OINSPC
C
RUN A SERIES OF COMPUTATIONS WITH PH FIXED FROM PH=(1 - 14)
DO 2222 J=1,14
ITER=0
GK(IADY(50))=J
10 CONTINUE
CALL SOLID
CALL SOLVE
CALL SOLIDX(K)
IF (K.NE.0) GO TO 10
CALL OUTCMP
CALL OUTSPC
CALL OUTPC(THRSH)
2222 CONTINUE
STOP
END
SUBROUTINE ERROR(I)
CHARACTER*4 MSG(40)
DATA MSG/
1'COMP', 'PONE', 'NTS ', '> NX', 'DIM ',
2'SPEC', 'IES ', '> NY', 'DIM ', ' ',
3'ID ', 'NOT ', 'FOUN', 'D: I', 'NPUT',
4'ID ', 'NOT ', 'FOUN', 'D: I', 'ADY ',
5'ID ', 'NOT ', 'FOUN', 'D: I', 'ADX ',
6'PHAS', 'E RU', 'LE V', 'IOLA', 'TION',
7'ITER', 'ATIO', 'NS >', 'ITMA', 'X ',
8'SING', 'ULAR', ' Z M', 'ATRI', 'X '/
C
I1=(I-1)*5+1
I2=I*5
WRITE(6,600) I, (MSG(II), II=I1, I2)
600 FORMAT('1', '**** EXECUTION TERMINATED ** ERROR', I3, ' ** ', 5A4)
C
CALL OUTCMP
CALL OUTSPC
STOP
END

```

	SUBROUTINE EXCOL(J0,JJ)	00000500
	COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN	00000510
	COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30),	00000520
	1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400)	00000530
C	IV=IDX(JJ)	00000540
	IDX(JJ)=IDX(J0)	00000550
	IDX(J0)=IV	00000560
	V=X(J0)	00000570
	X(J0)=X(JJ)	00000580
	X(JJ)=V	00000590
	V=GX(J0)	00000600
	GX(J0)=GX(JJ)	00000610
	GX(JJ)=V	00000620
	V=T(J0)	00000630
	T(J0)=T(JJ)	00000640
	T(JJ)=V	00000650
	DO 603 I=1,NYDIM	00000660
	V=A(I,J0)	00000670
	A(I,J0)=A(I,JJ)	00000680
	A(I,JJ)=V	00000690
603	CONTINUE	00000700
	RETURN	00000710
C	ENTRY EXROW(IO,II)	00000720
	IV=IDY(II)	00000730
	IDY(II)=IDY(IO)	00000740
	IDY(IO)=IV	00000750
	DO 46 J=1,NXDIM	00000760
	V=A(IO,J)	00000770
	A(IO,J)=A(II,J)	00000780
	A(II,J)=V	00000790
46	CONTINUE	00000800
	V=GK(IO)	00000810
	GK(IO)=GK(II)	00000820
	GK(II)=V	00000830
	RETURN	00000840
	END	00000850
	FUNCTION IADY(IDYT)	00000860
	COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN	00000870
	COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30),	00000880
	1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400)	00000890
C	II=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)+NN(6)	00000900
	DO 10 I=1,II	00000910
	IF(IDY(I).EQ.IDYT) GO TO 20	00000920
10	CONTINUE	00000930
	CALL ERROR(4)	00000940
20	IADY=I	00000950
	RETURN	00000960
C	ENTRY IADX(IDXT)	00000970
	JJ=NNN	00000980
	DO 110 J=1,JJ	00000990
	IF(IDX(J).EQ.IDXT) GO TO 120	0001000
110	CONTINUE	0001010
	CALL ERROR(5)	0001020
120	IA DX=J	0001030
	RETURN	0001040
	END	0001050
		0001060
		0001070
		0001080
		0001090


```

SUBROUTINE INION                                00001100
COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN 00001110
COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30),    00001120
1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400) 00001130
DIMENSION IONZ(200)                             00001140
C                                                    00001150
GF0=0.0                                           00001160
READ(11,500) (IONZ(J),J=1,200)                  00001170
500 FORMAT(40I2)                                00001180
C                                                    00001190
RETURN                                           00001200
ENTRY CONCOR(XMU)                                00001210
NC=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)+NN(6)          00001220
NX=NNN                                           00001230
C                                                    00001240
ET=-0.5                                           00001250
SI=SQRT(XMU)                                     00001260
GF=ET*(SI/(1.0+SI)-0.2*XMU)                     00001270
C                                                    00001280
DGF=GF-GF0                                       00001290
GF0=GF                                           00001300
C                                                    00001310
WRITE(6,60) XMU,GF0                             00001320
60 FORMAT('0',' IONIC STRENGTH = ',1PE9.2,5X,' LOG F(Z=1) =',0PF6.2) 00001330
C                                                    00001340
DO 20 I=1,NC                                     00001350
VJ=0.0                                           00001360
VI=0.0                                           00001370
DO 30 J=1,NX                                     00001380
VJ=VJ+A(I,J)*IONZ(IDX(J))*IONZ(IDX(J))          00001390
VI=VI+A(I,J)*IONZ(IDX(J))                      00001400
30 CONTINUE                                       00001410
GK(I)=GK(I)+DGF*(VJ-VI*VI)                     00001420
20 CONTINUE                                       00001430
RETURN                                           00001440
END                                               00001450
SUBROUTINE INPUT                                00001460
COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN 00001470
COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30),    00001480
1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400) 00001490
C                                                    00001500
DIMENSION IADXT(200),IAT(4),IDT(4)              00001510
C                                                    00001520
C INITIALIZE ADDRESS                             00001530
DO 20 J=1,200                                    00001540
20 IADXT(J)=0                                     00001550
C                                                    00001560
C INPUT PROBLEM DATA                           00001570
J=0                                               00001580
10 READ(5,500) IDXT,GXT,TT                      00001590
IF(IDXT.EQ.0) GO TO 90                          00001600
J=J+1                                            00001610
IDX(J)=IDXT                                      00001620
GX(J)=GXT                                        00001630
T(J)=TT                                          00001640
X(J)=10**GXT                                    00001650
IADXT(IDXT)=J                                    00001660
GO TO 10                                         00001670
90 CONTINUE                                       00001680
IF (J.GT.NXDIM) CALL ERROR(1)                  00001690

```

	NNN=J	00001700
C		00001710
C	INITIALIZE NN	00001720
	DO 100 L=1,6	00001730
100	NN(L)=0	00001740
C	INITIALIZE A	00001750
	DO 110 I=1,NYDIM	00001760
	DO 110 J=1,NXDIM	00001770
110	A(I,J)=0.0	00001780
C	INPUT BASIS IN A MATRIX	00001790
	JJ=NNN	00001800
	DO 200 I=1,JJ	00001810
	IDY(I)=IDX(I)	00001820
	A(I,I)=1.0	00001830
200	GK(I)=0.0	00001840
	NN(1)=JJ	00001850
C		00001860
C	INPUT THERMODYNAMIC DATA	00001870
	I=NN(1)	00001880
	DO 400 L=2,6	00001890
	IO=I	00001900
	READ(10,510) IN	00001910
	IF(IN.EQ.0) GO TO 400	00001920
	DO 300 II=1,IN	00001930
	READ(10,510) IDYT,GKT,(IDT(J),IAT(J),J=1,4)	00001940
	DO 310 J=1,4	00001950
	JTEST=IDT(J)	00001960
	IF(JTEST.EQ.0) GO TO 310	00001970
	IF(IADXT(JTEST).EQ.0) GO TO 300	00001980
310	CONTINUE	00001990
	I=I+1	00002000
	IDY(I)=IDYT	00002010
	GK(I)=GKT	00002020
	DO 320 J=1,4	00002030
	JTEST=IDT(J)	00002040
	IF(JTEST.EQ.0) GO TO 320	00002050
	A(I,IADXT(JTEST))=IAT(J)	00002060
320	CONTINUE	00002070
300	CONTINUE	00002080
	NN(L)=I-IO	00002090
400	CONTINUE	00002100
C		00002110
C	READ SPECIES MODIFICATION & TYPE SPECIFICATIONS	00002120
	ENTRY INTYPE	00002130
2000	READ(5,510) LTYPE	00002140
	IF(LTYPE.EQ.0) GO TO 1000	00002150
3000	READ(5,510) IDYT,GKT,(IDT(J),IAT(J),J=1,4)	00002160
	IF(IDYT.EQ.0) GO TO 2000	00002170
C	SEARCH	00002180
	II=0	00002190
	DO 710 L=1,6	00002200
	IF>NN(L).EQ.0) GO TO 710	00002210
	IO=II+1	00002220
	II=II+NN(L)	00002230
	DO 720 I=IO,II	00002240
	IF(IDY(I).NE.IDYT) GO TO 720	00002250
	IF(GKT.NE.0.0) GK(I)=GKT	00002260
	GO TO 900	00002270
720	CONTINUE	00002280
710	CONTINUE	00002290

```

C      SEARCH UNSUCCESSFUL: ENTER NEW SPECIES                                00002300
      IALLO=0                                                                00002310
      DO 810 J=1,4                                                            00002320
      JTEST=IDT(J)                                                            00002330
      IF (JTEST.EQ.0) GO TO 810                                              00002340
      IF (IADX(JTEST).EQ.0) CALL ERROR(3)                                    00002350
      IALLO=1                                                                00002360
810    CONTINUE                                                              00002370
      IF (IALLO.EQ.0) CALL ERROR(3)                                          00002380
      I=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)+NN(6)+1                             00002390
      NN(6)=NN(6)+1                                                         00002400
      L=6                                                                    00002410
      IDY(I)=IDYT                                                            00002420
      GK(I)=GKT                                                              00002430
      DO 820 J=1,4                                                            00002440
      JTEST=IDT(J)                                                            00002450
      IF (JTEST.EQ.0) GO TO 820                                              00002460
      A(I,IADX(JTEST))=IAT(J)                                               00002470
820    CONTINUE                                                              00002480
C      MOVE SPECIES I FROM TYPE L TO LTYPE                                  00002490
900    CONTINUE                                                              00002500
      IF (L.EQ.LTYPE) GO TO 910                                              00002510
      K=1                                                                    00002520
      II=0                                                                    00002530
      DO 940 LL=1,L                                                           00002540
940    II=II+NN(LL)                                                          00002550
      IF (LTYPE.GT.L) GO TO 920                                              00002560
      K=-1                                                                    00002570
      II=II-NN(L)+1                                                          00002580
920    CONTINUE                                                              00002590
      NN(L)=NN(L)-1                                                         00002600
      NN(LTYPE)=NN(LTYPE)+1                                                 00002610
930    CALL EXROW(I,II)                                                      00002620
      L=L+K                                                                  00002630
      I=II                                                                    00002640
      II=II+NN(L)*K                                                          00002650
      IF (L.NE.LTYPE) GO TO 930                                              00002660
910    CONTINUE                                                              00002670
      GO TO 3000                                                            00002680
C                                                                              00002690
1000   CONTINUE                                                             00002700
      RETURN                                                                00002710
500    FORMAT(I5,2X,F7.2,E7.2)                                              00002720
510    FORMAT(I5,2X,F7.2,4(I4,I3))                                          00002730
      END                                                                    00002740
      SUBROUTINE OUTPUT                                                      00002750
      CHARACTER*4 NAME(200),TYPE(42)
      COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN
      COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30),
1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400)
      DIMENSION IAT(4),IDT(4)
      DATA NAME/
1'CA ','MG ','SR ','K ','NA ','FE3 ','FE2 ','MN2 ','CU2 ',
2'BA ','CD ','ZN ','NI ','HG ','PB ','CO2 ','CO3 ','AG ',
3'CR ','AL ','CS ','LI ','BE ','SC ','TIO ','SN2 ','SN4 ',
4'LA ','CE3 ','AU1 ','TH4 ','UO2 ','CU1 ','X34 ','X35 ','X36 ',
513*','H ','48*','E-',
6'CO3-','SO4 ','CL ','F ','BR ','I ','NH3 ','S ','PO4 ',
7'P2O7','PO10','SIO3','S2O3','CN ','AC ','ACAC','CIT ','OX ',
8'SAL ','TART','EN ','DIP ','SUSA','GLY ','GLU ','PIC ','NTA ',

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9'EDTA','DCTA','CYST','NOR ','PHTH','ARG ','ORN ','LYS ','HIS ','      00002880
&'ASP ','SER ','ALA ','TYR ','MET ','VAL ','THR ','PHE ','ISO ','      00002890
1'LEU ','PRO ','BOH4','SO3 ','SCN ','NHOH','MOO4','WO4 ','ASO4','      00002900
2'VO4 ','SEO4','NO3 ','TRIS','X159','X160','X161','X162','X163','      00002910
337*' '/'      00002920
DATA TYPE/
1'I - ','COMP','ONEN','TS ','3*' ',      00002940
2'II - ','COM','PLEX','ES ','3*' ',      00002950
3'III ','- FI','XED ','SOLI','DS ','2*' ',      00002960
4'IV - ','PRE','CIPI','TATE','D SO','LIDS',' ',' ',      00002970
5'V - ','DISS','OLVE','D SO','LIDS','2*' ',      00002980
6'VI - ','SPE','CIES','NOT','CON','SIDE','RED'/'      00002990
C      00003010
C INPUT DATA COMPONENTS      00003020
ENTRY OINCOMP      00003030
JJ=NNN      00003040
WRITE(6,700)      00003050
WRITE(6,610)      00003060
DO 10 J=1,JJ      00003070
10 WRITE(6,620) IDX(J),X(J),GX(J),T(J),NAME(IDX(J))      00003080
RETURN      00003090
C      00003100
C INPUT DATA SPECIES      00003110
ENTRY OINSPC      00003120
II=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)+NN(6)      00003130
JJ=NNN      00003140
L=0      00003150
M=1      00003160
DO 50 I=1,II      00003170
IF(M.NE.I) GO TO 30      00003180
20 L=L+1      00003190
IF (NN(L).EQ.0) GO TO 20      00003200
M=M+NN(L)      00003210
L2=L*7      00003220
L1=L2-6      00003230
WRITE (6,600)      00003240
WRITE (6,630) (TYPE(N),N=L1,L2)      00003250
30 K=0      00003260
DO 40 J=1,JJ      00003270
IF (A(I,J).EQ.0) GO TO 40      00003280
K=K+1      00003290
IDT(K)=IDX(J)      00003300
IAT(K)=A(I,J)      00003310
40 CONTINUE      00003320
50 WRITE(6,640) IDY(I),GK(I),(NAME(IDT(J))),IAT(J),J=1,K)      00003330
RETURN      00003340
C      00003350
C COMPONENT OUTPUT      00003360
ENTRY OUTCMP      00003370
II=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)+NN(6)      00003380
JJ=NNN      00003390
WRITE(6,650) ITER      00003400
WRITE(6,660)      00003410
DO 60 J=1,JJ      00003420
60 WRITE(6,670) IDX(J),X(J),GX(J),T(J),Y(J),NAME(IDX(J))      00003430
RETURN      00003440
C      00003450
C SPECIES OUTPUT      00003460
ENTRY OUTSPC      00003470
II=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)+NN(6)      00003480

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	JJ=NNN	00003490
	L=0	00003500
	M=1	00003510
	DO 100 I=1,II	00003520
	IF (M.NE.I) GO TO 80	00003530
70	L=L+1	00003540
	IF (NN(L).EQ.0) GO TO 70	00003550
	M=M+NN(L)	00003560
	L2=L*7	00003570
	L1=L2-6	00003580
	WRITE (6,600)	00003590
	WRITE (6,680) (TYPE(N),N=L1,L2)	00003600
80	K=0	00003610
	DO 90 J=1,JJ	00003620
	IF (A(I,J).EQ.0) GO TO 90	00003630
	K=K+1	00003640
	IDT(K)=IDX(J)	00003650
	IAT(K)=A(I,J)	00003660
90	CONTINUE	00003670
100	WRITE(6,690) IDY(I),C(I),GC(I),GK(I), (NAME(IDT(J))), IAT(J), J=1,K)	00003680
	RETURN	00003690
C		00003700
C	OUTPUTS PERCENTAGES	00003710
	ENTRY OUTPC(TOL)	00003720
	JJ=NNN	00003730
	WRITE (6,730)	00003740
	DO 110 J=1,JJ	00003750
	WRITE (6,600)	00003760
	WRITE (6,710) NAME(IDX(J))	00003770
	V=T(J)	00003780
	IF(NN(3).EQ.0) GO TO 140	00003790
	I1=NN(1)+NN(2)+1	00003800
	I2=NN(1)+NN(2)+NN(3)	00003810
	DO 150 I=I1,I2	00003820
	V=V-A(I,J)*C(I)	00003830
150	CONTINUE	00003840
140	CONTINUE	00003850
	IF(V.EQ.0) GO TO 110	00003855
	I1=1	00003860
	I2=NN(1)+NN(2)	00003870
	DO 160 LL=1,2	00003880
	IF (LL.EQ.1) GO TO 170	00003890
	I1=NN(1)+NN(2)+NN(3)+1	00003900
	I2=NN(1)+NN(2)+NN(3)+NN(4)	00003910
170	CONTINUE	00003920
	DO 120 I=I1,I2	00003930
	PC=A(I,J)*C(I)/V	00003940
	IF (PC.LT.TOL) GO TO 120	00003950
	PC=PC*100	00003960
	L=0	00003970
	DO 130 K=1,JJ	00003980
	IF (A(I,K).EQ.0) GO TO 130	00003990
	L=L+1	00004000
	IDT(L)=IDX(K)	00004010
	IAT(L)=A(I,K)	00004020
130	CONTINUE	00004030
	WRITE (6,720) PC, IDY(I), (NAME(IDT(M))), IAT(M), M=1,L)	00004040
120	CONTINUE	00004050
160	CONTINUE	00004060
110	CONTINUE	00004070

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RETURN 00004080
C 00004090
600 FORMAT('0') 00004100
610 FORMAT('0',' ID',10X,'X',4X,'LOGX',10X,'T',5X,'COMPONENTS') 00004110
620 FORMAT('0',I5,2X,1PE9.2,2X,OPF6.2,2X,1PE9.2,5X,A4) 00004120
630 FORMAT('0',' ID',4X,'LOGK',5X,'SPECIES: TYPE ',7A4) 00004130
640 FORMAT('0',I5,2X,OPF6.2,5X,4(A4,1X,I3,4X)) 00004140
650 FORMAT('1',' OUTPUT DATA: ITERATIONS = ',I3) 00004150
660 FORMAT('0',' ID',10X,'X',4X,'LOGX',10X,'T',10X,'Y',5X, 00004160
1'SPECIES') 00004170
670 FORMAT('0',I5,2X,1PE9.2,2X,OPF6.2,2(2X,1PE9.2),5X,A4) 00004180
680 FORMAT('0',' ID',10X,'C',4X,'LOGC',4X,'LOGK',5X, 00004190
1'SPECIES: TYPE ',7A4) 00004200
690 FORMAT('0',I5,2X,1PE9.2,2(2X,OPF6.2),5X,4(A4,1X,I3,4X)) 00004210
700 FORMAT('1',' INPUT DATA') 00004220
710 FORMAT('0',A4) 00004230
720 FORMAT('0',7X,F6.1,5X,'PERCENT BOUND IN SPECIES #',I5,3X,4(A4,1X, 00004240
1I3,4X)) 00004250
730 FORMAT('1',' PERCENTAGE DISTRIBUTION OF COMPONENTS') 00004260
END 00004270
SUBROUTINE SIMQ(Z,Y,N,NXDIM) 00004280
DIMENSION Z(NXDIM,NXDIM),Y(NXDIM) 00004290
C PROVISION FOR N=1 00004300
IF(N.NE.1) GO TO 50 00004310
Y(1)=Y(1)/Z(1,1) 00004320
RETURN 00004330
50 CONTINUE 00004340
C ELEMENT OF ELIMINATION 00004350
N1=N-1 00004360
DO 10 M=1,N1 00004370
ZMAX=0. 00004380
IMAX=0 00004390
C FIND MAX OF COLUMN 00004400
DO 20 I=M,N 00004410
IF(ABS(Z(I,M)).LE.ZMAX) GO TO 20 00004420
IMAX=I 00004430
ZMAX=ABS(Z(I,M)) 00004440
20 CONTINUE 00004450
C ERROR RETURN 00004460
IF(IMAX.NE.0) GO TO 30 00004470
CALL ERROR(8) 00004480
30 CONTINUE 00004490
C ROW INTERCHANGE 00004500
IF(IMAX.EQ.M) GO TO 35 00004510
V=Y(M) 00004520
Y(M)=Y(IMAX) 00004530
Y(IMAX)=V 00004540
DO 40 J=M,N 00004550
V=Z(M,J) 00004560
Z(M,J)=Z(IMAX,J) 00004570
Z(IMAX,J)=V 00004580
40 CONTINUE 00004590
35 CONTINUE 00004600
C DIAGONALIZE 00004610
M1=M+1 00004620
DO 70 I=M1,N 00004630
V=Z(I,M)/Z(M,M) 00004640
Y(I)=Y(I)-V*Y(M) 00004650
DO 70 J=M,N 00004660
Z(I,J)=Z(I,J)-V*Z(M,J) 00004670

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70    CONTINUE                                00004680
10    CONTINUE                                00004690
C     BACK SUBSTITUTE                        00004700
      Y(N)=Y(N)/Z(N,N)                        00004710
      N1=N-1                                00004720
      DO 100 K=1,N1                          00004730
      I=N-K                                  00004740
      I1=I+1                                00004750
      DO 90 J=I1,N                          00004760
90    Y(I)=Y(I)-Y(J)*Z(I,J)                  00004770
100   Y(I)=Y(I)/Z(I,I)                      00004780
      RETURN                                00004790
      END                                    00004800
      SUBROUTINE SOLID                        00004810
      COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN 00004820
      COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30), 00004830
      1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400) 00004840
C
      IF(NN(3)+NN(4).EQ.0) RETURN            00004850
      LL=NN(3)+NN(4)                        00004860
      IO=NN(1)+NN(2)+NN(3)+NN(4)+1          00004880
      JO=NNN+1                              00004890
      DO 601 L=1,LL                         00004900
      IO=IO-1                               00004910
      JO=JO-1                               00004920
C FIND JEXC FOR IO                          00004930
      DO 602 J=1,JO                         00004940
      IF (ABS(A(IO,J)-(J-1))).GT.0.001) GO TO 603 00004950
602   CONTINUE                              00004960
      CALL ERROR(6)                          00004970
603   JEXC=JO-(J-1)                        00004980
      CALL EXCOL(JEXC,JO)                   00004990
      NXS=JO-1                              00005000
      NCS=IO-1                              00005010
C MODIFY A,B,T                             00005020
      DO 604 I=1,NCS                        00005030
      DO 604 J=1,NXS                        00005040
604   A(I,J)=A(I,J)-A(IO,J)*A(I,JO)/A(IO,JO) 00005050
      DO 605 J=1,NXS                        00005060
605   T(J)=T(J)-A(IO,J)*T(JO)/A(IO,JO)      00005070
      DO 606 I=1,NCS                        00005080
606   GK(I)=GK(I)-A(I,JO)*GK(IO)/A(IO,JO) 00005090
601   CONTINUE                              00005100
C                                             00005110
      RETURN                                00005120
      END                                    00005130
      SUBROUTINE SOLIDX(KK)                  00005140
      COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN 00005150
      COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30), 00005160
      1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400) 00005170
C                                             00005180
      KK=0                                  00005190
C                                             00005200
      IF(NN(3)+NN(4).EQ.0) GO TO 470        00005210
      LL=NN(3)+NN(4)                        00005220
      II=NN(1)+NN(2)                        00005230
      IO=NN(1)+NN(2)+1                      00005240
      JO=NNN-NN(3)-NN(4)+1                  00005250
      JJ=NNN                                00005260
C MOLE BALANCE MINUS SOLIDS                00005270

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DO 440 J=J0,JJ	00005280
Y(J)=--T(J)	00005290
DO 440 I=1,II	00005300
Y(J)=Y(J)+A(I,J)*C(I)	00005310
440 CONTINUE	00005320
C AMOUNT OF SOLIDS	00005330
DO 460 L=1,LL	00005340
C(I0)=--Y(J0)/A(I0,J0)	00005350
GC(I0)=ALOG10(ABS(C(I0)))	00005360
DO 450 K=J0,JJ	00005370
Y(K)=Y(K)+A(I0,K)*C(I0)	00005380
450 CONTINUE	00005390
C UNMODIFY A,B,T,GX,X	00005400
NXS=J0-1	00005410
NCS=I0-1	00005420
V=GK(I0)	00005430
DO 571 J=1,NXS	00005440
571 V=V+A(I0,J)*GX(J)	00005450
GX(J0)=--V/A(I0,J0)	00005460
X(J0)=10.**GX(J0)	00005470
DO 61 I=1,NCS	00005480
DO 61 J=1,NXS	00005490
61 A(I,J)=A(I,J)+A(I0,J)*A(I,J0)/A(I0,J0)	00005500
DO 62 J=1,NXS	00005510
62 T(J)=T(J)+A(I0,J)*T(J0)/A(I0,J0)	00005520
DO 63 I=1,NCS	00005530
63 GK(I)=GK(I)+A(I,J0)*GK(I0)/A(I0,J0)	00005540
I0=I0+1	00005550
J0=J0+1	00005560
460 CONTINUE	00005570
470 CONTINUE	00005580
C SOLUBILITY PRODUCTS	00005590
IF(NN(5)+NN(6).EQ.0) GO TO 220	00005600
I0=NN(1)+NN(2)+NN(3)+NN(4)+1	00005610
II=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)+NN(6)	00005620
JJ=NNN	00005630
DO 210 I=I0,II	00005640
V=GK(I)	00005650
DO 200 J=1,JJ	00005660
200 V=V+A(I,J)*GX(J)	00005670
GC(I)=V	00005680
C(I)=10.**v	00005690
210 CONTINUE	00005700
220 CONTINUE	00005710
C	00005720
C CHECK FOR DISSOLUTION	00005730
C	00005740
IF(NN(4).EQ.0) GO TO 45	00005750
IMIN=0	00005760
VMIN=0	00005770
I0=NN(1)+NN(2)+NN(3)+1	00005780
II=NN(1)+NN(2)+NN(3)+NN(4)	00005790
DO 44 I=I0,II	00005800
IF(C(I).GT.VMIN) GO TO 44	00005810
VMIN=C(I)	00005820
IMIN=I	00005830
44 CONTINUE	00005840
IF(IMIN.EQ.0) GO TO 45	00005850
WRITE(6,610) ITER,IDY(IMIN)	00005860
610 FORMAT('0',' ITERATIONS= ',I3,': SOLID ',I5,' DISSOLVES')	00005870

CALL EXROW(IMIN,II)	00005880
NN(5)=NN(5)+1	00005890
NN(4)=NN(4)-1	00005900
KK=-1	00005910
RETURN	00005920
45 CONTINUE	00005930
C	00005940
C CHECK FOR PRECIPITATION	00005950
C	00005960
IF (NN(5).EQ.0) GO TO 22	00005970
VMAX=0	00005980
IMAX=0	00005990
IO=NN(1)+NN(2)+NN(3)+NN(4)+1	00006000
II=NN(1)+NN(2)+NN(3)+NN(4)+NN(5)	00006010
DO 47 I=IO,II	00006020
IF(GC(I).LT.VMAX) GO TO 47	00006030
VMAX=GC(I)	00006040
IMAX=I	00006050
47 CONTINUE	00006060
IF(IMAX.EQ.0) GO TO 22	00006070
WRITE(6,600) ITER,IDY(IMAX)	00006080
600 FORMAT('0',' ITERATIONS= ',I3,': SOLID ',I5,' PRECIPITATES')	00006090
CALL EXROW(IMAX,IO)	00006100
NN(4)=NN(4)+1	00006110
NN(5)=NN(5)-1	00006120
KK=1	00006130
22 CONTINUE	00006140
RETURN	00006150
END	00006160
SUBROUTINE SOLVE	00006170
COMMON/PAARM/NXDIM,NYDIM,ITMAX,ITER,EPS,NN(6),NNN	00006180
COMMON/VAR/GX(30),X(30),T(30),Y(30),Z(30,30),	00006190
1C(400),GC(400),GK(400),A(400,30),IDX(30),IDY(400)	00006200
C	00006210
NC=NN(1)+NN(2)	00006220
NX=NNN-NN(3)-NN(4)	00006230
1000 CONTINUE	00006240
C COMPLEXES	00006250
DO 2 I=1,NC	00006260
V=GK(I)	00006270
DO 3 J=1,NX	00006280
3 V=V+A(I,J)*GX(J)	00006290
GC(I)=V	00006300
C(I)=10.**GC(I)	00006310
2 CONTINUE	00006320
C MOLE BALANCE	00006330
DO 201 J=1,NX	00006340
V=-T(J)	00006350
DO 200 I=1,NC	00006360
200 V=V+A(I,J)*C(I)	00006370
Y(J)=V	00006380
201 CONTINUE	00006390
C COMPUTE Z	00006400
DO 300 I=1,NX	00006410
DO 300 J=1,NX	00006420
300 Z(I,J)=0	00006430
DO 400 J=1,NX	00006440
DO 400 I=1,NC	00006450
DO 400 K=1,NX	00006460
400 Z(J,K)=Z(J,K)+A(I,J)*A(I,K)*C(I)/X(K)	00006470

C CONVERGENCE TEST	00006480
DO 800 J=1,NX	00006490
VMAX=ABS(T(J))	00006500
DO 810 I=1,NC	00006510
IF(ABS(A(I,J)*C(I)).LT.VMAX) GO TO 810	00006520
VMAX=ABS(A(I,J)*C(I))	00006530
810 CONTINUE	00006540
IF (ABS(Y(J))/VMAX.GT.EPS) GO TO 840	00006550
800 CONTINUE	00006560
RETURN	00006570
840 ITER=ITER+1	00006580
IF(ITER.GT.ITMAX) CALL ERROR(7)	00006590
C ITERATE	00006600
CALL SIMQ(Z,Y,NX,NXDIM)	00006610
DO 500 J=1,NX	00006620
X(J)=X(J)-Y(J)	00006630
IF(X(J).GT.0.) GO TO 501	00006640
X(J)=(X(J)+Y(J))/10.	00006650
501 GX(J)=ALOG10(X(J))	00006660
500 CONTINUE	00006670
GO TO 1000	00006680
END	00006690

APPENDIX C

APPENDIX C

Potential Migratory Activity Levels of Co-58, Sr-85, and Ce-141,
for the Loading Levels Utilized in this Study, in the Event
of Leachate Migration into the Surrounding Environment

- Radionuclides Considered

Co-58; Half Life = 70.8 days

Sr-85; Half Life = 64.8 days

Ce-141; Half Life = 32.5 days

Assumptions:

1. Co-58, Sr-85, and Ce-141 are codisposed with municipal refuse at levels of 0.91 mCi, 1.09 mCi, and 5.78 mCi, respectively. Thus, the loading levels of Co-58, Sr-85, and Ce-141 utilized in this study, on a dry weight of refuse basis, were 1.6×10^{-7} Ci/kg, 2.0×10^{-7} Ci/kg, and 10.6×10^{-7} Ci/kg, respectively.
2. During the period of investigation, the maximum levels of Co-58, Sr-85, and Ce-141 present in leachate from the landfill with provisions for leachate recycle (Cell B), expressed as a percent of the initial loading levels, were 12.7%, 15.7%, and 0.03%, respectively. The levels of Co-58, Sr-85, and Ce-141 present in the leachate from the landfill operated in a single pass mode were lower than corresponding levels in leachate from Cell B. Thus, the maximum levels of Co-58, Sr-85, and Ce-141 present in leachate, from a landfill receiving both municipal refuse and these radionuclides at the aforementioned loading levels, are assumed as 13%, 16%, and 0.03%, respectively.

3. Infiltration into the landfill is limited to a maximum height of 0.3 m (1 ft) above the bottom layer of the landfill.
4. The as-placed porosity of the refuse within the landfill is 0.4.

In accordance with 10 CFR Part 20, Appendix B, the maximum concentrations of Co-58, Sr-85, and Ce-141 in water above the natural background are 1.0×10^{-3} Ci/m³, 3×10^{-3} Ci/m³, and 3×10^{-3} Ci/m³, respectively.

The maximum levels of Co-58, Sr-85, and Ce-141 present in the leachate are 13%, 16%, and 0.03% of the initial loading levels. Therefore, the maximum levels of Co-58, Sr-85, and Ce-141 present in the leachate are 0.12 mCi, 0.17 mCi, and 0.002 mCi, respectively. Assuming, leachate in the landfill is limited to a 0.3 m (1 ft) head over the bottom layer and an as-placed porosity value of 0.4 for the refuse, the volume of leachate contained in the landfill with provisions for leachate recycle (Cell B) is 1.11 m³. The actual volume of leachate in Cell B was 1.32 m³.

Assuming, in the event of a liner failure the entire leachate from Cell B migrated into the surrounding environment and neglecting any dilution, the migratory activity levels of the radionuclides will be:

$$\text{Co-58: } \frac{0.12 \text{ mCi}}{1.32 \text{ m}^3} = 0.09 \text{ mCi/m}^3$$

$$\text{Sr-85: } \frac{0.17 \text{ mCi}}{1.32 \text{ m}^3} = 0.13 \text{ mCi/m}^3$$

$$\text{Ce-141: } \frac{0.002 \text{ mCi}}{1.32 \text{ m}^3} = 0.0015 \text{ mCi/m}^3$$

These values of 0.09 mCi/m^3 , 0.13 mCi/m^3 , and 0.0015 mCi/m^3 for Co-58, Sr-85, and Ce-141, respectively, are lower than the maximum permissible concentrations of Co-58 (1 mCi/m^3), Sr-85 (3 mCi/m^3), and Ce-141 (3 mCi/m^3) in water above the natural background.

APPENDIX D

Table D-1. Sampling Date and Corresponding Day Since
Leachate Generation Was Initiated

Sampling Date	Time Since Leachate Production Began, Days
7/27/82	0
8/19/82	13
8/16/82	20
8/24/82	28
8/30/82	34
9/07/82	41
9/14/82	48
9/21/82	55
9/27/82	61
10/04/82	68
10/11/82	75
10/18/82	82
10/20/82	90
11/02/82	97
11/08/82	103
11/15/82	110
11/22/82	117
11/29/82	124
12/07/82	132
12/14/82	139
12/27/82	152
1/07/83	163
1/12/83	168
1/18/83	174
1/25/83	181
2/08/83	196
2/17/83	205
3/03/83	219
3/15/83	231
3/26/83	241
4/08/83	254
4/14/83	260
4/20/83	266
4/29/83	275
5/14/83	290
7/13/83	350
7/23/83	360

continued...

Table D-1. Continued...

Sampling Date	Time Since Leachate Production Began, Days
8/01/83	369
8/09/83	377
8/20/83	388
9/01/83	400
9/15/83	414
9/27/83	426
9/30/83	429
10/15/83	444
10/29/83	458
11/12/83	472
11/22/83	482
12/07/83	497
12/26/83	516
1/19/84	540
2/04/84	556
2/18/84	570
3/15/84	594
4/15/84	625
5/15/84	655
6/15/84	686
7/15/84	716
8/15/84	747
9/15/84	778
10/15/84	808
11/15/84	839
12/15/84	869
1/15/85	900
2/15/85	931
3/27/85	970

Table D-2. Maximum and Minimum Ambient Temperatures and Temperatures Within Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	<u>Ambient Temperature*</u>		<u>Internal Temperature</u>	
	Maximum	Minimum	Cell A	Cell B
	°C	°C	°C	°C
97	25.0	14.4	12	12
103	18.8	2.7	10	10
110	13.3	1.1	10	10
117	20.5	13.8	14	14
124	19.4	10.0	12	12
132	16.6	2.7	5	6
139	10.0	-3.3	4	4
152	21.6	14.4	13	13
163	15.5	2.7	7	7
168	1.1	-3.3	4	4
174	2.7	-5.5	3	3
181	10.0	-0.5	5	5
196	10.0	-3.3	3	3
205	17.7	6.6	8	8
219	23.8	8.3	10	10
231	23.3	8.8	14	14
241	12.7	1.6	14	14
254	19.4	15.0	14	15
260	18.3	15.0	18	18
266	14.4	-1.1	18	18
275	27.2	13.8	25	25
290	29.4	16.6	29	30
350	34.4	21.6	33	35
360	37.2	25.0	37	39
369	30.0	21.1	33	35
377	33.3	22.2	33	37
388	37.2	23.8	38	39
400	25.0	20.5	30	32
414	23.8	15.5	27	28
426	22.7	11.6	22	23
429	23.3	12.7	22	22
444	21.6	8.3	21	22
458	24.4	7.2	23	23
472	11.1	0.0	20	21
482	22.7	5.5	20	22

continued....

Table D-2. Continued...

Time Since Leachate Production Began, Days	<u>Ambient Temperature*</u>		<u>Internal Temperature</u>	
	Maximum	Minimum	Cell A	Cell B
	°C	°C	°C	°C
497	7.7	-1.1	18	18
516	-8.3	-17.7	7	8
540	2.7	-6.6	5	5
556	13.8	4.4	10	13
570	21.1	5.5	20	23
594	26.1	8.3	25	26
625	22.2	8.8	22	23
655	23.3	11.1	22	25
686	32.7	21.1	30	33
716	30.0	22.2	30	33
747	31.1	21.1	32	34
778	28.3	17.7	30	30
808	26.1	18.3	28	28
839	20.5	3.3	22	22
869	20.0	9.4	20	21
900	3.8	-3.3	10	12
931	6.6	-4.4	10	11
970	22.2	8.3	18	18

*Obtained from the National Oceanic and Atmospheric
Administration (NOAA), Asheville, NC

Table D-3. Variations in the Moisture Content of the
Municipal Refuse Within Cell A (Single Pass)
and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Moisture Content of Refuse (Based on as-placed weight of refuse)	
	Cell A %	Cell B %
0	65.3	60.3
14	63.4	64.4
21	62.1	64.4
28	66.1	68.7
49	67.2	70.9
56	69.2	73.9
61	69.0	73.9
68	68.8	73.9
75	68.6	73.9
82	68.4	73.9
90	68.2	73.9
97	67.1	73.8
110	67.3	73.8
117	67.1	73.8
124	66.7	73.8
132	66.1	73.8
139	65.4	73.8
152	65.5	73.7
174	63.8	73.7
181	62.6	73.7
196	62.2	73.7
205	61.9	73.7
219	61.7	73.7
241	61.8	73.7
260	61.5	73.6
290	61.4	73.6
350	61.3	73.5
369	61.3	73.5
388	61.2	73.4
414	61.1	73.4
444	61.0	73.4
472	60.9	73.3
497	60.8	73.2
540	60.7	73.1

continued...

Table D-3. Continued...

Time Since Leachate Production Began, Days	Moisture Content of Refuse (Based on as-placed weight of refuse)	
	Cell A %	Cell B %
570	60.6	73.0
594	60.5	73.0
625	60.5	73.0
686	60.3	72.9
747	60.2	72.8
808	60.1	72.7
869	60.0	72.7
970	59.8	72.5

Table D-4. Chemical Oxygen Demand (COD) of Leachate Samples From Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Chemical Oxygen Demand	
	Cell A	Cell B
	mg/L	mg/L
0	15,100	13,975
13	26,956	19,468
20	28,224	16,600
28	24,120	19,620
34	26,880	17,920
48	35,276	34,138
55	40,256	38,327
61	35,526	35,200
68	26,800	21,760
75	21,280	22,080
82	19,000	25,087
90	22,450	21,185
97	21,240	18,540
103	19,228	23,859
110	18,685	24,809
117	8,333	22,333
124	6,630	24,751
132	4,397	23,874
139	2,541	25,856
152	3,680	20,210
163	4,530	18,877
181	3,811	23,762
196	3,674	24,530
205	3,661	16,744
212	3,240	23,650
219	3,716	18,583
231	2,888	21,066
241	1,803	23,644
254	2,564	21,544
260	3,716	20,654
266	2,231	18,583
279	2,894	23,500
283	3,324	17,540
350	2,656	7,614
360	1,562	3,593
369	1,708	3,978

continued...

Table D-4. Continued...

Time Since Leachate Production Began, Days	Chemical Oxygen Demand	
	Cell A	Cell B
	mg/L	mg/L
377	1,532	3,548
388	1,640	3,785
400	1,551	1,689
414	1,482	1,689
426	1,448	1,655
444	1,812	2,202
458	1,525	2,116
472	1,680	2,634
482	1,834	2,864
497	1,538	2,402
516	1,437	2,216
540	1,290	2,450
556	1,063	2,553
570	1,276	2,340
594	1,137	1,962
625	1,261	1,872
655	1,098	1,921
686	1,100	1,831
716	1,061	1,771
747	1,256	1,662
778	1,324	1,692
808	1,231	1,620
839	1,251	1,738
869	1,267	1,644
900	1,452	1,623
931	1,441	1,457
970	1,349	1,522

Table D-5. Biochemical Oxygen Demand (BOD₅) of Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Biochemical Oxygen Demand	
	Cell A	Cell B
	mg/L	mg/L
0	6,849	7,025
13	12,120	13,317
20	14,066	9,904
28	15,215	12,200
34	15,502	11,913
48	15,000	14,003
55	22,200	19,400
61	12,000	12,900
68	15,900	15,000
75	12,360	13,980
82	12,300	12,900
90	12,100	13,725
97	12,060	11,700
103	13,700	18,000
117	7,800	15,000
124	6,000	15,600
132	2,700	14,100
139	2,350	17,700
152	2,150	15,650
181	2,950	18,650
196	3,530	14,500
205	3,515	10,550
219	3,010	12,500
241	1,700	12,500
254	2,307	14,649
266	1,940	13,937
283	3,024	13,865
350	2,100	4,900
360	1,020	1,860
369	1,308	2,090
377	958	1,740
388	1,100	1,980
400	1,012	1,210
414	924	1,026
426	725	980
444	980	1,216

continued...

Table D-5. Continued...

Time Since Leachate Production Began, Days	Biochemical Oxygen Demand	
	Cell A	Cell B
	mg/L	mg/L
458	815	1,370
472	905	1,495
482	925	1,612
497	705	1,465
540	625	1,250
556	465	1,165
625	255	374
686	265	352
747	339	415
778	370	389
808	356	437
839	350	364
869	380	386
900	428	421
931	373	313
970	370	342

Table D-6. Total Organic Carbon (TOC) of Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Total Organic Carbon	
	Cell A	Cell B
	mg/L	mg/L
0	5,300	4,500
13	7,367	6,722
20	7,081	5,704
28	6,978	5,771
34	7,081	5,274
48	8,580	7,020
55	8,372	8,450
61	7,359	7,827
68	8,220	9,150
75	7,020	7,430
82	7,150	8,300
90	10,200	8,870
97	8,670	9,340
103	8,210	9,280
110	6,270	8,490
117	5,800	9,220
124	5,050	10,050
132	4,425	9,660
152	3,022	10,000
181	2,250	8,600
196	2,500	7,500
205	2,032	6,300
212	2,500	7,750
219	1,825	7,926
231	1,219	6,703
241	1,117	8,536
254	1,315	7,653
260	1,865	6,715
266	1,121	6,423
279	1,560	7,983
283	1,710	6,121
350	1,790	1,950
360	1,366	1,566
369	1,564	1,790
377	1,350	1,540
388	1,450	1,790

continued...

Table D-6. Continued...

Time Since Leachate Production Began, Days	Total Organic Carbon	
	Cell A	Cell B
	mg/L	mg/L
400	797	980
426	907	943
444	1,095	1,275
458	975	1,065
472	1,165	1,238
482	1,210	1,289
497	853	1,306
540	709	1,029
556	829	1,302
570	1,050	1,159
594	807	853
625	741	668
655	712	835
686	733	732
716	818	656
747	800	665
778	757	691
808	733	720
839	711	698
869	768	650
900	800	615
931	775	610
970	767	609

Table D-7. Individual and Total Volatile Fatty Acids Present
in Leachate Samples from Cell A (Single Pass)

Time Since Leachate Production Began, Days	Volatile Fatty Acids					Total, expressed as mg/L Acetic Acid
	Acetic mg/L	Propionic mg/L	Isobutyric mg/L	Butyric mg/L	Valeric mg/L	
0	3,931	1,482	232	5,303	1,984	10,073
13	2,842	844	922	2,022	1,190	6,233
28	4,479	1,342	624	3,822	1,552	9,518
34	4,034	1,883	607	3,465	1,396	9,408
41	15,854	1,803	521	3,376	1,556	20,892
48	7,689	1,549	501	3,288	1,419	12,402
55	5,275	3,147	527	3,168	1,464	11,288
61	6,465	1,946	324	3,150	1,290	11,182
68	5,761	2,912	587	2,239	1,483	11,542
75	4,773	1,552	183	2,083	954	8,248
82	6,180	1,835	238	1,887	1,070	9,849
90	4,780	1,700	235	1,887	1,017	8,203
103	3,218	970	236	1,326	844	5,565
110	2,216	1,523	203	547	467	4,237
117	1,133	2,063	275	347	394	3,461
124	891	1,735	352	300	128	2,818
134	681	738	71	206	156	1,560
142	150	569	61	32	204	794
153	245	1,248	51	67	228	1,472
163	77	841	N.D. *	N.D.	N.D.	759
174	161	112	97	61	38	405
181	139	494	78	36	42	643
205	134	390	N.D.	43	287	650
212	130	398	N.D.	N.D.	N.D.	453

continued...

Table D-7. Continued...

Time Since Leachate Production Began, Days	Volatile Fatty Acids					Total, expressed as mg/L Acetic Acid
	Acetic mg/L	Propionic mg/L	Isobutyric mg/L	Butyric mg/L	Valeric mg/L	
219	110	366	14.	7	5	424
231	63	283	N.D.	N.D.	N.D.	292
254	178	57	33	42	14	295
266	105	50	22	31	16	183
290	150	65	31	30	19	256
350	27	30	10	11	14	75
360	N.D.	33	3	N.D.	N.D.	30
369	31	33	5	N.D.	N.D.	62
377	21	21	16	17	10	64
388	37	38	10	10	10	66
400	N.D.	4	N.D.	1	N.D.	3
426	N.D.	8	N.D.	N.D.	N.D.	6
444	N.D.	N.D.	N.D.	N.D.	N.D.	N.D.
458	N.D.	N.D.	N.D.	N.D.	N.D.	N.D.
472	N.D.	N.D.	N.D.	N.D.	N.D.	N.D.

Table D-8. Individual and Total Volatile Fatty Acids Present
in Leachate Samples from Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Volatile Fatty Acids					Total, expressed as mg/L Acetic Acid
	Acetic mg/L	Propionic mg/L	Isobutyric mg/L	Butyric mg/L	Valeric mg/L	
0	3,071	1,144	143	4,029	1,243	7,573
13	3,045	1,233	958	2,312	1,446	7,124
28	3,595	1,097	339	1,983	875	6,582
34	3,829	1,216	649	2,345	1,105	7,506
41	5,603	815	544	2,724	1,858	9,584
48	4,025	1,436	518	2,552	1,144	7,955
55	4,296	3,089	533	2,739	1,218	9,747
61	4,501	1,319	302	2,494	919	8,017
68	6,301	3,630	348	3,903	1,216	12,857
75	3,943	2,414	241	2,292	774	8,082
82	5,163	1,818	199	2,420	698	8,833
90	3,050	1,285	192	1,734	643	5,783
103	4,309	1,193	169	1,957	719	7,149
110	2,313	1,109	153	1,508	497	4,637
117	2,508	1,013	466	2,631	965	6,008
124	2,114	1,731	512	2,998	829	6,398
134	3,404	2,086	547	2,786	903	7,899
142	1,356	2,046	627	3,861	891	6,599
153	2,854	1,725	424	2,964	1,183	7,258
163	1,859	1,334	295	2,849	832	5,574
174	1,926	1,592	667	2,369	768	5,727
181	1,913	1,856	372	1,752	276	5,029
205	2,470	965	223	1,205	308	4,408
212	2,605	1,774	364	1,926	83	5,654

continued...

Table D-8. Continued...

Time Since Leachate Production Began, Days	Volatile Fatty Acids					Total, expressed as mg/L Acetic Acid
	Acetic mg/L	Propionic mg/L	Isobutyric mg/L	Butyric mg/L	Valeric mg/L	
219	4,144	1,324	314	1,839	2,419	8,109
231	4,426	1,381	329	1,805	771	7,457
254	4,066	1,330	320	1,809	906	7,127
266	3,569	1,201	225	1,700	865	6,364
290	3,961	1,301	325	2,000	953	7,161
350	181	306	23	32	106	530
360	347	670	34	19	104	988
369	225	356	19	31	78	594
377	355	482	33	22	90	831
388	350	550	30	19	96	886
400	9	1	N.D.*	N.D.	N.D.	10
426	1	N.D.	N.D.	N.D.	N.D.	1
444	N.D.	N.D.	N.D.	N.D.	N.D.	N.D.
462	N.D.	N.D.	N.D.	N.D.	N.D.	N.D.

*N.D. - None Detectable

Table D-9. Biochemical Oxygen Demand to Chemical Oxygen Demand (BOD₅/COD) Ratio of Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	BOD ₅ /COD	
	Cell A	Cell B
0	0.45	0.50
13	0.45	0.68
20	0.50	0.59
28	0.63	0.66
34	0.57	0.66
48	0.43	0.41
55	0.55	0.50
61	0.34	0.36
68	0.59	0.68
75	0.58	0.63
82	0.64	0.51
90	0.53	0.64
97	0.56	0.63
103	0.71	0.75
117	0.93	0.67
124	0.90	0.63
132	0.61	0.59
139	0.92	0.68
152	0.58	0.77
181	0.77	0.78
196	0.96	0.59
205	0.95	0.63
219	0.81	0.67
241	0.94	0.53
254	0.89	0.68
266	0.87	0.75
283	0.91	0.79
350	0.79	0.91
360	0.65	0.51
369	0.76	0.52
377	0.62	0.49
388	0.67	0.52
400	0.65	0.71
414	0.62	0.61
426	0.50	0.59

continued...

Table D-9. Continued...

Time Since Leachate Production Began, Days	BOD ₅ /COD	
	Cell A	Cell B
444	0.54	0.55
458	0.53	0.65
472	0.54	0.56
482	0.50	0.56
497	0.45	0.61
540	0.48	0.51
556	0.44	0.45
625	0.20	0.20
686	0.24	0.19
747	0.27	0.25
778	0.28	0.23
808	0.29	0.27
839	0.28	0.21
869	0.30	0.23
900	0.29	0.26
931	0.26	0.21
970	0.27	0.22

Table D-10. Chemical Oxygen Demand to Total Organic Carbon (COD/TOC)
Ratio of Leachate Samples from Cell A (Single Pass) and
Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	COD/TOC	
	Cell A	Cell B
0	2.85	2.63
13	3.65	2.89
20	3.98	2.91
28	3.45	3.40
34	3.79	3.40
48	4.11	4.86
55	4.80	4.53
61	4.82	4.49
68	3.26	2.38
75	3.03	2.97
82	2.65	3.02
90	2.20	2.39
97	2.44	2.09
103	2.34	2.57
110	2.98	2.92
117	1.43	2.42
124	1.31	2.46
132	1.09	2.47
152	1.21	2.02
181	1.69	2.76
196	1.47	3.27
205	1.80	2.66
212	1.30	3.05
219	2.04	2.34
231	2.37	3.14
241	1.61	2.77
254	1.95	2.82
260	1.99	3.08
266	1.99	2.89
279	1.86	2.94
283	1.94	2.87
350	1.48	3.90
360	1.14	2.29
369	1.09	2.22
377	1.13	2.30

continued...

Table D-10. Continued...

Time Since Leachate Production Began, Days	COD/TOC	
	Cell A	Cell B
388	1.13	2.11
400	1.95	1.72
426	1.60	1.75
444	1.65	1.73
458	1.56	1.99
472	1.44	2.12
482	1.52	2.22
497	1.05	1.84
540	1.82	2.38
556	1.19	1.96
570	1.21	2.02
594	1.40	2.30
625	1.70	2.80
655	1.54	2.30
686	1.54	2.50
716	1.30	2.70
747	1.57	2.50
778	1.75	2.45
808	1.68	2.25
839	1.76	2.49
869	1.65	2.53
900	1.82	2.64
931	1.86	2.39
970	1.76	2.46

Table D-11. Conductivity of Leachate from Cell A (Single Pass)
and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Conductivity	
	Cell A	Cell B
	μmhos	μmhos
13	10,500	9,200
20	7,700	6,200
28	10,400	9,000
34	10,200	8,200
41	6,400	5,400
48	6,800	6,400
55	8,950	8,400
61	10,600	9,400
68	10,900	10,000
75	10,200	10,000
82	9,250	8,800
90	10,000	9,800
97	10,700	10,600
103	9,800	9,700
110	9,300	9,600
117	9,200	9,900
124	8,900	10,700
132	9,400	10,300
139	10,600	11,500
152	10,900	12,600
163	9,800	12,500
168	11,200	11,500
181	10,200	12,000
196	9,800	11,500
205	9,900	11,600
219	9,800	11,500
241	9,500	11,200
254	9,900	11,200
260	10,200	11,500
266	9,800	10,700
275	9,600	10,900
283	9,900	10,600
350	11,500	12,000
360	11,200	12,100
369	10,800	11,100
377	10,400	9,600

continued...

Table D-11. Continued...

Time Since Leachate Production Began, Days	Conductivity	
	Cell A	Cell B
	μmhos	μmhos
388	10,100	9,400
400	8,500	7,900
414	8,100	7,300
426	7,200	6,100
429	6,100	5,400
444	6,200	6,200
458	5,400	4,900
472	4,320	4,110
482	4,450	3,210
497	3,650	3,150
516	3,450	2,050
540	2,950	2,910
556	3,255	2,450
570	2,855	2,350
594	2,655	2,710
625	2,455	2,650
655	2,855	2,720
686	2,950	2,790
716	2,855	2,400
747	2,750	2,010
778	2,610	1,850
808	2,420	1,965
839	2,250	2,150
869	2,550	2,215
900	2,605	1,985
931	2,250	1,995
970	2,150	1,855

Table D-12. pH and Oxidation-Reduction Potential (ORP) of Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	pH		ORP	
	Cell A	Cell B	Cell A	Cell B
			E_C , mV	E_C , mV
0	5.20	5.40	300	240
13	5.00	5.20	210	200
20	5.10	5.20	-20	-10
28	5.80	5.60	10	20
34	5.80	5.60	-30	-10
41	5.30	5.00	-50	-20
48	5.10	5.20	-80	-40
55	5.60	5.50	-150	-90
61	5.10	5.00	-155	-90
68	5.01	5.01	-40	-10
75	5.06	5.07	-270	-298
82	5.16	5.10	-139	-104
90	5.42	5.38	-178	-57
97	5.28	5.19	-178	-140
103	5.45	5.20	-220	-80
110	5.80	5.15	-215	-105
117	6.71	5.24	-238	-88
124	6.75	5.17	-200	-30
132	7.11	5.34	-209	-57
139	7.14	5.40	-218	-75
152	6.89	5.36	-246	-127
163	6.86	5.36	-210	-101
168	6.93	5.43	-209	-78
174	6.81	5.36	-215	-60
181	6.87	5.28	-245	-45
196	6.87	5.41	-245	-67
205	7.20	5.68	-239	-70
219	7.09	5.76	-250	-107
241	6.86	5.45	-210	-95
254	6.90	5.60	-225	-97
260	6.80	5.60	-220	-105
266	6.80	5.60	-225	-110
275	6.83	5.58	-220	-107
283	6.88	5.55	-210	-103

continued...

Table D-12. Continued...

Time Since Leachate Production Began, Days	pH		ORP	
	Cell A	Cell B	Cell A	Cell B
			E _C , mV	E _C , mV
350	7.03	6.98	-224	-231
360	7.15	7.03	-205	-235
369	7.10	6.95	-210	-231
377	7.09	6.90	-192	-206
388	7.10	6.98	-205	-232
400	7.10	7.01	-221	-225
414	7.05	7.03	-216	-240
426	7.12	7.10	-225	-236
444	7.10	7.06	-200	-224
458	7.09	7.14	-190	-215
472	7.11	7.13	-205	-225
482	7.06	7.08	-198	-240
497	7.01	7.12	-201	-231
516	7.10	7.20	-183	-215
540	7.16	7.18	-180	-210
556	7.14	7.15	-175	-210
570	7.12	7.21	-143	-230
594	7.11	7.26	-160	-210
625	7.16	7.23	-152	-203
655	7.18	7.21	-159	-199
686	7.12	7.28	-140	-210
716	7.21	7.34	-145	-205
747	7.15	7.21	-130	-188
778	7.10	7.15	-136	-198
808	7.16	7.21	-148	-189
839	7.09	7.21	-135	-195
869	7.17	7.25	-136	-188
900	7.16	7.21	-145	-182
931	7.11	7.23	-142	-197
970	7.10	7.27	-139	-191

Table D-13. Ratio of the Organic Carbon Present in Total Volatile Acid to the Total Organic Carbon (TOC) Content of Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Organic Carbon Present in Total Volatile Acid/TOC	
	Cell A	Cell B
0	0.76	0.67
13	0.36	0.43
28	0.54	0.45
34	0.53	0.57
48	0.58	0.45
55	0.54	0.46
61	0.61	0.42
68	0.56	0.56
75	0.47	0.44
82	0.55	0.43
90	0.32	0.26
103	0.27	0.31
110	0.27	0.22
117	0.23	0.26
124	0.22	0.25
132	0.15	0.32
152	0.19	0.29
181	0.11	0.23
205	0.13	0.28
212	0.07	0.29
219	0.09	0.41
231	0.09	0.44
254	0.09	0.37
266	0.07	0.40
283	0.06	0.47
350	0.02	0.11
360	0.01	0.25
369	0.02	0.13
377	0.02	0.21
388	0.02	0.20
400	-	0.01
426	-	-

Table D-14. Iron, Manganese, Zinc, and Nickel Concentrations in Leachate
Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Cell A				Cell B			
	Iron	Manganese	Zinc	Nickel	Iron	Manganese	Zinc	Nickel
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
13	140	45	39	3.5	83	35	26	4.0
34	143	44	20	3.5	20	35	25	3.2
41	208	42	16	3.7	270	39	23	3.5
55	395	51	9	-	395	50	21	3.7
68	300	51	8.4	3.5	466	46	19	2.9
82	150	51	4.5	2.9	480	49	19	2.6
96	110	7	1.0	1.8	450	51	8.5	2.9
110	50	9	1.0	2.4	550	49	10.0	4.0
124	32	8	1.5	2.0	475	56	12.0	3.5
139	50	12	1.5	2.6	575	58	10.0	4.0
152	120	29	1.1	4.0	625	46	8.0	3.5
168	120	10	1.3	3.2	450	74	9.0	6.6
181	70	37	0.6	6.2	285	79	11.0	7.3
205	120	27	0.9	4.0	450	59	7.5	4.6
219	70	16	0.4	3.7	285	57	6.0	3.2
241	59	17	0.5	3.4	260	54	7.0	3.1
260	55	15	0.4	3.3	275	50	6.0	3.5
275	60	17	0.5	3.4	255	52	7.0	3.2
350	13	-	-	-	38	17	0.6	2.7
360	18	-	-	2.2	38	19	0.5	2.7
369	16	22	0.15	2.2	47	18	0.6	3.2
377	12	21	0.25	3.3	32	17	0.4	3.3
388	10	16	0.15	2.7	21	18	0.5	2.2
400	8	12	0.2	2.1	25	16	0.3	2.1

continued...

Table D-14. Continued...

Time Since Leachate Production Began, Days	Cell A				Cell B			
	Iron	Manganese	Zinc	Nickel	Iron	Manganese	Zinc	Nickel
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
426	6	10	0.1	2.3	15	14	0.4	2.1
444	5	8	0.2	1.5	12	8	0.2	0.6
472	6.5	5.4	0.1	0.5	11	6	0.1	0.5
516	7.6	6.3	N.D.*	N.D.	12.6	5.3	0.2	0.4
540	6.5	5.1	N.D.	N.D.	13	6.2	0.1	0.2
570	7.9	6.7	N.D.	N.D.	11.4	4.3	0.1	N.D.
594	10.1	5.2	N.D.	N.D.	10.7	3.6	0.2	N.D.
625	12	4.2	N.D.	N.D.	15.2	2.6	0.2	N.D.
655	10	3.2	N.D.	N.D.	18.6	2.8	N.D.	N.D.
686	8	4.6	N.D.	N.D.	17.6	3.1	N.D.	N.D.
716	7.5	5.1	N.D.	N.D.	12	3.7	N.D.	N.D.
747	7.9	4.8	N.D.	N.D.	8.5	4.1	N.D.	N.D.
778	8.5	4.9	N.D.	N.D.	8.2	4.0	N.D.	N.D.
808	6.7	4.5	N.D.	N.D.	8.9	4.1	N.D.	N.D.
839	7.2	4.1	N.D.	N.D.	9.1	3.2	N.D.	N.D.
869	8.4	3.6	N.D.	N.D.	10.2	2.9	N.D.	N.D.
900	6.1	2.7	N.D.	N.D.	8.1	2.1	N.D.	N.D.
931	7.2	3.6	N.D.	N.D.	7.5	3.1	N.D.	N.D.
970	8.1	2.1	N.D.	N.D.	8.6	2.0	N.D.	N.D.

*N.D. - None Detectable

Table D-15. Chloride Concentrations in Leachate Samples from
Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Cell A		Cell B	
	Chloride	(C_t/C_o)*	Chloride	(C_t/C_o)**
	mg/L		mg/L	
13	667	1.00	470	1.00
34	581	.87	445	0.94
41	544	0.81	421	0.89
55	519	0.77	372	0.79
68	507	0.75	372	0.79
82	534	0.80	604	1.28
96	719	1.07	654	1.39
110	704	1.05	713	1.52
124	654	0.98	654	1.39
139	587	0.88	528	1.12
152	555	0.83	565	1.20
168	515	0.77	540	1.15
181	489	0.73	526	1.12
205	469	0.70	549	1.16
219	459	0.69	519	1.10
231	458	0.69	528	1.12
254	425	0.64	501	1.06
275	401	0.60	535	1.13
283	415	0.62	516	1.10
350	360	0.54	491	1.05
369	315	0.47	453	0.96
388	320	0.48	486	1.03
414	301	0.45	501	1.06
429	285	0.43	466	0.99
444	266	0.40	428	0.91
472	226	0.34	401	0.85
497	254	0.38	408	0.87
540	221	0.33	418	0.89
570	245	0.37	429	0.91
594	201	0.30	406	0.86
625	211	0.32	385	0.82
686	206	0.31	366	0.78
716	216	0.32	345	0.74
747	180	0.27	341	0.73
778	196	0.29	328	0.70

continued...

Table D-15. Continued...

Time Since Leachate Production Began, Days	Cell A		Cell B	
	Chloride	$(C_t/C_o)^*$	Chloride	$(C_t/C_o)^{**}$
	mg/L		mg/L	
808	175	0.26	331	0.70
839	178	0.27	321	0.68
869	179	0.27	315	0.67
900	180	0.27	309	0.66
931	172	0.26	301	0.64
970	165	0.25	300	0.63

* C_o = 667 mg/L

** C_o = 470 mg/L

Table D-16. Radioactivity in the Different GPC Fractions of Equilibrated Samples

Sample	Counts Per Minute in Different GPC Fractions									
	1 (0mL-5mL)	2 (5mL-10mL)	3 (10mL-15mL)	4 (15mL-20mL)	5 (20mL-25mL)	6 (25mL-30mL)	7 (30mL-35mL)	8 (35mL-40mL)	9 (40mL-45mL)	10 (45mL-50mL)
Blank (pH=6.80)	N.D.*	N.D.	N.D.	N.D.	N.D.	N.D.	2,991	2,991	679	113
A-2HA (pH=7.00)	N.D.	23	540	85	444	9,455	10,819	3,492	375	N.D.
A-2HA (pH=5.50)	N.D.	64	190	N.D.	406	5,088	10,198	4,686	1,092	173
A-2HA (pH=8.50)	N.D.	462	125	142	6,408	9,348	2,105	203	N.D.	N.D.
Leachate (pH=7.27)	N.D.	N.D.	N.D.	N.D.	1,632	7,434	4,910	360	N.D.	N.D.

*N.D. - None Detectable.

Table D-17. Percent Relative Radioactivity in Different
GPC Fractions of Equilibrated Samples

GPC Fractions ↓	Sample →	Blank*	A-2HA pH=7.0	A-2HA pH=5.5	A-2HA pH=8.5	Leachate** pH=7.27
1		-	-	-	-	-
2		-	0.1	0.3	2.5	-
3		-	2.1	0.7	0.7	-
4		-	0.3	-	0.8	-
5		-	1.8	1.9	34.1	11.3
6		-	37.5	23.2	49.7	59.9
7		36.5	42.9	46.6	11.2	34.3
8		50.2	13.8	21.4	1.0	2.5
9		11.4	1.5	4.9	-	-
10		1.9	-	-	-	-

*Deionized Water

**Leachate from Cell B; Day 970

Table D-18. Cobalt Concentrations in Leachate from Cell A
(Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Cobalt	
	Cell A	Cell B
	mg/L	mg/L
13	0.90	0.78
20	0.42	0.71
34	0.56	0.65
50	0.51	0.55
61	0.52	0.56
82	0.47	0.91
96	0.34	0.99
110	0.27	1.01
124	0.29	0.97
139	0.41	0.82
152	0.54	0.88
168	0.38	0.85
181	0.61	0.82
205	0.42	0.85
241	0.42	0.72
260	0.68	0.68
275	0.52	0.73
350	0.48	0.63
360	0.23	0.55
369	0.15	0.46
377	0.15	0.48
388	N.D.*	0.45
400	N.D.	0.43
426	N.D.	0.26
444	N.D.	0.07
472	N.D.	0.12
516	0.05	0.15
540	N.D.	0.13
570	N.D.	0.11
594	N.D.	0.15
625	N.D.	0.15
655	N.D.	0.23
686	N.D.	0.27
716	N.D.	0.29
747	N.D.	0.40

continued...

Table D-18. Continued...

Time Since Leachate Production Began, Days	Cobalt	
	Cell A	Cell B
	mg/L	mg/L
778	N.D.	0.26
808	N.D.	0.20
839	N.D.	0.30
869	N.D.	0.22
900	N.D.	0.21
931	N.D.	0.21
970	N.D.	0.20

*None Detectable

Table D-19. Chloride Based Correction Factor (C.F.) for Dilution

Time Since Leachate Production Began, Days	Correction Factor, $[\text{Cl}^-]_t/[\text{Cl}^-]_{\text{max}}$	
	Cell A	Cell B
	mg/L	mg/L
13	0.93	0.65
34	0.81	0.62
41	0.76	0.58
55	0.72	0.52
68	0.70	0.52
82	0.75	0.84
96	1.00	0.91
110	0.98	1.00
124	0.91	0.91
139	0.82	0.74
152	0.77	0.79
168	0.68	0.76
181	0.68	0.74
205	0.65	0.76
219	0.64	0.72
231	0.64	0.74
254	0.59	0.69
275	0.56	0.74
283	0.58	0.72
350	0.44	0.69
369	0.44	0.63
388	0.45	0.68
414	0.42	0.68
429	0.40	0.65
444	0.37	0.60
472	0.32	0.57
497	0.35	0.57
540	0.31	0.58
570	0.35	0.60
594	0.28	0.56
625	0.30	0.54
686	0.29	0.51
716	0.30	0.49
747	0.25	0.48
778	0.27	0.46
808	0.24	0.46

continued...

Table D-19. Continued...

Time Since Leachate Production Began, Days	Correction Factor, $[Cl^-]_t/[Cl^-]_{max}$	
	Cell A	Cell B
	mg/L	mg/L
839	0.25	0.44
869	0.25	0.44
900	0.25	0.43
931	0.24	0.42
970	0.23	0.41

Table D-20. "As-Measured" COD and "Dilution-Effect-Corrected" COD of Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Cell A		Cell B	
	COD	Corrected	COD	Corrected
	"As-Measured" mg/L	COD mg/L	"As-Measured" mg/L	COD mg/L
13	26,956	28,984	19,468	29,950
34	26,880	33,185	17,920	28,903
48	35,276	46,415	34,138	58,858
55	40,256	55,911	38,327	73,705
68	26,800	38,285	21,760	73,625
82	19,000	25,333	25,087	30,158
96	21,240	21,240	18,540	23,340
110	18,685	19,066	24,809	19,066
124	6,630	7,285	24,751	27,198
139	2,541	3,099	25,856	34,940
152	3,680	4,779	20,210	25,582
163	4,530	6,661	18,877	24,838
181	3,811	5,604	23,762	32,110
205	3,661	5,632	16,744	22,031
219	3,716	5,806	18,583	25,809
231	2,888	4,512	21,066	28,467
254	2,564	4,345	21,544	31,223
283	3,324	5,731	17,540	24,361
350	2,656	5,312	7,614	11,034
369	1,708	3,881	3,978	6,314
388	1,640	3,644	3,785	5,566
414	1,482	3,528	1,689	2,483
426	1,448	3,620	1,655	2,546
444	1,812	4,897	2,202	3,670
472	1,680	5,250	2,634	4,621
497	1,538	4,394	2,402	4,214
540	1,290	4,161	2,450	4,224
570	1,276	3,646	2,340	3,900
594	1,137	4,060	1,962	3,503
625	1,261	4,203	1,872	3,466
686	1,100	3,793	1,831	3,590
716	1,061	3,536	1,771	3,614
747	1,256	5,024	1,662	3,462
778	1,324	4,903	1,692	3,678

continued...

Table D-20. Continued...

Time Since Leachate Production Began, Days	Cell A		Cell B	
	COD "As-Measured" mg/L	Corrected COD mg/L	COD "As-Measured" mg/L	Corrected COD mg/L
808	1,231	5,129	1,620	3,521
839	1,251	5,004	1,738	3,950
869	1,267	5,068	1,644	3,736
900	1,452	5,808	1,623	3,774
931	1,441	6,004	1,451	3,454
970	1,342	5,834	1,522	3,712

Table D-21. Ionic Strength (I) of Leachate from Cell A
(Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Ionic Strength (I)	
	Cell A	Cell B
	M	M
13	0.168	0.147
20	0.123	0.099
28	0.166	0.144
34	0.163	0.131
41	0.102	0.086
48	0.108	0.102
55	0.143	0.134
61	0.169	0.150
68	0.174	0.160
75	0.163	0.160
82	0.148	0.140
90	0.160	0.156
97	0.171	0.169
103	0.156	0.155
110	0.148	0.153
117	0.147	0.158
124	0.142	0.171
132	0.150	0.164
139	0.169	0.184
152	0.174	0.201
163	0.156	0.200
168	0.179	0.184
181	0.163	0.192
196	0.156	0.184
205	0.158	0.185
219	0.156	0.184
241	0.152	0.179
254	0.158	0.179
260	0.163	0.184
266	0.156	0.171
275	0.153	0.174
283	0.158	0.169
350	0.184	0.192
360	0.179	0.193
369	0.172	0.177

continued...

Table D-21. Continued...

Time Since Leachate Production Began, Days	Ionic Strength (I)	
	Cell A	Cell B
	M	M
377	0.166	0.153
388	0.161	0.150
400	0.136	0.126
414	0.129	0.116
426	0.115	0.097
429	0.097	0.086
444	0.099	0.099
458	0.086	0.078
472	0.069	0.065
482	0.071	0.051
497	0.058	0.050
516	0.055	0.032
540	0.047	0.046
556	0.052	0.039
570	0.045	0.037
594	0.042	0.043
625	0.039	0.042
655	0.045	0.043
686	0.047	0.044
716	0.045	0.038
747	0.044	0.032
778	0.041	0.029
808	0.038	0.031
839	0.036	0.034
869	0.040	0.035
900	0.041	0.031
931	0.036	0.031
970	0.034	0.029

Table D-22. Activity Coefficient (γ_i) for Monovalent, Divalent, and Trivalent Ions in Leachate Samples from Cell A (Single Pass) and Cell B (Recycle)

Time Since Leachate Production Began, Days	Activity Coefficient, (γ_i)					
	Cell A			Cell B		
	Z=1	Z=2	Z=3	Z=1	Z=2	Z=3
13	0.75	0.33	0.08	0.76	0.34	0.09
20	0.77	0.35	0.09	0.78	0.38	0.11
28	0.75	0.33	0.08	0.76	0.34	0.09
34	0.75	0.33	0.08	0.77	0.35	0.09
41	0.78	0.37	0.11	0.79	0.39	0.12
48	0.78	0.37	0.11	0.78	0.37	0.11
55	0.76	0.34	0.09	0.77	0.35	0.09
61	0.75	0.33	0.08	0.76	0.34	0.08
68	0.75	0.32	0.08	0.76	0.33	0.08
75	0.75	0.33	0.08	0.76	0.33	0.08
82	0.76	0.34	0.09	0.76	0.34	0.09
90	0.76	0.33	0.08	0.76	0.33	0.08
97	0.75	0.32	0.08	0.75	0.32	0.08
103	0.76	0.34	0.09	0.76	0.34	0.09
110	0.76	0.34	0.09	0.76	0.33	0.08
117	0.76	0.34	0.09	0.76	0.34	0.09
124	0.76	0.34	0.09	0.75	0.32	0.08
132	0.76	0.34	0.08	0.75	0.33	0.08
139	0.75	0.33	0.08	0.75	0.32	0.07
152	0.75	0.32	0.08	0.75	0.31	0.07
163	0.76	0.34	0.09	0.75	0.31	0.07
168	0.75	0.32	0.08	0.75	0.32	0.07
181	0.75	0.33	0.08	0.75	0.32	0.07
196	0.76	0.34	0.09	0.75	0.32	0.07
205	0.76	0.34	0.09	0.75	0.32	0.07
219	0.76	0.34	0.09	0.75	0.32	0.07
241	0.76	0.33	0.08	0.75	0.32	0.08
254	0.76	0.34	0.09	0.75	0.32	0.08
260	0.75	0.33	0.08	0.75	0.32	0.07
266	0.76	0.34	0.09	0.75	0.32	0.08
275	0.76	0.33	0.08	0.75	0.32	0.08
283	0.76	0.34	0.09	0.75	0.35	0.08
350	0.75	0.32	0.07	0.75	0.32	0.07
360	0.75	0.32	0.08	0.75	0.32	0.07

continued...

Table D-22. Continued...

Time Since Leachate Production Began, Days	Activity Coefficient, (γ_i)					
	Cell A			Cell B		
	Z=1	Z=2	Z=3	Z=1	Z=2	Z=3
369	0.75	0.32	0.08	0.75	0.32	0.08
377	0.75	0.33	0.08	0.76	0.33	0.08
388	0.76	0.33	0.08	0.76	0.34	0.08
400	0.76	0.35	0.09	0.77	0.36	0.10
414	0.77	0.35	0.10	0.78	0.36	0.10
426	0.78	0.36	0.10	0.78	0.38	0.11
429	0.78	0.38	0.11	0.79	0.39	0.12
444	0.78	0.38	0.11	0.78	0.38	0.11
458	0.79	0.39	0.12	0.80	0.41	0.13
472	0.81	0.42	0.14	0.81	0.43	0.15
482	0.80	0.42	0.14	0.82	0.46	0.17
497	0.82	0.44	0.16	0.82	0.46	0.18
516	0.82	0.45	0.17	0.85	0.52	0.23
540	0.83	0.47	0.18	0.83	0.47	0.18
556	0.82	0.46	0.17	0.84	0.49	0.21
570	0.83	0.48	0.19	0.84	0.50	0.21
594	0.83	0.48	0.20	0.83	0.48	0.20
625	0.84	0.49	0.21	0.83	0.48	0.20
655	0.83	0.48	0.19	0.83	0.48	0.20
686	0.83	0.47	0.18	0.83	0.48	0.19
716	0.83	0.48	0.19	0.84	0.49	0.21
747	0.83	0.48	0.19	0.85	0.52	0.23
778	0.83	0.49	0.20	0.85	0.53	0.24
808	0.84	0.50	0.21	0.85	0.52	0.23
839	0.84	0.51	0.21	0.84	0.51	0.22
869	0.84	0.49	0.20	0.84	0.51	0.22
900	0.84	0.49	0.20	0.85	0.52	0.23
931	0.84	0.51	0.21	0.85	0.52	0.23
970	0.84	0.51	0.22	0.85	0.53	0.24

Table D-23. Total Sulfide Concentrations in Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Total Sulfide (S_T)	
	Cell A	Cell B
	mg/L	mg/L
27	N.D.*	N.D.
43	0.23	N.D.
64	0.31	N.D.
110	0.15	N.D.
134	0.23	N.D.
152	0.30	N.D.
168	0.21	N.D.
183	0.34	N.D.
198	0.36	N.D.
205	0.23	N.D.
219	0.32	N.D.
241	0.38	N.D.
260	0.29	N.D.
283	0.85	N.D.
350	0.81	0.56
369	0.15	0.45
377	0.45	0.65
388	0.65	0.95
400	0.61	1.50
426	0.43	3.25
444	0.23	4.50
472	0.15	2.25
516	0.28	1.95
540	0.31	1.65
570	0.26	1.25
594	0.21	1.15
625	0.17	0.85
655	0.19	0.95
686	0.22	0.65
716	0.19	0.55
747	0.11	0.25
778	0.10	0.15
808	N.D.	0.11
839	N.D.	0.13

continued...

Table D-23. Continued...

Time Since Leachate Production Began, Days	Total Sulfide (S_T)	
	Cell A	Cell B
	mg/L	mg/L
869	N.D.	N.D.
900	N.D.	N.D.
931	N.D.	N.D.
970	N.D.	N.D.

Table D-24. Total Alkalinity of Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Total Alkalinity*	
	Cell A	Cell B
	mg/L as CaCO ₃	mg/L as CaCO ₃
13	4,717	3,547
20	5,559	4,380
28	8,500	10,000
34	16,900	12,700
55	8,466	7,042
61	9,169	7,127
68	9,504	9,750
75	6,598	6,384
82	7,485	6,889
97	4,229	5,910
103	4,500	7,800
110	4,350	4,850
117	4,200	6,500
124	4,100	6,350
132	4,500	6,800
139	4,300	7,150
152	4,750	7,500
163	4,440	6,500
168	4,800	6,100
181	5,000	6,750
196	4,600	6,250
205	4,800	6,600
219	4,700	6,670
241	4,300	6,120
260	4,100	5,255
266	4,200	5,735
283	4,100	5,825
350	3,450	3,650
360	4,050	4,595
377	3,583	3,666
400	3,683	3,253
414	3,215	3,050
426	2,950	3,151
429	2,655	3,156
444	2,450	2,985

continued...

Table D-24. Continued...

Time Since Leachate Production Began, Days	Total Alkalinity*	
	Cell A	Cell B
	mg/L as CaCO ₃	mg/L as CaCO ₃
458	2,150	2,465
472	2,250	2,215
482	2,200	2,015
497	2,106	2,050
516	2,195	1,985
540	2,000	1,865
556	2,050	1,915
570	2,065	1,975
594	1,965	1,915
625	1,985	1,995
655	2,125	1,925
686	2,225	1,985
716	2,067	1,915
747	2,095	1,929
778	2,165	1,967
808	2,087	1,927
839	2,015	1,815
869	2,070	1,867
900	2,055	1,895
931	2,035	1,950
970	1,990	1,825

*Titrated to pH=3.5

Table D-25. Strontium Concentrations in Leachate from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Strontium	
	Cell A	Cell B
	mg/L	mg/L
13	4.5	5.0
34	6.5	9.8
50	5.8	12.3
61	5.4	11.8
82	5.2	16.2
96	5.9	13.9
110	5.9	14.9
114	6.0	15.6
139	5.4	17.3
152	5.1	18.2
168	5.7	18.6
181	5.1	18.1
205	4.2	17.9
241	4.3	18.6
260	4.2	17.3
275	4.3	17.2
350	3.8	7.9
377	3.7	3.9
400	3.5	3.7
472	3.1	2.9
540	3.2	2.5
570	2.8	2.1
594	2.5	2.5
625	2.3	2.6
655	2.1	2.1
686	2.3	1.9
716	2.2	1.8
747	2.3	1.9
778	2.1	2.0
808	2.1	1.8
839	2.1	1.7
869	2.0	1.7
900	1.9	1.7
931	1.8	1.6
970	1.7	1.5

Table D-26. Total Inorganic Carbon (TIC) Present in Leachate Samples from Cell A (Single Pass) and Cell B (Leachate Recycle)

Time Since Leachate Production Began, Days	Total Inorganic Carbon	
	Cell A	Cell B
	mg/L	mg/L
117*	200	N.D.
124	100	N.D.
132	300	N.D.
152	250	N.D.
181	300	N.D.
196	450	N.D.
205	400	N.D.
212	526	N.D.
219	415	N.D.
231	470	N.D.
241	575	N.D.
254	625	N.D.
260	650	100
266	620	126
273	585	250
283	500	115
350	615	450
360	525	500
369	490	575
377	515	515
388	525	565
400	495	522
414	450	587
426	425	606
429	415	585
444	410	479
458	395	450
472	350	415
482	385	395
497	295	401
516	265	358
540	301	373
556	315	395
570	310	315
594	327	287

continued.....

Table D-26. Continued...

Time Since Leachate Production Began, Days	Total Inorganic Carbon	
	Cell A	Cell B
	mg/L	mg/L
625	310	315
655	327	287
686	287	215
716	265	295
747	292	263
778	311	257
808	301	287
839	265	299
869	295	216
900	327	192
931	343	187
970	301	199

*TIC was below detectable limits prior to Day 117.

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