Environmental Protection Agency

DCN 200-187-41-08

STUDY OF NON-HAZARDOUS WASTES
FROM COAL-FIRED ELECTRIC UTILITIES

DRAFT FINAL REPORT
15 DECEMBER 1978

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1.0 EXECUTIVE SUMMARY

The production of electrical energy in the U.S. in the near future will place greater emphasis on the utilization of coal. It has been reported that between 1976 and 1986 the coal-fired capacity in the U.S. will increase by 60% or 160,000 Mw. Emissions from these plants are controlled by the Clean Air Amendment of 1977 and result in the production of large masses of fly ash and scrubber sludge. In lieu of commercial utilization, disposal of these solid wastes must be conducted in a manner which is consistent with the requirements of the Resource Conservation and Recovery Act of 1976 (RCRA).

Radian Corporation was contracted by the Environmental Protection Agency (EPA) to provide the EPA Office of Solid Waste with a preliminary basis for responsible agencies to evaluate the definitions and possible regulations required for assessing the impact of implementation of RCRA Section 4004 as it may apply to the utility industry.

The objective of this study was accomplished through the completion of four tasks. The first task consisted of a characterization of the utility industry, including the number of plants, distribution by size and EPA region, and the general economic status of the industry. Results of this task can be used to assess the impact of implementation of RCRA Section 4004 on federal and state regulatory agencies and on the utility. States and utilities in the Ohio Valley will be hit the hardest by implementation.

The second task determined quantities of flue gas cleaning (FGC) wastes - fly ash, bottom ash and scrubber sludge - as

produced by a number of possible scenarios for a "model" 1000 Mw
plant. The chemical, physical, structural and leaching pro-
perties of the waste were discussed. The characteristics and
mass of the FGC waste affect disposal methodology, disposal costs
and potential environmental effect. Approximately 62 million dry
metric tons of ash and 2.7 million dry metric tons of sludge
were produced in 1977. Fly ashes from western coals can
stabilize scrubber sludges better than ashes from eastern coals
because of the higher calcium content and alkalinity of western
coals. Stabilization consists of lower permeability and higher
structural strength.

Disposal practices for FGC wastes were addressed in
Task 3. A distinction was made between ponding versus land-
filling. An average distance of 3 miles from the plant was
was determined. The economic impact of RCRA Section 4004 was
assessed to the degree possible consistent with the scope of work
and available information. Information gained in this task
provides part of the basis for determination of the impact of
regulations pertaining to solid waste disposal. Preliminary
assessment of the data indicate that RCRA could have a signifi-
cant impact on the current and future disposal practices of the
utilities and could have economic impact on the utility industry
as a whole. Alternate disposal methods such as mine disposal,
ocean disposal and landfarming were also examined. Mine disposal
either in surface or deep mines is the most technically possible
and environmentally acceptable alternate to ponding and landfilling.

The final task area involved evaluation of the methods
of the utilization of fly ash, bottom ash and scrubber sludge.
Both current and future recovery methods were identified and
evaluated with respect to technical, geographical, environmental
and economic considerations. Although regenerable processes

are in full-scale operation at several U.S. plants in 1978, it is not foreseen that these regenerable processes will significantly reduce the industry-wide quantity of FGD scrubber sludge in the near term. In specific instances where the sulfur or sulfuric acid by-products can be marketed, regenerable processes can reduce the FGC waste quantities and allow collection and utilization of dry fly ash without the need for using fly ash to fix scrubber sludge. Utilization of FGC waste products is important in that it reduces the quantity requiring disposal, conserves natural resources and can generate revenues to offset collection and disposal costs. Ash utilization was 21 percent in 1977 while sludge utilization was nil. Research and incentives to increase utilization of the FGC by-products are needed.

A summary of the findings of this study follows:

1.1   Characterization of Electric Utility Industry

The characterization of the utility industry consisted of the number of coal-fired plants organized according to inventories within the ten EPA regions and also within each of the states. The total coal-fired capacity was also delineated in the same manner. The financial status of the utility industry was also assessed. These characterizations will provide information to each of the states and each of the EPA regions in assesing their level of effort in implementing RCRA Section 4004. Results of this task also summarize the financial status of the utility industry and provide a partial basis for interpreting the economic impact of Section 4004 on the utility.

Utility owned, coal-fired generating plants in the United States were inventoried for the year 1976. There were 171 utilities owning approximately 399 coal-fired power plants. These 399 plants had a total coal-fired capacity of 202,380 megawatts (Mw).

The distribution of this coal-fired generating capacity by state in 1976 is shown in Figure 1. The number of utility-owned, coal-fired plants within each state is indicated in parenthesis. The EPA regions are indicated by heavier border lines. Most are in EPA regions III (60 plants), IV (72), V (151) and VII (49). These same four regions also accounted for 85 percent of the electrical energy produced from coal. Region IV (50,000 Mw) and Region V (69,000 Mw) accounted for over 50 percent of the total U.S. coal-fired capacity. Significant growth is anticipated in the southern and western EPA regions.

In 1976, approximately 25 percent of the plants had less than 100 Mw capacity. Approximately 40 percent of 157 plants had capacities between 100 and 500 Mw. There were 72 plants with coal-fired capacities of 500 to 1000 Mw and 57 plants in the 1000-2000 Mw range. Only 12 plants had capacities greater than 2000 Mw. The average coal-fired capacity of all plants was approximately 500 Mw.

Approximately 78 percent of the installed generating capacity in the U.S. is operated by investor-owned utility companies, serving about 78 percent of the nation's utility customers. Public power systems (10 percent of capacity, 12 percent of customers), rural electric cooperatives (2 percent, 10 percent) and TVA (10 percent, 0 percent) own and serve the remainder. Each electric utility company, public power system, or cooperative operates as regulated monopoly in a designated service area. The industry is highly capital investment-intensive. It has experienced diffuculty in raising capital during recent years due to decreasing bond ratings and increasing interest charges. During these years, investor-owned utility companies have experienced difficulties in raising capital through stock sales; price-book ratios less than 1.0 have been consistently
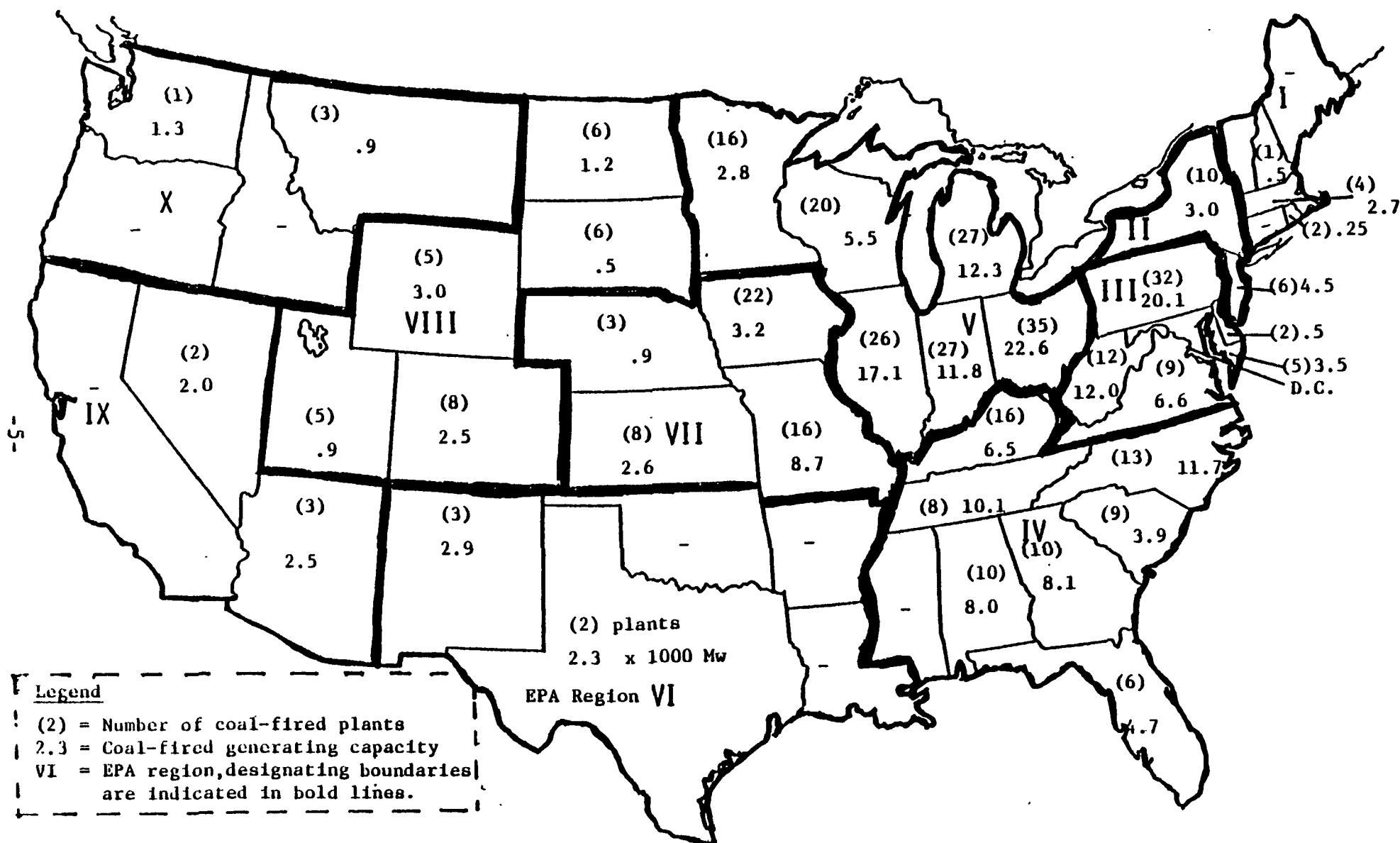
Figure 1-1. Distribution of coal-fired generating stations and coal-fired capacity by states (1976)

reported. While the industry's financial condition has improved somewhat since 1974 (the worst year), it has not sufficiently recovered to be considered healthy. Financial analysts believe that (1) inflation levels of 4 percent per year could cause improvements, (2) inflation levels of 5 to 6 percent per year are only tolerable, and (3) inflation rates in excess of 10 percent per year could restrict investment to the point that additional rate increases would be required in order to finance capital expenditures.

Utility regulation affects both plant construction and rates charged for services. Typical licensing times for coal-fired power plants range from 33 to 49 months. The licensing procedure may require up to 50 permits from a number of different regulatory agencies. Customer rate bases approved by rate commissions between 1974 and 1977 have reflected returns on common equity ranging from 10.6 to 11.8 percent. In early 1978 a return on common equity of about 12 percent was considered reasonable by financial analysts. Current capital costs for coal-fired plants reflected in these rate bases range from $450 to $800 per installed kilowatt capacity. Credit for costs of construction work in progress (CWIP) may or may not be reflected in the rate base, depending on the discretion of the governing rate commission. Generally, credit for CWIP, if granted, is based upon the financial state of the company, and it allows financing through rate increases rather than through the money market. Future revenue requirements, which will eventually be reflected in rate bases, have been estimated to range from about 3¢/kwh in 1978 to 3.5-4¢ per kwh sold in 2000.

In summary, over 50 percent of the U.S.'s coal-fired electricity is produced by the states in EPA regions IV and V during 1976. Regions III and VII also have significant coal-fired

capacity (35%). Region VI and other parts of the west are forecast to have significant growth during the next few years. Consequently, the states' agencies and EPA region staff of these regions will bear the bulk of the implementation of Section 4004. The bond ratings of the utility have been low since 1973 and price-book ratios of less than 1.0 have been consistently reported. Consequently the utility industry is not financially healthy.

1.2   Characteristics of Solid Wastes

The quality of coal fired in the boiler is the single most important variable affecting both the quality and quantity of ash and scrubber sludge produced. The quantities and qualities of ash and sludges produced by burning western and eastern coals in a "model" 1000 Mw power plant are shown in Table 1. The coal is the source of the minerals, both major and trace elements which comprise the fly ash and bottom ash. The quantity of scrubber sludge produced is largely a function of the sulfur content of the coal. The coal is also a major source of the chlorides and volatile trace elements found in the scrubber liquor. The makeup water and sorbent (lime or limestone) are other sources of dissolved species in the scrubber liquor.

The environmental impact and consequently the disposal methodology for reducing or minimizing this environmental impact is significantly affected by the quantity and characteristics of the FGC waste by products: fly ash, bottom ash and scrubber sludge. The factors influencing quantity which have been characterized by a number of scenarios include coal quality, scrubber type and operation, and emission regulations. The

TABLE 1-1   TYPICAL QUANTITIES AND QUALITIES OF
             ASH AND SCRUBBER SLUDGE FOR MODEL 1000 MW
             POWER PLANT

	Western Coal	Eastern Coal
Quantities (Dry Metric Tons/yr)		
Current Regulations*		
Fly Ash & Bottom Ash	270,000	408,000
Scrubber Sludge	17,000	336,000
Future Regulations**		
Scrubber Sludge	102,000	
Qualities		
Chemical		
Fly Ash	High calcium, high alkalinity	Low calcium, low alkalinity, even acidic in some cases.
	Cement-like properties can stabilize scrubber sludge	Cement-like properties with addition of lime.
Scrubber sludge	Higher sulfate content	Higher sulfite content
Structural		
Fly Ash	Stable	Stable with adequate moisture control and compaction.
Fly Ash/Scrubber Sludge in Mixture	Stable	Stable with adequate moisture control and compaction
Naturally Oxidized Scrubber Sludge	Unstable	Unstable
Leaching		
Fly Ash	High pH	pH alkaline to acidic generally higher solubilities

*Current regulations (1978) Maximum SO_2 emissions of 1.2 lbs $SO_2/10^6$ Btu.
 Particulate emissions reduced by 99%.

**Future regulations proposed 9/19/78 -SO_2 concentration in flue gas reduced
  by 85% or to .2 lbs $SO_2/10^6$ Btu.

chemical, physical, structural and leaching properties of each of the three wastes have been discussed based on available information.

Chemical and physical properties of the material dictate to a large extent the structural and leaching properties of the material. Structural properties describe the structural stability of the disposal material and future uses of the reclaimed land. Leaching properties are a major consideration in evaluating the potential for groundwater or surface water contamination. Results of this task are summarized in the following paragraphs.

The firing of western coal (9,000 Btu/lb, 8% ash, and 0.8% sulfur) in a 1000 Mw generating plant can result in 218,000 dry metric tons of fly ash per year (99% removal) and 54,000 dry metric tons of bottom ash per year. Approximately 17,000 dry metric tons or 50,000 wet metric tons (35% solids) of scrubber sludge would be produced with current SO_2 regulations of a maximum emission of 1.2 lbs $SO_2/10^6$ Btu and representative scrubber operations. Stricter particulate regulations will not significantly impact the quantities of fly ash produced either from eastern or western coal. An expected SO_2 regulation of 85 percent removal will increase the scrubber sludge quantity of this 1000 Mw plant from the 17,000 to 102,000 dry metric tons to 113,000 dry tons per year. The firing of eastern coal (11,000 Btu/lb, 15% ash, and 3% sulfur) in the same 1000 Mw plant results in 408,000 dry metric tons/year of ash and 336,000 dry metric tons/year of scrubber sludge under current regulations.

The effect of scrubber operation on the quantity of by-products was evaluated. The use of forced oxidation in a limestone scrubbing process will increase the quantity of dry scrubber sludge by 20 to 30 percent, but can be offset by

improved dewatering and settling properties. Approximately 20 to 30 percent more scrubber sludge will be produced with a lime or limestone stoichiometry of 1.5 (67% utilization) than with a stoichiometry of 1.1 (90% utilization).

Fly ash is made of fine particulates similar in texture to Portland cement, with specific gravities ranging from 2 to 3. Chemically, it primarily consists of silicon, aluminum and iron compounds. In addition, western ashes have significant quantities of calcium, magnesium, sodium and sulfur compounds. The calcium compounds of the fly ash are responsible for the cement-like qualities the western fly ash displays when mixed with water or with wet scrubber sludge.

These western fly ash disposal materials have displayed a higher degree of structural stability and impermeability than equivalent mixtures made with eastern bituminous fly ashes. The unconfined compressive strength of western fly ash disposal materials are reported to range from less than 100 psi to over 5000 psi. For comparison, a car exerts 30 psi on the surface while a person walking exerts 5 psi. Permeability coefficients of the western fly disposal materials varied from 10^{-5} to 10^{-10} cm/sec. Permeability coefficients less than 10^{-6} cm/sec are considered low. For example, permeability coefficients of 10^{-7} cm/sec and lower are specified as suitable for lining.

Adequate structural stability for disposal can be achieved using eastern fly ashes with moisture control and compaction. Because of the pozzolanic nature of the eastern fly ashes, lime can be added to reduce the permeability further and to enhance the structural stability.

Bottom ash is collected in the bottom of the boiler
as a coarse, angular particle ranging from 50 microns to 1 inch
in diameter, similar to fine gravel and sand. Consequently,
it has a high porosity with permeability coefficients, typically
ranging from 10^{-1} to 10^{-3} cm/sec. It displays no cohesive qual-
ities. With respect to major species, it is chemically very
similar to its respective fly ash. However, leachable constituents
are significantly lower, in part due to its fused condition and
reduced surface area.

The principal components found in scrubber solids are
the calcium sulfite coprecipitate, gypsum and calcium carbonate.
The scrubber liquor will be composed primarily of calcium,
magnesium, sodium, chloride, sulfite, carbonate and sulfate.
The relative concentration of the liquid and solid phase chemical
species will be dependent on the SO_2 and oxygen concentration in
the flue gas, the type of scrubber system, the quality of sorbent
and makeup water, and the operation of the scrubber.

Scrubber sludges as received from a clarifier are not
structurally stable and have relatively high permeabilities. In
general, scrubber sludges behave as cohesiveless materials dis-
playing unconfined compressive strengths less than 20 psi. In
many cases the scrubber sludges are not structurally stable.
Permeability coefficients of 10^{-3} to 10^{-4} cm/sec are reported
for settled and drained FGD scrubber sludges. By applying more
extensive dewatering techniques (e.g., vacuum filtration or
centrifugation) and compaction techniques the permeability
coefficients can be reduced to 10^{-4} to 10^{-5} cm/sec, and the
structural properties enhanced. Upon stabilization with either
western fly ash, eastern fly ash and lime, or with some commer-
cial fixation techniques, unconfined compressive strengths in
excess of 100 psi can be acheived. Disposal materials with these
properties are structurally stable and relatively impermeable.

The leaching of fly ash, bottom ash and scrubber sludge is important with respect to possible groundwater or surface water contamination. The concentration of major species in the FGD scrubber sludge is high particularly in comparison with that of the ash leachate. During the active stages of sludge or ash/ sludge disposal, the characteristics of the leachate will be dominated by the quality of the adherent scrubber liquor and will contain high concentrations of chloride, sulfate and total dis- solved solids (sodium and magnesium in some cases). After the original adherent scrubber liquor is leached, the solubilities of calcium sulfite hemihydrate, gypsum and calcium carbonate will control the concentration of major species.

Trace element concentrations in the leachate are highly variable and are dependent upon the chemical characteristics of the leaching solution, the pH, temperature, the solubility of the compound containing the trace element and the concentration in the solid phase. A review of available field leaching data of FGC waste products fly ash, bottom ash and scrubber sludge has revealed that the average reported concentration of all chemical species were below the September 12, 1978 RCRA criteria for toxicity. The average concentration of the following chemical species have exceeded or were near the federal drinking water standards or irrigation water quality parameters:

Arsenic	Chromium	Manganese	Molybdenum
Boron	Fluorine	Mercury	Selenium

Application of the same evaluation criteria to FGD sludge liquors' and elutriates indicates that the average concentration of the same elements plus cadmium exceeded either the drinking or irrigation water quality criteria. These water criteria were not meant for evaluation of FGC waste leachates but are used here for reference in lieu of applicable concentration standards for FGC leachates.

In summary, western subbituminous coal produces less ash and scrubber sludge than eastern bituminous coal does in producing the same amount of electricity. If the expected air regulation of 85 percent SO_2 removal is applied, the quantity of scrubber sludge produced by an installation firing western coal will be increased substantially. This effect is minimal for units firing eastern coal. Western coals, in general, produce a fly ash which has significantly higher calcium concentrations and higher alkalinity than that produced by eastern coal. The higher calcium concentrations and available alkalinity have been correlated with the capability of the western fly ashes to stabilize structurally the scrubber sludge without the addition of lime and other fixation additives.

Permeabilities below 10^{-6} cm/sec and unconfined compressive strength in excess of 100 psi can be achieved upon stabilization with either western fly ash, eastern fly ash and lime, or other fixation techniques. The quality of leachates from FGD scrubber sludge will initially be governed by the quality of the adherent scrubber liquor and later by the solubility of the solid phase. The average reported concentration of arsenic, boron, chloride, chromium, fluorine, manganese, mercury, molybdenum, selenium and sulfate exceeded federal drinking and irrigation water criteria.

1.3   Waste Disposal Practices

This task investigated current disposal methodologies employed by the utility industry in the disposal of their FGC by-products, fly ash, bottom ash and scrubber sludge. The prevalence of these practices in 1978 was determined and then projected into the future. Alternate disposal technologies which

may be applicable in the future were also considered. An analysis of the transportation distance from point of origin to the final disposal site was made in order to assess the economic impact of disposal.

The economic impact of implementation of RCRA was estimated by interpreting the RCRA relative to the disposal of nonhazardous solid wastes from the electric utility industry. In the absence of a sufficient data base, the cost of RCRA compliance was estimated by (1) making "reasonable" assumptions regarding existing and new plant situations and (2) applying engineering and cost data for a typical coal-fired plant to calculate the nationwide cost of compliance for a hypothetical enforcement scenario.

1.3.1   Current Disposal Practices

Current practices for disposing of solid waste products generated by coal-fired utility plants were determined from two data sources. One source was the Federal Power Commission Form 67 data tape which contained information reported by the utility companies to the FPC for the year ending December 31, 1974. Selected utility plants were contacted and the literature was searched to supplement the Form 67 data and provide more current information.

Data for 1978 were obtained from 64 plants producing over one-third of the fly ash and bottom ash concerning their mode of disposal, quantities, utilization, distance to disposal site and disposal costs. Information on sludge disposal practices was obtained from other recent reports.

Disposal of FGC by-products can be placed in two general categories, ponding or landfilling. Significantly more fly ash is currently disposed of in landfills (51 percent) than in ponds (34 percent). However, more plants use ponding of the fly ash than landfilling. Current bottom ash disposal practices favor ponding (44 percent) over landfilling (29 percent). The rest of the ash is either utilized or disposed of in undisclosed methods (paid disposal or other).

Future trends are anticipated to favor landfilling. Sufficient data was not available to determine the percentages of disposal sites which are utilizing linings and/or monitoring wells.

Information on the amounts of scrubber sludge ponded versus landfilled or other modes was not available on a consistent and meaningful basis either from the FPC data tape or from contacts with the utilities. This is principally due to the fact that scrubber sludge is produced as a wet product in slurry form. There was lack of consistency in reporting amounts based upon a dry product versus a slurry with varying percentages of solids on a weight basis. However, applicable documentation estimated that in 1977 approximately three million tons of FGD scrubber sludge on a dry weight basis were produced. Of the 30 plants producing sludge, 60 percent employed ponding and 40 percent employed landfill. Five of these plants stabilize the scrubber sludge by mixing with fly ash and lime (three have contracted with I. U. Conversion Systems). Two plants stabilize by blending with alkaline ash (one has contracted with Research Cottrell). One plant uses Dravo's Calcilox method for stabilization of the scrubber sludge followed by ponding. Of the 37 utilities who have scrubber

plans announced, 51 percent will use landfilling as opposed to ponding. This increase in percentage (40 percent to 51 percent) of plants choosing landfilling over ponding indicates a trend towards landfilling.

In order to characterize current disposal practice in terms of distance from generating plant to ultimate disposal site (whether pond or landfill), selected utilities responsible for producing over one-third of the fly ash and bottom were contacted. The tabulation of this data indicated that approximately 93 percent of fly ash and bottom ash is transported less than five miles from generation point to disposal point. The mean distance from plant to disposal site was 3.0 miles. The average distance each ton of ash was transported was slightly less than 2.7 miles. This information is more valuable in predicting costs than just a breakdown between on-site vs off-site costs. However, it is highly probable that the majority of disposal sites within five miles of the generating plant are located on-site.

1.3.2   Economic Impact of RCRA

The enforcement function of RCRA is intended to be performed by the states. The state agencies will have considerable flexibility in classifying sites as sanitary landfills or open dumps and determining both compliance options and compliance schedules for facilities classified as open dumps. Under current circumstances, it was not possible to determine the number of facilities potentially affected by RCRA or the course of action facilities will have to take. For these reasons, the economic evaluation was based on hypothetical enforcement scenarios as opposed to an estimate of the cost of compliance for the whole industry.

The economic evaluation was accomplished by estimating the costs for (1) potential enforcement options relating to RCRA and (2) current disposal practices. The analysis involved scenarios for existing and planned (or future) plants. The cost estimates were based on costing various disposal options for a "typical" 1,000 MW coal-fired plant. These cost factors were then used to determine the cost of compliance by applying them to the entire generating capacity assumed to be affected.

Several key assumptions were necessary to estimate the economic impact of RCRA. Some of these assumptions have a sound basis. However, others are subject to question. The assumptions are presented only as being reasonable and should not be construed as being the only reasonable assumption. Furthermore, the economic estimate presented here should be evaluated in light of these key assumptions.

The potential impact of RCRA on existing plants was assumed to be limited to plants beginning operation after 1970. The plants were assumed to have to move disposal from the current distance of 4.8 kilometers (3 miles) to 16.1 kilometers (10 miles). New landfill sites were assumed to be necessary to protect groundwater. Ponds were assumed to be lined. It was assumed that (1) the landfill and ponding cost factors developed for the "typical" 1000 MW coal-fired plant could be applied to the 1970-1976 distribution of existing generating capacity and (2) one-half of the existing capacity would have to move their disposal sites.

For planned (or future) plants, the primary assumption was that all plants would be affected by RCRA. The projected capacity to be constructed in the 1975-1985 period was taken to be 160,000 Mw, or 160 "typical" 1000 MW plants. The primary

effect of RCRA was assumed to be distance from plant to disposal. The average distance from plant to disposal was assumed to be 8.0 kilometers (5 miles) for RCRA as opposed to 4.8 kilometers without RCRA. As with existing plants, all ponds were assumed to be lined and the cost factors for the "typical" 1000 MW plants were assumed applicable.

A summary of the estimated cost of compliance with RCRA is presented in Table 1-2. The cost of well monitoring ($4,000,000/yr) has been included in the revenue requirements for the existing and planned facilities.

It is estimated that the capital investment costs related to disposal of electric utility nonhazardous solid waste will increase 36 percent or approximately one billion dollars due to compliance with RCRA. Annual revenue requirements, however, are estimated to increase only 5-6 percent, from 1.20 billion to 1.27 billion dollars per year.

In general, the cost data used indicate that the cost of disposal rises sharply with the distance from the plant. Ponding is economical only at very short distances. At greater distances, the cost of pumping is very high. The cost of liners is also very important. The cost differences between clay liners and synthetic liners may vary by 50 percent or more. When the capital investment is in the range of 10 to 50 million dollars per plant, a difference of 50 percent is large.

The assumptions that have the greatest effect on the cost estimates are the ones concerning (1) how the plants would comply with RCRA associated regulations, (2) what those regulations might be, and (3) the applicability of the cost data

TABLE 1-2. SUMMARY OF ESTIMATED COST OF COMPLIANCE

(MID 1979 DOLLARS)

	Capital Investment Costs ($)		Revenue Requirements ($/yr)	
	Estimated Cost of Compliance with RCRA	Estimated Current with Predicted Cost - No RCRA	Estimated Cost of Compliance with RCRA	Estimated Current or Predicted Cost - No RCRA
Existing* Plants (1970-1978 Construction)	49,075,000	0	68,725,000	55,350,000
Planned and Future Facilities (1978-1985)	3,631,750,000 -	2,699,650,000	1,202,700,000.	1,148,140,000
Total Costs	3,680,825,000 -	2,699,650,000	1,271,425,000 -	1,203,490,000
Net Costs	981,175,000		67,935,000	

*Assuming 50% out of Compliance

for the generalized calculations made in this study. Several
important factors were not included in the analysis of the cost
of compliance. One important question is the fate of any existing
disposal facilities abandoned because of danger to groundwater.
If such facilities are forced to remove the waste materials
and recover the site, the costs would be very high. If the
facility is allowed to cover ponds or landfills and abandon them,
the impact will be limited to the costs associated with the mature
retirement of the facility.

Another important cost consideration that was not in-
cluded in the study was the cost of building a levee to protect
ponds from flooding. As the potential impact of the levee on
flood water retention is a major factor, a study must first be
conducted to determine if such structures can provide protection
while allowing passage of flood water. If levees can be constructed
that will be in compliance with these restrictions, the cost of
construction and maintenance will be highly site dependent.

1.3.3   Alternate Disposal Technologies

Three alternate disposal techniques - mine disposal,
ocean disposal and landfarming - have been considered.

Of the three techniques, disposal in mine is the most
promising. Disposal in surface mines is the most technically
feasible for several reasons. Surface mines offer substantial
capacity, ease of disposal, future resource recovery, and in many
cases close proximity to sludge sources. One potential benefit
of disposal in deep mines is in preventing mine subsidence.
Mine disposal in general should be considered a viable disposal
alternative. The suitability of a mine will be site-specific

and depend on factors such as geographical location, site geology and site hydrology. Provided transportation distances are not excessive and lining is not required, surface mine disposal can be less expensive than ponding or landfilling. More work is needed to study waste/mine interaction both chemical and physical. Other studies could discuss the potential for mine disposal on a regional basis.

Significant questions have been raised as to the environmental acceptability of ocean dumping of FGC wastes. Four principal impacts have been identified: Benthic sedimentation, suspended solids, increased chemical oxygen demand and trace element contamination. Existing regulations and regulatory initiatives do not favor any new ocean dumping However, the disposal of stabilized FGD sludge blocks in ocean as manmade reefs appears promising. Costs of disposal have been estimated to range from $4.00 to $12.50 per dry ton of sludge. Additional research is required to characterize the chemical, physical and biological interaction with the ocean environment.

Landfarming does not seem to be a viable option because of a lack of nutrients, the potential for excessive concentrations of trace elements and the large volume of solid waste involved.

In summary, disposal practices of the utilities equally favor ponding and landfilling. Future trends slightly favor landfilling. Disposal in surface and underground mines is technically and economically feasible and consequently may see increased usage as a disposal site. The mean distance from the plant to disposal site is 3 miles. RCRA Section 4004 will result in increased disposal costs to the utility industry as a whole. Current practices of some utilities will meet RCRA Section 4004 requirements and consequently will feel little economic impact. Sites of other utilities will require substantial upgrading which will result in significantly higher disposal costs.

1.4   Utilization of Utility Wastes

For coal-fired utility plants, recovery practices
are directed toward the utilization of the fly ash, bottom ash
and/or boiler slag and flue gas desulfurization scrubber sludge
in some form. Utilization of the by-produces would be desirable
in that

1) it reduces the quantity of wastes requiring
    disposal thus reducing disposal costs and
    potential adverse environmental impacts
    of disposal, and

2) it conserves other U.S. natural resources
    and can result in income to offset the
    costs of disposal.

In 1977 total U.S. production of fly ash, bottom ash
and boiler slag was 67.8 million tons with 14.0 million tons
successfully recovered and utilized for a utilization of 20.7
percent.

Several areas of fly ash, bottom ash and boiler slag
utilization have been proven technically feasible. These areas
are summarized in Table 1-3.

The utilization of calcium sulfite/sulfate based FGD
scrubber sludge has been much more limited than that of ash.
Utilization of scrubber sludge is more one of conceptual
development and testing for recovery rather than actual wide-
spread utilization. Possible uses for scrubber sludges are (1)
recovery of chemicals, (2) manufacture of building materials,
(3) structural fill, (4) paving materials, (5) soil stabilization
in agriculture and (6) environmental pollution control. Only the

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TABLE 1-3. U.S. ASH UTILIZATION IN 1977

	Fly Ash	Bottom Ash	Boiler Slag
Total Ash Collected (10^6 tons)	48.5	14.1	5.2
Total Ash Commercially Utilized (10^6 tons)	4.2	2.8	3.0
Utilization (%)			
Raw Material - Portland Cement	10	-	3
Type 1 - P Cement	7	3	-
Partial Replacement of Cement in Concrete	37	-	-
Lightweight Aggregate	3	5	-
Structural Fills & Embankments	30	33	8
Road Base and Subbase	4	8	2
Asphalt Filler	3	-	-
Ice Control	-	36	14
Blasting Grit and Roofing Material	-	-	50
Miscellaneous	3	15	23

production of gypsum wallboard and the utilization of gypsum in making Portland cement have any possibility as a significant outlet for scrubber sludge in the foreseeable future.

Regenerable SO_2 processes have the capability of reducing the tonnage of sludge while producing commercial by-products with demonstrated markets. Regenerable SO_2 processes eliminate the calcium sulfite/sulfate scrubber sludge while producing either sulfur or sulfuric acid. Another advantage of regenerable processes is that they make more ash available for utilization, since some of the ash would otherwise be required to stabilize the scrubber sludge.

Marketability of sulfur and/or sulfuric acid would appear to be limited for the next 10-12 years on any widespread basis. In general, existing sources of sulfur and sulfuric acid are available to meet demand. However, in certain locations unique conditions might create a local market. After 1990, marketability of sulfur and sulfuric acid derived from fossil fuel SO_2 may very well be enhanced because deposits of sulfur presently being mined along the Gulf Coast are expected to be depleted.

Regenerable processes do require a pre-scrubber system prior to the SO_2 removal system to remove fly ash and chlorides. The pre-scrubber waste stream will be small (5-10 percent) compared to the typical scrubber sludge waste stream, but will also have to be evaluated carefully as an environmental hazard due to its high proportion of soluble species.

In summary, utilization of FGC by-products is impor-
tant in terms of reducing environmental impact, conserving
natural resources and boosting the economy. The majority of the
fly ash is used as a fill material. Bottom ash is principally
used as a fill material and for skid control. The largest
portion of the boiler slag is used as blasting grit and roofing
material. Current utilization of FGD scrubber sludge is
practically zero. Regenerable processes in a limited scale can
replace the scrubber sludge with either sulfur or sulfuric acid.

1.5   Conclusions and Recommendations

The solid wastes produced by coal-fired electric
utilities were examined with regard to the requirements of
Section 4004 of the Resource Conservation and Recovery Act.
The study involved four major tasks: (1) characterization of
the electric utility industry, (2) characterization of the solid
wastes, (3) waste disposal practices, and (4) utilization of
utility wastes.

The following conclusions have been drawn based upon
the results of this study:

1)   Implementation of Section 4004 of the
     RCRA will be borne primarily by the
     states and EPA staffs in Regions III,
     IV, V and VII. These regions account
     for over 85 percent of the coal-fired
     capacity in 1976. The coal-fired
     capacity of Region VI and other western
     states will undoubtedly increase in the
     near future.

based on

2) Technical information available at this
    time *it appears* that the majority of utility
    solid wastes are nonhazardous and, as such,
    will require disposal in sanitary landfills.
    Furthermore, fly ash, bottom ash, and
    scrubber sludge mixtures can be stabilized
    to improve structural strength and reduce
    permeability. However, sufficient hydrologi-
    cal data from existing disposal sites are
    not available for adequate assessment of the
    degree to which existing sites qualify as
    sanitary landfills.

3) The use of western coal is more desirable
    in that its combustion produces 33 percent
    less fly ash and 95 percent less scrubber
    sludge than eastern coal under current SO_2
    and particulate regulations. The potential
    future regulation of 85 percent SO_2 removal
    remove part of this incentive to use western
    coal. Western coal has the added advantage
    of producing alkaline fly ashes which are
    capable of stabilizing FGD scrubber sludges.

4) The economic impact of RCRA was estimated
    to be roughly 1 billion dollars in capital
    investment and 70 million dollars in annual
    revenue requirements for a hypothetical
    enforcement scenario involving the modifica-
    tion of disposal sites for approximately
    20,000 Mw of existing capacity and 160,000 Mw
    of planned capacity. It should be emphasized
    that these costs are only rough estimates and
    highly dependent upon the scenario basis and
    key assumptions.

5) <u>Utilization of utility wastes should be strongly encouraged</u> because it reduces the environmental impact of disposal, conserves natural resources and could boost the economy. Utilization may be increased by eliminating the three major areas of resistance: (1) technical limitations compared to alternative materials, (2) institutional barriers related to poor understanding or inadequate market development, and (3) possible environmental concerns related to some uses. Regenerable SO_2 control processes will probably not significantly reduce the industry-wide production of solid wastes, although these processes can reduce the solid waste disposal requirements for a particular utility where local markets exist for sulfur and sulfuric acid.

Additional effort can be recommended in many areas examined by the study. Three major areas requiring further work are listed below:

1) Additional effort is required to determine the probability of existing disposal sites being classified as open dumps and thus needing upgrading. This will require documentation of existing information and then an intensive characterization of all existing disposal sites.

2) Further research is needed to correlate the leachability of solid wastes with coal quality, degree of coal pulverization, type and operation of boiler, type and operation of control equipment, and disposal methodology. Results of this research will allow the selection of a disposal technology (prior to construction of the power plant) which will insure no adverse effects on the environment.

3) An aggressive program is needed to eliminate the three areas of resistance to utility waste utilization. First, technical limitations in comparison to alternative materials must be surmounted through research and development. For example, in light of an impending mineral crisis, basic research and development should be funded to develop extraction processes for minerals and trace elements from FGC by-products. Second, institutional barriers related to a poor industry or user industries must be removed. Marketing of these by-products needs encouragement, perhaps through financial credits of some sort. Third, where potential environmental concern exists, policy decisions based upon the relative merits of a particular application compared to the environmental impact of landfilling or ponding must be made.

2.0 CHARACTERIZATION OF ELECTRIC UTILITY INDUSTRY

For the purposes of this study, the United States electric utility industry was characterized in terms of two items: fuels and economics. Of particular interest in the fuels area is that fraction of power plants which burn coal, either exclusively or in combination with other fuels. In terms of economics, both the general state of the industry and the regulatory climate under which it functions are of interest. These topics will be discussed in the following subsections.

2.1 Fuels Characterization

Electric power is usually produced commercially through the operation of AC generator, or alternator, which is driven by a prime mover. Almost 70 percent of these prime movers are steam turbines powered by boilers which consume fossil fuel (either coal, oil, or gas). Figure 2-1 shows the relative importance of all prime movers in 1976, 1981, and 1986, as reported by the National Electric Reliability Council (NERC).[1] As can be seen, coal is singularly the most important fuel, and it will probably remain the most important through this period.

For this study, all utility-owned coal-fired power plants* existing in 1976 were inventoried. This inventory was based on 1974 FPC (now FERC) Form 67 data updated through 1976 by utilizing data in the 1977-1978 Electrical World Directory of Electric Utilities.[2] In some cases, data were verified through

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*For this study, a coal-fired power plant is defined as a facility consisting of one or more generating units, which burns coal alone or in combination with other fuels and which produced electricity for sale to the general public. Some boilers in the plant may not be equipped to burn coal. A coal-fired unit, for this study, is a unit equipped to burn coal, regardless of actual fuels consumed.
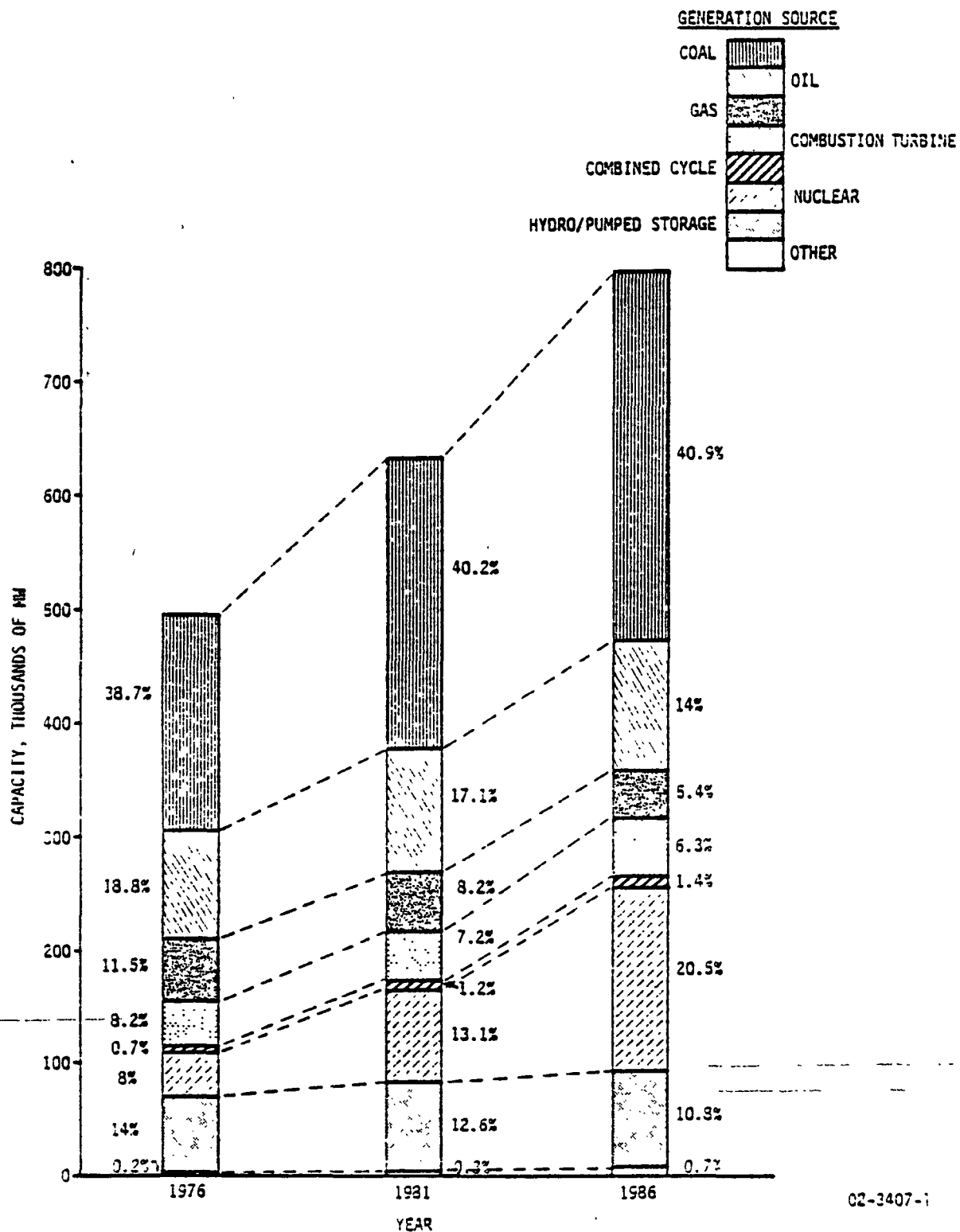
FIGURE 2-1.   EXISTING AND PLANNED GENERATING
CAPABILITY AS ESTIMATED BY NERC (SOURCE: REF. 1)

comparison with other reported data[1,3,4,5] and Radian proprietary
data. Results of this inventory are summarized for the 10 EPA
regions in the 48 contiguous United States. These regions are
as follows:

Region I   - Maine, Vermont, New Hampshire, Rhode
                Island, Massachusetts, Connecticut

Region II   - New York, New Jersey

Region III   - Pennsylvania, Delaware, West Virginia,
                District of Columbia, Maryland, Virginia

Region IV   - Kentucky, Tennessee, North Carolina,
                South Carolina, Georgia, Alabama, Florida,
                Mississippi

Region V   - Wisconsin, Michigan, Illinois, Indiana,
                Ohio, Minnesota

Region VI   - Texas, Arkansas, Louisiana, Oklahoma,
                New Mexico

Region VII   - Nebraska, Kansas, Iowa, Missouri

Region VIII   - Montana, North Dakota, South Dakota,
                Wyoming, Colorado, Utah

Region IX   - Arizona, California, Nevada

Region X   - Washington, Oregon, Idaho

Table 2-1 shows the number of facilities with coal-fired units,
the electric megawatt (Mw) capacity of these facilities, the
amount of coal-fired capacity in Mw, and the average size in Mw
of a coal-fired unit for each region. Table 2-2 gives the actual
size distribution of coal-fired capacity for each region. For
the nation, there were 399 plants inventoried, with 65 percent
having coal-fired capacity less than 500 Mw. There were 57 plants
with coal-fired capacity from 1000 to 1999 Mw and there were 12
plants whose coal-fired capacity was in excess of 2000 Mw. These
largest plants were in Region III (2 plants), Region IV (4 plants),
Region V (3 plants), and Regions VI, VII, IX (1 plant each).
Most of the coal-fired plants were in Regions V (151), IV (72),
III (60), and VII (49). These plants account for about 85 percent
of the coal-fired capacity in the nation and are located in the
Midwest and South (areas which have traditionally relied upon

TABLE 2-1
INVENTORIED U.S. COAL-FIRED CAPACITY IN 1976

EPA Region	No. Plants with Coal-Fired Units	Total Plant Capacity (Mw)	Total Coal-Fired Capacity (Mw)	Average Coal-Fired Capacity Per Plant (Mw)
I	7	3,441.	3,295.89	470.84
II	16	7,438.09	6,333.96	395.87
III	60	43,580.19	39,474.80	657.91
IV	72	53,063.06	50,153.7	696.58
V	151	73,197.81	68,832.58	455.84
VI	5	5,172.2	5,172.2	1,034.44
VII	49	15,505.17	14,264.76	291.12
VIII	33	9,177.7	9,171.2	277.92
IX	5	4,444.	4,351.	870.2
X	1	1,329.8	1,329.8	1,329.8
U.S. Total Inventoried for 1976	399	216,349.02	202,379.89	507.22

TABLE 2-2

DISTRIBUTION OF INVENTORIED COAL-FIRED CAPACITY WITHIN EPA REGIONS

EPA Region	No. Plants with Coal-Fired Units	Number of Plants with Given Coal-Fired Capacity				
		<100 Mw	100-499 Mw	500-999 Mw	1000-1999 Mw	>2000 Mw
I	7	2	3	1	1	0
II	16	2	9	4	1	0
III	60	6	30	7	15	2
IV	72	9	29	16	14	4
V	151	46	55	27	20	3
VI	5	1	0	1	2	1
VII	49	19	18	10	0	1
VIII	33	15	11	6	1	0
IX	5	1	2	0	1	1
X	1	0	0	0	1	0
U.S. Total Inventoried for 1976	399	101	157	72	57	12

coal). Data by state for capacity and size distribution for
these plants and all others inventoried are given in Appendices
A and B and all plants are itemized by EPA region and state in
Appendix C. Nationwide total coal-fired generating capacity as
inventoried was 202,380 Mw in 1976. This megawatt figure is 5.8
percent higher than the widely accepted NERC estimate[1] of 191,336
Mw in 1976, probably because (1) nameplate data used may be differ-
ent in some cases, (2) the Radian data may have included some
plants under construction in 1976, and (3) the Radian data did
not include some plants which, while equipped to burn coal,
burned little or none during 1976. Consequently, while some of
these Radian data may be subject to review and revision, it is
believed that this inventory provides reasonable data on size and
location of coal-fired plants existing in 1976.

Industry based projections of coal-fired capacity and
coal consumption in 1981 and 1986 have been made by NERC[1] for the
contiguous NERC regions (shown in Figure 2-2).* These projections
are shown for capacity in Table 2-3 and consumption in Table 2-4.
As can be seen from these tables, the use of coal will vary widely
across the country, with NPCC (most of EPA Regions I and II)
utilizing the least amount and ECAR (including most of EPA Region
V and parts of Regions III and IV), the most. The fastest growing
coal-using regions, in terms of reliance on coal as a fuel, are
ERCOT (17.3 percent per year) and SPP (12.4 percent per year),
both of which are primarily in EPA Region VI. In some regions,
the relative importance of coal is seen to be decreasing, although
total nationwide installed coal capacity is forecasted by NERC
to be increasing at the rate of approximately 5.8 percent per
year between 1976 and 1981 and approximately 5.2 percent per year
between 1981 and 1986. Reliance on western coal is also seen to

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*NERC estimates are not delineated by individual states. Cross-
reference can be made using the EPA regions in Section 2.1 to
obtain very rough estimates of coal capacity in each EPA region.

MARCA
5

9
WSCC

4
MAIN

1
ECAR

6
NPCC

3
MAAC

SPP
8

ERCOT
2

7
SERC
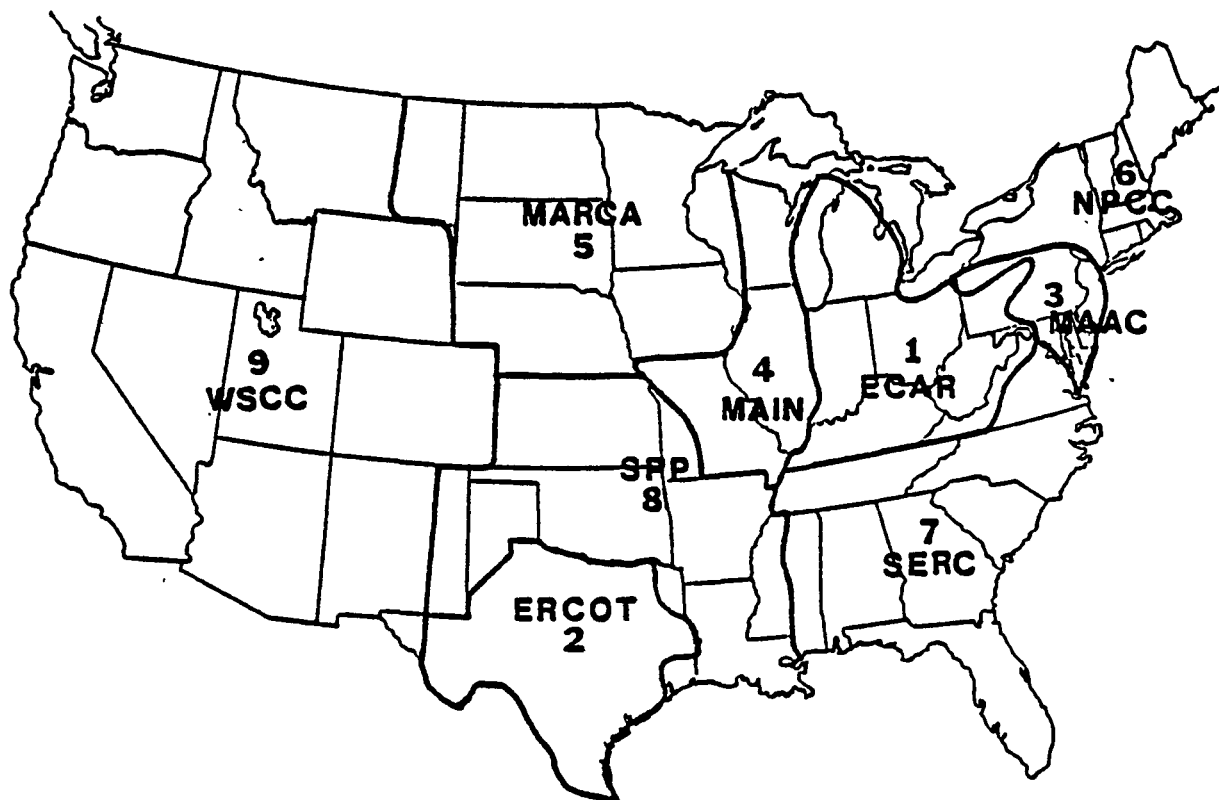
1 - ECAR - East Central Area Reliability Coordination Agreement
2 - ERCOT - Electric Reliability Council of Texas
3 - MAAC - Mid-Atlantic Area Council
4 - MAIN - Mid-America Interpool Network
5 - MARCA - Mid-Continent Area Reliability Coordination Agreement
6 - NPCC - Northeast Power Coordinating Council
7 - SERC - Southeastern Electric Reliability Council
8 - SPP - Southwestern Power Pool
9 - WSCC - Western Systems Coordinating Council

Figure 2-2

NERC RELIABILITY REGIONS

TABLE 2-3
NERC GENERATION PROJECTIONS, 1976-1986

NERC Region	1976* Coal-Fired Capacity (Mw)	1976* % of Total Capacity for NERC Region	1981 Coal-Fired Capacity (Mw)	1981 % of Total Capacity for NERC Region	1986 Coal-Fired Capacity (Mw)	1986 % of Total Capacity for NERC Region
1-ECAR	62,620	81.7	76,154	79.0	88,101	73.8
2-ERCOT	2,360	6.9	10,711	23.5	18,999	34.1
3-MAAC	14,057	33.7	14,730	29.8	16,989	27.1
4-MAIN	26,014	66.1	30,574	58.2	37,297	56.4
5-MARCA	9,964	49.3	17,704	62.8	21,422	60.8
6-NPCC	3,737	7.6	4,035	7.6	5,337	8.4
7-SERC	51,936	52.1	58,585	44.9	74,574	43.4
8-SPP	5,084	11.8	15,954	27.2	30,384	38.1
9-WSCC	15,564	17.3	24,736	21.3	32,978	23.0
U.S. TOTAL	191,336	38.7	253,362	40.2	326,081	40.9

*Actual capacity.

Source: Ref. 2.

TABLE 2-4

ANNUAL COAL REQUIREMENTS (MILLIONS OF TONS) FOR THE U.S.

UTILITY INDUSTRY AS ESTIMATED BY NERC

NERC Region	1976* Western Lignite	1976* Western Coal	1976* Other Coal	1976* Total Coal	1981 Western Lignite	1981 Western Coal	1981 Other Coal	1981 Total Coal	1986 Western Lignite	1986 Western Coal	1986 Other Coal	1986 Total Coal
1-ECAR	0	6.7	135.5	142.2	0	12.2	157.5	169.7	0	20.9	198.2	219.1
2-ERCOT	12.	0	0	12.0	30.3	15.7	0	46.0	58.0	18.2	0	76.2
3-MAAC	0	0	32.9	32.9	0	0	38.6	38.6	0	0	41.6	41.6
4-MAIN	0	12.9	38.5	57.3	0	21.7	52.2	74.0	0	30.9	57.2	88.1
5-MARCA	9.9	12.4	6.1	24.4	15.8	30.6	5.7	52.1	24.7	39.5	5.8	70.0
6-NPCC	0	0.8	7.0	7.8	0	1.4	8.7	10.1	0	1.8	12.0	13.8
7-SERC	0	0	104.3	104.3	0	1.4	121.6	123.0	0	9.2	137.9	147.1
8-SPP	0	9.1	0	9.1	0	57.7	0	57.7	7.0	113.4	0	120.4
9-WSCC	0	39.9	0	39.9	0	78.1	0	78.1	0	103.1	0	103.1
U.S. TOTAL	21.9	87.8	324.3	434.0	46.2	218.8	384.3	649.3	89.8	336.9	452.7	879.4

*Actual coal requirements.

Source: Ref. 3

increase at a rapid rate during this period, while total coal consumption is expected to increase at a rate of about 8.4 percent per year until 1981, when it will increase at a rate of about 6.3 percent per year until 1986. Any effects of a revised New Source Performance Standard for SO_2 on western coal development were not estimated.

Thus it can be seen that the use of coal as a boiler fuel in the electric power industry will continue to be highly important, especially in some regions. While overall nationwide coal growth is expected, reliance will decrease in some regions, although these decreases will probably be offset by rapid increases in reliance in other regions, especially in the Southwest. Presently, about 400 plants, with a total capacity of about 200,000 Mw, utilize coal as a fuel. Most of these plants are in the Midwest and South. Consequently, coal for use as a boiler fuel is now and will probably continue to be of great importance to the U.S. electric power industry.

2.2   Economic Characterization

The electric utility industry is different from most other industries in the United States because it is one of the most, if not the most, capital investment-intensive industry.* In the past, the ratio of investment to revenue for the electric utility industry has been reported to be at least 15 times higher than that for general business.[6] Total installed fixed plant facilities (generating units, transmission lines, etc.) were valued at between 150 and 180 billion dollars in 1976.[7] This capital investment intensity requires that the utility industry utilize considerably more long-term financing (such as bond) than does general industry. Consequently, the effects of inflation

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*Only the railroads might have as much capital investment in fixed physical plant.

and uncertainty in the economy are considerably more important than for general industry. Moreover, the industry is highly regulated and cannot make investment decisions in the same manner as general business. Therefore, the industry cannot be fairly compared with and must be viewed apart from general business. Over the last several years, the financial strength of the industry has significantly declined with falling bond ratings and stock selling for less than book value. While the financial state of the industry has improved somewhat since the low point in 1974, the general state is still only marginal, with reduced investment at high interest rates, diminished buyer interest, and extremely high sensitivity to inflation and uncertainty in both the economy and the regulatory process.

Four types of electric utilities exist in the United States today: (1) the investor-owned utility company, (2) the municipal or state-owned utility or public power district (PPD), (3) the rural electric cooperative (co-op) and (4) the Federal agency. Table 2-5 illustrates the market shares of each of the nation's customers and operating 95.6 percent of the nation's generating capacity.[8] As can be seen, investor-owned utilities account for 78.3 percent of nationwide installed capacity and serve 77.6 percent of the customers. Municipal and state agencies and PPD's constitute the next largest market share. Each individual utility (company, agency, or co-op) is franchised to operate as a monopoly within a certain specified geographical area, and the rates which it may charge its customers are regulated by government at some level. Consequently, the economic structure of the electric utility industry is very unique. In this section, the economic status of the industry will be discussed.

TABLE 2-5
RELATIVE MARKET SHARES OF ELECTRIC UTILITY
TYPES IN THE UNITED STATES FOR 1977

Type	Percent of Installed Capacity	Percent of Customers Served
Investor-Owned	78.3	77.6
Muni, State, and PPD's	10.2	12.4
Co-op's	1.8	10.0
Federal Agencies	9.7	0.0
U.S. Total	100.0	100.0

Source: Ref 8, p. 75.

2.2.1   Capital Expenditures

As previously mentioned, the fixed plant requirements of the electric utility industry are enormously high relative to those experienced by general business. For example, prior to 1973, a fossil-fueled single or twin unit 250-1300 Mw power plant costs about $150 per kilowatt installed capacity. In 1977, the cost had tripled to about $450 per kilowatt, and it is estimated to be about $800 per kilowatt in 1985, although some costs in this latter range have already been reported.[9] This capital cost escalation from 1973 to 1977 was roughly four times the annual rate of inflation (in percent per year) for the same period.[10] As a further example of the high capital costs faced by the industry, a 345 KV* double circuit transmission line generally costs about $165,000 per mile and up, depending on terrain, right-of-way costs, labor costs, etc.[11]

Capital expenditures for 1977 and 1978 are shown in Table 2-6 according to geographical region. These regions are given by state as follows:

New England - Maine, Vermont, New Hampshire, Rhode Island, Massachusetts, Connecticut

Middle Atlantic - New York, New Jersey, Pennsylvania

South Atlantic - West Virginia, Virginia, Maryland, Delaware, District of Columbia, North Carolina, South Carolina, Georgia, Florida

East South Central - Kentucky, Tennessee, Mississippi, Alabama

West South Central - Arkansas, Louisiana, Oklahoma, Texas

East North Central - Wisconsin, Michigan, Illinois, Indiana, Ohio

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*345 KV is a median transmission voltage. In general, transmission voltages presently range from 138 KV to 765 KV in the United States, with higher voltage lines presently under development.

TABLE 2-6

ELECTRIC UTILITY CAPITAL EXPENDITURES - 1977 AND 1978

Geographic Region	1977 Total Maintenance Costs Millions of Dollars	1978 Total Budgeted Maintenance Costs Millions of Dollars	Percent Growth in Maintenance Costs
New England	766.6 (97.4%)	963.0 (96.0%)	+25.6
Middle Atlantic	3,106.2 (91.8%)	3,318.0 (98.7%)	+ 6.8
East North Central	5,515.7 (92.3%)	6,406.0 (89.4%)	+16.1
West North Central	3,071.9 (42.9%)	4,052.9 (46.0%)	+31.9
South Atlantic	3,905.3 (84.6%)	4,613.4 (86.9%)	+18.1
East South Central	2,466.7 (32.9%)	2,670.1 (28.1%)	+ 8.2
West South Central	3,376.0 (75.1%)	3,939.0 (72.8%)	+16.7
Mountain	1,821.1 (64.8%)	2,183.1 (67.4%)	+19.9
Pacific	3,592.4 (53.5%)	4,542.2 (60.8%)	+26.4
U.S. Total	27,621.8 (71.5%)	32,687.6 (71.0%)	+18.3

Source: Ref (6), pp. 82-82

*Percent of spending in investor-owned sector in parentheses.

West North Central - North Dakota, South Dakota, Kansas,
Nebraska, Minnesota, Iowa, Missouri

Mountain - Montana, Idaho, Wyoming, Nevada, Utah, New
Mexico, Colorado, Arizona

Pacific - Washington, Oregon, California

The breakdown of capital expenditures according to the type of
utility (e.g., investor owned, co-op, municipalities, etc.) is
delineated in Appendix D. As can be seen in Table 2-6, capital
spending was about 60 billion dollars in 1977 and 1978, with
about 71 percent of the spending in the investor-owned sector.
During this two-year period, capital spending requirements were
growing for almost all types in almost all regions, with a new
national growth rate of 18.3 percent per year, or about three
times the rate of inflation as measured by the GNP deflator.[12]
Consequently, Table 2-6 illustrates the large capital expendi-
tures are primarily financed through borrowed money. Investor-
owned utilities generally borrow money through the sales of
securities, such as common and preferred stock, bonds, etc..
Figure 2-3 shows these sales, as well as capital expenditures,
for the years 1967 through 1977. It may be seen that total
securities sales for investor-owned companies are increasing at an
average rate of about 12 percent per year, with stock sales pre-
sently about equal to bond sales. However, stock sales are
increasing at a rate slightly less than total sales, while public
bond sales are increasing at a rate of only about 8 percent per
year. Capital expenditures, on the other hand, are increasing at
about 18 percent per year. Other types of electric utilities
usually finance their capital expenditures through the sale of bonds.

Because of the importance of bonds in capital formation,
the interest rate paid is a good measure of industry strength -
the stronger the company, the lower the interest rate. This
strength is measured by the bond rating of the company and/or
its fixed charge-coverage ratio. The bond rating, which influences
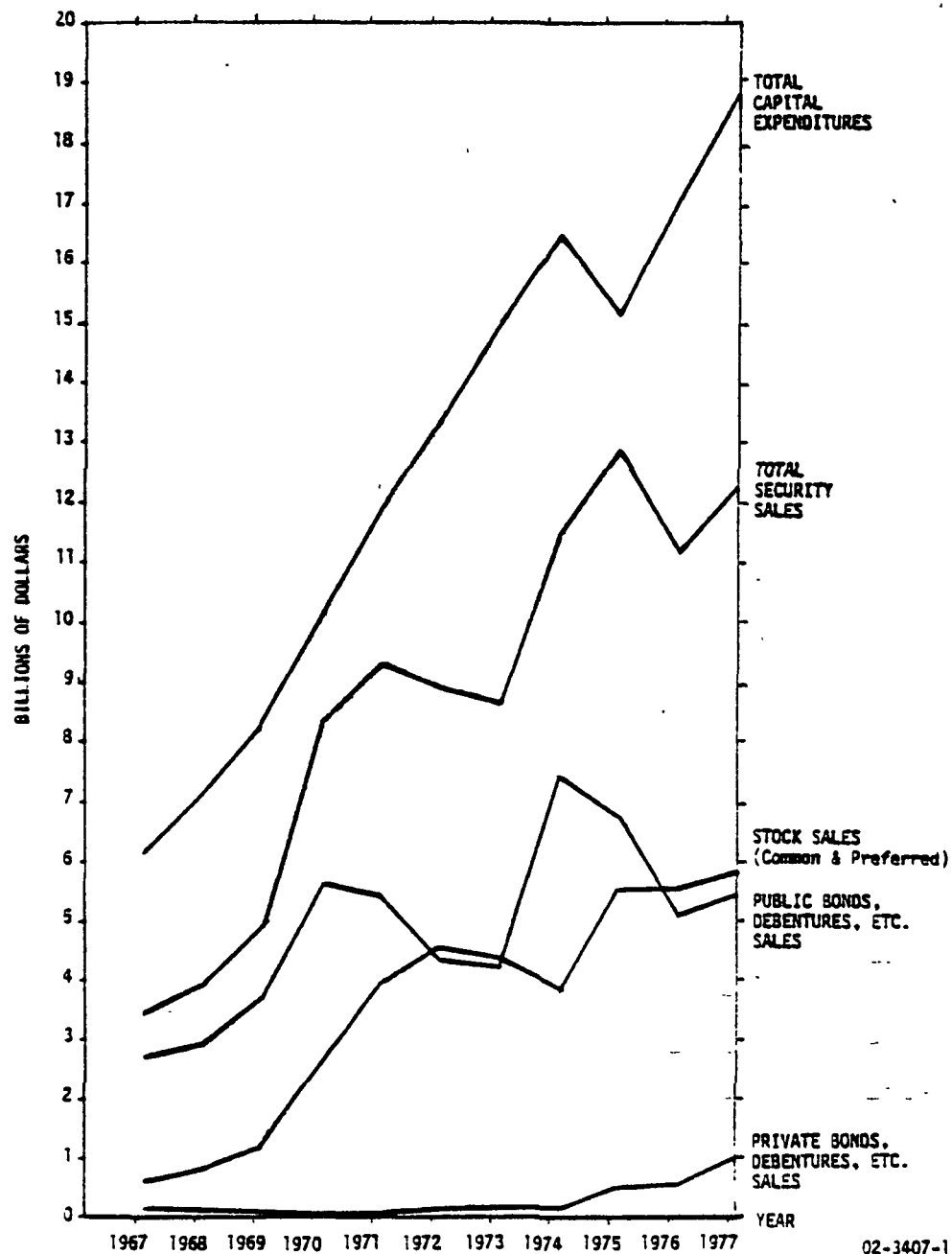
Figure 2-3

INVESTOR-OWNED ELECTRIC UTILITY
SECURITY SALES AND CAPITAL EXPENDITURES
(BILLIONS OF DOLLARS)
(Source: Ref. 6, pp. 79 & 101)

the interest rate paid, reflects marketplace conditions and historical payback records and is reported by Standard & Poor's (S&P) and Moody's. Prior to the early 1970's, utility bonds were considered extremely safe investments; most had AA or AAA S&P ratings, reflecting about the lowest interest rates existing for utility bonds.[13] By the early 1970's, some bond ratings had dro.pped slightly (namely several issues of Consolidated Edison and. Public Service Gas and Electric of Newark, New Jersey). However, the fuel crisis of 1973-1974 caused a drastic change in the utility bond market, and culminated in 1974 with Con Ed's first skipping of a quarterly dividend since 1885.[14] This financial crisis, illustrated by the Con Ed case, became widespread in the industry. Utility bond ratings began to fall until, in 1978, Texas Utilities, Inc., was the sole remaining investor-owned utility with an AAA bond rating.[13]

This reduction in the ability to raise capital is also evident in the fixed charge-coverage ratio for bonds. The pre-tax fixed charge-coverage ratio is defined as

$$FCCR = \frac{pre\text{-}tax\ income}{capital\ charges}$$

The pre-tax FCCR represents the ability to pay back borrowed funds. Generally an AAA bond rating corresponds to a pre-tax FCCR of 4 (i.e., the company can pay back the interest on its investments 4 times) and an A bond rating corresponds roughly to an FCCR of about 3. If the pre-tax FCCR is below 2.3, the utility reaches a limitation preventing it from selling additional bonds.[14] The nationwide average pre-tax FCCR's for investor-owned utilities for 1969 to 1977 are shown in Figure 2-4. As can be seen, the average FCCR dipped close to 2.3 in 1974 from a high of 3.64 in 1969. The FCCR for Con Ed actually dipped below this tolerance level.[14] While the nationwide FCCR's have been increasing, the financial status of the industry is still nowhere near that prior to the 1970's in terms of bond ratings.

PRE-TAX
FIXED CHARGE-COVERAGE
RATIO (FCCR)

"NO-SALE
THRESHOLD"

YEAR
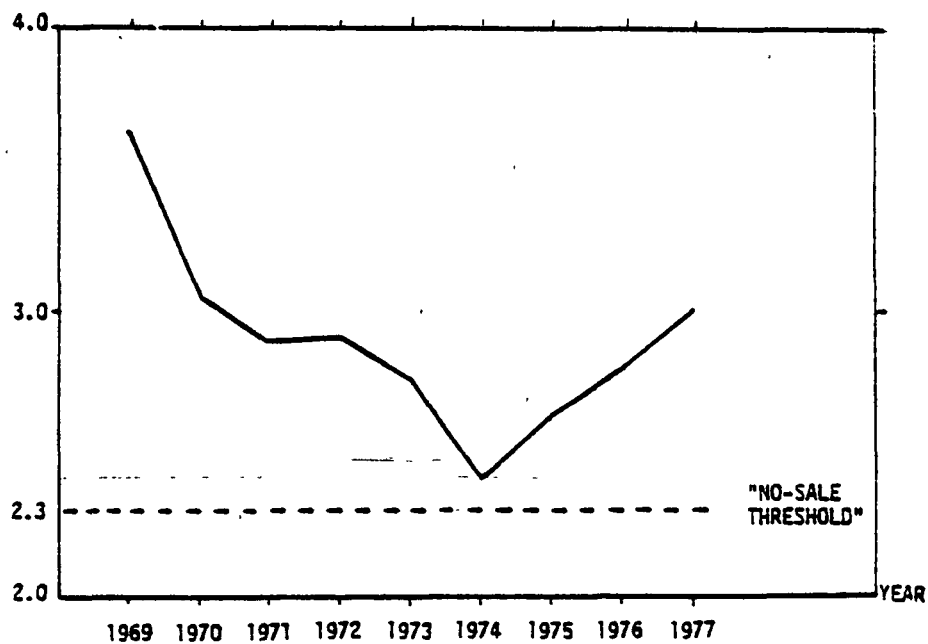
02-3407-1

Figure 2-4

FIXED COST-COVERAGE RATIO FOR INVESTOR-OWNED
ELECTRIC UTILITIES 1969-1977
(Sources: Ref. 6, Ref. 15, Ref. 16)

As for common stock, the same trends are evident in the price-book ratio.[14] The price-book ratio is defined as

$$PBR = \frac{\text{(current selling price of common stock)}}{\text{(book value of common stock)}}$$

The book value of the stock is derived from common equity and is defined[14] as

$$BV = \frac{\text{(total capital investment in plant)-(accumulated depreciation)}}{\text{(number of shares issued)}}$$

A price-book ratio greater than 1.00 is required to raise new capital. By 1974, PBR's in the Northeast had declined to 0.55; Con Ed's PBR was 0.20, the lowest in the nation. The national average PBR was 0.76. The highest PBR (Southwestern Public Service) was 1.40. Table 2-7 shows the PBR's for investor-owned utilities during this period. As can be seen, all the utilities except those in Texas, Oklahoma, and parts of Arkansas, Louisiana, Montana, Wyoming, Oregon, and Washington had PBR's less than 1.00. Those utilities in the Midwest, East, and South, and in southern California and Nevada had the lowest PBR's. While PBR's, like bond ratings, have improved somewhat since this time (up to a nationwide average of 1.01 at the end of 1977)[8], the average PBR was 0.98 for 36 new issues of common stock in the first half of 1978,[15] and 15 percent of the issues had PBR's between 0.82% and 0.89.

Although utility stocks have fared well recently when compared to the S&P 500 industrials,[16] it is evident that the industry is still not as strong financially, from a capital standpoint, as it has been in the past, and that it will be faced with rising capital costs in the future. Furthermore, the ability to raise capital is closely connected to the inflation rate. Standard and Poors reports that inflation was the most important financial factor in the period 1965-1974, causing significant increases in interest rates and financing requirements.[7] While inflation rates of 4 percent per year are desirable and

TABLE 2-7

PRICE-BOOK RATIOS (PBR'S) FOR

INVESTOR-OWNED UTILITIES IN JUNE, 1974

Geographic Region	PBR(> 1.00 is acceptable)
New England	
Maine	0.60-0.69
Remainder	<0.60
Middle Atlantic	
Pennsylvania	0.80-0.99
Remainder	<0.60
South Atlantic	
West Virginia	0.70-0.79
Virginia	<0.60
Delaware	0.80-0.99
Parts of Florida	0.70-0.99
Remainder	0.60-0.69
East South Central	
Parts of Kentucky	0.80-0.99
Remainder	0.70-0.79
West South Central	
Parts of Arkansas	0.70-0.79
Parts of Arkansas and Louisiana	0.80-0.99
Remainder including parts of Arkansas and Louisiana	>1.00
East North Central	
Michigan	<0.60
Missouri, Parts of Iowa and Kansas	0.70-0.79
Parts of Kansas and Iowa	0.60-0.69
Remainder including part of Iowa	0.80-0.99
Mountain	
Parts of Wyoming and Montana	>1.00
New Mexico and part of Arizona	0.60-0.69
Nevada	<0.60
Remainder	0.70-0.79
Pacific	
Parts of California and Washington	<0.60
Parts of California and Oregon	0.60-0.69
Parts of California	0.70-0.79
Parts of California, Oregon and Washington	0.80-0.99
Parts of Washington and Oregon	>1.00

Source: Ref. 14, p. 41

TABLE 2-8

DISTRIBUTION OF OPERATING EXPENSES (PERCENT)

	1977*	1976	1975	1974	1973
Total Operating Expenses (millions of dollars)	31,410	26,175	23,293	19,542	13,512
Constituent Percentages:					
Fuel	63.7%	62.1%	62.1%	60.8%	49.5%
Maintenance	11.1	11.1	10.8	11.5	14.5
Other	24.2	26.8	27.1	27.7	36.0

*Estimated by Edison Electric Institute

Source: Ref (8), pp. 100-101.

rates of 5 to 6 percent per year are tolerable given present
regulatory trends, inflation rates exceeding 10 percent per year
(i.e., "dougle-digit inflation") could jeopardize dividend rates,
further restrict common stock pricing, and would require signifi-
ficantly higher equity returns obtainable only through additional
rate increases. In short, the industry is not out of economic
trouble yet and may face more problems.

2.2.2   Operating and Maintenance Expenditures

Other major utility expenditures are in the operations
area. Table 2-8 shows the distribution of operating expenses
for the years 1973 through 1977. It may be seen that fuel cost
is the largest constituent of these expenses, rising from 49.5
percent in 1973 to 64.7 percent in 1977, with an abrupt jump in
1974 at the time of the fuel crisis. Fuel cost constituent
growth is about 2 percent per year as a fraction of total operat-
ing costs. Fuel costs presently account for about 37 percent of
before-taxes expenditures, or about $20.3 billion in 1977, [8] and
these costs are expected to increase in the future.

Maintenance costs, accounting for about 11 percent of
total operating expenses, are detailed for 1977 and 1978 in
Table 2-9. Over these 2 years, about $10 billion has been spent
on maintenance of physical plant. Growth in maintenance charges
escalated about 10.5 percent over the period 1977-1978. The vast
majority of these costs (almost 80 percent nationwide) were borne
by the investor-owned segment.

2.2.3   Income and General Economic Impacts of the Utility
Sector

Table 2-10 shows investor-owned electric utility income
for 1973 through 1977. As can be seen, while total income is

TABLE 2-9

ELECTRIC UTILITY MAINTENANCE COSTS - 1977 AND 1978

Geographic Region	1977 Total Maintenance Costs Millions of Dollars	1978 Total Budgeted Maintenance Costs Millions of Dollars	Percent Growth in Maintenance Costs
New England	200.0 (87.9%)	213.6 (90.1%)	+ 6.8%
Middle Atlantic	963.2 (99.4%)	1,044.6 (99.4%)	+ 8.5
East North Central	920.7 (90.1%)	1,020.1 (91.4%)	+10.8
West North Central	357.7 (66.7%)	388.7 (68.0%)	+ 8.7
South Atlantic	691.5 (89.9%)	785.7 (88.7%)	+13.6
East South Central	365.4 (34.1%)	429.6 (34.4%)	+17.6
West South Central	335.6 (70.6%)	355.2 (74.4%)	+ 5.8
Mountain	197.9 (66.6%)	224.9 (68.0%)	+13.7
Pacific	535.7 (53.8%)	583.6 (51.8%)	+ 8.9
U.S. Total	4,567.7 (79.0%)	5,046.0 (79.1%)	+10.5

Source: Ref (8), p. 97.

*Percent borne by investor-owned companies in parentheses.

TABLE 2-10

INVESTOR-OWNED UTILITY INCOME (MILLIONS OF DOLLARS)

	1977*	1976	1975	1974	1973
Total Electric Revenue	55,175	47,080	41,855	34,970	27,526
Other Income	2,800	3,105	2,666	3,515	1,982
Extraordinary Income	0	0	19	0	0
Total Income	57,975	50,185	44,540	38,485	29,508
Operating Expenses	31,410	26,175	23,293	19,542	13,512
Depreciation	4,700	4,240	3,814	3,360	3,012
Taxes	8,520	7,221	6,212	4,898	4,553
Capital Charges	5,320	5,566	5,184	4,615	3,642
Extraordinary Expenses	0	7	0	76	62
Dividends	5,810	5,118	4,415	3,818	3,424
Retained Earnings	2,215	1,872	1,587	1,328	1,427
Dividends - percent of total income	10%	10.2%	9.9%	9.9%	11.6%
Retained Earnings - percent of total income	4%	4%	3.8%	3.8%	5.2%

*Estimated by Edison Electric Institute.

Source: Ref (8), p. 100.

rising, the percent paid out in dividends or retained as earnings has decreased from about 16 percent to about 14 percent. This factor indicates that capital is more difficult to raise than in 1973, although the situation is seen to have improved slightly since 1974. The table also indicates that, while electric revenue is increasing at a rate of about 15 percent per year, operating costs are increasing at about 18 percent per year. These data further indicate that the industry is not as well off financially as in the past and may face more problems in the future.

NERC has estimated that 20 percent of the generation needed by 1986 is not yet under construction and is subject to potential unintentional delay.[1] One of the reasons cited is the uncertain financial climate, which affects the utility segment more than general business because of high long-term capital investment requirements. This uncertain climate consequently has secondary effects on a number of other industries including equipment suppliers and large consumers due to uncertainty in construction,[17] and can thus have significant effects on the national economy.

2.3   Regulatory Characterization

As previously mentioned, the electric utility industry is a regulated industry, both economically and otherwise. However, the total regulatory process eventually effects the industry's economic state, since all costs must be either passed through or absorbed.

Future revenue requirements for U.S. electric utilities have been estimated.[18] These requirements are shown in Figure 2-5 and are reflected in the consumer's rate base.* This rate

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*The rate base is schedule of charges for electric service (usually in cents per KWH) which are reflected in the consumer's electric bill.

AVERAGE
REVENUE REQUIREMENTS
¢/KWH
(1976 DOLLARS)
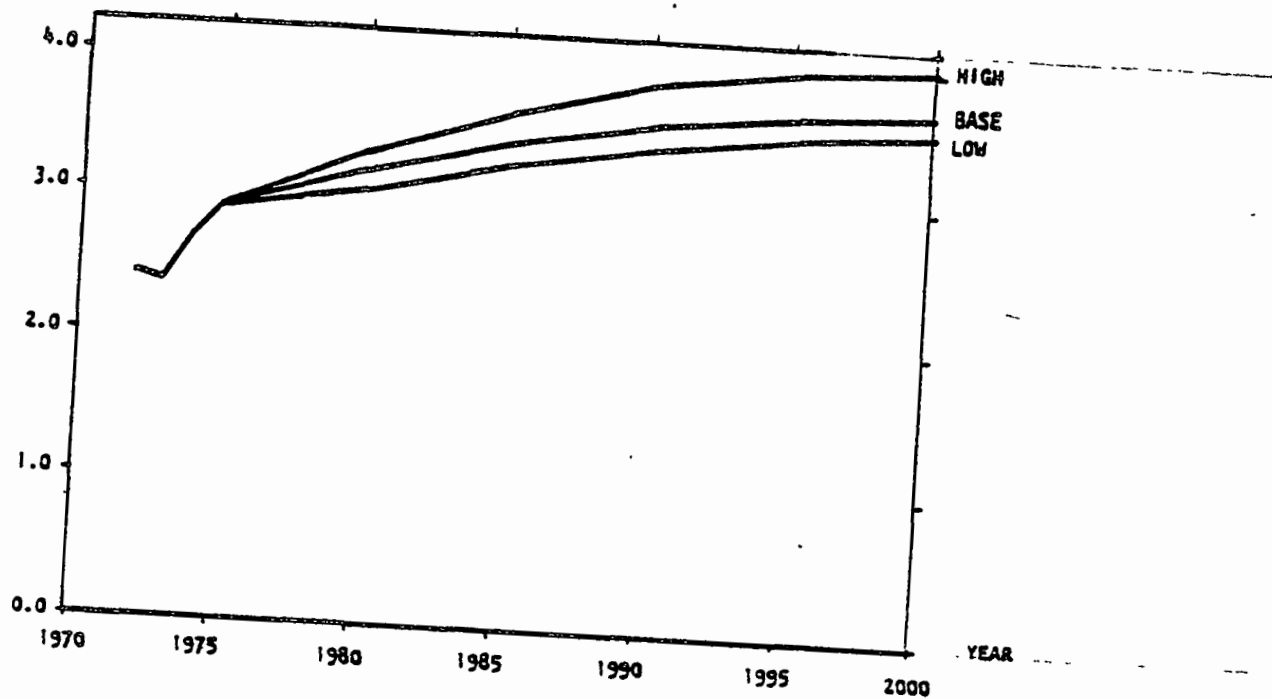
Figure 2-5
ESTIMATED ELECTRIC UTILITY
REVENUE REQUIREMENTS - 1972-2000
(Source: Ref. 11)

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-54-

base should provide a return on common equity sufficient to attract investors; the allowed rate is usually about 12 percent.[14] However, return on common equity was reported to have been 10.6 percent in 1974, 11.5 percent in 1976, and 11.8 percent in 1977 on a nationwide basis,[8] less than the rate normally allowed. Furthermore, lately some utilities have been required to pass through savings obtained through capital spending to increase efficiency rather than being allowed to earn a return in that investment.

Capital expenditures inevitably will affect rate bases, since interest and dividends must be paid to finance capital construction. Operating costs also affect rate bases as witnessed by the recent rate escalations due to rising fuel costs. Generally rate regulation agencies or commissions require some form of justification for new expenditures (either capital or operating) prior to inclusion in a rate base, and regulators may order utilities to take steps to reduce costs or to pass savings through to consumers.

The rate-setting procedure generally includes an evaluation of the company's general financial condition, operating costs, present capital investment (i.e., value of equipment in service) and, in some cases, the cost of construction work in progress (CWIP).[19] Proposed project costs are never allowed for inclusion in a rate base. In general, a certain percentage of the costs for CWIP may be allowed to be included in a rate base depending on the financial condition of the company. However, some states do not allow any inclusion of CWIP costs in rate bases; they require that equipment be installed and operating or that capital plant be built and in service before any returns on equity are allowed. This latter practice requires that the utility finance capital expenditures through the money market prior to

construction rather than through rate increases, effectively reducing actual return on equity during the construction period. Any costs incurred in either this or any other fashion are usually made up eventually through rate increases.

As was previously mentioned, electric rates must allow a return on equity sufficient to attract investors; otherwise, spiraling rate increases and reduced financial integrity are the results. In 1977, 88 investor-owned electric utility companies received rate increases amounting to $2.6 billion, and 12 companies in 10 states were ordered to cut rates by about $167 million. This compares to 1976, when 83 companies were granted increases of $2.4 million. The ratios of amounts received to amounts requested were 52 percent in 1976 and 53.8 percent in 1977. After rates are set, if the utility or consumers believe that the regulatory action was unjust, legal avenues are generally available for appeal.

In terms of plant design and construction itself, most regulatory agencies require that permits be granted by appropriate agencies before a plant may be built. In some cases, this process can require up to 50 permits from numerous agencies, with many permits requiring public hearings, legal findings, and extensive studies.[20] This permitting procedure affects the timing and cost of plant construction, especially if mistakes in studies, bad scheduling, or failure to meet agency needs occur. Careful planning and preliminary work with key agencies can serve to reduce the potential for difficulties in the licensing procedure, thus reducing overall plant cost and hastening completion. For a typical coal-fired power plant, the licensing procedure requires from 45 to 48 months in states with siting regulations and 39 to 45 months in states with no siting regulations. Site selection

generally requires an additional 12 months prior to licensing for both of the above cases. After licensing, construction requires about 42 months. For units added at existing sites, site selection updating requires from 6 to 12 months and licensing requires from 33 to 44½ months.[20]

2.4 Summary

Coal-fired plants in the United States were inventoried for the year, 1976. There were 399 plants with coal-fired capacity of 202,380 Mw, about 5.8 percent higher than widely accepted, industry-based NERC estimates. This discrepancy is probably due to differences in data and in definitions. Of these 399 plants, most are in EPA Regions III, IV, V, and VII, although significant future growth is anticipated to Region VI and other parts of the West. Total coal-fired generation in 1986 has been estimated to be 326,081 Mw, and constitutes 40.9 percent of total capacity.

In terms of economics, the electric utility industry operates as a regulated monopoly, with designated service areas. The industry is probably the most capital investment-intensive in the nation, and it has experienced difficulty in raising funds during recent years with decreasing bond ratings and price-book ratios less than 1.0 (stock selling at prices below book value). Utility regulation affects rates, plant construction, and the general economic health of the industry. Typical plant licensing times range from 33 to 49 months for coal-fired power plants, and rate bases approved between 1974 and 1977 have reflected returns on common equity ranging from 10.6 to 11.8 percent, below the 12 percent rate usually considered to be reasonable by industry and most regulators. Because of the long-term capital investment requirements of the industry, its general economic state is more sensitive to inflation and uncertainty than is that for general

business. The uncertain regulatory climate also places the
industry in a less sound position than unregulated general
industry. Reduced investor confidence has caused higher interest
rates, lower bond ratings, and most common stock sales at prices
below book value. Consequently, the economic state of the
industry is not as healthy as in the past. While the economic
situation has improved somewhat since 1974, it appears that the
recent financial problems faced by the industry are not over yet.
Furthermore, current capital costs for coal-fired plants reflected
in these present rate bases range from $450 to $800 per installed
kilowatt capacity, and are rising rapidly, thus causing further
uncertainty and making capital requirements even more important.

3.0 CHARACTERISTICS OF SOLID WASTES

With the trend towards increased utilization of the vast coal reserves across the United States, more and more attention is being focused on the chemical and physical characterization of residual by-products of coal combustion. Coal-fired power plants produce voluminous quantities of fly ash, bottom ash, and scrubber sludge each day. Full characterization of these flue gas cleaning (FGC) by-products is required if potential hazards associated with waste emission into the natural and populated environments are to be understood.

This section presents the state of the technology regarding raw wastes generated from a coal-fired power plant. The types and quantities of the principal FGC by-products will be presented for a number of typical scenarios using a "model" 1000 Mw coal-fired power plant in order to place the disposal problem in perspective. The source of combustion products, the coal, will be characterized so that the correlations between source and product can be discussed. A comprehensive description of the chemical, physical, engineering and leaching properties of FGC wastes--fly ash, bottom ash, and scrubber sludge--will be presented. Distinguishing characteristics of coals and their combustion products as they relate to their places or origin (i.e., eastern vs. western U.S. coal fields) will be emphasized.

3.1 Types and Quantities of FGC By-Products

There are a number of small waste streams or materials associated with any steam-electric generating plant. However, with coal-fired power stations there are three major by-products of coal combustion and flue gas cleaning: fly ash, bottom ash, and scrubber sludge. The relationship of these waste streams with a typical coal-fired generating station is depicted by Figure 3-1. Input materials are also identified.
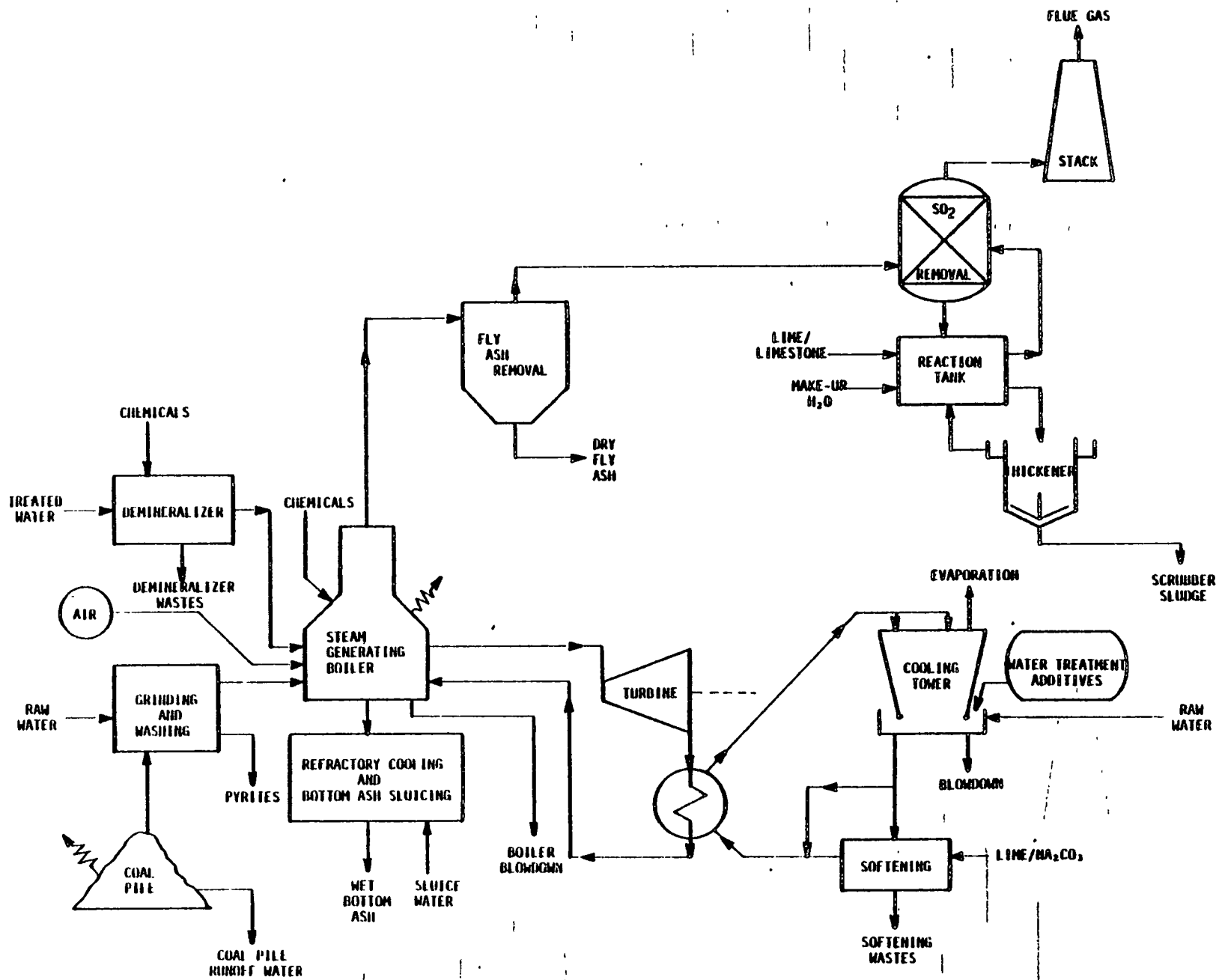
Figure 3-1.   Waste streams of a typical steam electric generating station.

02-3126-2

The three coal combustion products--fly ash, bottom ash, and scrubber sludge--pose a major solid waste disposal problem if for no other reason than the large quantity produced by a single generating plant. For perspective, the quantities of fly ash, bottom ash, and scrubber sludge produced by a "model" 1000 Mw coal-fired generating station have been calculated for a number of scenarios. Factors affecting waste quantities which have been considered are coal quality, scrubber type and operation, and both current (1978) and future emission standards (as proposed in Federal Register, September 19, 1978)[21] for SO_2 and particulates. A summary of the assumptions for the various scenarios follows.

TABLE 3-1. BASIC ASSUMPTIONS IN BY-PRODUCT SCENARIOS

	Western Sub-bituminous	Eastern Bituminous
Rated Electrical Capacity	1000 Mw	1000 Mw
Thermal Efficiency	35%	35%
Stream Factor	80%	80%
Heating Value	9000 Btu/lb	11,000 Btu/lb
Ash Content (as burned)	8%	15%
Sulfur Content	0.8%	3%
Fraction of Sulfur Emitted as SO_2	0.95%	0.95%
Fly Ash:Bottom Ash Ratio	80:20	80:20

Eight scenarios have been selected to illustrate the effects of pertinent variables on the production quantities of fly ash, bottom ash, and scrubber sludge as they leave the collection system. The design basis for these scenarios is summarized in Table 3-2. The eight scenarios are shown in Figures 3-2 through 3-9. Production rates of the fly ash, bottom ash, and scrubber sludge are summarized in Table 3-3.

TABLE 3-2. DESIGN BASIS OF EIGHT SCENARIOS FOR WASTE QUANTITIES

| Scenario | Type[1] | Coal Characteristics | | | Scrubber Parameters | | | Regulations | |
		Btu/lb	Ash[2] (%)	Sulfur (%)	Sorbent[3]	Stoichiometry[4]	Oxidation (%)	Fly Ash	SO_2
1	E	11,000	15	3.0	L	1.1	15	99% removal	1.2 lb $SO_2/10^6$ Btu Allowable[5]
2	E	11,000	15	3.0	L.S	1.5	15	"	"
3	E	11,000	15	3.0	L	1.1	15	"	85% removal[6]
4	E	11,000	15	3.0	L.S	1.1	100	"	"
5	W	9,000	8	0.8	L	1.1	15	"	1.2 lb $SO_2/10^6$ Btu[5]
6	W	9,000	8	0.8	L.S	1.5	15	"	"
7	W	9,000	8	0.8	L	1.1	15	"	85% removal[6]
8	W	9,000	8	0.8	L.S	1.1	100	"	"

[1]W = Western subbituminous; E = Eastern bituminous

[2]Residue after combustion in boiler

[3]L = Lime; L.S. = Limestone

[4]Molar ratio of lime or limestone (calcium) to mole of SO_2 removal

[5]Current 1978 New Source Performance Standards

[6]Proposed New Source Performance Standards (1978)

BASIS SCENARIO 1

Eastern Coal - 11000 BTU/lb, 15% Ash, 3% Sulfur
   15% Oxidation
   1.1 Lime stoichiometry
   99% Fly ash collection
   1.2 lb $SO_2/10^6$ BTU allowable

1.2 lbs $SO_2/10^6$ BTU

Stack

SO_2 Removal

Fly Ash Removal

Reaction Tank

Thickeners

8.6 lbs/10^6 BTU
38 dry metric tons/hr

6.8 lbs dry sludge/10^6 BTU
86 metric tons/hr @ 35% solids

Steam Generating Boiler

Coal

2.2 lbs/10^6 BTU
13 metric tons/hr @ 75% solids

Dewatered Bottom Ash
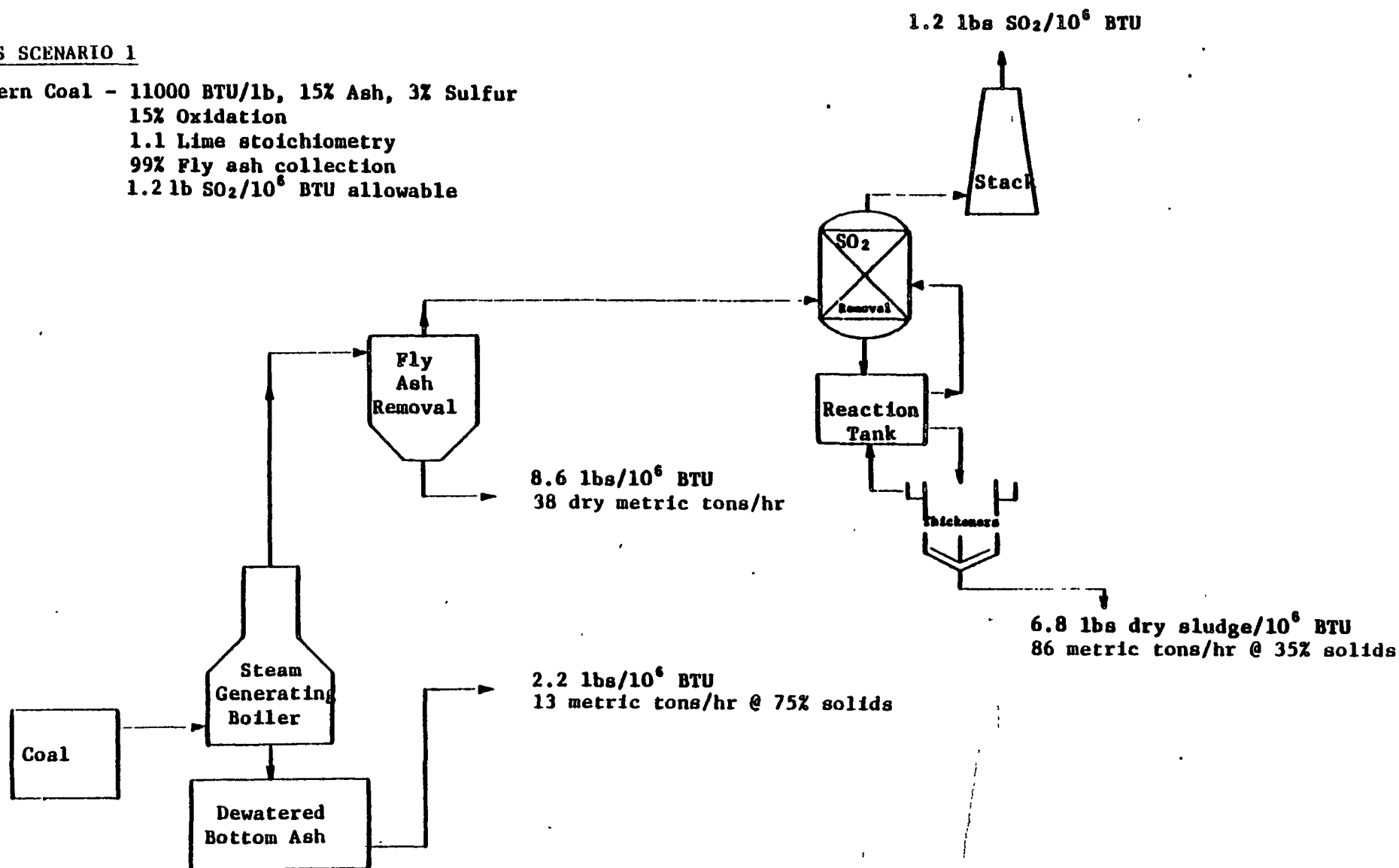
Figure 3-2. Byproduct production rates with eastern coal, low oxidation, high utilization, and current regulations.

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BASIS SCENARIO 2

Eastern Coal – 11000 BTU/lb, 15% Ash, 3% Sulfur
15% Oxidation
1.5 Limestone stoichiometry
99% Fly ash collection
1.2 lbs $SO_2/10^6$ BTU allowable

1.2 lbs $SO_2/10^6$ BTU

Stack

SO_2
Removal

Fly
Ash
Removal

Reaction
Tank

Thickeners

8.6 lbs/10^6 BTU
38 dry metric tons/hr

8.6 lbs dry sludge/10^6 BTU
110 metric tons/hr @ 35%
solids

Steam
Generating
Boiler

2.2 lbs/10^6 BTU
13 metric tons/hr @ 75% solids

Coal

Dewatered
Bottom Ash

Figure 3-3. Byproduct production rates with eastern coal, low oxidation, low utilization,
and current regulations.

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U7-3138-2
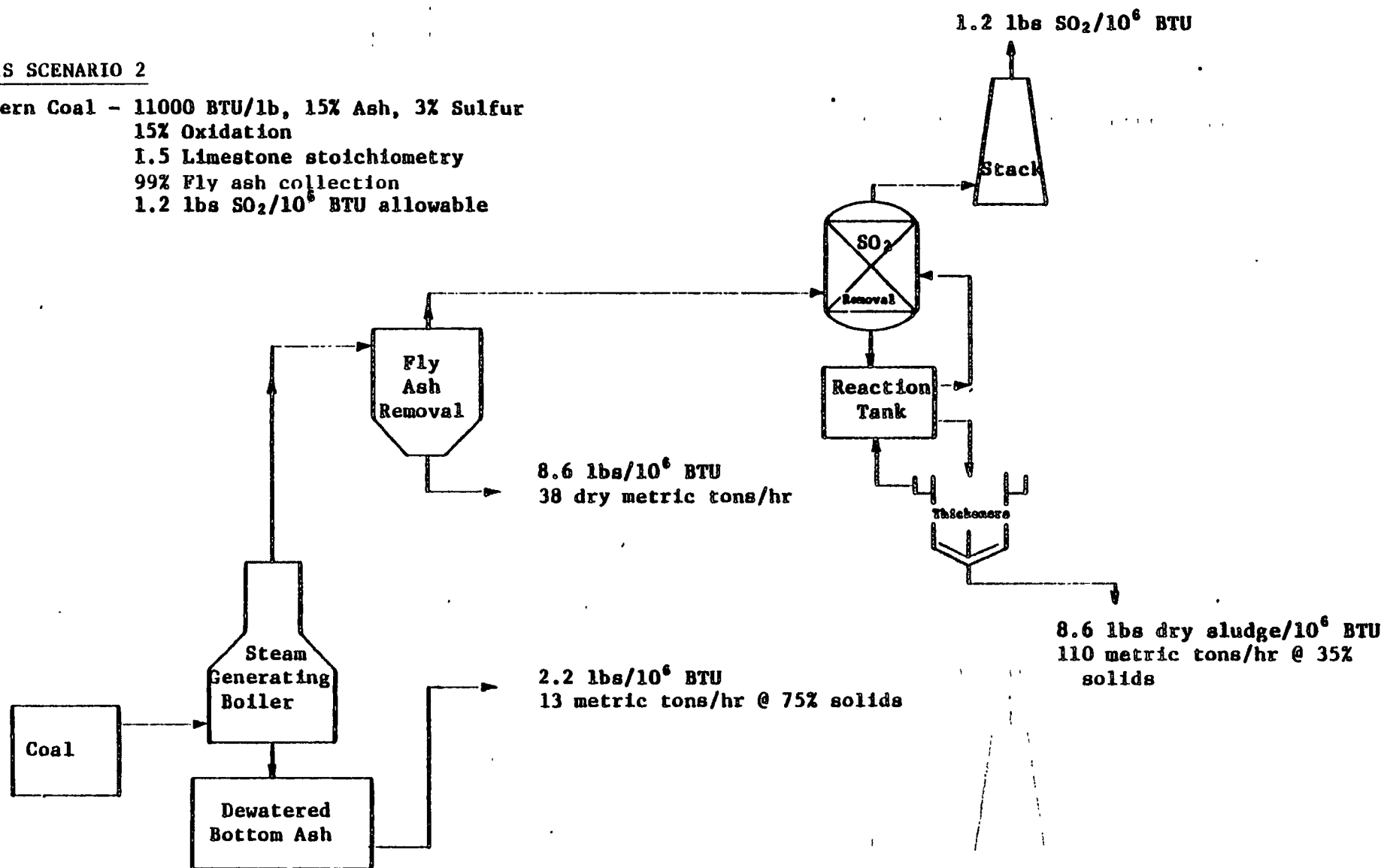
Figure 3-4. Byproduct production rates with eastern coal, low oxidation, high
utilization, and future regulations.

BASIS SCENARIO 4

Eastern Coal — 11000 BTU/lb, 15% Ash, 3% Sulfur
          100% Oxidation
          1.1 Limestone stoichiometry
          99% Fly ash collection
          85% SO$_2$ removal

.41 lbs SO$_2$/10^6 BTU

Stack

SO$_2$
Removal

Fly
Ash
Removal

Reaction
Tank

Thickener

8.6 lbs/10^6 BTU
38 dry metric tons/hr

10.1 lbs dry sludge/10^6 BTU
128 metric tons/hr @ 35% solids

Steam
Generating
Boiler

2.2 lbs/10^6 BTU
13 metric tons/hr @ 75% solids

Coal

Dewatered
Bottom Ash
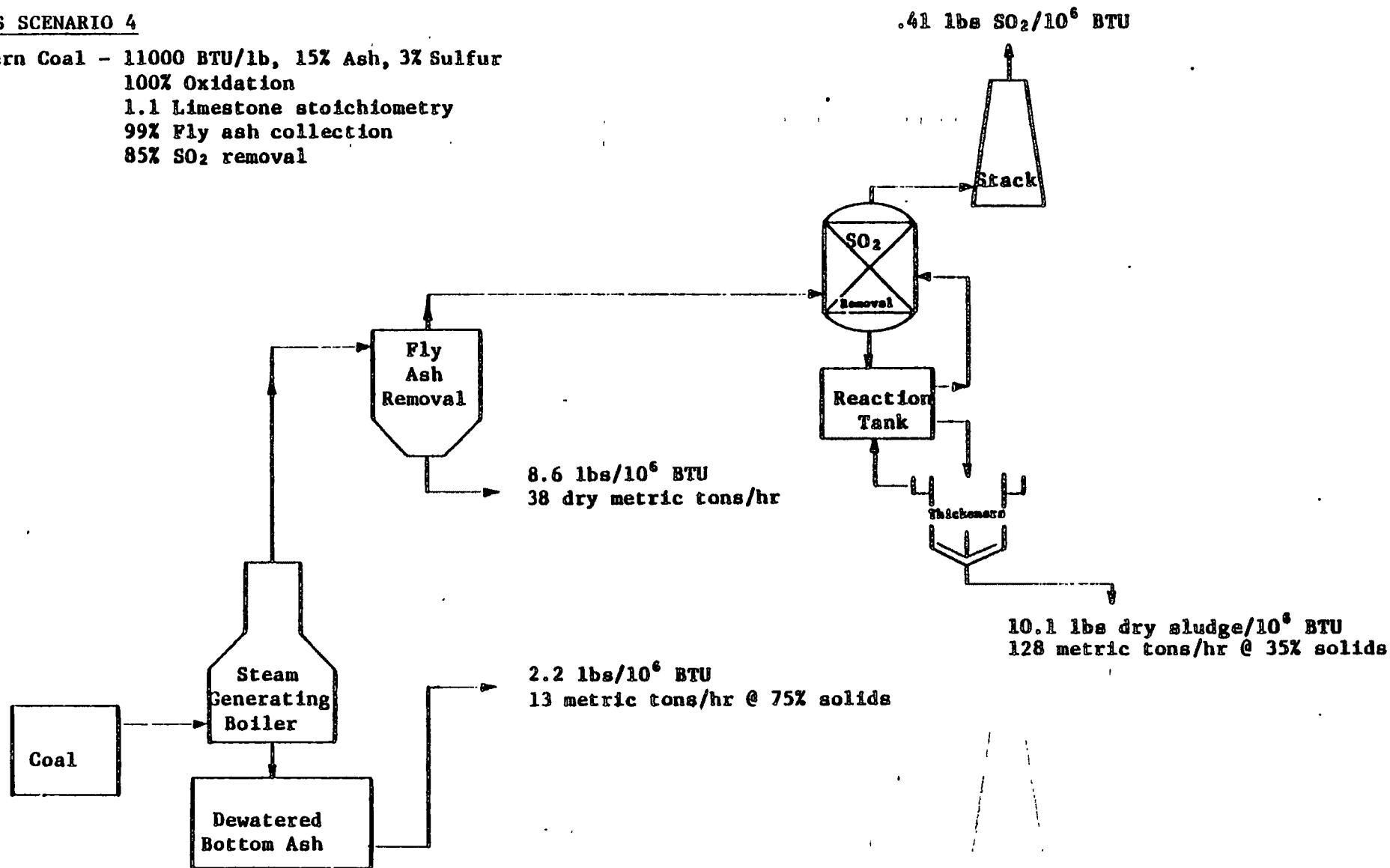
Figure 3-5.   Byproduct production rates with eastern coal, high oxidation, high
              utilization, and future regulations.

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BASIS SCENARIO 5

Western Coal – 9000 BTU/lb, 8% Ash, .8% Sulfur
           15% Oxidation
           1.1 Lime stoichiometry
           99% Fly ash collection
           1.2 lbs $SO_2/10^6$ BTU allowable

1.2 lbs $SO_2/10^6$ BTU

Stack

SO_2

Removal

Fly
Ash
Removal

Reaction
Tank

Thickener

5.6 lbs/10^6 BTU
25 dry metric tons/hr

.35 lbs dry sludge/10^6 BTU
4.5 metric tons/hr @ 35% solids

Steam
Generating
Boiler

Coal

1.4 lbs/10^6 BTU
8.4 metric tons/hr @ 75% solids

Dewatered
Bottom Ash
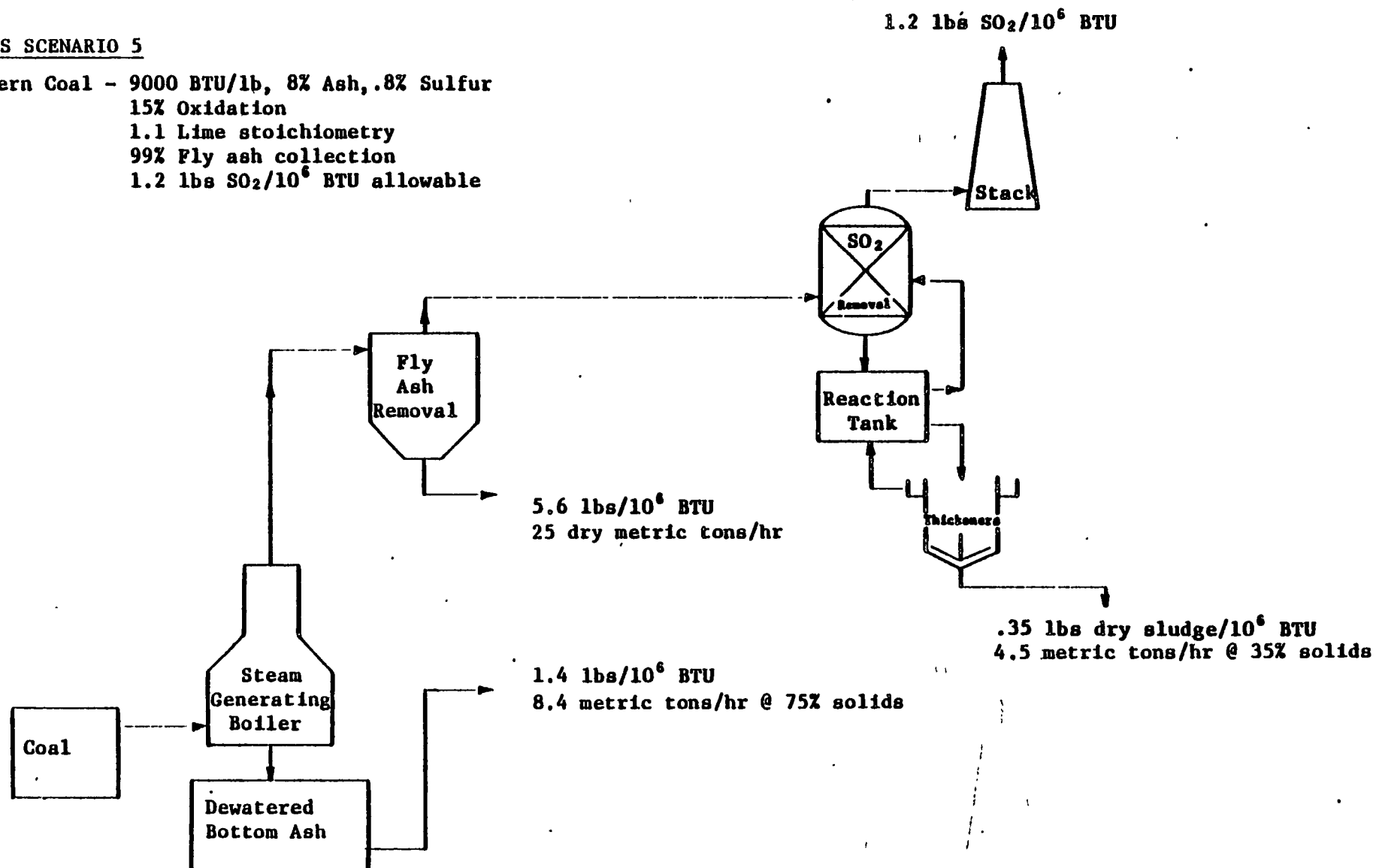
Figure 3-6. Byproduct production rates with western coal, low oxidation, high utilization,
            and current regulations.

u7-2178-7

BASIS SCENARIO 6

Western Coal - 9000 BTU/lb, 8% Ash, .8% Sulfur
15% Oxidation
1.5 Limestone stoichiometry
99% Fly ash collection
1.2 lbs $SO_2/10^6$ BTU allowable

1.2 lbs $SO_2/10^6$ BTU

Stack

SO_2 Removal

Fly Ash Removal

5.6 lbs/10^6 BTU
25 dry metric tons/hr

Reaction Tank

Thickener

.45 lbs dry sludge/10^6 BTU
5.7 metric tons/hr @ 35% solids

Steam Generating Boiler

1.4 lbs/10^6 BTU
8.4 metric tons/hr @ 75% solids

Coal

Dewatered Bottom Ash
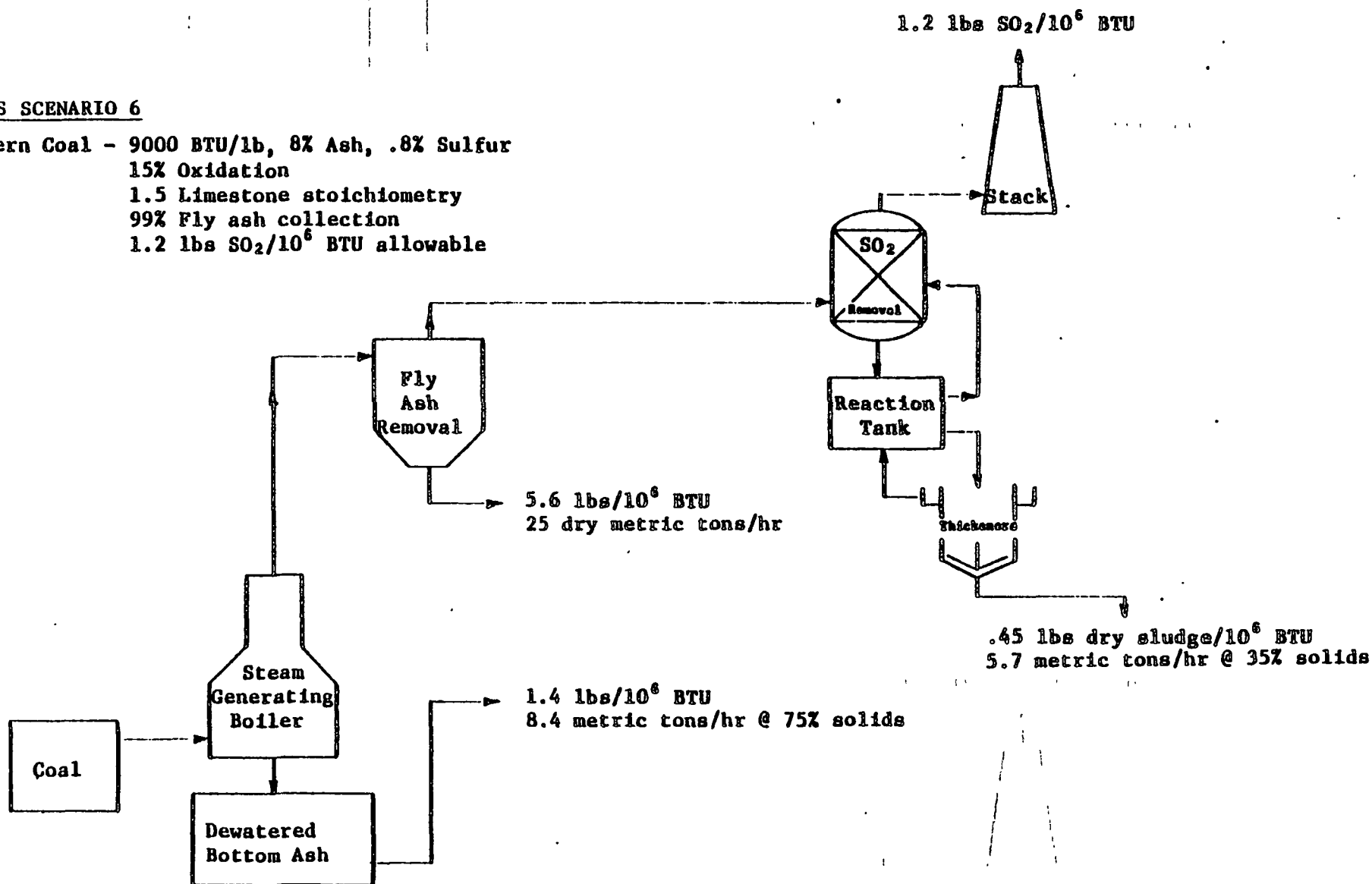
Figure 3-7. Byproduct production rates with western coal, low oxidation, low utilization, and current regulations.

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Western Coal - 9000 BTU/lb, 8% Ash, .8% Sulfur
   15% Oxidation
   1.1 Lime stoichiometry
   99% Fly ash collection
   85% SO_2 Removal

.2 lbs $SO_2/10^6$ BTU

Stack

SO_2
Removal

Fly
Ash
Removal

5.6 lbs/10^6 BTU
25 dry metric tons/hr

Reaction
Tank

Thickeners

2.6 lbs dry sludge/10^6 BTU
34 metric tons/hr @ 35% solids

Steam
Generating
Boiler

1.4 lbs/10^6 BTU
8.4 metric tons/hr @ 75% solids

Coal

Dewatered
Bottom Ash

Figure 3-8.   Byproduct production rates with western coal, low oxidation, high utilization,
              and future regulations.

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u7-3176-7

BASIS SCENARIO 8

Western Coal - 9000 BTU/lb, 8% Ash, .8% Sulfur
          100% Oxidation
          1.1 Limestone stoichiometry
          99% Fly ash collection
          85% SO_2 Removal

.2 lbs $SO_2/10^6$ BTU

Stack

SO_2 Removal

Fly Ash Removal

Reaction Tank

Thickener

5.6 lbs/10^6 BTU
25 dry metric tons/hr

3.3 lbs dry sludge/10^6 BTU
42 metric tons/hr @ 35% solids

Steam Generating Boiler

1.4 lbs/10^6 BTU
8.4 metric tons/hr @ 75% solids

Coal

Dewatered Bottom Ash
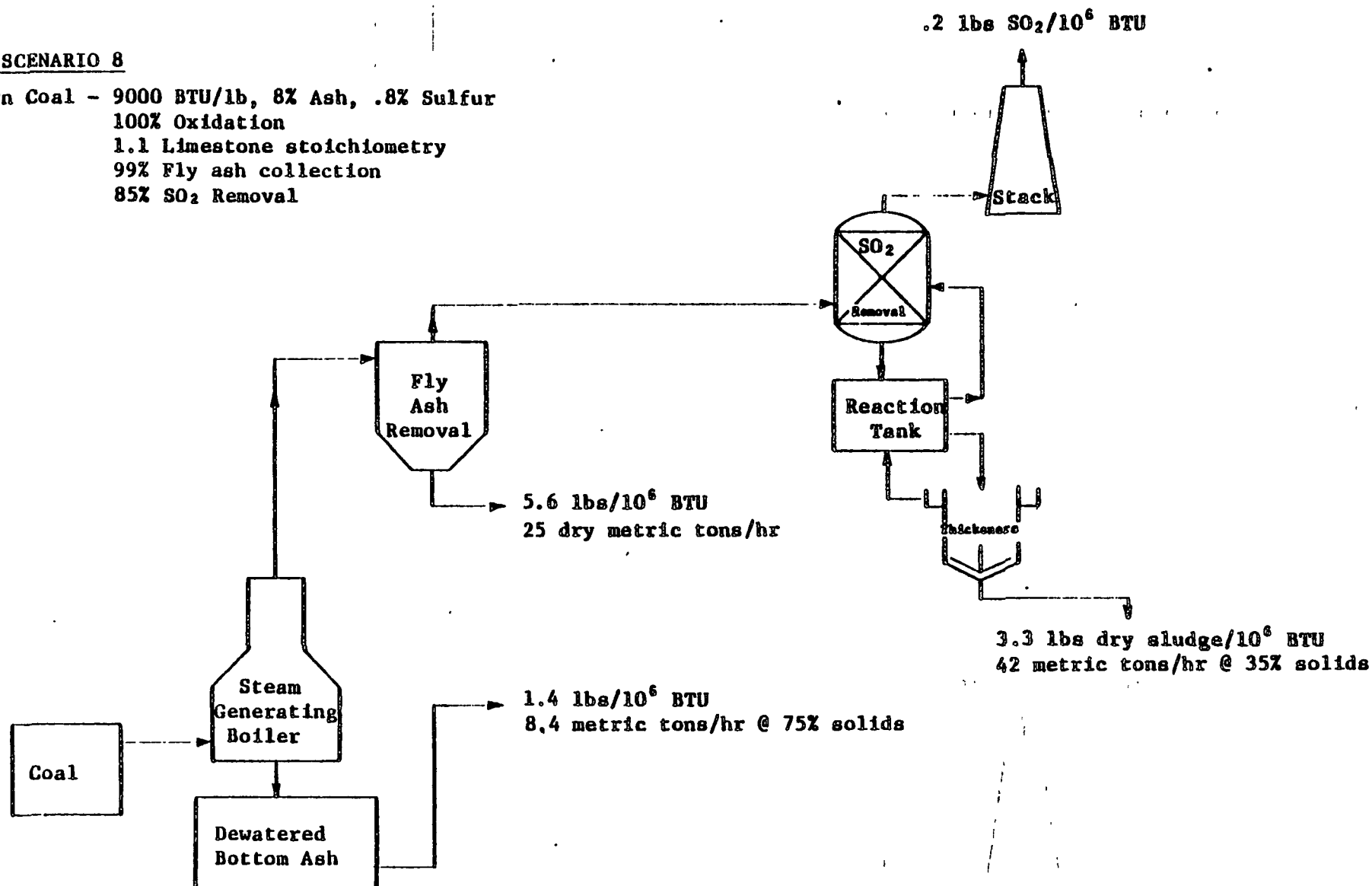
Figure 3-9. Byproduct production rates with western coal, high oxidation, high utilization, and future regulations.

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TABLE 3-3. BY-PRODUCT PRODUCTION RATES FOR EIGHT SCENARIOS

Scenario	Coal Type	Oxidation	Utilization	Regulations	Fly Ash lbs/10⁶ Btu (dry metric tons/hr)	Bottom Ash lbs/10⁶ Btu (metric tons/hr @ 75% solids)	Scrubber Sludge dry lbs/10⁶ Btu (metric tons/hr @ 35% solids)	Stack Emissions	
								lbs Particulates/ 10⁶ Btu	lbs SO₂/ 10⁶ Btu
1	E	Low	High	Current	8.6 (38)	2.2 (13)	6.8 (86)	.086	1.2
2	E	Low	Low	Current	8.6 (38)	2.2 (13)	8.6 (110)	.086	1.2
3	E	Low	High	Future	8.6 (38)	2.2 (13)	8.1 (103)	.086	0.41
4	E	High	High	Future	8.6 (38)	2.2 (13)	10.9 (128)	.086	0.41
5	W	Low	High	Current	5.6 (25)	1.4 (8.4)	0.35 (4.5)	.056	1.2
6	W	Low	Low	Current	5.6 (25)	1.4 (8.4)	0.45 (5.7)	.056	1.2
7	W	Low	High	Future	5.6 (25)	1.4 (8.4)	2.6 (34)	.056	0.2
8	W	High	High	Future	5.6 (25)	1.4 (8.4)	3.3 (42)	.056	0.2

Dry collection of the fly ash is assumed either by electrostatic precipitation or baghouses. A wet bottom boiler is assumed for the collection of bottom ash. Data presented in Table 3-3 indicate that over 90% of fly ash is collected dry and over 80% of bottom ash is collected wet. SO_2 is typically removed in a wet limestone or lime scrubber, precipitated as calcium sulfite or sulfate salts, and then thickened with a clarifier to 35% solids. Simultaneous collection of fly ash in a wet scrubber has not been depicted but can easily be calculated by taking the production rate in dry tons and dividing it by .35 (35% solids) and then adding it to the production rate of wet SO_2 scrubber sludge.

Current federal New Source Performance Standards[22] (NSPS, 1978) require at least 99% reduction in particulate loading and a maximum allowable SO_2 emission of 1.2 lbs of SO_2 per 10^6 Btu. Proposed regulations require an 85% reduction in the flue gas SO_2 concentration.[21] Future regulations for reducing particulate emissions have not been considered as they will not significantly impact waste quantities of ash.

Lime and limestone wet SO_2 scrubbers have been considered because they represent the principal technologies being used by industry to control SO_2 emission. A purity of 98% has been selected for the lime and limestone.

For limestone as the sorbing reagent, a stoichiometric ratio of 1.5 moles of limestone to 1.0 mole of SO_2 sorbed has been chosen as representative of current technology with 1.0 stoichiometry for future reference when improvements in scrubbing technology allow a reduction in the excess of sorbent. A stoichiometric ratio of 1:1 has been assumed for lime as currently feasible in 1978. The effect of 15% oxidation has been estimated for both current and future performance standards (proposed in September

19, 1978 Federal Register).[21] Forced oxidation has been proposed as a trend in future scrubber operation.

The following conclusions have been drawn from an examination of the eight scenarios (Figures 3-2 through 3-9) which depicted the effects of coal type, scrubber type and operation, and NSPS regulations.

- The firing of eastern coal in a 1000 Mw generating station results in significantly larger quantities of both ash and scrubber sludge.

- Under current regulations (1978) for maximum permissible emissions, eastern coal results in 2000% more scrubber sludge and 50% more ash than western coal.

- Future regulation (proposed in September 19, 1978 Federal Register) will not significantly affect ash production rates from either eastern or western coal or with the scrubber sludge generated by the firing of eastern coal.

- Assuming a required 85% removel in future SO_2 emission regulations (proposed in September 19, 1978 Federal Register), seven times as much scrubber sludge will be produced by a 1000 Mw western coal-fired power plant than under current regulations of 1.2 lb $SO_2/10^6$ Btu. An 85% SO_2 removal regulation will not significantly increase the scrubber sludge rates produced from a unit firing eastern coal.

- Forced oxidation will increase the mass of scrubber sludge by 20-30%. This increase can be offset to some degree by improved dewatering and settling properties.

- Stoichiometries of 1.5 for either lime or limestone will result in approximately 20-30% higher scrubber sludge masses than with 1.1 stoichiometries.

3.2   Characterization of Coal

Coal is classified into "grades" on the basis of its fixed carbon, percent volatility and heat content, and its agglom-erating characteristics (ASTM Part 26, 1977).[23] The caloric content, i.e., its value as a fuel (Btu/lb), is a good general indication of its classification. Coal grades range in descending value of Btu content from bituminous to subbituminous to lignite, with certain categorial variations within.

Coal grade variability is highly correlative with the geographic location and geologic structure of a coal basin. These factors, together with the nature of the coalification process and timely environmental conditions, are what contribute to the chemical and physical properties of any given coal type. Such properties include:

- heat content (Btu value),
- moisture content,
- ash content,
- sulfur content,
- distribution of major and minor constituents, and
- trace element concentrations.

During the combustion of a coal, the combustion waste products are fractionated into three separate parts:

- fly ash - the ash fraction entrained with the flue gas,
- bottom ash - collected wet or dry at furnace bottom, and
- volatiles - either scrubbed or emitted to the atmosphere.

The chemical and physical properties of these different combination products and their proportionate ratios vary with respect to the coal type being fired. For example, eastern coals are characteristically higher in sulfur content as compared to western coals. This feature of a higher percent sulfur is particularly noteworthy because of the higher sulfur dioxide content in the flue gas emissions following combustion of an eastern coal. The characteristic differences in the waste products resulting from burning eastern vs. western coal will be discussed in the following sections.

The inorganic constituents of the coal are comprised of minerals derived from the surrounding lithology and incorporated into the coal seam. These inorganic constituents can be divided into three classes based on concentration expressed in the oxide form:

- Major - >5%,
- Minor - .1% → 5%, and
- Trace - <.1%.

The major elements found in the inorganic phase of all coals include silicon, aluminum, and iron. Calcium and, in some cases, sodium and magnesium occur in western coals as major inorganic constituents. Compounds of these major elements typically account for 90% of the total ash content. The concentration and distribution of these major species will have a major effect on the reactivity of the fly ash and, to some extent, on the distribution of coal sulfur between the flue gas and ash.

Minor inorganic constituents found in all coals include potassium, titanium, phosphorous, magnesium, sodium, and, in some cases, barium, strontium, manganese, and boron. Trace element concentrations, <.1% content, are highly variable among the different coal types.

3.3　Chemical Properties of FGC By-Products

Three types of chemical properties will be discussed. Major elements form the chemical compounds which are responsible for the chemical, physical, structural, and leaching properties of the flue gas cleaning (FGC) products--fly ash, bottom ash and scrubber sludge. Trace element concentrations are of environmental concern with respect to leaching and possible ground water or surface water contamination. Alkalinity and calcium content are of particular importance in the differentiation of western and eastern coal fly ash and its effect on the stabilization of scrubber sludge.

3.3.1　Chemical Characterization of Ash

The elements forming the major and minor chemical constituents of fly ash are summarized in Table 3-4 according to coal rank. Analyses of ashes have shown that the fly ash and bottom ash fractions contain a similar distribution of most of the major elements. The iron content has been reported to be noticeably higher in bottom ashes than in its respective fly ash. The sulfate (SO_3) levels were reported to be lower in bottom ash; calcium concentrations were very similar although the leachable calcium levels were higher with fly ash. This is in part due to the fused condition of the bottom ash particles.

The crystalline materials most often identified in fly ash by X-ray diffraction include silica (α-quartz), β-calcium sulfate (anhydrite), iron oxides (hematite, α-Fe_2O_3, and magnetite), and aluminum silicate (mullite). Sometimes calcium oxide (lime) and/or magnesium oxide (periclase) are present.

There can be major differences between the fly ash and the bottom ash in the quantities of certain trace elements they

TABLE 3-4. CHEMICAL COMPOSITION OF FLY ASHES ACCORDING
TO COAL RANK - MAJOR AND MINOR SPECIES
(WEIGHT PERCENT)

Chemical Species	Eastern Bituminous			Western Sub-bituminous			Western Lignite		
	Range	Median	Total No. of Observations	Range	Median	Total No. of Observations	Range	Median	Total No. of Observations
Sodium Oxide, Na₂O	0.05-2.04	0.53	21	0.15-2.14	1.04	8	0.60-8.10	3.45	8
Potassium Oxide, K₂O	0.92-4.00	2.53	20	0.50-1.80	0.99	8	0.20-1.02	0.50	8
Magnesium Oxide, MgO	0.50-5.50	1.24	23	1.10-5.90	2.96	12	3.3-12.75	6.79	10
Calcium Oxide, CaO	0.26-13.15	2.88	21	1.80-30.40	13.81	12	11.7-35.44	22.29	10
Silicon Dioxide, SiO₂	36.00-57.00	48.76	22	31.00-64.80	49.69	9	2.20-46.1	30.69	8
Aluminum Oxide, Al₂O₃	16.25-30.30	23.26	22	18.70-37.00	23.04	12	10.7-25.3	15.48	10
Iron Oxide, Fe₂O₃	3.88-35.40	16.44	23	3.07-21.50	6.48	12	2.9-14.15	8.87	10
Titanium Dioxide, TiO₂	1.00-2.50	1.45	19	0.68-1.66	1.09	11	0.52-1.60	0.74	8
Phosphorous Pentoxide, P₂O₅	<0.02-0.42	2.73	16	0.19-0.70	0.38	6	<0.02-0.76	0.25	5
Sulfur Trioxide, SO₃	0.09-3.30	0.78	17	0.10-5.23	1.66	12	0.32-7.20	3.14	8

Sources: 24, 26, 27, 28, 29, 30, 31, 32, 33, 34, 35, 36.

contain. For example, selenium concentration is generally an order of magnitude higher in fly ash than in bottom ash. Arsenic, boron, and fluorine are more concentrated in the fly ash than the bottom ash by a factor of two or more due to the volatility of these elements.

In recent years, several independent studies[24,25] have been made investigating the trace element composition of coal and its combustion products. Although the studies included sampling of systems varying drastically in size and design, the systems analyzed are generally consistent regarding trace elements, some of which are hazardous at high concentrations. Certain of these trace elements are preferentially concentrated in or on the fine fly ash particulates.[24]

arsenic	selenium	sulfur
cadmium	copper	zinc
lead	molybdenum	antimony
beryllium	gallium	

Other elements which tend to be emitted to the atmosphere as vapors are:

mercury	bromine
chlorine	sulfur

All other elements tend to be concentrated in approximately equal proportions in the fly ash and bottom ash residues.

The loss of weight on ignition is often taken to be a measure of unburned carbon in the fly ash. A higher loss occurs in the fine fraction. The typical range for loss of ignition of a fly ash is from <.05 to 7 percent.

Moisture content of fly ash is low and is usually less than one percent for fly ash collected dry.

3.3.2 Eastern vs Western Coal Fly Ashes

The grade of coal which is the major fuel source in the eastern half of the United States is bituminous coal of medium to high volatility. Western coal is mostly subbituminous and lignite. On this basis, generalizations can be made to distinguish between eastern and western coal fly ashes. Examples of the differences between eastern and western coal ashes are shown in Table 3-4.

Two of the most important differences between eastern and western coal fly ashes are the calcium concentration and available alkalinity. Western coal fly ashes in general have higher calcium concentrations and available alkalinity. These parameters have been correlated with the capability of western coal ashes to stabilize scrubber sludge without the addition of other fixation additives.[26]

3.3.3 Chemical Characterization of Scrubber Solids

The principal components of limestone or lime scrubber solids are the calcium sulfite coprecipitate, gypsum and calcium carbonate. The concentration of the species will be dependent upon the SO_2 and oxygen concentration in the flue gas prior to scrubbing, the type of scrubber system and the operation of the scrubber.

With low oxidizing conditions, calcium sulfite hemihydrate is the principal solid phase product. Small quantities of calcium sulfate will be coprecipitated with the sulfite. The calcium sulfite-sulfate coprecipitate has the formula,[37]

$$Ca \left\{ (1-x)\ SO_3 + (x)\ SO_4 \right\} \cdot \tfrac{1}{2} H\,O$$
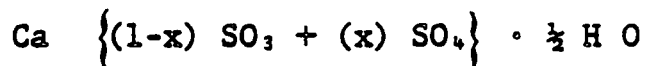
where:

$$0 \leq x \leq .16$$

Gypsum, i.e., calcium sulfate dihydrate will be formed when the natural oxidation rate is high or when forced oxidation is utilized. Calcium carbonate from either unreacted limestone or from carbonation of the lime (CaO) will be found in varying degrees. Small concentrations of other species are also found in the solid phase.

Trace element concentrations in the sludge will be dependent to a large extent on (1) the quality of the sorbent and the make-up water, and (2) the trace element content of the coal. Reported values are summarized in Table 3-5 as a function of coal type.

3.4   Physical Properties of FGC By-Products

The physical properties of importance with respect to disposal are:

- particle morphology
- particle size distribution
- specific gravity
- solids content

These parameters will have direct bearing on compaction, settling rate, and dewaterability and an indirect bearing on the permeability.

TABLE 3-5. TRACE ELEMENT COMPOSITION OF FGD SLUDGE SOLIDS (CONCENTRATION IN ppm)

Identification: Coal Type: Sorbent:	Plant A Western Low-Sulfur Lime	Plant B Western Low-Sulfur Limestone	Plant C Eastern High-Sulfur Limestone	Plant D Eastern High-Sulfur Limestone	Plant E Eastern High-Sulfur Limestone	Plant F Eastern High-Sulfur Lime
Antimony	4.3		7.5	6.7		
Arsenic	4.0		12.	6.7		
Barium	500		4400	<20		
Beryllium	1.5	23.7	1.0	1.8	3.9	10.4
Boron	68.7		211.	41.8		
Cadmium	0.40	17.7	1.1	25	12.9	2.8
Chromium	1.6	131.9	4.0	5.2	90.6	98.2
Copper	38.9	239.8	104.	65	169.7	87.3
Fluorine	1017		950.	266		
Germanium	<1.0		2.4	5.9		
Lead	1.6	287.8	2.4	190	155.3	54.1
Manganese	56.	595.7	147.	340	251.8	159.2
Mercury	<0.010		0.46	0.101		
Molybdenum	81.	140.5	8.0	9.6	147.1	23.6
Nickel	13.	202.4	26.0	75.2	215.3	137.4
Selenium	4.13		3.8	2.1		
Vanadium	<50		<100	<100		
Zinc	13.9	333.7	169	2050	330.8	166.1
Cobalt		46.7			4.4	35.8
COMMENTS:	Electrostatic precipitator for fly ash control upstream of scrubber		Electrostatic precipitator for fly ash control upstream of scrubber	Scrubber used for both particulate and SO2 control		

Sources: 38, 39.

3.4.1   Physical Properties of Fly Ash

The particle size distribution of a fly ash will depend upon chemical composition, boiler type, and firing conditions. The size distribution of the collected fly ash will also be affected by the method of collection. Ashes collected by electrostatic precipitation and baghouses tend to have a higher percentage of fine particles than those collected mechanically.

Fly ash particles are very fine, varying in size from ~ 0.5 to 100 microns and averaging from 8 to 30 microns. The morphological characteristics of the fly ash are diverse and are particle size-dependent. One recent study has identified eleven morphological particle types in the fly ash produced by a low sulfur, high ash, high moisture western coal.[40] The finest fraction is composed primarily of unopaque solid spheres. The coarsest fraction is composed of cenospheres (lightweight hollow spheres) 20 to 100 microns in diameter. These cenospheres are caused by evolution of nitrogen and carbon dioxide inside fused silicate particles in the boiler. Because of their low density, these "floaters" become a suspended solids problem in disposal ponds.

Specific gravities of representative U.S. fly ashes ranged from 1.97 to 2.85.[26] One of the primary factors correlated with specific gravity is the iron oxide content.[41,42] Minnich also indicated that particle size distribution was important in addition to the iron oxide content.[41] The specific gravity, i.e., density, of the particle will have a bearing on the bulk density of the compacted or settled by-product and consequently will affect disposal volumes.

3.4.2   Physical Properties of Bottom Ash

Bottom ash is collected in either ash or slag form depending on the design of the boiler. The ash produced in a dry bottom boiler is comprised of coarse, angular particles with porous surfaces. Wet bottom boilers produce heavy, angular particles which are termed slag. These typically black particles have a glass-like appearance.

Bottom ash in comparison to fly ash is much coarser, ranging from 50 microns to one inch in diameter. This is equivalent to the range bounded by fine gravel to fine sand.

The specific gravity of 6 eastern bottom ashes has been reported to range from 2.28 to 2.78 with the iron oxide content an important factor.[41] The range for bottom ashes in the U.S. can be expected to be broader, ~2 to 3.

3.4.3   Physical Properties of FGD Sludge

Particle morphology, particle size distribution, specific gravity, and solids content are particularly affected by the scrubber system design and operation and the chemical composition of the solids.

The coprecipitate of calcium sulfite hemihydrate formed under normal nonnucleating conditions will appear as platelets .5 to 2 microns in thickness and from 2 to 40 microns in length and width. The ratio of length to width will be approximately 2:1. Under nucleating conditions, e.g., rapid dissolution of lime, the calcium sulfite crystals may randomly nucleate on the surfaces of existing sulfite crystals, leading to a dendritic

3.5 Structural Properties

Compaction and settling properties give an indication of the disposal volume involved in disposal in landfills and ponds, respectively. Triaxial shear tests and unconfined compressive strength tests are indicative of the load bearing capacity of the disposal site. The load bearing capacity, in turn, indicates the structural stability of the solid waste material during and after disposal. Permeability coefficients are closely related to these structural properties and can directly affect the quantity of leachate from the disposal site.

3.5.1 Compaction Properties

Compaction tests are performed to determine the optimum water content for placing the material at maximum density. Placing of the landfilled material at the optimum moisture and maximum density will insure minimum settlement and permeability and maximum shear strength. It also reduces the land requirements for disposal. Optimum moisture contents and maximum unit dry densities, i.e., the compaction properties, of fly ash, bottom ash, scrubber sludges, and mixtures of fly ash and scrubber sludge have been reported by a number of investigators.[26,27,41,43,45,46] Compaction properties of fly ashes, bottom ashes, and scrubber sludges as determined by these investigators are summarized in Table 3-6. The addition of fly ash to the scrubber sludges at 1:1 ash to sludge ratio or higher reduced the optimum moisture content of the sludge by 5 to 20 percentage points and increased the compacted density of 80 to 320 kg/m³ (5-20 lb/ft³). Using these results, representative moistures and densities have been selected in calculating disposal volumes of the various disposal options discussed in Section 4. The moistures and densities for these options are presented in Table 3-7.

TABLE 3-7. ASSUMPTIONS OF DRY AND WET BULK DENSITIES OF FGC WASTE PRODUCTS

Product	Specific Gravity	Dry Bulk Density @				Wet Bulk Density @			
		Optimum	70% Solids	65%	50%	Optimum	70% Solids	65%	50%
		(lbs dry solids/ft³)				(lbs total/ft³)			
Fly Ash	2.55	95	76	67	45	112	108	103	89
Bottom Ash	2.55	90	76	67	45	110	108	103	89
Scrubber Sludge	2.55	80	76	67	45	104	108	103	89
Fly Ash + Scrubber Sludge	2.55	95	76	67	45	112	108	103	89
		(kilograms of dry solids/m³)				(kilograms of wet solids/m³)			
Fly Ash	2.55	1520	1220	1070	720	1790	1730	1650	1430
Bottom Ash	2.55	1440	1220	1070	720	1760	1730	1650	1430
Scrubber Sludge	2.55	1280	1220	1070	720	1670	1730	1650	1430
Fly Ash + Scrubber Sludge	2.55	1520	1220	1070	720	1790	1730	1650	1430

TABLE 3-6.   RANGE OF MOISTURE-DENSITY RELATIONS
OF FGC WASTE PRODUCTS

Product	Optimum Moisture Content*	Compacted Dry Density of Dry Solid kg/m³	lbs/ft³
Fly Ash	16-31%	1140-1650	71-103
Bottom Ash	14-25%	1170-1870	73-117
Sulfite Rich Sludge	31-52%	1040-1350	65-84
Sulfate Rich Sludge	14-32%	1310-1550	82-97

*Optimum Moisture Content - grams of water per gram of dry solid

3.5.2   Unconfined Compressive Strength

The load bearing capacity of the disposal material can
be determined by unconfined compressive strength measurements of
cohesive products and triaxial shear tests of cohesiveless mater-
ials. The load bearing capacity in pounds per square inch (psi)
is a measure of the structural stability of the disposal material.
The use of the disposal site after reclamation will depend in part
on the structural stability. That is, can it be used as an indus-
trial building site or must it be a park or undeveloped site. The
load bearing capacity also is a rough indication whether the mater-
ial can be worked as a landfill with typical earth-moving equipment
or must it be contained in a pond. For rough comparative purposes,
a person walking on the disposal material exerts approximately .35
kg/cm² (5 psi), a car, approximately 2.1 kg/cm² (30 psi), and a truck
or other heavy equipment, approximately 4.2 kg/cm² (60 psi). A
material having low or negligible unconfined compressive strength
is a cohesiveless material.

In general, scrubber sludges behave as cohesiveless materials, having unconfined compressive strengths less than 1.4 kg/ cm^2 (20 psi).[45] Some ashes and ash/sludge mixtures, particularly those eastern coal ashes containing little calcium or available alkalinity, display very little cohesive properties with unconfined compression strengths reported at less than 4.2 kg/cm^2 (60 psi).[26]

Unconfined compressive strength tests are applicable to those products displaying cohesive properties. Typically, such hydrated products are western coal fly ashes, eastern coal fly ashes to which lime has been added, mixtures of scrubber sludges with these ashes, and scrubber sludge stabilized with other commercially available techniques.

Unconfined compressive strengths of ash/sludge mixtures vary considerably but have been reported in the literature from less than 7 kg/cm^2 (100 psi) to as high as 350 kg/cm^2 (5000 psi).[26] The latter strength is equivalent to typical ready-mix Portland cement concrete. The calcium content and available alkalinity of the ash, type of sludge, the ash to sludge ratio, the amount of lime addition, the water content, and curing time were all factors instrumental to this wide range of unconfined compressive strengths.[26,27]

3.5.3   Permeability

The permeability coefficient of the waste product will depend upon

- particle morphology,
- particle size distribution,

- porosity or void ratio of in-situ material, and
- unconfined compressive strength, i.e., the reduction of pore volume which accompanies the formation of the hydration products.

The particle size distribution of bottom ash is similar to that of sand and fine gravel. Settled or compacted bottom ash has a high porosity (high void ratio) and displays no cohesive properties. It has the high permeability that one would expect. Permeability coefficients of 10^{-1} to 10^{-3} cm/sec are typically reported.[41] Permeabilities of compacted fly ash are significantly lower, ranging from 10^{-4} cm/sec to 10^{-10} cm/sec.[26] The higher value (10^{-4} cm/sec) is more typical of uncompacted eastern coal fly ashes and the lower permeabilities are representative of compacted hydrated alkaline western coal ashes.

Permeabilities of 10^{-3} to 10^{-4} cm/sec are reported as a typical range for settled and drained FGD sludges. By applying compaction and more extensive dewatering techniques, permeabilities of 10^{-4} to 10^{-5} and in some cases less than 10^{-5} cm/sec can be achieved. Fly ash addition decreases the permeability of a sludge even further by filling in interstitial spaces of the FGD sludge with fine fly ash particles. This limits the flow of liquids through the media and decreases the permeability to as low as 10^{-6} cm/sec.

Western coal ash addition to sludge or the addition of lime and eastern coal ash to the sludge can lower the porosity and permeability in yet another way. Hydration reactions consume the pore water replacing it with the solid phase hydration productions responsible for increased unconfined compressive strength. Permeability coefficients as low as 10^{-10} cm/sec have been observed. Reduction of one to two orders of magnitude are more typical.

3.6 Leaching Characteristics

The leaching behavior of coal ash and sludge is considered for several reasons. First, collected fly ash constitutes a major utility flue gas cleaning waste. Secondly, with the implementation of the Clean Air Act Amendments of 1977, increased quantities of SO_2 scrubber sludge will be produced. Thirdly, when combined ash and sludge disposal operations are practiced, certain leachate parameters are generally dominated by the ash constituents, while others are more strongly influenced by sludge components. The concentrations of many trace elements in ash-sludge liquor are derived primarily from the ash, while others are derived from the scrubber sludge. In addition, concentrations of major species and total dissolved solids will be governed by the scrubber solids and adherent scrubber liquor.

The disposal of these FGD wastes may result in a potential source of ground water and surface water contamination if improperly managed. The concentration of the major species in the leachate as well as the trace element content may pose a problem. In lieu of being able to predict the concentration of chemical species in the leachate based on the chemical properties of ash and sludge, available field data will be evaluated and tabulated for FGC by-products from eastern and western coals. Additional research is necessary for predicting leaching quality from a knowledge of chemical characteristics of the coal, ash and/or sludge, and from laboratory leaching data.

One of the major problems associated with the evalua-
tion of leaching data is the accuracy of the analytical results
particularly with respect to the trace elements determination.
Chemical characterizations of the same sample by different labora-
tories produce significantly different results with respect to
key trace elements. Errors in the results within a given labora-
tory can be from several sources, some of which are contamination
of sample or standards, analytical interference from the other
species present in the sample, sorption of the trace elements
into or onto the walls of the sample storage container, desorp-
tion of trace elements from the wall. These errors or uncertain-
ties in the analytical results can and do in many cases preclude
the meaningful assessment of the laboratory or field leaching
studies.

The following subsection summarizes readily available
data concerning leaching data from coal ashes and FGD sludges.
In many cases analytical methods and error analyses were not docu-
mented adequately.

3.6.1   Coal Ash Leachates

The concentration range and median values of major spe-
cies measured in studies of field ash leachates have been summar-
ized in Table 3-8. The sources of information for all leachate
tables are summarized at the bottom of the table. The data re-
ported in Table 3-8 have been organized, where possible, into
eastern coal ash leachates and western coal ash leachates. In-
terim drinking water standards and recommended water quality para-
meters for irrigation water are summarized in Table 3-9.

The concentrations of major species including calcium,
magnesium, sodium, potassium, and sulfate are higher in western
ash leachates than eastern ash leachates. The only exception

TABLE 3-8. CONCENTRATION OF MAJOR CHEMICAL SPECIES IN
COAL ASH LIQUORS* (CONCENTRATION IN mg/ℓ)

Species	Eastern Coal Ash				Western Coal Ash			
	Min.**	Avg.	Max.	Total No. of Observations	Min.**	Avg.	Max.	Total No. of Observations
Calcium (Ca^{+2})	8	183	856	25	39	452	1200	5
Magnesium (Mg^{+2})	.4	72	402	25	.09	26	66	5
Sodium (Na^{++})	.8	27	90	5	22	28	33	2
Potassium ($K^{+}1$)	.4	32	112	5	4.2	5.4	6.5	2
Iron (Fe^{+2})	.05	12	200	20	.07	2.9	10	4
Aluminum (Al^{+})	.05	4.6	43	20	1.1	6.0	14	4
Chloride (Cl^{-1})	.4	11	55	19	12	18	28	4
Sulfate (SO_4^{-2})	12	374	2000	24	61	318	807	4
Carbonate (CO_3^{-2})		27		1		89		1
Nitrate (NO_3^{-1})						15		1
Phosphate (PO_4^{-3})						.74		1
Silica (SiO_3^{-2})	1	18	53	5	40	59	79	2
TDS	142		3800	20		664		2
pH	2.9		11.5	21	7.7		12.5	2

*Includes analyses of fly ash, bottom ash, and combined fly ash/bottom ash liquors.

**Low values are mostly from bottom ash samples.

SOURCES: 24, 26, 46, 47, 48, 49, 50.

TABLE 3-9.   FEDERAL WATER QUALITY PARAMETERS

NATIONAL INTERIM PRIMARY DRINKING WATER STANDARDS[51]

Arsenic	0.05	mg/ℓ
Barium	1.0	mg/ℓ
Cadmium	0.01	mg/ℓ
Chromium	0.05	mg/ℓ
Lead	0.05	mg/ℓ
Mercury	0.002	mg/ℓ
Nitrate (as N)	10.0	mg/ℓ
Selenium	0.01	mg/ℓ
Silver	0.05	mg/ℓ
Fluoride (avg)	2.4	mg/ℓ[2]

PROPOSED NATIONAL SECONDARY DRINKING WATER REGULATIONS[52]

Chloride	250	mg/ℓ
Copper	1	mg/ℓ
Iron	0.3	mg/ℓ
Manganese	0.05	mg/ℓ
pH	6.5 - 8.5	
Sulfate	250	mg/ℓ
TDS	500	mg/ℓ
Zinc	5	mg/ℓ

LIMITS FOR IRRIGATION USE OF CONTINUOUSLY-USED WATER[53]

Aluminum	5.0	mg/ℓ
Arsenic	1.0	mg/ℓ
Beryllium	0.10	mg/ℓ
Boron	0.75	mg/ℓ
Cadmium	0.01	mg/ℓ
Chromium	0.10	mg/ℓ
Cobalt	0.05	mg/ℓ
Copper	0.2	mg/ℓ
Fluoride	1.0	mg/ℓ
Iron	5.0	mg/ℓ
Lead	5.0	mg/ℓ
Lithium	2.5	mg/ℓ
Manganese	0.2	mg/ℓ
Molybdenum	0.01	mg/ℓ
Nickel	0.2	mg/ℓ
Selenium	0.02	mg/ℓ
Vanadium	0.10	mg/ℓ
Zinc	2.0	mg/ℓ
TDS	5000	mg/ℓ
pH	4.5 - 9.0	

was an eastern ash leachate with an extremely low pH, 2.9-3.7.
In view of the inertness and low surface area of the bottom ash
relative to that of the fly ash, the lower end of each range is
more characteristic of the leachability of the bottom ash while
the median to higher end is more characteristic of fly ash leachate.

The data base on trace elements in coal ash leachates
is continuing to grow but, like trace element analysis or ash it-
self, the data remain sketchy in several respects. The concen-
tration ranges and median of selected trace elements are summar-
ized in Table 3-10 for western and eastern coal ash leachates.

The average concentration of inorganic chemical species
as measured in leaching samples collected from ash ponds and/or
landfills exceeded drinking water or irrigation water parameters
by a factor of ten. The concentration must be less than 10 times
the primary drinking water standards for the waste material to be
indicated as non-hazardous under RCRA. The following elements
had average concentrations which were near to the drinking and/or
irrigation water quality criteria presented in Table 3-9.

Arsenic	Fluorine	Molybdenum
Boron	Manganese	Selenium
Chromium	Mercury	

These species will bear monitoring. Zinc, cadmium, manganese,
and lead were near to the RCRA (10X) criteria in two eastern ash
disposal sites. These high concentrations were probably the re-
sult of the low pH (<3) of the leachate.

3.6.2   Leaching of FGD Scrubber Sludge

The quality and quantity of the leachate from the FGD
disposal material is expected to be the worst during the initial

TABLE 3-10. CONCENTRATION OF TRACE ELEMENT SPECIES
IN COAL ASH LIQUORS* (CONCENTRATION IN
mg/ℓ)

	Eastern Coal Ash Leachate				Western Coal Ash Leachate			
Species	Min.	Avg.	Max	Total No. of Observations	Min.	Avg.	Max.	Total No. of Observations
Antimony	0.003	0.014	0.033	7	0.002	0.018	0.04	6
Arsenic	<0.005	0.026	0.1	21	0.0001	0.003	.009	7
Barium	0.1	0.224	0.3	21	0.1	7.03	40	6
Beryllium	0.00026	0.007	0.01	21	0.00004	0.003	0.01	6
Boron	0.048	2.77	16.9	7	0.03	1.25	3.2	6
Cadmium	0.0005	0.003	0.037	21	0.0001	0.002	0.005	6
Chromium	<0.0006	0.076	1.0	21	0.0006	0.048	0.111	6
Cobalt					0.0003	0.005	0.011	5
Copper	<0.004	0.411	6.75	25	0.004	0.025	0.08	6
Cyanide	<0.01	<0.01	0.01	14				
Fluorine	0.35	3.33	17.3	7	0.2	7.0	18.7	6
Iron				7	0.01	2.3	4.6	2
Germanium	<0.01	0.026	<.1	7	<0.01	0.115	0.22	2
Lead	0.0027	0.035	0.3	24	0.0068	0.05	0.25	6
Manganese	<0.002	0.269	2.15	24	0.001	0.308	0.9	6
Mercury	0.0002	0.0045	0.038	21	0.0004	0.02	0.08	6
Molybdenum	0.004	0.123	0.69	7	0.002	0.028	0.056	6
Nickel	0.01	0.099	1.10	21	0.001	0.036	<0.05	6
Selenium	<0.0005	0.0336	0.47	21	0.001	0.033	0.12	6
Silver	<0.01	<0.01	0.01	14	<0.0003	0.002	0.01	5
Uranium					0.002	0.024	0.1	5
Vanadium	<0.1	0.143	<0.2	7	<0.005	0.099	0.23	6
Zinc	<0.005	2.1	5.1	25	0.01	0.124	0.4	6

* Includes analyses of fly ash, bottom ash and combined fly ash/bottom ash liquors.

Sources: 38,48,. 54.

stage of disposal and will be primarily a function of the quality of the scrubber liquor. The water present in the disposal site will be the adherent scrubber liquor which will be high in total dissolved solids. The total dissolved solids will increase the solubility of the solids due to increased ionic strength and ionic complexation. As the wastes age and leach, the chemical quality of the leachates will improve and will be governed by the dissolution of the solid phases as opposed to the quality of the adherent scrubber liquor.

The quality of the scrubber liquor will be significantly affected by the scrubber system design and operating parameters, the characteristics of system input meterials, and the nature of operation of the particulate control device.

The composition of the scrubber liquor will be significantly affected by the liquor residence time which is affected by the evaporation loss in the scrubber and the solids content of the sludge leaving the scrubber system. The major source of chloride in the scrubber liquor is the coal. Magnesium sources include the lime or limestone sorbent and in some cases, magnesium addition to control scaling. The quality of the make-up water will also be important in view of the large cycles of concentration, as high as 50 in some cases. Trace elements in the scrubber liquor are contributed by the coal, additives, and make-up water.

Concentrations of major and trace element species in FGD scrubber liquor and elutriates for lime and limestone systems are summarized in Table 3-11 for both eastern and western coals. The concentrations of chloride, sulfate, and total dissolved solids are consistently higher than the irrigation standards. Scrubber liquor cannot be used as irrigation water without some kind of treatment.

TABLE 3-11. LEVELS OF CHEMICAL SPECIES IN FGE SLUDGE LIQUORS AND ELUTRIATES[a]

	Eastern Coals			Western Coals		
	Range in Liquor (ppm)	Median (ppm)	Total No. of Observations	Range in Liquor (ppm)	Median (ppm)	Total No. of Observations
Aluminum	0.24-0.6	--[d]	2	2.1	--[d]	1
Antimony	0.46-1.6	--[d]	4	0.09-0.22	--[d]	2
Arsenic	<0.004-2.2	0.13	22	<0.004-0.2	0.016	10
Beryllium	<0.0005-0.05	0.013	23	0.0006-0.14	0.012	10
Boron	2-41	--[d]	2	8.0	--[d]	1
Cadmium	0.003-0.3	0.047	17	0.011-0.064	0.034	10
Calcium	470-2,600	960	22	240(-45,000)[c]	700	9
Chromium	0.001-2.8	0.16	21	0.024-0.4	0.082	10
Cobalt	<0.002-0.1	.035	3	0.1-0.17	--[d]	2
Copper	0.002-1.5	0.086	21	0.002-0.6	0.20	10
Iron	0.02-0.1	--[d]	5	0.42-8.1	--[d]	2
Magnesium	420-2750	--[d]	2	9-390	--[d]	2
Lead	0.002-0.35	0.12	24	0.0016-0.37	0.031	10
Manganese	<0.01-9.0	0.17	8	0.007-2.5	0.74	6
Mercury	0.0004-0.07	0.016	17	<0.01-0.07	<0.058	10
Molybdenum	5.3	--	1	0.91	--[d]	1
Nickel	0.03-0.91	0.13	11	0.005-1.5	0.09	6
Potassium	22-41	--[d]	3	8-100	--[d]	2
Selenium	<0.005-2.7	.01	19	<0.001-2.2	0.12	10
Sodium	22-20,000[b]	127.	10	320-29,000[b]	--[d]	5
Zinc	0.01-27	0.046	17	0.028-0.88	0.16	9
Chloride	420-5,000	2350	12	760-43,000[c]	--[d]	4
Fluoride	1.4-70	3.6	15	0.7-3.1	1.7	5
Sulfite	4.3-110	--[d]	3.	5,870	--[d]	1
Sulfate	720-30,000[b]	2280	20	2,100-18,500[b]	3,700	11
TDS	2,500-70,000	7,000	—	5,000-95,000[c]	12,000	—
pH	7.1-12.8	—	—	2.8-10.2	—	—

[a] Sample sources range from liquors directly from a wet scrubber to pond liquors of varying ages.

[b] Levels of soluble sodium salts in dual alkali sludge (filter cake) depend strongly on the degree of cake wash. The highest levels shown reflect single measurements on an unwashed dual alkali filter cake.

[c] Levels of soluble chloride components in sludges are dependent upon the chloride-to-sulfur ratio in the coal. The highest levels shown are single measurements for a western limestone scrubbing system operating in a closed-loop using cooling tower blowdown for process makeup water.

[d] Sufficient data points are not available for calculation of meaningful "median."

Sources: 44, 45, 48, 49, 50.

The major species present in the leachate after the leaching of the adherent scrubber liquor result from the dissolution of the gypsum, limestone, and calcium sulfite hemihydrate in the scrubber solids. For this reason, the concentration of the major species in the leachate will be limited by the solubility of these crystalline species. Experimental studies of scrubber sludge containing some gypsum have shown that the calcium concentration will be from 600 to 700 mg/liter and the sulfate concentration will be 1500 to 1600 mg/liter. Total dissolved solids will typically range from 2200 to 2600 mg/liter at steady state.[26] These steady-state concentrations will be unavoidable since they are dictated by the solubility of gypsum. Concentrations of calcium and sulfate can be lower than these values when working with a calcium sulfite sludge containing little or no gypsum. Here, concentrations will be limited by both the solubility of the calcium sulfite hemihydrate and oxidation.

Trace element concentrations are highly variable as evidenced by the wide range for most species. Here again, no chemical species exceeded RCRA criteria for toxicity. The concentration of arsenic, boron, cadmium, chromium, fluoride, manganese, mercury, and selenium do warrant monitoring in that the average concentration exceeded the drinking water or irrigation water quality parameters.

4.0 WASTE DISPOSAL PRACTICES

The various disposal methods currently employed by coal-fired utility plants are addressed in this section. The disposal methods commonly used are discussed followed by presentation of survey data establishing the number of existing and planned facilities either using or planning to use each type of disposal practice. The costs associated with the disposal methods are established and used to provide a rough cost estimate for implementation of RCRA. Alternative disposal practices, methods not in common used today, are also evaluated from technical, environmental and economic standpoints.

4.1 Disposal Methods and Practices

The common disposal practices currently in use in the coal-fired electric utility industry are described in the following pages. Survey data describing the number of plants using each type of method is also presented. The data describing the percentage of plants using each type of disposal method were used in Section 4.2 to define the costs of current and future disposal.

4.1.1 Disposal Methods

The various alternatives commonly used for fly ash and scrubber sludge collection and disposal are diagrammed in Figures 4-1, 4-2, and 4-3. These diagrams focus on fly ash and scrubber sludge to the exclusion of bottom ash because of the greater quantities of these materials generated. The alternatives for bottom ash disposal are more limited and less complicated in terms of disposal alternatives. Ultimate disposal of any of these wastes is in a pond or landfill, either lined or unlined.

```
                    ┌─────────────┐
                    │   Fly Ash   │
                    └──────┬──────┘
          ┌────────────────┴────────────────────┐
┌─────────────────┐                    ┌─────────────────┐
│ Dry Collection  │                    │ Wet Collection  │
│    (ESP or      │                    │   (Scrubber     │
│   Mechanical)   │                    │    System)      │
└────────┬────────┘                    └────────┬────────┘
   ┌─────┼──────────┐                      ┌─────┴──────┐
┌────────┐ ┌────────┐ ┌─────────────┐  ┌────────────┐ ┌────────┐
│  Ash   │ │ Dry Ash│ │ Dry Ash Mixed│  │Combined Fly│ │Fly Ash │
│Sluiced │ │Removal │ │With Scrubber │  │Ash & Scrubber│ Slurry │
│to Pond │ │        │ │   Sludge     │  │Sludge Sluiced│ to Pond│
└────────┘ └───┬────┘ └─────────────┘  │  to Pond   │ └────────┘
      ┌────────┼────────┐              └────────────┘
┌──────────┐┌─────────┐┌──────────────┐
│Hauled to ││Hauled Off││Ash Given Away│
│Landfill by││Site--Paid││to Agency or │
│ Company  ││Disposal ││Group at No   │
└──────────┘└─────────┘│Cost to Company│
                       └──────────────┘
```
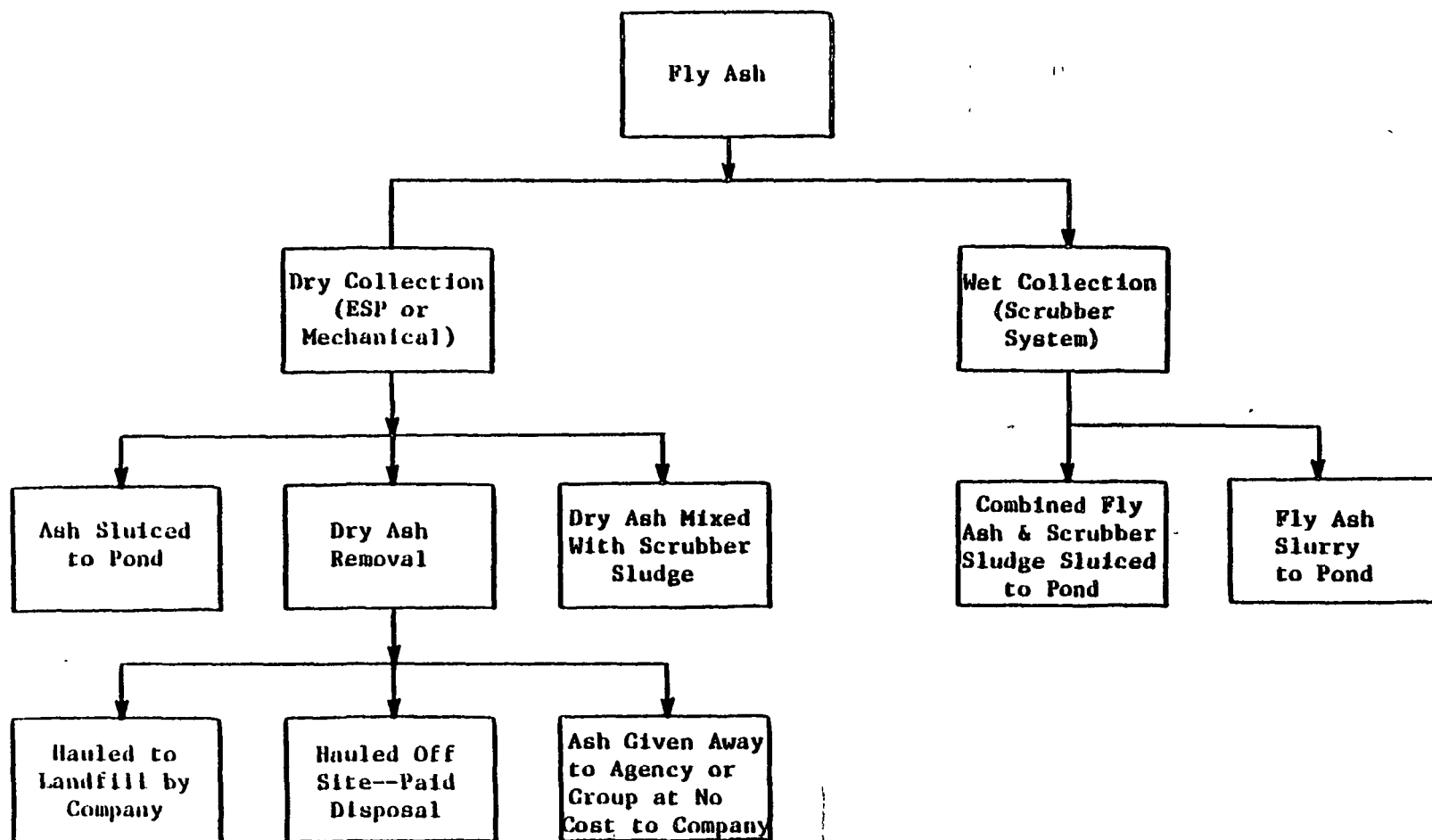
FIGURE 4-1

COMMON DISPOSAL ALTERNATIVES FOR FLY ASH

(SEPARATE COLLECTION OF FLY ASH AND SO_2)
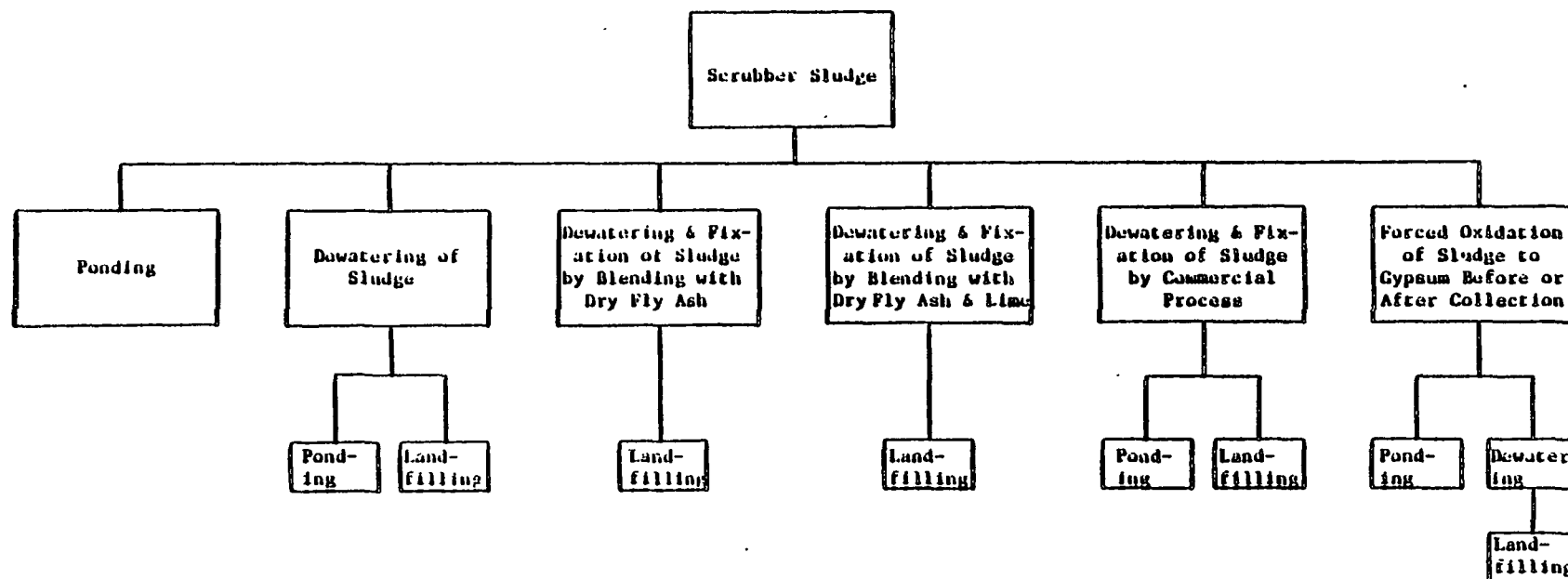
FIGURE 4-2

COMMON DISPOSAL ALTERNATIVES FOR FGD SCRUBBER SLUDGE

```
                    ┌─────────────────┐
                    │   Fly Ash and   │
                    │ Scrubber Sludge │
                    │    Combined     │
                    └─────────────────┘
                             │
        ┌────────────────────┼────────────────────────┐
        │                    │                         │
┌───────────────┐   ┌─────────────────┐     ┌──────────────────┐
│  Sluicing of  │   │   Dewatering    │     │      Forced      │
│    Mixture    │   │       and       │     │  Oxidation of    │
│   to Pond     │   │    Fixation     │     │   FGD Sludge     │
└───────────────┘   └─────────────────┘     │   to Gypsum      │
                             │              └──────────────────┘
                    ┌────────┴────────┐          │
                    │                 │     ┌────┴──────┐
              ┌─────────┐  ┌──────────┐  ┌────────┐ ┌──────────┐
              │  Pond-  │  │  Land-   │  │ Pond-  │ │ Dewater- │
              │  ing    │  │ filling  │  │ ing    │ │ ing      │
              └─────────┘  └──────────┘  └────────┘ └──────────┘
                                                         │
                                                   ┌──────────┐
                                                   │  Land-   │
                                                   │ filling  │
                                                   └──────────┘
```
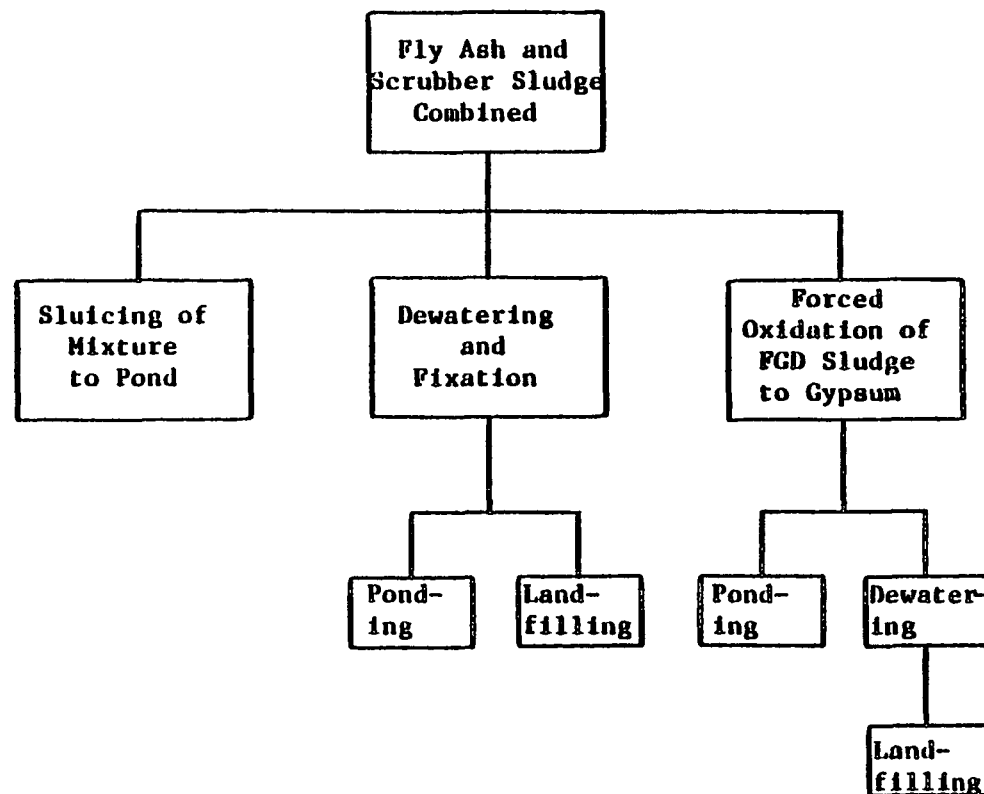
FIGURE 4-3.

COMMON DISPOSAL ALTERNATIVES FOR
COMBINED FLY ASH AND FGD SCRUBBER SLUDGE

4.1.1.1 Alternatives for Collection and Disposal

Collection and disposal of fly ash is commonly performed in either a wet or dry mode as shown in Figure 4-1. While the ash material may be collected dry, as with an electrostatic precipitator or a baghouse, wet sluicing may then be employed to convey the ash to a disposal pond. Where wet sluicing and ponding are not employed, the material is usually hauled by truck to landfill disposal. If the ash is collected with a wet system, as with a wet venturi, disposal is usually accomplished by subsequent pumping to a pond in slurry form. The choice of pumping a water mixture of the ash or transporting dry is often site specific and highly dependent upon the method of collection.

In addition to direct disposal of fly ash, it is often advantageous to mix fly ash and scrubber sludge together to take advantage of the increased stabilization and fixation resulting from the mixture. This material can be used as a very effective pond liner, as discussed in the section on chemical and physical properties.

In dry form, the ash is either hauled to a landfill by the utility company, or hauled offsite by a company paid to dispose of the material, or even given away or sold to a company using the material in some commercial process. Most of the dry material leaving the plant ends up in landfills. The landfill sites are rarely lined.

In the case of ponding of wet sluiced fly ash or combined fly ash and scrubber sludge, the ponds can be lined or unlined. Common liners include clay and synthetic liners. A stabilized mixture of fly ash and scrubber sludge can also serve as an effective pond liner.

Figure 4-2 illustrates various disposal options for
scrubber sludge. Basically, the options involve direct ponding
or dewatering. In either case, the sludge can be mixed with
(1) fly ash or fly ash and lime or (2) a number of other materials
used for commercial fixation processes. In the case of a pre-
dominantly sulfite sludge, material can also be fully oxidized
to gypsum before ponding or dewatering. If dewatering is not
used, the sludges are commonly pumped at 10-15 percent solids.
Sludges can undergo partial dewatering and pumped to ponds at
20-35 percent solids. The sludges can be vacuum filtered to 50-60
percent solids and trucked or otherwise hauled to the landfill.

The options for disposal of combined fly ash and scrubber
sludge, such as that generated from a combined particulate and SO_2
scrubber, are presented in Figure 4-3. The options for disposal
are basically the same as those presented in Figure 4-2. The
combined sludge can be pumped directly to a pond, either fully
oxidized to gypsum or as a sulfite sludge. The sludge can be de-
watered and fixed and, depending on the extent of dewatering,
either ponded or landfilled.

In every case where ponding is used, either for dewatered
sludges or the slurry pumped directly to ponds, the solids can be
removed and landfilled after settling. Common designs for ponds
and landfills are presented in the following pages.

4.1.1.2   Pond Designs

The common types of designs for ponds are the diked pond,
the incised pond, and the side hill pond. Figure 4-4 gives con-
figurations for the three types of pond designs. The side hill
pond shown in the figure is advantageous in an area of hilly
terrain where a level area for constructing a diked pond or
incised pond is not readily available. The advantage of the side
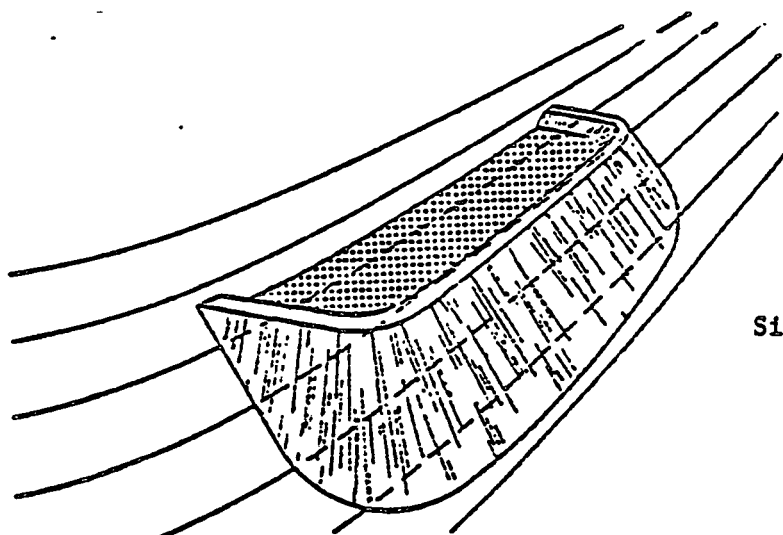
Side Hill Pond
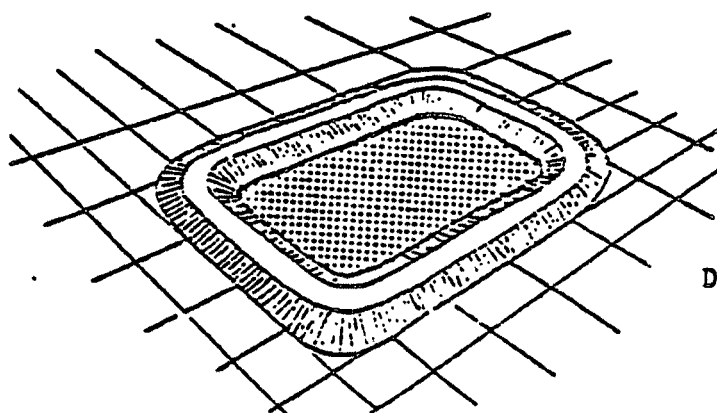
Diked Pond
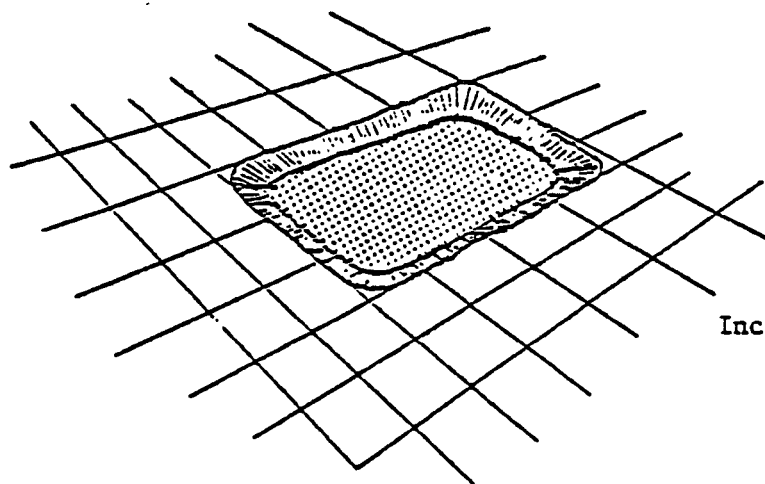
Incised Pond

FIGURE 4-4

POND DESIGNS

hill pond is that it uses the natural slope of the terrain to provide one or two sides of the pond. If the angle of the sloping terrain is too great, a large pond must be constructed utilizing the contour of the hill or the bank on the downhill side must be high.

Both the diked pond and incised pond are constructed on a relatively level site. The incised pond is such that the special volume of the pond is located entirely below the surrounding ground level. Since excavation is necessary for the incised pond, it is advantageous when bedrock is not encountered or where groundwater will not cause drainage problems. The incised pond does not require space for dike construction or a suitable material for construction and the incised pond may be an advantage where such factors are important. The diked pond is the most common type encountered. This type may be constructed above grade utilizing extraneous dike material or it may be constructed below ground level utilizing excavated material for full or partial dike construction.

Not shown in Figure 4-4, but sometimes used, is ponding in an existing basin such as an abandoned strip mine or a quarry.

4.1.1.3 Landfill Designs

The most common landfill designs are shown in Figure 4-5. The figure shows the side hill landfill. As in the case of the side hill pond, this type of landfill is advantageous and most often utilized in areas of hilly terrain where the natural slope of one side of a hill or valley may provide containment. The side hill landfill must be properly prepared to insure stability.

The structurally simplest form of landfill which may be utilized with level terrain is the heaped landfill. Even though this design is simplest in terms of site preparation and offers
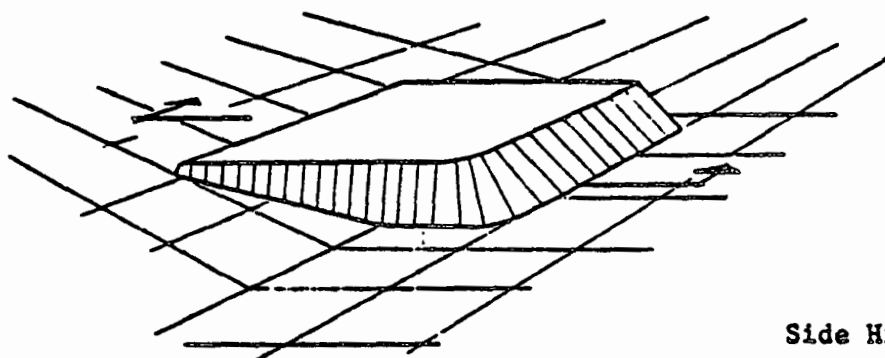
Side Hill Landfill
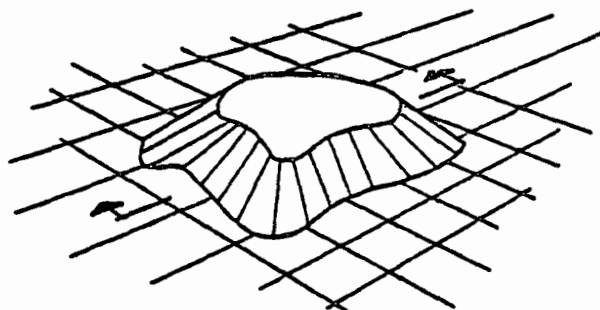
Heaped Landfill
Configuration
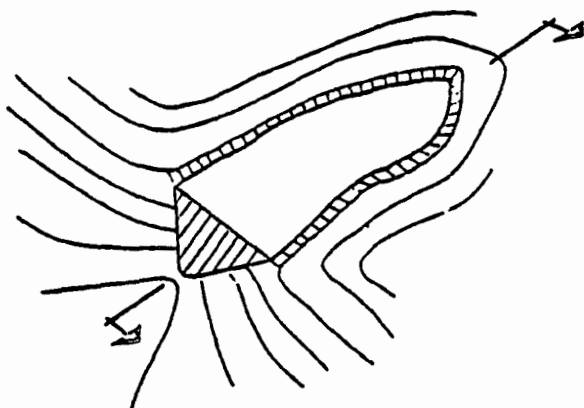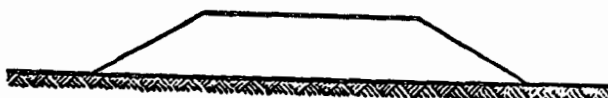
Valley Fill
Disposal
Configuration
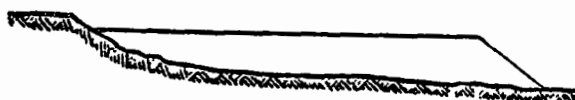
FIGURE 4-5

LANDFILL DESIGNS

advantages in terms of slope stability and groundwater pollution, it is often an "eye sore" in relation to the surrounding terrain and may not be preferred because of its visibility.

The valley fill design, which is the most common type of landfill used, is often the most complex in terms of original site preparation. Natural valleys or ravines are often sources of surface water runoff and may have springs along side slopes. In such cases, surface water and groundwater control is necessary to avoid accumulation of water and the development of a leachate problem. Drainage must be provided and often water is diverted under the landfill and collected in order to control erosion and/or pollution.

4.1.2   Disposal Practices

The previous discussion covered the methods of solid waste disposal. In the following pages, the current mix of disposal practices at coal-fired utilities is described. Fly ash, bottom ash and scrubber sludge are evaluated for percentage of each disposed by each of the methods addressed earlier.

Several sources of information were evaluated to provide the data presented in this section. The initial data collection effort involved obtaining a copy of the most current Federal Power Commission tape for their Form 67 data. This tape contained information for the year ending 31 December 1974. The FPC data indicated that some 390 plants burned some percentage of coal as a boiler fuel. The data covering disposal practices were not as specific in terms of how the wastes are handled as were needed. In addition, the cost data on disposal were a composite figure containing the costs of both collection and disposal. It was not possible to calculate disposal costs from these data.

In order to provide the needed data on disposal practices, Radian contacted 64 coal-fired utility plants. These plants, most of which began operation in the period 1970-1978, were characterized according to the disposal method used for each type of solid waste, the total coal-fired generating capacity of the plant, the solid wastes collection equipment and the type of FGD systems used. The data obtained in this effort were supplemented with in-house data and the results of studies done by other companies. The data obtained through the Radian initiated contact of selected utilities are more up-to-date and more extensive than the FPC data primarily due to the fact that plant personnel were questioned in some detail about the disposal method, the quantity of materials, the use of pond liners, and the mechanism of disposal. In this way, any confusion about the definition between landfill and ponding was resolved.

Sixty-four coal-fired utility plants representing 50,900 Mw of generating capacity were contacted. Most of the detailed data presented in this section come from 45 of the 64 plants. These 45 plants were able to provide the full extent of information requested. The only requested information not provided by the plants was cost data relating to disposal. This was primarily due to the fact that records are not kept to provide easy access to such data. Most of these plants are relatively large and represent a total generating capacity of 35,832 Mw. The mean per plant capacity is about 800 Mw.

4.1.2.1 Fly Ash Disposal

The data on fly ash covers the methods of collection, the amounts disposed of, and the methods of disposal. The collection methods and disposal practices for the 64 plants is presented in Table 4-1.

TABLE 4-1

FLY ASH COLLECTION AND DISPOSAL PRACTICES

Method	Number of Plants Reporting Method	Percent of Plants Reporting Method
I. Collection		
Dry Electrostatic Precipitator	39	61
Mechanical (Baghouse, etc.)	8	13
Wet Electrostatic Precipitator	4	6
Particulate Scrubber	2	3
Other	11	17
Total	64	100
II. Disposal		
Ash Pond	26	40
Conveyed to Landfill (Dry)	19	30
Paid Disposal	11	17
Sale of Fly Ash	3	5
Intermediate Ponding followed by Landfill	3	5
Other	2	3
Total	64	100

The data indicate that a majority of the plants surveyed use ESP's to collect dry fly ash. However, most of these plants dispose of the ash by ponding. Thirty percent haul the dry fly ash to landfill disposal. Five percent use landfill of settled ash after ponding.

The data indicate that approximately 17 percent pay another company to dispose of the ash and five percent sell the ash directly to some company. In addition, a significant amount of the fly ash removed by paid disposal may be sold and utilized. Thus, the total amount of ash utilized is difficult to determine from the disposal data. The fraction indicated "other" is fly ash given away or otherwise disposed at no cost to the utility.

The data on the amount of fly ash disposed by each method
is presented in Table 4-2. These data were obtained from the 45
plants providing more detailed data as compared to the whole sur-
vey of 64 plants.

TABLE 4-2

FLY ASH DISPOSAL PRACTICES BY QUANTITY OF ASH

Disposal Method	Amount (10^3 metric tons/yr.)	Percent of Total
Ponded	3,148	34
Landfill	4,763	51
Paid Disposal	1,415	15
Sold	--	--
Other	3	--
Total	9,329	100

A comparison of Table 4-1 and 4-2 indicates that while a
greater number of plants use ponding of fly ash, the greatest
quantity of material is disposed by landfill. Of the 45 plants
contacted, none sold their fly ash but some of the paid disposal
includes material that was eventually sold.

4.1.2.2   Bottom Ash Disposal

The data on bottom ash obtained from the survey covered
collection method, the amounts disposed of, and the methods of dis-
posal. Collection methods and primary disposal practices for the
64 plants are presented in Table 4-3.

TABLE 4-3

BOTTOM ASH COLLECTION AND DISPOSAL PRACTICES

Method	Number of Plants Reporting Method	Percent of Plants Reporting Method
I. Collection		
Wet Sluiced	52	81
Dry Conveyor	11	17
Other	1	2
Total	64	100
II. Disposal		
Ash Pond	24	38
Conveyed to Landfill (Dry)	17	27
Paid Disposal	8	12
Sale of Bottom Ash	6	9
Intermediate Ponding Followed by Landfill	6	9
Other	3	5
Total	64	100

The data indicate that, on a plant basis, the vast majority wet sluice bottom ash but less than half of these plants use ponding for disposal. It is not known why this discrepancy exists. Mechanical dewatering and dry disposal of bottom ash is not a common practice.

The data on the amount of bottom ash disposed of by each method are presented in Table 4-4. These data were obtained from the 45 plants providing more detailed data.

These data indicate that, from a quantity standpoint, the most common method of disposal is ponding. These data tend to confirm the data for disposal methods on a per plant basis presented in Table 4-3 as to the use of ponding over landfill for disposal.

TABLE 4-4

BOTTOM ASH DISPOSAL PRACTICES BY QUANTITY OF ASH

Disposal Method	Amount (10^3 metric tons/yr.)	Percent of Total
Ponded	1,763	44
Landfill	1,138	29
Paid Disposal	671	16
Sold	444	11
Total	4,016	100

4.1.2.3   Scrubber Sludge Disposal

The data obtained from the 1974 Form 67 or from additional contact with 64 plants were not sufficient to cover solid wastes from flue gas desulfurization processes. To enable evaluation of FGD disposal practices, additional data was gathered on 30 plants utilizing SO_2 scrubbers. The plants began operation in the period 1970 to 1978. Data sources used to generate the list of 30, include an EPRI report by Michael Baker, Jr., Inc.[55] a PEDCO report, [56] and various in-house information. The data provide information on the disposal practices, treatment prior to disposal and disposal site preparation. The data do not include waste generation rates on a per plant basis.

Of the 30 plants operating SO_2 removal systems, all utilize a wet scrubbing system. The source of alkalinity (calcium for the calcium-sulfur reaction) does not vary. The plants use either limestone, lime, fly ash or a combination of the three. The total amount of sludge generated is highly dependent on the source of calcium.

The total generating capacity of the 30 plants amounted to approximately 12000 Mw. Some 18 plants totaling 8,700 Mw utilize pond disposal of the sludges while 12 plants totaling 3,400 Mw utilize landfilling. In many cases, fly ash and scrubber sludge are removed

together. However, complete information on how many plants do this was not available. In many cases, the fly ash is removed in a dry collection system, such as an ESP, prior to the scrubber and landfilled while the scrubber sludge is ponded.

Of the 18 plants ponding scrubber sludge, eight pump the collected sludge to the pond site without any further treatment (dewatering, thickening, or commercial fixation). The remaining ten plants all utilize thickening processes prior to ponding of the sludge but only one plant is utilizing a commercial fixation process in conjunction with ponding while two utilize forced oxidation of the sulfite sludge to gypsum.

Among the 12 plants utilizing landfilling of scrubber sludge, seven use some method of stabilization involving either a commercial process or blending of sludge with fly ash or a fly ash/lime mixture. Three plants provide commercial fixation. For the five plants providing no stabilization prior to landfilling, one dewaters and landfills direct while the other four plants (located in arid climates) utilize intermediate ponds for evaporation and eventually excavate sludge from the holding ponds for landfill.

The same data sources mentioned earlier, provided data on the planned practices among some 38 power plants contracted for SO_2 removal systems or under a letter of intent to do so by April 1, 1978. These plants should begin operation by 1986. The generating capacity of these additional 38 plants is 26,700 Mw. The survey of the planned scrubber systems indicate a slight tendency to use landfilling of dry waste over ponding. Specifically, 19 units (11,500 Mw) will utilize landfill and 17 (13,700 Mw) will use ponding with two plants being undecided as to disposal alternative.

For future planned installations using ponding, ten of 17 will not treat the sludge in anyway prior to ponding. Of the

remaining seven, six will utilize dewatering techniques prior to ponding and one will use a settling pond. One plant will provide forced oxidation without thickening and another plans to contract for utilization of the Dravo commercial fixation process with ponding.

Some type of stabilization and/or fixation is planned for 16 of the 19 new units landfilling sludge. Four plan to contract for use of the IUCS commercial fixation process. The remaining units are planning to use some other form of stabilization. The methods are to blend either lime and fly ash with sludge (1-unit) or fly ash alone with sludge (11-units), all after some form of dewatering (centrifugation, vacuum filtration or mechanical thickening). The three installations not planning to use any means of stabilization will landfill sludge that has been dewatered by vacuum filtration.

4.1.2.4 Distance to Disposal Site

As the distance from the plant to the disposal site is a prime factor in the cost. of disposal and in characterizing existing disposal practices, data was obtained from 54 coal-fired power plants. These data are presented in Table 4-5. The data are presented by distance intervals. The mean distance from plant to disposal site was three miles.

The information was obtained in the form of .distance to disposal site as opposed to on-site or off-site disposal as this information is believed to be more relevant for the purposes of this report.

TABLE 4-5

DISTANCE FROM PLANT TO WASTE DISPOSAL SITE

	Percent of Plant in Interval	Percent of Waste in Interval
Less than 0.8 kilometers	26.8	7.22
0.8 kilometers to 1.4 kilometers	16.0	24.49
1.6 kilometers to 4.7 kilometers	21.4	36.50
4.8 kilometers to 7.0 kilometers	19.6	24.52
8.0 kilometers to 15.9 kilometers	7.2	4.77
16.1 kilometers to 32.0 kilometers	5.4	1.08
Greater than 32.1 kilometers	3.6	1.42
	100.0	100.00

A significant factor regarding the distance analysis is the fact that nearly 93 percent of all bottom ash and fly ash from the representative 54 plants is transported less than 8 kilometers (5 miles) form the generating plant to the ultimate disposal site. This is a strong indication that the cost of transporting large volume wastes over distances is generally avoided.

The data presented in Table 4-5 are intended to portray the practices currently used in the utility industry in the United States. In addition, the mean distance from the plant to disposal sites was 4.8 kilometers (3 miles) for this representative group and is intended to be realistic for the industry as a whole.

4.2   Economic Impact of RCRA

The economic impact of Section 4004 of RCRA on the electric utility industry is addressed in this section. The primary objective was to quantify the economics of compliance with RCRA. Inherent in this development was the assumption that fly ash, scrubber sludge and bottom ash resulting from the burning of coal in power plants are non-hazardous wastes under the definitions of the Act.

The basic approach used in this section involved development of an enforcement scenario, design of a typical plant to describe the quantities and types of wastes to be disposed of, development of the cost of disposal for the typical plant, and use of the disposal cost data to estimate the cost of the comliance scenario.

The analysis of the economics of compliance with RCRA was based on defining disposal options for a "typical" coal-fired power plant. The typical plant has a name plate capacity of 1000 Mw. Options for the plant included burning either high sulfur eastern coal or low sulfur western coal. The nationwide cost of compliance was estimated from using the costs developed for the various disposal options for the typical plant.

A direct calculation of the cost of compliance for the industry on a nationwide scale was not performed. This was not possible for essentially two reasons. First, there is considerable uncertainty in how the regulations will be interpreted and enforced. Second, determination of which plants are not in compliance required extensive information on disposal practices on a plant by plant basis. In the first case, there is considerable flexibility inherent in the enforcement mechanism preventing precise determination of how RCRA will be enforced. Ultimately the enforcement mechanism will be a state and local function and will be dictated by local disposal conditions based on options available to the plants. In the second case, a necessary level of detail concerning current disposal practices and the environmental effects of the practices was not available and therefore, determination of which plants or facilities are not in compliance was not possible.

The following discussion includes an analysis of the
Resource Conservation and Recovery Act as it may pertain to solid
waste disposal in the electric utility industry, the development
of several enforcement scenarios, a typical or base case for the
economics and some hypothetical impacts of the various enforce-
ment scenarios. This analysis does emphasize wet scrubbing as a
means of SO_2 and fly ash removal. As currently proposed (9/19/78)[21]
air regulations for coal-fired power plants tend to encourage the
use of scrubbers.

4.2.1   Interpretation of RCRA

The development of potential enforcement scenarios was
accomplished with input from both the written documents and from
conversations with the EPA regional offices. Although no claim
is made suggesting that the proposed regulations will be enforced
as outlined in this section, the enforcement scenarios were neces-
sary to define what the impacts might be on the typical plant.
The site-specific nature of the regulations as written and the
flexibility given to the states in both classifying facilities and
regulating possible clean-up make it impossible to define exactly
what actions will be taken. This situation made it necessary to
use the hypothetic enforcement scenarios as a means of assessing
potential impacts.

The general interpretations of the proposed regulation
were based on an analysis of the Resource Conservation and Recov-
ery Act of 1976[57] and the proposed classification criteria pub-
lished in the 6 February 1978 Federal Register.[58]

One of the stated objectives of RCRA is "to prohibit
future open dumping on the land and to require conversion of
open dumps to facilities which do not pose a danger to the en-
vironment or to health".

The objective is to be accomplished by state and federal cooperation through the development of state plans aided by federal economic and technical support. The responsibilities of the states in these "state plans" are to: (1) identify the responsibilities of state, local, and regional authorities in the implementation of the "state plan"; (2) to prohibit the establishment of new open dumps within the state; and (3) to require that all solid waste be disposed of in sanitary landfills or in an environmentally safe manner. The plans will provide for closing or upgrading of existing open dumps.

The main engineering point is the distinction between an open dump and a sanitary landfill. This point is further clarified in RCRA in that "a facility may be classified as a sanitary landfill and not as an open dump only if there is no reasonable probability of adverse effects on health or the environment from the disposal of solid wastes at such facilities."[58] The probability of adverse effects must then involve the character of the wastes, the manner of disposal and the location of the disposal site.

The mechanism of enforcement starts with the requirement, set forth in RCRA, that an inventory be made of all existing disposal facilities in the U.S. which are open dumps and that this inventory be published. The states, under the proposed rules published in the Federal Register, have responsibility for preparing the inventory.

As currently proposed, the considerations of importance in determining whether a site is an open dump or a sanitary landfill involve both location and performance factors. These are discussed below.

4.2.1.1 Location Considerations

The sites are not to be located, in general, "in environmentally sensitive areas when feasible alternatives exist unless it can be clearly demonstrated that no significant adverse impact on the ecosystem or human health" will result. The environmentally sensitive areas which are addressed in this section are: (1) wetlands, (2) flood plains, and (3) sole source aquifers. Those, then become the primary location judgements. While these environmental units have been defined, they have not been fully mapped for all the states. In general, it is advised not to locate new facilities or expand old facilities in such areas unless no other feasible and/or economical alternatives exist.

For wetlands, new disposal sites may not be placed in designated areas and existing operation may not be continued unless a NPDES permit is obtained under Section 402 of the FWPCA Amendment of 1972.[59]

For flood plains, the establishment of any new facility or expansion of any old facility in a flood plain must not cause increased flooding during the base flood or be constructed in such a way as to be inundated during flood episodes. Facilities can be located in such flood plains if it can be demonstrated that they will not adversely affect water quality or flood flow capacity.

Sole source aquifers are to be protected from any degradation of water quality. In general, no solid waste disposal facility is to be located in a sole source recharge zone unless no feasible alternatives exist.

4.2.1.2 Performance Considerations

The performance standards tend to emphasize two points: (1) permitting for new facilities (NPDES) and (2) proof that existing or planned facilities, particularly those within environmentally sensitive areas, are the only feasible options available to the industry and that they will have no adverse effects on the environment. Of particular concern is protection of groundwater, even if the site is not in an environmentally sensitive area.

In determining compliance with the regulations, application of the best practicable controls is specified in conjunction with environmental monitoring to determine any adverse impacts. All facilities must be "so located, designed, constructed, operated and maintained in order to emphasize the use of best practicable controls and to allow a determination of compliance based on site-specific evaluation of these control technologies..."[58] Although environmental monitoring is encouraged, it is pointed out that "the state may determine it is not necessary to monitor if the facility is such that no adverse effect is expected because of low volume or inert or innocuous wastes and because the control technologies and practices are considered (by the state) to achieve the environmental standards."[58] It is up to the states to determine a compliance schedule for any facilities judged as not in compliance.

From this review of the proposed regulation the following points were concluded:

- The states and local agencies will have considerable flexibility in determining what existing sites are open dumps, in the development of compliance schedules and in determination of what actions, designs, and controls are considered acceptable.

- The regulations emphasize both location
  and performance standards which make it
  impossible to determine how many plants
  will be affected.

4.2.1.3   EPA Opinions on Enforcement Options

All of the EPA regional offices were contacted in an
attempt to define three items: (1) if the environmentallly sen-
sitive areas have been mapped, (2) if any estimate of the number
of facilities affected has been made, and (3) what, if any, inter-
pretations the regional offices have made concerning enforcement.
The responses were used to develop reasonable enforcement scenarios.
They were solicited on an informal basis and are not to be con-
strued as official statements of policy.

The answers from the regional offices to these informally
presented questions demonstrated a considerable variance in inter-
pretation of the proposed regulations. There were some definite
general conclusions, however. While some of the states are cur-
rently developing or have recently developed maps of wetlands,
these are not generally available. The flood plains have been
mapped by HUD. Some sole source aquifers have been designated.
However, many of the regional offices are of the opinion that
additional aquifers may be designated in the future.

On the question of enforcement, many of the regional
offices expressed the opinion that there will be considerable
flexibility in how the various states enforce RCRA. This flexi-
bility will be caused as much by the local disposal conditions as
by the regulatory climate in the specific states. Most of the
regional offices expressed the opinion that enforcement will
necessarily proceed on a case-by-case basis and that generalized
conclusions regarding non-compliance are not possible under the
circumstances.

Another important opinion presented by many of the re-
gional offices is that case-by-case review of existing facilities
will probably be oriented toward proof of environmental damage as
opposed to a presumption of guild based on location. In other
words, just because a site is located in a wetland does not mean
that environmental damage will occur. The primary concern appears
to be with groundwater protection.

4.2.2   Costing Basis for Enforcement Scenarios

The development of a costing basis for estimating the
impact of RCRA was performed in three steps:

1)   Development of a potential enforcement scenario,

2)   Development of engineering data describing
      disposal options for a "typical" coal-fired
      plant, and

3)   Development of cost factors for the disposal
      options.

4.2.2.1   Development of Potential Enforcement Scenarios

As explained earlier, flexibility in the methods of en-
forcement is expected when RCRA is implemented. Compliance
options were developed for the purpose of providing a reasonable
enforcement scenario. The general scenarios are summarized
below. Plants are divided into existing facilities and planned
(future) facilities.

●   Existing disposal sites declared open dumps
     (located in wetlands, flood plains, sole source
     aquifers or known to be contamination ground-
     water) and the sites are closed. New disposal
     sites are located outside the environmentally
     sensitive areas and all ponds are lined to pro-
     tect groundwater.

- New plants to be constructed will generally avoid environmentally sensitive areas and will provide for maximum protection of groundwater through lining of all ponds.

The potential impact of RCRA on existing facilities was assumed to be limited to newer plants, those that began operation after 1970. The data for existing plants presented in Section 4.1.2 was used to describe existing operations at these plants. The plants were assumed to have to move disposal for the current distance of 4.8 kilometers or 3 miles (average) to 16.1 kilometers (10 miles) from the plants. The increase in the average distance to disposal is assumed to be necessary to protect groundwater. New landfill sites are assumed to be necessary to protect groundwater. New landfill sites are assumed to be located in impermeable areas or areas where groundwater quality is not in danger from such facilities. Ponds are assumed to be located outside environmentally sensitive areas, away from groundwater resources and are assumed to be lined for further protection.

For planned facilities, the primary assumption is that all plants will be affected by RCRA. The primary effect is assumed to be the distance from plant to disposal. As siting of disposal facilities will be a concern in the planning stage for the whole plant, it is assumed that the average distance from plant to disposal site will increase over the current average of 4.8 kilometers. As the plants can take this into account in siting the plant, the new distance is assumed to be less than the 16.1 kilometers used for the existing plants. The distance used was 8 kilometers. It was also assumed that all ponds would be lined, regardless of groundwater conditions.

4.2.2.2   Development of Engineering Data

The typical plant, characterized for the purpose of quantifying the impact of enforcement, was based on 1000 Mw "name

plate" capability with a 35 percent thermal efficiency. To cover
a typical range of coal properties and operating conditions, the
design data presented in Table 4-6 were used. The assumptions
made on all design data are based on typical data for the industry.
SO₂ scrubbers using limestone were assumed for the purpose of
meeting air quality regulations for SO_2 emissions.

Regulatory assumptions were made for both current regula-
tions and proposed future regulations. These regulatory conditions
were chosen to illustrate any impacts on solid waste generation
to be expected from future regulations. The final regulations for
new source performance have not been adopted. The regulatory
assumptions used are listed below.

	Current Regulations	Proposed Regulations
Fly Ash Removal	99%	99%
SO₂ Removal	$\frac{1.2 \text{ lb } SO_2}{10^6 \text{ Btu}}$ (Allowable Discharge)	85%

The operating conditions, coal properties and regula-
tions assumed were used to calculate the solid wastes generated
by the "typical" plant. These are presented in Table 4-7.

The dominant impact of the proposed regulations is for
scrubber sludge at plants burning low sulfur coal. This impact
is taken as the difference between existing sludge generation
rates and the rates to be generated assuming implementation of
the proposed regulations. The difference for high sulfur coal
is only slight.

The densities for the solid wastes listed in the pre-
vious table are summarized in Table 4-8. These figures were
used to estimate the volumes of solids generated. In addition
to the assumptions concerning density, the water content of the

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TABLE 4-6

DESIGN BASIS - FOR 1000 MW COAL-FIRED POWER PLANT

	Coal Type	
	Western Coal	Eastern Coal
Plant Design Data		
Stream Factor, hr/yr		
First Year	7,000	7,000
Average 30 yr. life	4,350	4,350
Thermal Efficiency, %	35	35
Fly Ash to Bottom Ash Ratio	80/20	80/20
Plant Life		
New, yr.	30	30
Existing, yr.	25	25
Coal Data		
Heat Value (Wet), Btu/lb	9,000	11,000
Ash Content (As Burned, %	8	15
Sulfur Content, %	0.8	3
Flue Gas Cleanup		
Percent of Sulfur in Coal Converted to SO_2, %	95	95
Fly Ash Removal		
Scrubber, %	99	99
Dry Removal (ESP, Baghouse), %	99	99
Scrubber Design		
Limestone Stoichiometry*	1.5	1.5
Percent Solids in Effluent from FGD System, %	15	15
Ratio of Sulfite to Sulfate in Scrubber Sludge	85/15	85/15

*Ratio of moles limestone added to moles SO_2 removal.

TABLE 4-7

SOLID WASTE GENERATION RATES FOR
1000 MW COAL-FIRED POWER PLANT

| | Solid Waste Rates, Metric tons/hr. | | | | | |
| | Western Coal | | | Eastern Coal | | |
	Dry Solids	50% Solids	65% Solids	Dry Solids	50% Solids	65% Solids
Fly Ash	25.0			38.2		
Bottom Ash	6.3			9.5		
Scrubber Sludge						
Current Regulations[22]	2.0	4.0	3.1	38.3	76.6	58.9
Proposed Regulations[21]	15.0	29.9	25.4	45.8	91.6	70.5
		65% Solids	70% Solids		65% Solids	70% Solids
Fly Ash and Scrubber Sludge Combined						
Current Regulations[22]	26.9	53.9	38.5	76.5	117.9	109.8
Proposed Regulations[21]	39.9	79.8	57.1	84.0	128.8	119.7

of the settled sludges were also assumed. The volume of material for ultimate disposal is very dependent on the percent solids. The volumetric waste generation rates are summarized in Table 4-9.

TABLE 4-8. DRY AND WET BULK DENSITIES OF FGD WASTE PRODUCTS

	Specific Gravity	Dry Bulk Density @				Wet Bulk Density @			
		Optimum	70%	65%	50%	Optimum	70%	65%	50%
			Solids				Solids		
		Kgs. Dry Solids/m^3				Kgs. Total/m^3			
Fly Ash	2.55	1520	1220	1070	720	1790	1730	1650	1430
Bottom Ash	2.55	1440	1220	1070	720	1760	1730	1650	1430
Scrubber Sludge	2.55	1280	1220	1070	720	1670	1730	1650	1430
Fly Ash and Scrubber Sludge	2.55	1520	1220	1070	720	1790	1730	1650	1430

TABLE 4-9. VOLUMETRIC GENERATION RATES FOR SOLIDS AND SLUDGES FROM A 1000 MW PLANT

Disposal Assumptions	Volumetric Rates, 10^{3}m^3/yr	
	Western Coal	Eastern Coal
Dry Fly Ash (Optimum H$_2$O for compaction)	144	221
Wet Sluiced Fly Ash (70% solids)	178	275
Wet Sluiced Bottom Ash (70% solids)	45	68
Scrubber Sludge (50% solids)		
Current Regulations	24	464
Proposed Regulations	181	158
Fly Ash and Scrubber Sludge (65% solids)		
Current Regulations	193 *	623
Proposed Regulations	326	685

*Exception - 70% solids.

There were two basic disposal practices used for the calculations in this study. They were ponding and landfill. Although ponding can be considered a form of landfill, the method of solids handling and equipment required are different. For landfill of dry solid or dewatered sludges, it is assumed that some form of trench filling will be used as described in the previous section. Landfills can be located in areas suitable for this type of disposal and not require liners.

Ponds are assumed to be 9.1 meters deep, rectangular shape and contained by dikes with a three to one slope. In actual practice, the depth and shape of the ponds will be determined by local conditions. The ponds are assumed to provide a thirty-year lifetime. The land area needed to dispose of sludges was calculated and are summarized in Table 4-10. The land requirements needed for ultimate disposal depends on the final settled volume which in turn depends on the water content. The data in the table reflect the considerable difference between eastern and western coal in terms of pond requirements.

In order to protect ponds from flooding, it was assumed that dikes or levees would be constructed. These would have to be designed so as not to interfere with flood waters or cause increased upstream flooding. Should it be impossible to design and construct a flood protection dike under these restrictions, it was assumed that disposal would have to be located outside the flood plain. The dike referred to for flood protection is a diversion structure and is different from the dikes forming the pond walls.

TABLE 4-10

LAND REQUIREMENTS IN SQUARE METERS FOR
A 1000 MW COAL-FIRED POWER PLANT

PONDING*

	Western Coal	Eastern Coal
Wet Sluiced Fly Ash	364	563
Wet Sluiced Bottom Ash	93	138
Scrubber Sludge		
Current Regulations[22]	49	951
Proposed Regulations[21]	372	1,141
Fly Ash and Scrubber Sludge Combined		
Current Regulations[22]	393	1,275
Proposed Regulations[21]	668	1,404

LANDFILL*

	Western Coal	Eastern Coal
Dry Disposal Fly Ash	295	453
Dry Disposal of Scrubber Sludge (Dewatered to 50% Solids)		
Current Regulations[22]	49	951
Proposed Regulations[21]	372	1,141
Dry Disposal of Fly Ash and Scrubber Sludge Combined (Dewatered to 65% Solids)		
Current Regulations[22]	393	1,275
Proposed Regulations[21]	668	1,404

*Both ponds and landfills are 9.1 meters deep (30 feet)

TABLE 4-11 COMPARISON OF ASSUMPTIONS
BETWEEN 1000 MW PLANT AND TVA COST EVALUATION

	TVA Study	Western Coal	Eastern Coal
Plant Size, Mw	500	1,000	1,000
Coal Data			
Heating Value (Wet), Btu/lb	10,000	9,000	11,000
Ash Content, %	16	8	15
Sulfur Content, %	3.5	0.8	3
Yearly Operating Time, Hrs.			
New Plant	7,000	7,000	7,000
Average Over Plant Lifetime	4,350	4,350	4,350
Life of New Plant, yrs.	30	30	30
Scrubber Data			
Limestone Stoichiometry for Scrubbing Operations	1.5	1.5	1.5
Percent of SO_2 Converted to:			
Calcium Sulfate, %	15	15	15
Calcium Sulfite, %	85	85	85
Solids Content of Slurry from Scrubber, %	15	15	15
Solids Content After Clarifier, %	35	35	35
Percent Solids of Settled Sludge:			
Scrubber Sludge Only, %	50	50	50
Fly Ash & Scrubber Sludge, %	50	65	65

4.2.2.3 Development of Cost Factors

The economics of the various disposal options were esti-
mated using published data. Most of the cost data used were taken
from the EPA report 600/7-78-023a "Economics of Disposal of Lime/
Limestone Scrubbing Wastes; Untreated and Chemically Treated
Wastes." [61]

While the exact cost to a specific power plant will vary
widely depending on the type of coal burned, available disposal
conditions and existing equipment at the plant, the cost figures
are presented as reasonable "ballpark" numbers and give a rough
cost estimate for compliance with the assumed enforcement scenario.
The published data used were for a specific coal and plant design.
These conditions were very nearly the same as for the eastern coal
used in this study. The coal properties for the western coal were
considerably different, however.

To compensate for the difference between eastern and
western coals, it was necessary to estimate the impact associated
with the lower ash and sulfur content of the western type coal.
The cost data used to develop the estimates for this study included
an assessment of the effect of ash content and sulfur content on
the economics of disposal. Linear interpolations were made using
these data to estimate the base cost for western coal. The
reductions in the costs of disposal resulting from the lower ash
and sulfur rates were partially offset by the slightly higher
coal rate required. The higher burn rate for coal resulted from
the lower heating value of the western coal as opposed to the coal
used in the cost study. A comparison of the significant assump-
tions between the various coals is presented in Table 4-11.

The cost of landfill disposal of dry solids is generally cheaper than wet sluicing of these materials if the disposal site (pond or landfill) is three miles or more from the plant. The cost of land for disposal can be expected to vary considerably from one location to another. The primary advantage of this method of disposal is the low capital cost resulting from the absence of expensive pumping or dewatering equipment. Although site preparation was taken into account, it was assumed that liners will not be necessary for landfill type operations.

Table 4-12 contains the cost estimates used for the evaluation of the disposal options. These cost figures were used to estimate the cost of compliance associated with the enforcement scenario. The table includes both capital and operating costs for each option. These figures are presented in dollars per kilowatt for capital investment and mills per kilowatt hour for revenue requirements. The data can be used to assess the impact on a specific plant given the capacity and operating characteristics of the plant. The cost figures represent reasonable "average" figures to the extent that 1000 Mw represents an average plant.

4.2.3   Estimated Cost of Compliance

The cost factors summarized in the previous table were used to estimate the cost of compliance of the enforcement scenario. The results of this analysis are presented in the following pages.

4.2.3.1   Assumptions

Several key assumptions were necessary to estimate the economic impact of RCRA. Some of these assumptions have a reasonable basis. However, others are subject to question.

TABLE 4-12

COST ESTIMATES[*] FOR SOLID WASTE
DISPOSAL FROM A 1000 MW PLANT

Design Cases	Western Coal				Eastern Coal			
	Capital Cost (10^3\$)	($/KW)	Revenue Requirements (10^3\$/yr)	(Mills/Kwhr)	Capital Cost (10^3\$)	($/KW)	Revenue Requirements (10^3\$/yr)	(Mills/Kwhr)
1. New Plant: Scrubber Sludge and Flyash — 15 Percent Solids Pumped 4.8 Kilometers to Clay-Lined Pond.	17,275	17.3	3,920	0.56	31,920	31.9	6,730	0.97
2. Five Year Old Plant: Same as Number 1.	12,920	12.9	3,400	0.49	23,880	23.8	5,840	0.84
3. New Plant: Scrubber Sludge (only) — 15 Percent Solids Pumped 4.8 Kilometers Clay-Lined Pond.	11,240	11.2	2,670	0.38	21,070	21.1	4,530	0.66
4. Five Year Old Plant: Same as Number 3.	8,400	8.4	2,310	0.33	15,760	15.8	3,920	0.57
5. New Plant: Same as Number 1 Except Pumped 8 Kilometers	21,050	21.1	4,920	0.70	38,890	38.9	8,440	1.21
6. New Plant: Same as Number 1 and 5 Except Pumped 16.1 Kilometers.	29,350	29.4	6,680	0.95	54,220	54.2	11,460	1.64
7. New Plant: Fly Ash Only. Trucked 4.8 Kilometers to Landfill.	390	0.4	1,560	0.22	592	0.6	2,380	0.34
8. New Plant: Fly Ash Only, Trucked to Disposal 16.1 Kilometers.	430	0.4	2,060	0.29	660	0.7	3,140	0.44
9. New Plant: Fly Ash Only, Wet Sluiced to Pond 4.8 Kilometers.	11,240	11.2	2,670	0.38	14,000	14.0	3,200	0.46
10. New Plant: Fly Ash Only, Wet Sluiced to Ponds 16.1 Kilometers.	19,100	19.1	4,550	0.65	23,780	23.8	5,450	0.78
11. New Plant: Same as Number 1 Except Clarified to 35 Percent Solids Before Pumping.	15,620	15.6	3,830	0.55	30,970	31.0	6,580	0.94
12. New Plant: Clarified and Filtered to 60 Percent Solids; Trucked 4.8 Kilometers.	6,670	6.7	3,790	0.54	12,330	12.3	6,490	0.93
13. New Plant: Same as Number 11 Except Pumped 16.1 Kilometers	19,520	19.5	5,740	0.78	45,580	45.6	9,850	1.41
14. New Plant: Same as Number 12 Trucked 16.1 Kilometers to Landfill.	7,250	7.3	5,180	0.74	13,400	13.4	8,890	1.27
15. New Plant: Same as Number 1 Except Pond Lined with Synthetic Pond Lining Costing $1.50/yd².	20,250	20.3	4,460	0.64	37,400	37.4	7,650	1.10
16. New Plant: Same as Number 15 Except Cost of Lining is $4.50/yd².	24,730	24.7	5,360	0.77	45,680	45.7	9,200	1.32

[*]Based on cost data from Barrier. J.W., et.al., "Economics of Disposal of Lime/Limestone Scrubbing Wastes: Untreated and Chemically Treated Wastes" National Fertilizer Development Center, Tennessee Valley Authority, TVA BULL 4-123, EPA-600/7-78-023a, February 1978.

Therefore, the assumptions are presented only as being reasonable and are not to be construed as being the only reasonable assumptions nor necessarily the best assumptions. Furthermore, the estimate of the economic impact of RCRA presented in this section should be evaluated in light of these key assumptions.

Two sets of assumptions were made: one set for existing plants and one for new plants. For existing plants, the assumptions are concerned with : (1) what plants are currently doing in terms of disposal, (2) the cost of current operations, (3) the number of plants affected by RCRA and (4) how the plants will react to the new regulations. For future plants, the assumptions cover (1) what disposal methods the plants will need, (2) how RCRA will influence disposal and (3) the number of plants affected.

The specific assumptions made to enable an estimate to be made of the cost of enforcement are presented below.

Assumptions for Existing Plants

1. Only plants starting operation since 1970 will be affected by RCRA.

2. The total capacity of plants starting operation in the period 1970-1976 is 42,500 Mw, based on the absolute increase in coal consumption by electric utilities over this time period (FEA report[61]). The average size of existing plants is assured to be 1000 Mw, resulting in an estimated 43 new plants.

3. Of these plants, 82 percent burn eastern-type coal and 18 percent burn western-type coal.

4. The distribution of disposal practices for the 42,500 Mw of facilities is the same as the results as presented in Section 4.1 of this report.

5. The economic data presented in the TVA report are applicable to the plants.

6. No estimate is made of the cost to cover and abandon existing facilities.

7. The average distance of current disposal practices is 4.8 kilo-
    meters (3 miles).

8. Plants will need to relocate their disposal an average of 16.1
    kilometers (10 miles) for the plant.

9. The plant will use the cheapest disposal method at the new
    disposal site.

10. The cheapest disposal method is assumed to be the cheaper
    for the TVA report.

11. The TVA data for 5-year old plants can be used to estimate the
    cost of current disposal.

12. The additional costs of RCRA will be the capital cost and
    moving the ponds an average of 16.1 kilometers. The additional
    revenue requirements will be the difference between these at
    16.1 kilometers (using liners) and current operations 4.8
    kilometers.

    Assumptions for New Plants (Future Construction)

1. The projected impact of RCRA on new plants is limited to those
    projected to be constructed by 1985.

2. The projected capacity to be constructed in the period 1975-
    1985 is 160,000 Mw or assuming 1,000 Mw plants approximately
    160 plants (FEA report[60]).

3. Of these plants, 77 percent are projected to burn an eastern-
    type coal and 23 percent are projected to burn a western-type
    coal.

4. The distribution of disposal methods for these plants will be
    the same as estimated for currently planned but not yet con-
    structed facilities.

5. The economic data for disposal practices presented in the TVA
    report can be used to predict disposal cost.

6. The addition of location criteria plus the other RCRA regula-
    tions will result in plants having to locate disposal an average
    of some 8 kilometers from the plant as opposed to 4.8 kilometers
    for no RCRA regulations.

7. The added cost associated with RCRA is the difference between
    requiring 3 meters of clay liner at all ponds and an average
    distance of 8 kilometers versus the cost of disposal as cur-
    rently performed.

8. Landfill will be located in areas where no danger to ground-
    water exists and, therefore, no liners will be required.

4.2.3.2 Existing Plants

The estimate of the cost of RCRA for existing plants involved several calculations. The first calculation was the estimate of the cost of current disposal practice. As the capital investment for a new disposal site is independent of any past capital expenses, the cost of current operations was based on only the revenue requirements (operation and maintenance costs). The mix of disposal practices established in Section 4.1.2 was used to estimate the total megawatt capacity of facilities disposing of each waste with each disposal method. The capacity figures were then multiplied by modified cost factors based upon the cost estimates from Table 4-12 (in mills/kw-hr) to give the revenue requirements in dollars per hour. An assumed operating factor of 7,000 hr/yr was then used to estimate the annual revenue requirements. The annual revenue requirements calculated in this manner thus represent annual costs during early plant life as opposed to costs averaged over the entire plant life. In later years, the annual revenue requirements will be lowered. The results of these calculations are presented below in Table 4-13.

TABLE 4-13

ESTIMATE OF REVENUE REQUIREMENTS FOR EXISTING DISPOSAL FACILITIES -
1970-1978

Material	Method of Disposal	Capacity (in Mw) of Plants Using Disposal Method	Revenue Requirements ($/yr)
Fly Ash	Ponded	10,400	$28,230,000
	Landfilled	15,600	$29,750,000
Scrubber Sludge	Ponded	5,220	$19,250,000
	Dewatered & Landfilled	2,100.	$ 7,350,000
Fly Ash & Scrubber Sludge	Ponded	3,480	$18,920,000
Combined	Dewatered & Landfilled	1,400	$ 7,200,000
	Total	38,200	$110,700,000

The estimate of the cost of existing disposal is $110,700,000 per year. This cost represents 38,200 Mw or 90 percent of the assumed total capacity of plants. The remaining 10 percent use paid disposal, sell the material, or do not specify the exact disposal method. This cost estimate presented has ignored this 10 percent of capacity and assumes that disposal costs are balanced by utilization revenues for this segment.

The next step involved estimating the cost of compliance with RCRA. The assumed impact on the plants was that the average distance to disposal would be 16.1 kilometers (10 miles), as opposed to 4.8 kilometers (3 miles) at present, and that all ponds would use clay liners. As established in the previous section on costs, as the distance to the disposal site increases, the costs go up dramatically. It was, therefore, assumed that the plants would use the most economical means of disposal for the new disposal facilities.

The cost data used in this study indicate that at a distance of 16.1 kilometers, the most economical disposal method is dewatering of sludges to 60 percent solids followed by trucking to landfill. It was, therefore, assumed that plants currently pumping sludges to ponds would install dewatering facilities. The cost of moving the facilities, both capital expenses and revenue requirements, are presented below in Table 4-14.

The net cost for the scenario is the estimated cost of compliance minus the existing costs. If all of the existing plants are assumed to be out of compliance, the net cost increase for RCRA is $98,150,000 (98,150,000-0) in capital investment and $24,600,000/yr (135,300,000-110,700,000) in revenue requirements.

TABLE 4-14

ESTIMATE OF THE COST OF DISPOSAL FOR MOVING THE
DISPOSAL SITE FOR 16.1 KILOMETERS FROM THE PLANT COST

Material	Disposal Method	Capacity in (Mw)	Investment Capital ($)	Revenue Requirement ($/yr)
Fly Ash	Landfill	26,000	11,960,000	$66,600,000
Scrubber Sludge	Dewatered and Trucked to Landfill	7,320	42,540,000	34,780,000
Fly Ash and Scrubber Sludge Combined	Dewatered and Trucked to Landfill	4,880	43,650,000	23,920,000
	Total	38,200	98,150,000	135,300,000

For comparative purposes, the cost of using the same
current mix of disposal methods but moved to 16.1 kilometers
from plant was estimated. For this scenario, the cost was
$467,290,000 for capital investment and $171,550,000/yr for
revenue requirements. For capital investments, this presents a
376 percent increase over the minimal cost estimate presented
above. The increase in revenue requirements was 27 percent.
Most of this difference is in expensive pumping equipment and
the cost of clay lining for the ponds.

One additional cost not included in the cost numbers is
the cost of monitoring wells. Such wells could be used to moni-
tor groundwater quality around the pond or landfill site to insure
that leachate contamination is not occurring. An estimated cost
for such monitoring is $25,000 per plant per year. This estimate
is based using 3 wells at an average depth of 100 feet. Samples
are assumed to be taken regularly (once per month) and analyzed
for major species and any trace toxic pollutants suspected as being

present. The first year cost includes the cost of drilling the well and any equipment associated with it. Thereafter, the cost may decrease if the sampling interval and number of samples to be analyzed remains the same. This additional yearly cost would total $1,075,000 for the 43 plants. While this represents a significant expense, the amount is rather small compared to the overall cost of the scenario.

For reasons of estimating an impact, it is assumed that (1) half of the existing capacity addressed in this study would have to move their disposal sites, and (2) all the plants are assumed to use groundwater monitoring. Thus, the cost impact of RCRA for existing plants is:

Capital Investment	$49,075,000
Revenue Requirements	13,375,000/yr

4.2.3.3   New Plants

The estimate of the cost of RCRA for planned or future facilities to come on line by 1985 involved two calculations. The first step involved estimating the cost of disposal assuming the plants will not be affected in any way by disposal regulations other than the current ones. In this case, the plants are assumed to use the mix of disposal methods found for currently planned facilities and presented in Section 4.1.2. These plants are assumed to comply with currently proposed air quality regulations as presented in Section 4.2.1.

The second step involved estimating the cost of disposal assuming RCRA would cause the facilities to locate disposal an average of 8 kilometers from the plant but that the mix of disposal methods would remain the same. All ponds are assumed to be lined with 0.3 meters of clay.

The projected increase in megawatt capacity of coal-fired units to the year 1985 is 160,000 Mw. This figure was taken from the Federal Energy Administration's "1976 National Energy Outlook."[60] The average capacity was assumed to be 1,000 Mw resulting in 160 plants covered by the projection period. It is assumed that only 20 percent of the facilities would line ponds in the absence of RCRA type regulations. The average distance from plant to disposal site was assumed to remain the same as is currently the case (4.8 kilometers).

The projected cost of disposal for the 160,000 Mw of capacity was calculated and is summarized below in Table 4-15. It should be noted that these capacity figures cannot be summed to yield 160,000 Mw. The distribution of plant capacity among the disposal methods listed in Table 4-15 is based upon the percentages of disposal practices obtained from the plant survey for planned construction. The disposal methods listed in Table 4-15 represent greater than 90 percent of the disposal options being considered. Since cost estimates were not available for these minority options, these costs (and the associated plant capacities) were ignored in the cost figures presented in Table 4-15. Thus, the cost estimates presented are equivalent to assuming that the disposal costs are balanced by the utilization credits for the minority options not considered. In any case, this assumption should introduce only a small error.

The cost of disposal assuming RCRA will result in an increase in the average disposal distance of 3.2 kilometers and require lining of all ponds to protect groundwater was calculated. The mix of disposal methods was assumed to be the same as the previous development. These results are presented in Table 4-16. The higher cost of pumping sludges as compared with landfill operations is readily seen in the case for fly ash disposal. In this development, the capital costs associated with landfill of dry solids is very low.

TABLE 4-15 ESTIMATED COST FOR DISPOSAL FOR PLANNED AND FUTURE FACILITIES TO THE YEAR 1985 WITH NO ADDITIONAL REGULATIONS

Material	Method of Disposal	Capacity (In Mw) of Plants Using Method	Capital Investment ($)	Revenue Requirements ($/yr)
Fly Ash	Ponded (15% solids)	48,000	556,310,000	174,020,000
	Landfill	48,000	26,600,000	104,960,000
Scrubber Sludge	Ponded (15% solids)	28,890	471,370,000	92,280,000
	Ponded (dewatered to 35% solids)	20,070	302,400,000	90,590,000
	Dewatered and Landfilled	41,280	245,270,000	188,120,000
Fly Ash and Scrubber Sludge Combined	Ponded (15% solids)	19,260	476,560,000	138,480,000
	Ponded (dewatered to 35% solids)	13,380	318,090,000	197,810,000
	Dewatered and Landfilled	27,520	303,050,000	161,880,000
	Total		2,699,650,000	1,148,140,000

TABLE 4-16   ESTIMATED COST FOR DISPOSAL FOR

PLANNED AND FUTURE FACILITIES TO THE YEAR 1985

WITH RESTRICTIONS IMPOSED BY RCRA

Material	Method of Disposal	Capacity (In Mw) of Plants Using Method	Capital Investment ($)	Revenue Requirements ($/yr)
Fly Ash	Ponded (15% solids)	48,000	769,930,000	177,850,00
	Landfill	48,000	26,600,000	114,280,000
Scrubber Sludge	Ponded (15% solids)	28,890	652,450,000	142,830,000
	Ponded (dewatered to 35% solids	20,070	430,250,000	96,420,000
	Dewatered and Landfilled	41,280	287,820,000	235,490,000
Fly Ash and Scrubber Sludge Combined	Ponded (15% solids)	19,260	659,690,000	141,200,000
	Ponded (dewatered to 35% solids)	13,380	440,760,000	95,700,000
	Dewatered and Landfilled	27,520	364,250,000	194,930,000
	Total		3,631,750,000	1,198,700,000

The estimated cost increase of RCRA for planned and future facilities is summarized in Table 4-17. These cost can be derived from the differences in total costs presented in Table 4-15 and 4-16. In addition, estimated costs of groundwater monitoring have been included in the total revenue requirement increase resulting from RCRA.

TABLE 4-17. ESTIMATED ADDITIONAL DISPOSAL COSTS RESULTING FROM RCRA FOR PLANNED AND FUTURE COAL-FIRED FACILITIES TO THE YEAR 1985.

	Capital Investment ($)	Revenue Requirements ($/yr)
Estimated Disposal Cost with RCRA	3,631,750,000	1,198,700,000
Estimated Disposal Costs without RCRA	2,699,650,000	1,148,140,000
Estimated Cost Increase of RCRA	932,100,000	50,560,000
Groundwater Monitoring Costs	-	4,000,000
Additional Costs of RCRA Including Monitoring	$ 932,100,000	$ 54,500,000

4.2.3.4   Costs for Existing Plus New Plants

Within the context of the stated limitations, the estimate of the cost of compliance with RCRA for coal-fired electric utilities was made by adding the cost of existing facilities to the costs for the planned and future facilities. The capital investment cost through the year 1985 (using mid-1979 dollars) is $981,175,000. The annual revenue requirements are $67,935,000/yr which includes the costs of groundwater monitoring. These costs represent an estimate of the added expense of RCRA, that is the cost over and above the cost of disposal in the absence of such regulations.

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One important factor that needs mention is the cost of liners. Much of the analysis of compliance with RCRA is based on the assumed need to line ponds with some impermeable layer to slow or prevent the intrusion of leachate into groundwater. The costs developed here used a layer of 0.3 meters of clay. The permeability of clay is dependent on the physical properties of the specific type of clay and a range of permeabilities are commonly reported for clay. These are typically in the range of 10^{-5} to 10^{-8} cm/sec. There are numerous liners available that can achieve this performance, including a mixture of fly ash and scrubber sludge in some cases. Synthetic liners such as ployethylene, polyvinyl chloride or butyrubber, and others can achieve greater protection from a leachate penetration standpoint. The cost of commercially available liners is rather high, however.

Using the TVA study[61] as a basis for cost comparison, the relative costs for the various liner options are presented in Table 4-18. Note that the capital investment for the synthetic liner costing $4.50 per square meter is some 86 percent greater than for no liner. It is some 55 percent greater than for clay liners while the operating costs are 77 percent greater than no liner and 116 percent greater than for clay liners.

TABLE 4-18

RELATIVE COMPARATIVE COST OF POND LINERS[61]

	No Liner (Base Case)	Clay	Synthetic Liners $1.50/m^2	$4.50/m^2
Relative Capital Cost	1.00	1.20	1.50	1.90
Relative Revenue Requirements	1.00	0.82	1.40	1.77

4.2.4   Summary of Cost Analysis

The assumptions that have the greatest effect on
the cost estimates are the ones concerning (1) how the plants
would comply with RCRA associated regulations, (2) what those
regulations might be, and (3) the applicability of the cost data
for the generalized calculations made in this study. Several
important factors were not included in the analysis of the cost
of compliance. One important question is the fate of any existing
disposal facilities abandoned because of danger to groundwater.
If such facilities are forced to remove the waste materials
and recover the site, the costs would be very high. If the
facility is allowed to cover ponds or landfills and abandon them,
the impact will be limited to the costs associated with the
mature retirement of the facility.

Another important cost consideration that was not in-
cluded in the study was the cost of building a levee to protect
ponds from flooding. As the potential impact of the levee on
flood water retention is a major factor, a study must first be
conducted to determine if such structures can provide protection
while allowing passage of flood water. If levees can be constructed
that will be in compliance with these restrictions, the cost
of construction and maintenance will be highly site dependent.

The cost estimate that was generated was based on the
approach of costing various disposal options for a "typical"
1,000 Mw coal-fired plant. These cost data were used to estimate
the cost for the entire generating capacity assumed to be affected.
This approach is in contrast to calculating the answer on a per
plant basis or per ton or cubic meter of waste material. The
cost data presented in the previous section along with the plant
design data can be used to generate these and other calculations
methods.

In general, the cost data used to indicate that the cost of disposal rises sharply with the distance from the plant. Ponding is economical only at very short distances. At greater distances, the cost of pumping is very high. The cost of liners is also very important. The cost differences between clay liners and synthetic liners may vary by 50 percent or more. When the capital investment is in the range of 10 to 50 million dollars per plant, a difference of 50 percent is large.

A summary of the cost evaluation used to estimate the cost of compliance with RCRA is presented in Table 4-19. The cost of well monitoring has been added to the revenue requirements for the two classifications of facilities.

4.3   Alternative Disposal Technologies

The discussion of disposal methods thus far has centered on the technologies in widest use today. The environmental impacts and disposal costs have been linked directly to them. In this section, alternative disposal technologies that show some potential for disposal of the large volume of wastes generated in coal-fired power plants are evaluated. These are mine disposal, ocean disposal, and landfarming. For each method, the advantages and disadvantages are addressed, the regulatory restrictions are discussed and the general cost considerations are evaluated.

4.3.1   Mine Disposal

The large number of mines in the United States represent a considerable potential for the disposal of power plant wastes and solid wastes in general. At this time, there are over 15,000 mines within the United States which produce over 0.5 billion tons/year

TABLE 4-19 SUMMARY OF ESTIMATED COST OF COMPLIANCE
(MID 1979 DOLLARS)

	Capital Investment Costs ($)		Revenue Requirements ($/yr)	
	Estimated Cost of Compliance with RCRA	Estimated Current with Predicted Cost – No RCRA	Estimated Cost of Compliance with RCRA	Estimated Current or Predicted Cost – No RCRA
Existing* Plants (1970-1978 Construction)	49,075,000	0	68,725,000	55,350,000
Planned and Future Facilities (1978-1985)	3,631,750,000 –	2,699,650,000	1,202,700,000.	1,148,140,000
Total Costs	3,680,825,000 –	2,699,650,000	1,271,425,000 –	1,203,490,000
Net Costs	981,175,000		67,935,000	

*Assuming 50% out of Compliance

of coal and 2.5 billion tons/year of minerals. Much of this capacity is not suitable for mine disposal purposes.[62] Any number of reasons may be sufficient to rule out a particular mine as a potential disposal site. The mining method, local geological conditions and hydrology must all be considered.

4.3.1.1 Process Description

The two primary variables in the overall process of mine disposal are the mine type or mining method and the actual waste handling method. In terms of technical feasibility, the following four mine types have been ranked in terms of disposal potential:[62]

- Surface coal mines,
- Underground room-and-pillar coal mines,
- Underground room-and-pillar limestone mines, and
- Underground room-and-pillar lead/zinc mines.

This ranking was based on the estimated capacity for sludge, ease of disposal, prevention of future resource recovery and general proximity to sludge sources. Mine configuration and mining methods have been found to have relatively little influence on the environmental acceptability and operational feasibility of the disposal method, except as they are related to local geology and hydrology.[49]

The method of waste placement must be compatible with ongoing mine operations where the mine is still active. Compatibility with existing physical conditions must also be a consideration in the case of abandoned mines. Sludge placement may be accomplished in surface mines either in the working pit, in the

spoil banks, or mixed in with the overburden. The exact method chosen depends on site specific mine conditions and operational procedures. In underground mines the primary methods available for waste placement are hydraulic backfilling in the case of a slurry and either mechanical or pneumatic stowing in the case of dry wastes.

In the feasibility study previously mentioned[49], three different mine sites were evaluated for the disposal of FGD sludge. One potential benefit from this disposal practice emphasized in the report was the prevention or reduction in mine subsidence problems associated with specific mining practices. The three mines studies in detail were all underground coal mines. After evaluating the physical nature of the three sites, the specific mine configurations, and local impacts and constraints, the following conclusions were drawn:

"Mine disposal of lime/limestone scrubber sludge produced from flue gas desulfurization processes is attractive from the standpoint of providing an environmentally acceptable disposal means.

The sludge material in combination with aggregate also appears to be of utility in the prevention of mine subsidence. Potential benefits are seen in controlling acid mine drainage and preventing mine fires both chemically and by sealing voids in the mine fill material.

Generally, there appear to be no insurmountable environmental obstacles to sludge disposal in deep mines. Each potential disposal site, however, must be investigated to determine the hydrogeologic controls on mine-water flow, and the location of recharge and discharge areas. In particular, either the geohydrologic environment must effectively contain the sludge and its associated water or the

formation fluids in the environment must be of
equal or poorer quality than that which is derived
from the emplaced material. Of concern in the
geohydrologic evaluation are existence of any
freshwater aquifers, fractures in the overburden,
local piezometric head relations, discharge points
to surface streams, and especially the water-
solids ratio of the sludge."

4.3.1.2   Technical Considerations

In evaluating the suitability of a particular mine
site for the disposal of any solid wastes, a number of specific
factors must be evaluated. The critical or limiting factor will
vary from site to site. The following factors are important in
evaluating any mine disposal plan:

- physiography and regional setting,

- site geology/stratigraphy,

- surface-water hydrology/quality,

- ground-water hydrology/quality,

- mine configuration,

- mine method,

- solid waste characterization (physical and chemical), and

- waste handling/placement method.

4.3.1.3   Economics

The economic analysis of mine disposal is based on an
evaluation of alternative FGD disposal practices[45]. The
analysis is based on waste sludge from a 500-megawatt power plant
burning eastern coal (3.0 percent sulfur, 10.0 percent ash and
0.39 kilograms coal/kwh). This plant produces 331,000 metric tons/

year of dry sludge including ash. The sludge and ash is available
either as dry filter cake (50 percent solids), as 35 percent solids
slurry (thickener underflow), or as treated sludge. No costs are
included for sludge processing, except in the case of treated
sludge where excavation costs are included. Capital costs are
based on a 1978 completion of construction.

Disposal costs were estimated for six different mine
disposal options. The options included both treated and un-
treated sludges, and both onsite and offsite mines. The costs
for the onsite disposal of untreated sludge (or treated, soil-
like sludges) including transfer and intermediate storage ranged
from $2.20 to $3.85 per dry metric ton. This compates to $7.15
to $8.80 per dry metric ton for offsite disposal. For the disposal
of treated sludges requiring the use of ponds or impoundments,
these costs increased by $2.20 to $2.75 per metric ton to account
for excavation costs. Costs do not include site monitoring.

4.3.1.4   Assessment of Information Needs

A good background of general information exists con-
cerning the disposal of FGD wastes in mines. Much of this infor-
mation is of a theoretical nature which therefore leaves many
practical questions unanswered. More work needs to be done con-
cerning waste/mine interaction, both chemical and structural.
Additional studies should address the potential of this disposal
method on a regional basis.

4.3.2   Ocean Disposal

The ocean disposal offers little potential for disposal
of power plant wastes. From a technical standpoint it is a
feasible method for disposal, but at the current time strict

regulations under both the Marine Protection Research Sanctuaries
Act of 1972 [63,64] and EPA Ocean Dumping Regulations [65]
tend to prohibit this method. There are many serious questions
concerning the short and long term harmful effects to benthic
communities as well as other biological communities resulting
from the dumping of large volumes of flue gas cleaning wastes.
At the present time it is unlikely that permits would be issued
for such disposal unless it can be proven that all other alter-
natives are more harmful.

While it is agreed that this method of disposal offers
little promise at this time, the process description, the
findings of several studies on the subject and the estimated
costs of disposal are addressed in this section.

4.3.2.1 Process Description

Several technological options are available for ocean
disposal. The sludge itself may be handled as a slurry or
dewatered previous to being transported. The dewatering may
be to heavy sludge (35% solids) or to a brick-like form (60-70%
solids). Transportation to the disposal site may be by pipeline
in the case of a slurry or by barge with dewatered sludges. The
disposal sites may be on the continental shelf (shallow water)
or in the deep ocean.

The nature of the sludge during and after disposal
must be considered. Dewatered sludges and slurries will dis-
perse and become diluted as they fall through the water column
during disposal operations. Brick-like treated sludges and
dewatered sludges with a sufficiently high solids content will
fall as a cohesive mass through the water until they reach the
bottom. In evaluating the option of ocean disposal for coal-
fired utility wastes, each of these factors must be considered.

In a recent study[45] by Arthur D. Little, Inc. for the
Office of Research and Development, U.S. Environmental Protection
Agency the following four ocean dumping scenarios were chosen
for analysis:

- dispersed dumping on the continental shelf,

- conventional dumping off the continental
  shelf (deep ocean),

- dispersed dumping off the continental shelf,
  and

- concentrated dumping of treated sludges.

Each of these scenarios were evaluated from a purely
mechanistic approach. No consideration was given to regulations
or legal constraints.

The preferred method for ocean dumping, in the absence
of regulatory constraints, involves the use of conventional
bottom-dump barges on the continental shelf.

There are four principal categories of potential
impact for FGD sludge disposal in the ocean. These categories are:
benthic sedimentation, sludge suspended in the water column,
sulfite-rich sludge and trace contaminants. The primary poten-
tial impact of benthic sedimentation is the creation of unsuitable
habitats on the ocean bottom due to the fine-grained nature of
most sludges and the lack of acceptable nutrient levels. The
impact of the suspended sediments in the water column is estimated
to be dependent on the chemical composition of the sludge and on
feeding habits and sensitivities of fish populations. The intro-
duction of sulfite-rich sludge into the ocean environment is of
interest for two reasons. First, sulfite is measurably toxic;

and second, it exerts an oxygen demand on surrounding waters.
There are several factors which would determine the severity of
impact from sulfite-rich sludge on the ocean environment,
including the dissolution rates of the sludge, water pH and
dilution in the area of the dump. In a similar fashion, the
impact from trace contaminants is estimated to be highly variable.

A study by the State University of New York, Stony
Brook is presently examining the use of stabilized FGD sludge
blocks to create artificial reefs for marine habitats.[56] In
a two-phase study both laboratory and in-situ tests are being
conducted. The study was performed in an estuarine area but
the results have some applicability to open ocean disposal as
well. Preliminary results indicate that:

- calcium and certain other species show
  rapid initial solubilization which slows
  as equilibrium is reached,

- no appreciable trace metal leaching was
  noted,

- sludge compressive strength increases with
  increased exposure to sea water, and

- in the in-situ tests, benthic organisms
  attached themselves to the submerged
  blocks within two weeks of submersion.

4.3.2.2 Economics

The Arthur D. Little, Inc., study[45] developed cost
estimates for five ocean disposal options. The five options
were:

- on-shelf disposal of untreated sludge,

- on-shelf disposal of treated, brick-like
  sludge,

- off-shelf disposal of untreated filter cake,

- off-shelf disposal of thickener under-flow, and

- off-shelf disposal of treated, brick-like sludge.

An eastern power plant with ready access to the ocean was assumed. The disposal of untreated filter cake (with ash) on the shelf was estimated at $4.40 to $5.50 per dry metric ton of sludge. Treated sludge requiring ponds or impoundments would cost $2.20 to $2.75 to cover the cost of excavation. Deep ocean (off-shelf) disposal costs $3.30 to $4.40 per dry metric ton more than on-self disposal. Thickener underflow disposal costs $1.10 more per metric ton than filter cake disposal. These cost estimates do not include monitoring or sludge processing costs.

4.3.2.3 Assessment of Information Needs

While a sufficient amount of information does not exist to justify the current restrictions as ocean disposal, there are several information gaps that could be addressed. Some estimates have been made concerning the chemical and physical interactions between sea water and sludge, but information concerning biological impacts of sludge disposal in the ocean environment is needed. It may be necessary to propose changes in existing ocean dumping regulations, should the results of ongoing research indicate that ocean disposal of power plant wastes is an acceptable method. The acceptability argument must be taken in light of the potentially harmful effects of land disposal as compared to ocean disposal of these waters.

4.3.3   Land Farming

Very little information is currently available on
the potential for landfarming, adding FGD sludges and solids
to the soil. As little or no nutrient material is available
from these wastes they are not a potential fertilizer. The
cost of spreading the material, particularly the large quantities
available, would be rather prohibitive when considering this
as a disposal option.

There may be some limited applicability as a soil
conditioner, however. Where soils tend to be too acidic, basic
fly ash material which has the opposite characteristics could
be used to neutralize the soil. High sodium soils and clays
can be stabilized to some degree with the addition of high
calcium fly ashes. In such cases the disposal costs could be
passed on to the user as opposed to the plant. The quantities
needed would not encompass the entire output, however. Such
cases would tend to be highly site specific and this method
is not believed to offer a potential for widespread use.

5.0 UTILIZATION OF UTILITY WASTES

For coal-fired utility plants, recovery practices may be directed toward fly ash, bottom ash and/or boiler slag and flue gas desulfurization scrubber sludge utilization in some form. In 1977 total U.S. production of fly ash, bottom ash and boiler slag was 67.8 million tons with 14.0 million tons success-fully recovered and utilized.[67] This is more than three times that used by any other world nation during the same time period. However, this 14 x 10^6 tons corresponded to only 21 percent utilization compared with values for England and Wales of 41 percent, for France of 42 percent, and for Germany of 60 percent. [68,69,70] There are, of course, inherent differences in avail-ability of raw materials and marketability in each country which account for differences.

Utilization of ash is expected to continue to in-crease in the United States as indicated in Table 5-1 with trends dating back to 1966.[67,71] The increasing reliance on coal as a utility fuel with attendant increases in ash produc-tion may very well mean that the percent of utilization will not increase, and may decrease despite efforts to promote ash utilization through increased market visibility and technolog-ical development. Competitive utilization of ash based upon market development or technological improvement in recovery or recycle methods would be desirable from both the standpoint of total resource utilization and of reduction in the quantity of waste requiring disposal.

The utilization of calcium sulfite/sulfate based FGD scrubber sludge is and has been much more limited even than that of ash. In addition, definitive data on the actual amount of material utilized is not readily available. In effect, the

TABLE 5-1

COMPARATIVE RESULTS

	1966*	1974	1975	1976	1977
ASH COLLECTED					
Fly Ash	17.1	40.4	42.3	42.8	48.5
Bottom Ash	8.1	14.3	13.1	14.3	14.1
Boiler Slag		4.8	4.6	4.8	5.2
TOTAL ASH COLLECTED - TONS x 10⁶	25.2	59.5	60.0	61.9	67.8
ASH UTILIZED					
Fly Ash	1.4	3.4	4.5	5.7	6.3
Bottom Ash	1.7	2.9	3.5	4.5	4.6
Boiler Slag		2.4	1.8	2.2	3.1
TOTAL ASH UTILIZED - TONS x 10⁶					
PERCENT OF ASH UTILIZED					
% Fly Ash	7.9	8.4	10.6	13.3	13.0
% Bottom Ash	21.0	20.3	26.7	31.5	32.6
% Boiler Slag		50.0	40.0	45.8	60.0
PERCENT OF TOTAL ASH UTILIZED	12.1	14.6	16.4	20.0	20.7

* First year that data was taken.

** 1967-1973 data omitted from tabulation because of space limitation.

Source: Reference 67.

situation with regard to scrubber sludge utilization is more one of conceptual development and testing rather than actual widespread utilization. On recent summary of conceptual developments[72] listed the following possible uses for scrubber sludges: (1) recovery of chemicals, (2) manufacture of building materials, (3) structural fill, (4) paving materials, (5) soil stabilization in agriculture and (6) environmental pollution control. Among all of the concepts evaluated, it was concluded that only the production of gypsum wallboard and the utilization of gypsum in making Portland cement have any possibility as a significant outlet for scrubber sludge in the foreseeable future. This conclusion, of course, assumes that the collection process in the FGD system involves forced oxidation of the scrubber sludge to gypsum with separate disposal of waste product in a manner that allows recovery.

It can be seen, that many conceived uses of scrubber sludge parallel uses of ash. Since ash utilization is more developed and ash is a superior product to FGD scrubber sludge in many of these parallel uses, significant utilization of FGD wastes is not expected when it is unlikely that the percentage utilization of ash itself will increase. This conclusion is not to minimize the significance of the total quantity of FGD scrubber sludge that might be utilized in a given location where market considerations create a need for the raw material in such processes as Portland cement production and in wallboard manufacture.

5.1   Fly Ash Utilization Practices

Fly ash utilization practices in the United States during 1977 are summarized in Table 5-2. The two major areas of commercial utilization were in the partial replacement of cement in concrete and concrete products and in the structural landfill embankments and road construction. Lightweight

TABLE 5-2

ASH COLLECTION AND UTILIZATION 1977

(Million Tons)

	Fly Ash Tons x 10⁶	Bottom Ash Tons x 10⁶	Boiler Slag (if separated from Bottom Ash Tons x 10⁶
1. TOTAL ASH COLLECTED	48.5	14.1	5.2
2. ASH UTILIZED	6.3	4.6	3.1
3. UTILIZATION PERCENTAGE			
A. COMMERCIAL UTILIZATION			
a. Mixed with raw material before forming cement clinker	7	--	3
b. Mixed with cement clinker or mixed with cement (Type 1-P cement)	5	2	--
c. Partial replacement of cement in concrete and blocks	25	--	--
d. Lightweight aggregate	2	3	--
e. Fill material for roads construction sites, land reclamation, ecology dikes, etc.	20	20	8
f. Stabilizer for road bases, parking areas, etc.	3	5	2
g. Filler in asphalt mix	2	--	--
h. Ice control	--	22	13
i. Blast grit and roofing granules	--	--	48
j. Miscellaneous	3	9	22
B. ASH REMOVED FROM PLANT SITES AT NO COST TO UTILITY	7	17	4
C. ASH UTILIZED FROM DISPOSAL SITES AFTER DISPOSAL COSTS	26	22	--
	100	100	100

Source: Reference 67.

aggregate use in the United states was surprising low, 120,000 tons, accounting for only 2 percent of the total utilized. This is in contrast to the significant use within England and Wales of 353,000 tons or 8 percent of total utilized.[68] Other uses in the U.S. included mixing with cement or cement clinker, road base stabilizer, and asphalt mix filler.

5.1.1   Partial Replacement of Cement in Concrete and Concrete
        Products

Fly ash used as replacement material for cement in concrete has been found to produce beneficial effects provided good quality control is maintained along with specifications for the fly ash utilized. Some of the beneficial effects are: (1) reduced water requirements (2) improved workability of the concrete, (3) improved finishing qualities, (4) reduction in heat generation during hardening reactions (5) increased strength for a given cement composition, (6) reduction in permeability and absorption, (2) improved resistance to chemical attack, (8) improvement in uniformity of strength development and (9) decreased cost of concrete manufacturer at a given quality.[73] However. the realization of some or all of these benefits rather than detrimental effects such as difficulty in control of air content, slow strength grain, poor freezing and thawing resistance depend upon attention to physical specification and quality control regarding such factors as fineness uniformity, moisture content, air entrainment, and composition and particle size of the fly ash utilized.

Recently, the American Society for Testing and Materials adopted and published standard specifications (ASTM C618-77)[74] for the use of fly ash as a pozzolan in concrete manufacture. Fly

ashes were classified in these specifications as either derived
from burning (1) anthracite or bituminous coal or (2) lignite .
or subbituminous coal with physical and chemical requirements
given for each class. The specifications covered such parameters
as combined percentages of SiO_2, Al_2O_3 and Fe_2O_3, maximum SO_3,
maximum moisture content, maximum loss on ignition, maximum MgO
content, maximum Na_2O, particle fineness, pozzolanic activity
index, soundness and uniformity requirements. The adoption and
acceptance of these standards should enhance this avenue for
fly ash utilization.

It is reasonable to expect an increase in use of fly
ash as an admixture in concrete production provided there is
aggressive marketing and attention to quality control. However,
a major factor in the economic competitiveness of fly ash as a
pozzolan is transportation costs. This factor may limit wide-
spread utilization in the foreseeable future to those areas of
the country where there is reasonably close proximity of plants
generating fly ash as a waste product to areas of active con-
struction utilizing significant quantities of fly ash-based
concrete. It is unlikely that fly ash transported over long
distances would be economically competitive with other available
materials.

5.1.2   Base and Subbase in Road Construction

Demand for aggregate materials in base and subbase
construction in the United States runs into billions of tons
each year.[75] In some parts of the country there exists both
a shortage of suitable material and existing material on a
limited basis that is increasingly costly. A potential market
for fly ash exists in areas of the county where large amounts of
bottom ash, boiler slag or fly ash are produced and there is
a limited supply of road construction material.

However, in spite of the fact that tests on fly ash used in road construction have shown that it is comparable to other aggregates in terms of structural specifications, resistance to its use still must be overcome because of a lack of knowledge of its beneficial properties on the part of engineers. There is a natural tendency to utilize experience-proven natural aggregate where it is available even though testing has demonstrated the validity of fly ash use. Nonetheless, more than thirty states under the increasing pressure to find low-cost road construction materials have developed specifications for the use of power plant ash in new construction as well as maintenance programs. In addition, the Federal Highway Administration has sponsored research programs designed to increase the use of ash in road construction.[71]

Even though technical utilization of fly ash for road construction as a subbase or base material seems adequately demonstrated, there seem to be two limiting factors to its use. In the limited sense, transportation from a generating plant to site use is a controlling factor. It has been found that transportation distances of greater than 50 miles by truck or 100 miles by rail make the use of fly ash unattractive.[75] In the more general sense, there is concern over possible contamination of water supplies by leachate that might result from water in contact with fly ash used in road construction. Unless the fly ash was effectively "sealed" from surface water contact by an asphalt or concrete covering, then concern over leachate potential will continue to exist. However, the actual threat from leaching is not known and testing would be required to establish actual results. Any such testing would likely be site-specific to a high degree and would not answer questions of leachate potential in any general sense. Despite possible limitations, the use of fly ash in base and subbase construction is definitely expected

to increase and be especially significant in areas of the country where natural aggregate materials are in low supply and coal-fired electrical generation occurs on a large scale.

5.1.3   Cement Manufacture

Fly ash has been used in cement manufacture both by mixing fly ash with finished Portland cement and by intergrinding with Portland cement clinker prior to forming a finished product. Several companies now manufacture a type I-P cement containing fly ash. There are ASTM and Federal specifications covering the required properties of such cements and basically permit any suitable pozzolan (including fly ash) to be utilized along as specifications for the final product are met. The I-P type of cement is used for general construction purposes upon which specifications are built. These specifications permit both interblending or intergrinding in the manufacture of cement but stipulate that the pozzolan constitute from 15-40 weight percent (ASTM)[77] or 15-35 weight percent (Federal) of the finished pro-duct.

In addition to the above, fly ash has been and is used in several locations as a raw material in the manufacture of Portland cement. In most cases, the fly ash is used to compensate for deficiencies in material composition such as alumina and usually constitutes no more than 5-10 percent by weight of the final product. Fly ash as a raw material is not bound by any specifications, since specifications are applied rigorously to the final product and the fly ash looses its physical identity in the final product. When fly ash is used as a raw material to provide certain chemical constituents, its use is limited by the fly ashes variable composition particularly as it relates to possible high values of magnesium, sulfate and alkalies.

Limiting factors to the use of fly ash in cement manufacture either as a pozzolan or raw material are often the same as those affecting fly ash use in concrete manufacture, namely, market inpenetrability due to low confidence in a by-product, concern over variable composition of fly ash, constancy of supply and economic concern over transportation costs if long distance hauling is involved. As in the case of concrete manufacture, use should continue to increase with aggressive marketing and acceptance of specifications provided the manufacture is taking place in a geographical location where supplies are available from power plants without inhibiting transportation costs.

5.1.4 · Lightweight Aggregate

Production of mineral aggregates in the U.S. now amounts to more than 2 billion tons on an annual basis and all aggregates combined constitute the largest single mined commodity in the United States.[75] However, the supply is decreasing and in certain areas of the country (particularly some industrial or metropolitan areas) demand for natural mineral aggregates for construction purposes exceeds locally available supplies. Consequently, the search has been made for substitute materials with fly ash and bottom ash coming under both consideration and actual use. With increasing energy costs and subsequent costs in transportation there has been a driving force to find suitable replacement aggregates in the local area rather than transport over large distances. Therefore, there is definite potential for growing use of fly ash as a lightweight aggregate in many parts of the country where there is a coupling of shortages and coal-fired generating plants in the same area. There are apparently few significant technical limitations to the use of fly ash for lightweight aggregate, and it is feasible

that this particular application holds promise for the greater
percentage increase in fly ash utilization among all categories
of use. Possible limitations which may require further atten-
tion in the future are variable density among fly ash sources
and a potential for water contamination through leaching for some
applications of lightweight aggregate use.

5.1.5   Filler in Asphalt Mix

Mineral fillers have been used for some time to in-
crease the stability and durability of asphalt paving surface.
Normally the mineral filler constitutes no more than four percent
by weight.[78]   Materials which have been used for this purpose
include limestone dust, volcanic ash, hydrated lime, powdered
shale, Portland cement, bentonite clay, mineral sludges and
fly ash.

From a technical view point, the major limitation on
use of fly ash as a filler is its poor traction characteristics.
However, it does provide adequate overall paving characteristics
and since filler specifications allow the use of a low grade
fly ash from the standpoint of quality control, its cost is
attractive in comparison with other filler options. Consequently,
in areas where a reasonably constant supply is available from
the utility in close proximity to the point of use, a signi-
ficant outlet market for fly ash should be sustained. However,
since the use is largely seasonal and the percentage use of
filler is small, the asphalt filler market could not be an
outlet for much more than one million tons per year and would
be localized in certain areas of the country.

5.1.6   Future Uses

Beyond the "major" outlets for power plant fly ash mentioned above, numerous other uses have been conceptualized or demonstrated to some degree with several deserving special mention.   In some instances, these other uses await the surmounting of technical problems, and in other cases market awareness of the feasibility and advantages of the product or particulate use must be developed.   An example of a future potential use requiring technological breakthrough for economic competitiveness is mineral recovery from fly ash.   Much research has been done and methods for extracting alumina, magnetite or other minerals do exist.   However, these processes are not capable of competing with more established processes in some cases and in others the market does not exist because of a readily available supply of the mineral.

By the year 2000, the minerals deficit in the United States will exceed the energy deficit; the trade deficit in minerals may be as high $100 billion dollars witnin 25 years.[79] Currently, U.S. sources for 22 of the 74 non-energy essential minerals are almost completely dependent on foreign sources.   Of the crucial 12 elements, seven are imported in quantities greater than 50% of our need.[79] Utilization of the aluminum content of the fly ash can completely offset bauxite imports currently.[80] Zinc concentrations in the fly ash are equivalent to zinc mined from commercial sources.   In light of an impending minerals crisis and a critical shortage of certain industrially crucial trace elements, the presence of these constituents in fly ash and the large amounts of fly ash available are strong reason and justification for basic research and development of extraction processes and incentives on the state and national level for the effective use of these constituents as a valuable natural resource.

A possible use, requiring market development in the U.S., is the production of gas concrete or lightweight concrete as a building material. Large amounts of fly ash are utilized for this purpose in Europe where this product has become a widely used and popular construction material. In Europe, the material has been found to reduce construction costs by as much as 20-30 percent over conventional brick and the use has steadily grown in over twenty countries.[78] Were this use to gain favor in the U.S., manufacture of gas concrete could become a major use of fly ash in the future.

Other uses providing a significant outlet for fly ash are soil amendment, production of cenospheres and utilization in mineral based insulation.

All of the above discussion of actual and potential fly ash utilization assume that the material is available in an unmixed form through dry collection procedures such as electro-static precipitation or mechanical collection. There is some-what of a tendency to date to find air quality control systems at increasing numbers of utility plants in which fly ash is collected in large portions along with SO_2 in a wet scrubbing system with limestone or lime as the sorbent material. The resulting mixture of wet fly ash and scrubber sludge does not have the same properties as separately collected fly ash and could not be used in many of the applications discussed above. If fly ash utilization is a desirable goal and proves financially beneficial in some instances, the altering of properties through wet collection along with SO_2 should be avoided.

5.2   Bottom Ash Utilization Practices

In many instances, bottom ash can be used in a parallel fashion to fly ash. There are general similarities to con-

stituents on a chemical basis, but there are significant dif-
ferences in composition on the basis of those constituents.
In addition, there are significant differences in particle
morphology and physical characteristics. For example, fly ash
is typically a fine material of glassy spheres, whereas bottom
ash is typically a more coarse material of a cinder-like nature.
Therefore, some of the uses are significantly different from
those of the fly ash.

In addition to the bottom ash obtained from dry boiler
operations, boiler slag is another distinctive mineral byproduct
of combustion resulting from wet boiler operations. The uses
of bottom ash and boiler slag overlap to a large extent, and
consequently, their uses will be considered together. On a
percentage basis the utilization of bottom ash and boiler slag
exceed that of fly ash. This is shown in Table 5-2 for utili-
zation of ash for 1977.[67] However, it must be realized that
the uses of bottom ash and boiler slag have been limited in
nature. The significantly lower quantities of bottom ash and
boiler slag being produced result in a higher percentage of
utilization than fly ash rather than more extensive outlets
for these materials over fly ash.

5.2.1   Use in Manufacture of Cement

Bottom ash finds some use as a mineral filler in the
manufacture of cement. In actuality, the bottom ash is used
as a raw material in the production of cement clinker and as a
means of providing required mineral mass balance. In 1974, this
particular use amounted to less than one million tons[71], but this
outlet is still one of the major utilizations of bottom ash in
the United States.

Bottom ash as a raw material ingredient in cement manu-
facture parallels the advantages and disadvantages given for

fly ash used in this manner, because both serve as an extender
for mineral constituents. The market for this use should not
only be sustainable in the future, but should see growth with
a corresponding increase in all types of construction.

5.2.2   Lightweight Aggregate and Aggregate in Road Construction

The growth in use of lightweight aggregates (a
doubling of use is expected in the next 10 to 20 years) and the
shortages expected to occur in certain urban areas has been
mentioned in the section above on use of fly ash as lightweight
aggregate.

Bottom ash has found use as a lightweight aggregate
in construction needs and has been utilized as an aggregate
(gravel) material in road base construction. Extensive testing
has been done on the physical and structural characteristics
of bottom ash and boiler slag for various purposes in road
construction.[81] This has included use for nonstabilized bases,
stabilized bases (either Portland cement or bituminous stabilized),
bituminous paving mixtures and underdrain filter material. It
has been generally concluded that bottom ash and slag can provide
properties comparable to those resulting from natural aggre-
gates, but that the specifications developed for the use of
material aggregates require modification to achieve the best
results when using power plant aggregates. With modification,
performance of the power plant aggregates in road construction
can equal or exceed that found for natural aggregates.

As in the case of fly ash, the increasing acceptance
of bottom ash and boiler slag as aggregate materials and the
development of appropriate specification for their particular
unique properties will provide an outlet in those areas of the
country where large amounts of coal-fired wastes are being

produced. Fortuitously, a number of urban areas expected to have shortages of natural aggregates are located in regions of the country where large amounts of power plant aggregate are being produced. The proximity should reduce transportation costs to the point where the ash products are economically attractive. Consequently, aggregate use represents one of the better alternatives for increasing utilization in the future.

5.2.3   Filler in Asphalt Mix

Both bottom ash and boiler slag have been used in asphaltic concrete for paving purposes, although boiler slag has a more extensive history of such use. Boiler slag received considerable promotion after World War II as a desirable surfacing aggregate under the trade name of "Black Beauty" with the heaviest use coming in the midwestern United States.[82] Even though more extensive quantitative data is needed, asphaltic surfaces with boiler slag as an aggregate poses anti-skid properties as compared to such a natural aggregate as limestone sand.

Bottom ash in asphaltic paving has been most extensively used in West Virginia where since 1972, dry bottom ash has been cold mixed with emulsified asphalt and used to pave rural secondary roads.[81] The performance of such surfaces has been reported as favorable to this point, with good experience also being reported in supply of the materials from the power plants involved. In addition, extensive testing of bottom ash and boiler slag for asphaltic surface production has been conducted at West Virginia University. That testing clearly established the feasibility of both materials in asphalt mix provided specifications are modified to accomodate the unique nature these by-products. Additional laboratory and field testing should be conducted with bottom ash and slag from the burning of

subbituminous and lignite coals because the bulk of experience has been obtained with eastern bituminous coals. However, the West Virginia results have paved the way for acceptance of this use and there is no reason why this market cannot be successfully developed.

5.2.4 Abrasive Material for Skid Control Under Snow or Icing Conditions on Roads

Because bottom ash is a cinder-like material with angular particles, it has found rather extensive use by highway and road departments to improve traction as a gut material on icy roads and bridges. In fact, several plants gave away bottom ash at no cost to highway or city road departments for the purpose of skid control at a number of locations. As natural aggregates become less available and more expensive, it is likely that power plant bottom ash and boiler slag will become more attractive because of their excellent traction properties, and utilities will be able to move beyond "give-away" of this product to active sale.

There is no major technical limitation for this use of bottom ash. As it gains acceptance by engineers and as natural aggregate use becomes less attractive, this outlet will grow where power plants are advantageously situated and can guarantee constant and adequate supply during crucial periods of need.

5.2.5 Other Uses

A growing interest is apparent at present in the use of boiler slag grit material in cleaning - blasting replacing sand. This interest is not just a matter of unavailability of sand or other commonly used material, but there are apparently

tangible technical advantages for boiler slag based upon its properties. Bottom ash has been used to control mine subsidence, to neutralize and abate acid mine drainage, and as a structural fill material particularly in enbankment areas where drainage is a problem.

Parallel to the case with fly ash, both bottom ash and boiler slag are candidate precursor materials for mineral extraction or trace element extraction. The demand for this use, the technical limitations, the potential for future development and the need for incentives are identical to those points discussed earlier in regard to fly ash.

5.3   Utilization of FGD on Scrubber Sludge

The basic overview of potential uses of FGD sludge and actual practices has already been given at the beginning of the discussion on waste utilization. Actual use at the present time is nil and no widespread efforts are being made to develop utilization. There is active interest in some recovery possibilities such as the production of elemental sulfur. However, that particular potential is one awaiting research and development that will lead to a process that is economically competitive with more conventional sulfur production processes. The production of sulfur and sulfuric acid will be discussed in Section 5.5.

As more FGD systems begin to utilize stabilization procedures with landfill of sludge, the modified and stabilized sludge might find an outlet for use in structural fills and enbankments. Often, the stabilized sludge contains significant amounts of fly ash either as a result of the stabilization process or through mixing of stabilized sludge and fly ash at disposal. The structural properties of stabilized sludge are such that it would frequently be advantageous for landfill involving landscaping architecture or strip mine reclamation.

5.4   Factors Limiting Waste Utilization

A survey of current utilization of by-product material
from flue gas cleaning processes at coal-fired utility plants
indicates that there are at least three areas of resistance or
limitation to their increased use. These areas are (1) technical
limitations in comparison to alternative materials, (2) insti-
tutional barriers related to poor understanding of the by-
products and failure to develop markets by either the utility
industry or user industries, and (3) possible environmental con-
cerns related to some uses.

When one considers the growing volume of waste material
that must be dealt with in an economically and environmentally
satisfactory manner, removing or reducing these barriers becomes
a high priority item. In some instances, technical limitations
preclude further significant progress for a given application,
but in other cases basic research and development will surmount
obstacles. The most damaging barriers are apparently human and
corporate, but seemingly can be overcome by concerted efforts.
Utility companies have previously been concerned primarily with
the production of electricity. Marketing of waste by-products
has been secondary. Wastes have been viewed as a nuisance and
liability rather than as a potential asset to be sold and produced
along with electricity. Not only must the utility companies
envision the importance of marketing their waste products, but
they must aggressively develop markets with the educational pro-
cess being a key role in overcoming the reluctance on the part
of many users to use a waste product because of fears involving
chemical and physical variability of the material, lack of
specifications for its use in their manufacturing processes, and
fear of a lack of constant supply depending upon the vagaries of
plant operation. Once a market is developed, plant operations

must be geared to accommodate the customer using the waste product and provide maximum cooperation in regard to access to waste facilities of loading and transportation and cooperative efforts in monitoring quality control.

Finally, for those uses where a potential environmental concern exists, policy decisions based upon the relative merits of use in a particular application compared to the environmental impact of landfilling or ponding must be made so that a climate of uncertainty regarding future regulations does not exert an inhibiting effect on use.

5.5   Regenerable FGD Processes

This section will discuss regenerable FGD processes and the possible impact that this technology could have on the quantity and type of waste produced through flue gas cleaning associated with coal-fired electrical generation. Regenerable processes combine absorption of SO_2 by a suitable alkaline reagent system with a second step which regenerates the sorbent material and produces a by-product such as sulfuric acid or elemental sulfur. In comparison, the more widely used wet scrubbing systems throw away any solid waste produced. The previous examination of the use of calcium sulfate/sulfite scrubber sludge led to the conclusion that the beneficial utilization was limited in the immediate future with no large scale use at the present time.

In essence, the regenerable processes described here replace large volumes of scrubber sludge with an alternative by-product (H_2SO_4 or S). Given the limited utilization of scrubber sludge, the comparison of throw-away systems with regenerable processes must be made not only between the economics of operating the systems, but the overall product processing from regenerable processes, and specifically, the potential market for by-products.

A brief description of selected regenerable processes will be given for those processes which have the most successful operating history. The following five processes have been considered as they have or soon will be demonstrated and tested on a coal-fired boiler of 50 megawatts rating or higher.

- Wellman-Lord

- Magnesium Oxide Slurry Adsorption

- Aqueous Absorption with Citrate Buffering

- Integrated Cat-Ox Process

- Aqueous Carbonate Process

Following the discussion of these demonstrated systems, selected "advanced" technology processes which might improve the state-of-the-art of regenerable process technology will be identified.

Reduction of disposal waste volume is a major driving force for regenerable processes. In part, the advantage of a regenerable process is dependent upon this waste reduction, i.e. a marketable by-product taking the place of a throwaway product. The combination of growth in coal-fired capacity and more stringent environmental standards will lead to very large increases in total waste (both ash and FGD scrubber sludge) to be managed through the balance of the century. It has been estimated,[83] for example, based upon current SO_2 emission standards, that if all new units install nonregenerable FGD systems, that the wastes produced from those plants will be approximately 20 million dry tons annually (ash and FGD sludge) by 1980 with nearly one-half being ash. This compares with the approximately 3 million tons of dry FGD sludge being produced at 1977 levels. The total wastes would grow to be about 80 million dry tons in 1990 and 155 million tons by 1998 if nonregenerable systems are utilized. Also, the total waste volume would conservatively be doubled from these dry amounts if 50 percent moisture in the wastes is assumed.

Use of regenerable FGD systems can substantially reduce the volume of sludge produced even though total wastes (because of ash) will still be high. Again, with current emission standards, it has been estimated[83] that if half the new units install regenerable systems and one-half lime/limestone throwaway systems,

then the total waste volume in 1990 would be 60 million tons
(dry) rather than the 80 million given above with approximately
40 million tons of the waste being ash. The total quantity of
waste still represents a significant technological challenge for
proper disposal environmentally or from the standpoint of new
developments in waste product utilization. Nonetheless, the
impact of regenerable systems is seen by the estimated 20 million
ton reduction in FGD sludge produced by the installation of re-
generable systems in only one-half the units between the present
and 1990.

Beyond reduction in the amount of FGD waste, regenerable
processes must ultimately be demonstrated as advantageous from the
standpoint of the following factors which will be discussed in
the final section dealing with the possible impact of regenerable
processes:

- Waste streams which are produced in regenerable
  systems and requiring disposal

- Immediate and longer-range marketability of
  possible by-products (H_2SO_4 or S)

- Feasibility of stockpiling by-products such
  as elemental sulfur in anticipation of a
  future market

5.5.1   Wellman-Lord

The Wellman-Lord recovery process coupled with the
Allied Chemical SO_2 to S process has been in operation since
June, 1977, at Northern Indiana Public Service, Dean H. Mitchell
Plant with 115 Mw capacity. Performance testing was completed
prior to June, 1977. Actual performance data during a 12-day
acceptance test period revealed (1) 91 percent removal efficiency
for SO_2, (2) particulate emissions of 0.04 lbs/10^6 Btu, and (3)
production of a sulfur product of 99.9 percent purity.

The Wellman-Lord process is based upon the reaction of a sodium sulfite solution with SO_2 to produce sodium bisulfite. Fly ash and chlorides are removed separately and prior to SO_2 scrubbing. Some sodium sulfate is produced in the process and must be purged in the regeneration process before recovered Na_2SO_3 is returned to the absorber. Thermal regeneration in an evaporator to produce Na_2SO_3, water and a concentrated stream of SO_2 is used. The SO_2 stream can be used to produce either sulfuric acid or elemental sulfur.

5.5.2   Magnesium Oxide Slurry Adsorption

The magnesium oxide process has been in operation since September, 1975, at Philadelphia Electric's Eddystone No. 1A Plant with a generating capacity of 120 Mw. In this particular operation, performance testing demonstrated SO_2 removal efficiencies exceeding 95 percent. A 98 percent H_2SO_4 product was produced along with regenerated MgO to be used in the absorption process. The H_2SO_4 product was marketable. Unit reliability was disappointing during performance testing with cumulative availability being only 32 percent with a longest continuous run of 140 hours.

A wet scrubbing system composed of an aqueous slurry of magnesium oxide sorbs SO_2 to produce magnesium sulfite. The slurry is dried and calcined to regenerate the MgO and produce an SO_2 stream. Fly ash and chlorides are removed upstream to avoid contamination of the regenerative process. When used for scrubbing with utility boilers, the process is fairly mechanically complex which has contributed to maintenance problems and low availability.

5.5.3 Aqueous Absorption with Citrate Buffering

The citrate buffered absorption process was developed by the Bureau of Mines for industrial boiler applications with Bureau of Mines operation on a 1,000 scfm pilot unit on a lead sintering furnace tail gas for a 2-year period. An EPA/Bureau of Mines co-sponsored demonstration unit is scheduled to begin operation in 1978 involving a coal-fired utility unit.

In this process, flue gas is cleaned prior to SO_2 absorption to remove fly ash and chlorides. The aqueous stream carrying absorbed SO_2 is transported to a regenerator where it is converted to elemental sulfur by the reaction: $SO_2 + 2H_2S \rightleftarrows 3S + 2H_2O$. The H_2S for this regeneration reaction can be produced by reacting two thirds of the sulfur produced with a reducing gas.

5.5.4 Integrated Cat-Ox Process

This particular regenerable process was tested extensively during a demonstration program in which the Cat-Ox system was retrofitted to the Unit No. 4 boiler at the Illinois Power Company's Wood River Station. The Cat-Ox process utilizes catalytic oxidation of SO_2 to SO_3 using vanadium pentoxide catalyst in a bed fixed arrangement. The SO_2 is then passed into a 75 to 80 percent sulfuric acid stream for absorption.

An upstream electrostatic precipitator operated at 850° to 900° (400° to 500°F above normal) removes particulates and the heated flue gas passes across the vanadium pentoxide bed where SO_2 is converted to SO_3 with 95 to 99 percent efficiency. Even though several operational problems were encountered at the Wood River Plant, the problems should be solvable

by design and construction changes since the components of the process have been successfully tested. The process is not complex and has low raw material and utility requirements among demonstrated regenerable processes. However, the developer seems to have discontinued marketing of the process.

5.5.5   Aqueous Carbonate Process

A 100 Mw FGD demonstration plant using the aqueous carbonate process is under program planning sponsored jointly by the Empire State Electric Energy Research Corporation and EPA. The particular process is the one developed by Atomics International and uses an aqueous solution of sodium carbonate to remove SO_2 in a spray dryer scrubber. The process produces elemental sulfur as a by-product utilizing solid carbon as the reducing agent.

Even though a detailed history of demonstration does not exist in this country the basic process has been evaluated in detail for large coal-fired generating plants (500 and 1000 Mw) in terms of capital investment and operating costs and the process holds promise an an advanced regenerable FGD process.

5.5.6   Advanced Technology Processes

The regenerable processes described above while having a degree of demonstrated success do have additional technological problems to be improved upon or overcome. Beyond reliability and efficiency in SO_2 removal, the systems require the maximum mechanical and chemical simplicity possible, retrofit capability, scale-up potential, load following capability and sub-process separability. In addition, the economics of operation, particularly raw material requirements and utility (energy) requirements

for the regeneration step, must be competitive with the
non-regenerable processes which are currently in more wide-
spread use.

The following selected processes are new technologies
which might improve upon existing technologies.

- Aqueous Adsorption/Steam Stripping to
  produce Concentrated SO_2 Stream.

- Reduction Processes for Producing Sulfur
  from SO_2

- Direct Reduction of Sodium Sulfur Oxides

- Advanced Technology Involving Forced
  Oxidation in Throw-Away Systems

5.5.7   Economic Comparison of Regenerable Processes with Wet
         Scrubbing Lime/Limestone Throwaway Systems

One important factor governing the advantages of regen-
erable processes in comparison to the more widely used throwaway
systems is cost, both in terms of initial capital investment and
also in terms of annual operating expenses. In computing the cost
factor for a regenerable process, a revenue figure for by-product
sale is normally credited against the costs of collection and dis-
posal. This assumes, of course, that a market for the product
exists and that the product is of a quality to be sold as comparable
in nature to the same product from more established sources. This
may not be a valid assumption as will be discussed later, but such
revenue credits are normally assumed in computing costs for regen-
erable processes.

Table 5-3 gives a comparison of both capital costs and annualized operating costs for a selected group of regenerable processes and also for limestone scrubbing. Disposal costs have been included in the limestone scrubbing case while a credit for product sale has been given to the regenerable processes. The comparisons are for the same base case. Both the source of the information and base case assumptions are given. While projections of costs are not without areas of uncertainty, the data does illustrate that regenerable processes in many instances are competitive economically with limestone scrubbing. As is the case with nearly any developing technology, these processes have met with varying degrees of success in terms of system reliability and on line availability. In most cases the technical base exists to "de-bug" the systems and overcome chemical and mechanical problems provided an incentive exists.

If technical problems are surmountable and costs are roughly comparable, then the decision "for" or "against" regenerable processes must be made on the basis of total by-product utilization and the feasibility of marketing both bottom ash and fly ash as well as sulfur or sulfuric acid as a by-product of regeneration.

5.5.8   Possible Impact of Regenerable Processes

It has been reported[84] that between the present and 1990, coal-fired electrical generating capacity will expand by at least 120,000 Mw, through approximately 250 new units consuming 360 million tons of coal annually. The necessity of meeting air quality standards (1978) on the coal-fired plants will add a very significant increment to cost of generation. The additional cost is estimated to average 40 percent of new unit capital costs for control of all pollutants and an annual

TABLE 5-3

COMPARISON OF CAPITAL INVESTMENT AND OPERATING COSTS FOR
FGD SYSTEMS 500 MW COAL-FIRED GENERATING PLANT*

	Capital Costs ($/kw)	Annualized Operating Costs (mills/kwh)
1. Limestone Scrubbing	96	4.27
2. Double Alkali	102	4.64
3. MgO	104	4.26
4. Wellman-Lord	113	5.89

*Generating Plant, 90% SO_2 removal, 35 wt % sulfur coal (dry).
Source: Reference 85

operation and maintenance cost increment of 100 percent while
creating a heating rate penalty of 15 percent. For the entire
U.S. utility industry, these additional control requirements
add 60 billion to capital costs, one billion in annual opera-
ting costs and require that an additional 50 million tons of
coal be burned annually in the time period. These estimates
dramatically reveal the fact that even moderate improvements
in costs and technological efficiencies of pollution control
systems (e.g. - regenerable processes) translate into the bil-
lion dollar economic scale.

In terms of waste volume, the combination of growth
in coal-fired capacity and more stringent environmental stan-
dards will lead to very large increases in total waste (both ash
and FGD scrubber sludge) to be managed through the balance of
the century. It has been estimated,[83] for example, based upon
current SO_2 emission standards that if all new units install
nonregenerable FGD systems, that the wastes produced from those
plants will be approximately 20 million dry tons annually (ash
and FGD sludge) by 1980 with nearly one-half being ash. This
compares with the approximately 3 million tons of dry FGD sludge
being produced at 1977 levels. The total wastes would grow to
be about 80 million dry tons in 1990 and 155 million tons by
1998 if nonregenerable systems are utilized. Also, the total
waste volume would conservatively be doubled from these dry
amounts if 50 percent moisture in the wastes is assumed.

Use of regenerable FGD systems can substantially
reduce the volume of sludge produced even though total wastes
(because of ash) will still be high. Again, with current emis-
sion standards, it has been estimated[82] that if one-half the new
units install regenerable systems and one-half lime/limestone
throwaway systems, then the total waste volume in 1990 would be

60 million tons (dry) rather than the 80 million given above
with approximately 40 million tons of the waste being ash. The
total quantity of waste still represents a significant techno-
logical challenge for both proper disposal and for waste product
utilization. Nonetheless, the impact of regenerable systems is
seen by the estimated 20 million ton reduction in FGD sludge
produced by the installation of regenerable systems in only one-
half the units between the present and 1990.

Generally speaking, regenerable processes are designed
in such a way that fly ash must be removed from the flue gas prior
to process of SO_2 removal and regeneration. Consequently, fly
ash and bottom ash are collected separately and may be maintained
as separate products from the by-product of regeneration. There-
fore, all products are potentially marketable on a segregated
basis which may be a distinct advantage. In addition, since
scrubber sludge is essentially eliminated, fly ash which might
be needed to stabilize sludge can now be marketed. In the
future, it may be necessary to utilize greater and greater
amounts of potentially saleable fly ash to stabilize scrubber
sludge if throwaway processes are being used. Therefore, it is
possible that one resultant advantage of regenerable processes
would be that more fly ash could be utilized in a direct fashion
rather than being used to stabilize FGD scrubber waste.

Beyond the need of regenerable processes to demonstrate
technical and economic competitiveness with throwaway FGD systems,
there are other limitations to be considered. As regenerable
processes undergo further development it is likely that a pre-
scrubber system (prior to SO_2 removal) will be necessary to
remove certain chemical species from the flue gas which would
interfere in the regeneration step with by-product manufacture.

If such species, for example, as chloride are not removed they ultimately would become a contaminant to a sulfur or sulfuric acid by-product stream. The waste from such a prescrubber system must also be disposed of in an environmentally acceptable manner, and although it would be a smaller stream (probably on the order of five to ten percent as large as a typical wet scrubbing FGD stream) it will also be expected to contain more soluble species than a typical calcium sulfate/sulfite FGD stream. Consequently, the prescrubber waste system must be considered as a significant limitation on regenerable processes even though it represents a smaller quantity. Research will be required to determine the impact of this waste.

Perhaps, the single most significant limitation on regenerable processes taken as a whole, is the finiteness of markets for both sulfur and sulfuric acid as by-products. The ability to sell by-products at a competitive price is obviously a crucial principle in the desirability and feasibility of any regenerable process. Beyond certain specific locations where a local market may appear to exist, there does not appear to be a sufficient market in the immediate future for the sulfur or sulfuric acid that would be produced if all of the current units equipped with throwaway FGD systems were replaced with a regenerable process. This can be illustrated in the case of sulfuric acid.

Projections indicate that the United States consumption of sulfuric acid will be about 43 million tons[86] by 1980 and that existing technology for its production can meet this need. There are approximately 100 producers now at 200 locations. In an earlier section, two base case electrical generating plants were designed. Both were for 1000 Mw coal-fired generating plants with 80 percent on-stream plant factors and 35 percent thermal

efficiency. The amount of sulfuric acid that could be produced each year for each plant was calculated assuming that both could be operated at 90 percent efficiency for SO_2 collection and at 90 percent efficiency for conversion of SO_2 to sulfuric acid by-product. Case 1 assumed the burning of an eastern bituminous coal (11,000 Btu/lb heating value and 3% sulfur with 95% of S emitted as SO_2) and Case 2 the burning of a western sub-bituminous coal (9,000 Btu/lb heating value and 0.80% sulfur with 95% of S emitted as SO_2). The amounts of sulfuric acid produced annually under each case with the assumptions given above were:

> Case 1 - 220,000 tons annually
> Case 2 - 72,000 tons annually

For Case 1, the 220,000 tons of H_2SO_4 produced represents 0.5 percent of the expected 1980 consumption of H_2SO_4 and the 72,000 tons for Case 2 represents 0.17 percent of that total. It has been estimated that coal-fired capacity will increase by 120,000 Mw between the present and 1990. If all new capacity involved SO_2 removal by regenerable processes producing H_2SO_4 with conditions approximating those above, then the H_2SO_4 produced under Case 1 conditions would be 26,400,000 tons and 8,640,000 tons under Case 2. These amounts would represent 61.3 percent and 20.1 percent of the 1980 H_2SO_4 consumption. While it is expected that a growth in H_2SO_4 consumption will occur between 1980 and 1990, the above amounts indicate that total conversion of a coal-fired capacity to H_2SO_4 producing regenerable processes would have the effect of producing approximately 40 percent of the total consumed for a rough average.

Because existing sources of H_2SO_4 production are expected to keep pace with needs through 1990, it is probable that the marketing of power plant by-product H_2SO_4 will be

highly site-specific. There will undoubtedly be situations where some utility plants will find a strong local market for by-product H_2SO_4 because existing manufacturers are not supplying a given local need. In other instances, it is likely that the market would be saturated, thus making penetration difficult.

Production of sulfur and sulfuric acid must be considered as interrelated. Approximately 81 percent of H_2SO_4 is produced with elemental sulfur as the raw material. Sulfur also has large outlets for use in diesel fuel treatment, fertilizers, hydrotreating, insecticides and as a vulcanizing agent in rubber manufacture. Presently, most U.S. sulfur is mined by the Frasch process from underground deposits along the Gulf Coast. It is expected that these deposits will be largely depleted by the end of the century. Therefore, beyond 1980, it is expected that sulfur from the burning of fossil fuels or extracted from large deposits of gypsum will be looked to as primary sources. Because of the importance of H_2SO_4 and hence sulfur to the U.S. economy, it is possible that the marketability of by-product sulfur or sulfuric acid from regenerable SO_2 removal processes could become considerably enhanced. Therefore, stockpiling might be one alternative for utilities which could not find an immediate market outlet for by-product sulfur. Stockpiling of sulfuric acid is not considered highly likely because of technical problems, but stockpiling of elemental sulfur would be feasible if incentives existed. Based upon realistic needs by the end of the century, stockpiling of elemental sulfur produced in a regenerable FGD process on coal-fired electrical generating plants could very well be in the national interest. Governmental policy might be needed to create an incentive for the utilities to handle SO_2 emissions in such a manner.

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APPENDIX A

INVENTORIED COAL-FIRED CAPACITY
BY STATE FOR 1976

(Includes some units which may have been
under construction and some which, while equipped
to burn coal, had little or
no coal consumption.)

State	EPA Region	No. Plants With Coal-Fired Units	Total Plant Capacity (MW)	Total Coal-Fired Capacity (MW)	Average Coal-Fired Capacity per Plant (MW)
Alabama	IV	10	7991.02	7975.74	797.57
Arizona	IX	3	2534	2441	813.67
Arkansas	VI	0	0	0	0
California	IX	0	0	0	0
Colorado	VIII	8	2519.75	2514.25	314.28
Connecticut	I	0	0	0	0
Delaware	III	2	459.5	459.5	229.75
District of Columbia	III	0	0	0	0
Florida	IV	6	4705.7	3992.8	665.47
Georgia	IV	10	8138.9	7775.2	777.52
Idaho	X	0	0	0	0
Illinois	V	26	17,131.46	16,033.56	616.67
Indiana	V	27	11,790.21	11,558.85	428.11
Iowa	VII	22	3200.58	3077.32	139.88
Kansas	VII	8	2631.4	2157.4	269.68
Kentucky	IV	16	6555.1	6431.1	401.94
Louisiana	VI	0	0	0	0
Maine	I	0	0	0	0
Maryland	III	5	4345.3	3353	670.6
Massachusetts	I	4	2692.12	2642.39	660.6
Michigan	V	27	12,299.03	11,038.45	408.83
Minnesota	V	16	2773.95	2747.95	171.75
Mississippi	IV	0	0	0	0
Missouri	VII	16	8735.19	8092.04	505.75
Montana	VIII	3	939.54	939.54	313.18
Nebraska	VII	3	938	938	312.67
Nevada	IX	2	1910	1910	955
New Hampshire	I	1	506	459	459
New Jersey	II	6	4470.94	3373.71	562.29
New Mexico	VI	3	2872.2	2872.2	957.4
New York	II	10	2967.15	2960.25	296.03
North Carolina	IV	13	11,711	10,905	838.85
North Dakota	VIII	6	1205.5	1205.5	200.92
Ohio	V	35	23,692.61	22,203.34	634.38
Oklahoma	VI	0	0	0	0
Oregon	X	0	0	0	0
Pennsylvania	III	32	20,133.82	18,027.62	563.36
Rhode Island	I	2	242.88	194.5	97.25
South Carolina	IV	9	3870.94	2983.46	
South Dakota	VIII	6	542.9	542.9	90.5
Tennessee	IV	8	10,090.4	10,090.4	1261.3
Texas	VI	2	2300	2300	1150
Utah	VIII	5	923.65	923.65	184.73
Vermont	I	0	0	0	0
Virginia	III	9	6618.12	5629.82	625.54
Washington	X	1	1329.8	1329.8	1329.8
West Virginia	III	12	12,023.45	12,004.86	1000.41
Wisconsin	V	20	5510.55	5250.45	262.52
Wyoming	VIII	5	3046.36	3045.36	609.07
U. S. TOTAL Inventoried for 1976		399	216,349.02	202,379.89	507.22

APPENDIX B

SIZE DISTRIBUTION OF COAL-FIRED CAPACITY
BY STATE FOR 1976

(Includes some units which may have been
under construction and some which, while equipped
to burn coal, had little or
no coal consumption.)

State	EPA Region	No. Plants with Coal-Fired Units	No. of Plants with Given Coal-fired Capacity per Plant				
			<100 MW	100-499 MW	500-999 MW	1000-1999 MW	≥2000 MW
Alabama	IV	10	2	2	2	4	0
Arizona	IX	3	1	1	0	0	1
Arkansas	VI	0	0	0	0	0	0
California	IX	0	0	0	0	0	0
Colorado	VIII	8	3	3	2	0	0
Connecticut	I	0	0	0	0	0	0
Delaware	III	2	0	2	0	0	0
District of Columbia	III	0	0	0	0	0	0
Florida	IV	6	1	1	2	2	0
Georgia	IV	10	1	4	2	2	1
Idaho	X	0	0	0	0	0	0
Illinois	V	26	3	10	6	7	0
Indiana	V	27	8	8	8	3	0
Iowa	VII	22	12	9	0	1	0
Kansas	VII	8	3	3	2	0	0
Kentucky	IV	16	4	7	4	1	0
Louisiana	VI	0	0	0	0	0	0
Maine	I	0	0	0	0	0	0
Maryland	III	5	0	2	1	2	0
Massachusetts	I	4	1	1	1	1	0
Michigan	V	27	11	9	4	2	1
Minnesota	V	16	8	7	1	0	0
Mississippi	IV	0	0	0	0	0	0
Missouri	VII	16	4	4	7	0	1
Montana	VIII	3	1	1	1	0	0
Nebraska	VII	3	0	2	1	0	0
Nevada	IX	2	0	1	0	1	0
New Hampshire	I	1	0	1	0	0	0
New Jersey	II	6	0	3	2	1	0
New Mexico	VI	3	1	0	1	0	1
New York	II	10	2	6	2	0	0
North Carolina	IV	13	0	7	2	2	2
North Dakota	VIII	6	2	3	1	0	0
Ohio	V	35	9	11	6	7	2
Oklahoma	VI	0	0	0	0	0	0
Oregon	X	0	0	0	0	0	0
Pennsylvania	III	32	5	17	3	7	0
Rhode Island	I	2	1	1	0	0	0
South Carolina	IV	9	1	7	1	0	0
South Dakota	VIII	6	5	1	0	0	0
Tennessee	IV	8	0	1	3	3	1
Texas	VI	2	0	0	0	2	0
Utah	VIII	5	2	3	0	0	0
Vermont	I	0	0	0	0	0	0
Virginia	III	9	1	4	2	2	0
Washington	X	1	0	0	0	1	0
West Virginia	III	12	0	5	1	4	2
Wisconsin	V	20	7	10	2	1	0
Wyoming	VIII	5	2	0	2	1	0
U. S. TOTAL Inventoried for 1976		399	101	157	72	57	12

APPENDIX C

INVENTORIED COAL-FIRED CAPACITY
BY EPA REGION, STATE AND PLANT FOR 1976

(Includes some units which may have been
under construction and some which, while equipped
to burn coal, had little or
no coal consumption.)

EPA REGION I

(Connecticut, Maine, Massachusetts,
New Hampshire, Rhode Island, Vermont)

STATE: Connecticut

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity.

STATE: Massachusetts

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Riverside	Holyoke Water Power Co.	47.39	27.25[1]
Brayton Point	New England Power Co.	1611.25	1600.25
Salem Harbor	New England Power Co.	805.25	805.25
West Springfield	Western Mass. Electric Co.	228.23	209.64

[1] Riverside Plant has 2 700 lb. coal boilers and 2 200 lb. oil boilers on a single low pressure header to produce low pressure steam. Electricity is generated only in the winter.

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Merrimack	Public Service Co. of N. H.	506	459

STATE: Rhode Island

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Manchester Street	Narragansett Electric Co.	132	132
South Street	Narragansett Electric Co.	110.88	62.5

STATE: Vermont

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity.

EPA REGION II

(New Jersey, New York)

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
England	Atlantic City Electric Co.	437.605	275
Deepwater	Atlantic City Electric Co.	306	226
Bergen	Public Service Elec. & Gas Co.	712	650.432
Burlington	Public Service Elec. & Gas Co.	1017.65	455 ·
Hudson	Public Service Elec. & Gas Co.	1229.68	1114.48
Mercer	Public Service Elec. & Gas Co.	768	652.8

STATE: New York

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Goudey	New York State Elec. & Gas Co.	145.75	145.75
Greenidge	New York State Elec. & Gas Co.	170	170
Hickling	New York State Elec. & Gas Co.	70	70
Jennison	New York State Elec. & Gas Co.	60	60
Milliken	New York State Elec. & Gas Co.	275.5	270
Huntley	Niagara-Mohawk Power Co.	720.7	720
Dunkirk	Niagara-Mohawk Power Co.	560.7	560
Rochester #3 (Beebe)	Rochester Gas & Electric Corp.	215.2	215.2
Rochester #7 (Russell)	Rochester Gas & Electric Corp.	252.6	252.6
Lovett	Orange & Rockland Utilities, Inc.	496.7	496.7

EPA REGION III

(Delaware, District of Columbia, Maryland,
Pennsylvania, Virginia, West Virginia)

STATE: Delaware

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Indian River	Delmarva Power & Light	340	340
Delaware City	Delmarva Power & Light	119.5	119.5

District of Columbia

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity.

STATE: Maryland

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Morgantown	Potomac Electric Power Co.	1360	1112
Chalk Point	Potomac Electric Power Co.	1310	1262
Dickerson	Potomac Electric Power Co.	561	548
Smith	Potomac Edison Co.	109.5	109.5
Wagner	Baltimore Gas & Electric Co.	1004.8	321.5

STATE: Pennsylvania

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Elrama	Duquesne Light Co.	425	425
Phillips	Duquesne Light Co.	315	315
Cheswick	Duquesne Light Co.	525	525
Portland	Metropolitan Edison Co.[1]	426.7	426.7
Titus	Metropolitan Edison Co.[1]	225	225
Crawford	Metropolitan Edison Co.	116.7	46.7
Front Street	Pennsylvania Electric Co.	118.8	118.8
Homer City	Pennsylvania Electric Co.[2]	1326	1320
Seward	Pennsylvania Electric Co.	280.23	280.23
Shawville	Pennsylvania Electric Co.	646	640
Warren	Pennsylvania Electric Co.	135.7	84.6
Williamsburg	Pennsylvania Electric Co.	25	25
Conemaugh	Pennsylvania Power & Light Co.[3]	1882.9	1872
Brunner Island	Pennsylvania Power & Light Co.	1566.98	1558.73
Holtwood	Pennsylvania Power & Light Co.	183.2	75
Martins Creek	Pennsylvania Power & Light Co.	1168.5	1163
Sunbury	Pennsylvania Power & Light Co.	415.28	409.78
Montour	Pennsylvania Power & Light Co.	1641.7	1641.7
New Castle	Pennsylvania Power Co.	431.3	425.8
Eddystone	Philadelphia Electric Co.	1463	665
Southwark	Philadelphia Electric Co.	422.75	356
Barbadoes	Philadelphia Electric Co.	192.75	134
Chester	Philadelphia Electric Co.	174.75	124
Cromby	Philadelphia Electric Co.	353.75	351
Delaware	Philadelphia Electric Co.	318.75	250
Richmond	Philadelphia Electric Co.	909	275
Armstrong	West Penn Power Co.	326.4	326.4
Mitchell	West Penn Power Co.	448.7	299.2
Springdale	West Penn Power Co.	215.38	215.38
Hatfield's Ferry	West Penn Power Co.[4]	1728	1728
Hunlock Creek	UGI Corp.	45.6	45.6
Keystone	Jersey Central Power & Light[5]	1680	1680

[1] Co-owner: Pennsylvania Electric Co.

[2] Co-owner: New York State Electric & Gas Co.

[3] Co-owners: Pennsylvania Electric Co.; Potomac Electric Power Co.; Atlantic City Electric Co.; Baltimore Gas & Electric Co.; Delmarva Power & Light Co.; Public Service Electric & Gas of N.J.; Metropolitan Edison Co.; Philadelphia Electric Co.

[4] Co-owner: Potomac Edison Co.; Monongahela Power Co.

[5] Co-owners: Pennsylvania Power & Light; Atlantic City Electric Co.; Baltimore Gas & Electric; Public Service Electric & Gas Co.; Delmarva Power & Light; Philadelphia Electric Co.

STATE: Virginia

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Clinch River	Appalachian Power Co.	669	669
Glen Lyn	Appalachian Power Co.	337.5	337.5
Brantly	City of Danville, Water, Gas & Electric	39.13	29
Potomac River	Potomac Electric Power Co.	458	458
Bremo Bluff	Virginia Electric Power Co.	254.28	254.28
Chesterfield	Virginia Electric Power Co.	1484.4	1484.4
Portsmouth	Virginia Electric Power Co.	649.64	649.64
Possum Point	Virginia Electric Power Co.	1469	1373
Yorktown	Virginia Electric Power Co.	1257	375

STATE: West Virginia

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Cabin Creek	Appalachian Power Co.	170	170
Kawawha River	Appalachian Power Co.	426	426
Amos	Appalachian Power Co.[1]	2775.2	2775.2
Philip Sporn	Appalachian Power Co.[1]	1060	1060
Albright	Monongahela Power Co.	209.25	209.25
Fort Martin	Monongahela Power Co.[2]	1152	1152
Rivesville	Monongahela Power Co.	109.75	109.75
Willow Island	Monongahela Power Co.	215	215
Harrison	Monongahela Power Co.[2]	2052	2052
Kammer	Ohio Power Co.	675	675
Mitchell	Ohio Power Co.	1498.18	1498.18
Mt. Storm	Virginia Electric Power Co.	1681.07	1662.48

[1] Co-owner: Ohio Power Co.

[2] Co-owner: West Penn Power Co.
Potomac Edison Co.

EPA REGION IV

(Alabama, Florida, Georgia, Kentucky,
Mississippi, North Carolina, South Carolina,
Tennessee)

STATE: Alabama

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
McWilliams	Alabama Electric Co-op	55.28	40
Tombigbee	Alabama Electric Co-op	75	75
Barry	Alabama Power Co.	1525	1525
Chickasaw	Alabama Power Co.	120	120
Gaston	Alabama Power Co.	880	880
Gadsen	Alabama Power Co.	120	120
Gorgas	Alabama Power Co.	1341.25	1341.25
Greene County	Alabama Power Co. [1]	500	500
Colbert	Tennessee Valley Authority	1396.5	1396.5
Widows Creek	Tennessee Valley Authority	1977.99	1977.99

[1] Co-owner: Mississippi Power Co.

STATE: Florida

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Crystal River	Florida Power Co.	964.3	523.8
Crist	Gulf Power Co.	1045	970
Smith	Gulf Power Co.	344.4	305
Scholz	Gulf Power Co.	80	80
Gannon	Tampa Electric Co.	1076	1062
Big Bend	Tampa Electric Co.	1196	1052

STATE: Georgia

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Arkwright	Georgia Power Co.	190.58	160
Atkinson	Georgia Power Co.	318.72	240
Hammond	Georgia Power Co.	800	800
McDonough	Georgia Power Co.	568.8	490
Mitchell	Georgia Power Co.	288.2	170
Yates	Georgia Power Co.	1250	1250
Bowen	Georgia Power Co.	2319.4	2280.0
Wansley	Georgia Power Co.[1]	833	833
Harllee Branch	Georgia Power Co.	1539.7	1539.7
Plant Crisp & Crisp Hydro	Crisp County Power Comm.	30.5	12.5

[1] Co-owner: Munic. Electric Authority of Georgia

STATE: Kentucky

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Coleman	Big Rivers Electric Corp.	455	455
Reid	Big Rivers Electric Corp.[1]	370	370
Cooper	East Kentucky Power Co-op	354	354
Dale	East Kentucky Power Co-op	194	194
Henderson #1	Henderson Municipal Power & Light	51.5	49.5
Big Sandy	Kentucky Power Co.	1002.6	1002.6
Brown	Kentucky Utilities Co.	706	706
Ghent	Kentucky Utilities Co.	525	525
Green River	Kentucky Utilities Co.	242	242
Pineville (4 mile)	Kentucky Utilities Co.	34	34
Tyrone	Kentucky Utilities Co.	137	75
Cane Run	Louisville Gas & Electric Co.	1004	988
Paddy's Run	Louisville Gas & Electric Co.	360	316
Mill Creek	Louisville Gas & Electric Co.	660	660
Smith	Owensboro Public Utilities	415	415
Owensboro #1	Owensboro Public Utilities	45	45

[1] Co-owner: Henderson Municipal Power & Light

STATE: Mississippi

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity.

STATE: North Carolina

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Cape Fear	Carolina Power & Light Co.	465	381
Asheville	Carolina Power & Light Co.	392	392
Lee	Carolina Power & Light Co.	498	407
Roxboro	Carolina Power & Light Co.	1720	1705
Weatherspoon	Carolina Power & Light Co.	314	176
Sutton	Carolina Power & Light Co.	652	607
Belews Creek	Duke Power Co.	2200	2200
Buck	Duke Power Co.	488	364
Cliffside	Duke Power Co.	770	770
Dan River	Duke Power Co.	369	284
Marshall	Duke Power Co.	2025	2025
River Bend	Duke Power Co.	678	454
Allen	Duke Power Co.	1140	1140

STATE: South Carolina

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Robinson	Carolina Power & Light Co.	854	174
Lee	Duke Power Co.	383	323
Wateree	South Carolina Electric & Gas Co.	771.8	771.8
Parr	South Carolina Electric & Gas Co.	87.38	72.5
Canadys	South Carolina Electric & Gas Co.	489.6	489.6
McMeekin	South Carolina Electric & Gas Co.	293.76	293.76
Urquhart	South Carolina Electric & Gas Co.	250	250
Grainger	South Carolina Public Serv. Auth.	163.2	163.2
Jefferies	South Carolina Public Serv. Auth.	578.2	445.6

STATE: Tennessee

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Allen	Tennessee Valley Authority	990	990
Bull Run	Tennessee Valley Authority	950	950
Gallatin	Tennessee Valley Authority	1255.2	1255.2
Sevier	Tennessee Valley Authority	846.5	846.5
Kingston	Tennessee Valley Authority	1723.5	1723.5
Watts Bar	Tennessee Valley Authority	240	240
Cumberland	Tennessee Valley Authority	2600	2600
Johnsonville	Tennessee Valley Authority	1485.2	1485.2

EPA REGION V

(Illinois, Indiana, Michigan, Minnesota,
Ohio, Wisconsin)

STATE: Illinois

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Coffeen	Central Ill. Pub. Serv.	1005.46	1005.46
Grandtower	Central Ill. Pub. Serv.	194.64	194.64
Hutsonville	Central Ill. Pub. Serv.	200	150
Meredosia	Central Ill. Pub. Serv.	564.06	354.36
Edwards	Central Ill. Light Co.	725	725
Wallace	Central Ill. Light Co.	275	275
Kincaid	Commonwealth Edison Co.	1319.4	1319.4
Powerton	Commonwealth Edison Co.	1785.6	1785.6
Will County	Commonwealth Edison Co.	1268.85	1268.85
Fisk	Commonwealth Edison Co.	860.58	546.58
Crawford	Commonwealth Edison Co.	805.12	597.52
Dixon	Commonwealth Edison Co.	122.2	119
Joliet	Commonwealth Edison Co.	1944.8	1787.4
Waukegan	Commonwealth Edison Co.	1084.75	932.75
Joppa	Electric Energy, Inc.	1041	1041
Hennepin	Ill. Power Co.	311	311
Vermillion	Ill. Power Co.	186	186
Baldwin	Ill. Power Co.	1815	1815
Wood River	Ill. Power Co.	651	651
Marion	South Ill. Power Co-op	114	114
Dallman	Springfield Water, Light & Power	160	160
Lakeside	Springfield Water, Light & Power	141	141
Fairfield	Fairfield Munic. Elec. Dept.	16.5	16.5
Venice #2	Union Electric Co.	500	500
Winnetka	Village of Winnetka	25.5	25.5
Mt. Carmel	Mt. Carmel Public Utility Co.	15	15

STATE: Indiana

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
State Line	Commonwealth Edison Co.	967.69	967.69
Crawfordsville	Crawfordsville Elec. Lt. & Power	23.5	23.5
Washington Ave.	Frankfort City Lt. & Power Dept.	50	32.5
Ratts	Hoosier Eng. Div. Ind. State. Rec.	233.2	233.2
Breed	Indiana & Michigan Elec. Co.	450	450
Tanners Creek	Indiana & Michigan Elec. Co.	1098	1098
Twin Branch	Indiana & Michigan Elec. Co.	244.76	237.5
Clifty Creek	Indiana Kentucky Elec. Co.[1]	1290	1290
Stout	Indianapolis Power & Light Co.	777.85	704.35
Pritchard	Indianapolis Power & Light Co.	364	361.25
Perry	Indianapolis Power & Light Co.	58.5	58.5
Petersburg	Indianapolis Power & Light Co.	649.58	641.43
Logansport	Logansport Munic. Utilities	56	39
Bailly	Northern Indiana Public Serv.	649.5	615.6
D. H. Mitchell	Northern Indiana Public Serv.	581.6	529.4
Michigan City	Northern Indiana Public Serv.	736	736
Peru	City of Peru Utilities	37	37
Edwardsport	Public Service Co. of Indiana	165	165
Noblesville	Public Service Co. of Indiana	106	106
Gallagher	Public Service Co. of Indiana	637	637
Wabash River	Public Service Co. of Indiana	889	881
Cayuga	Public Service Co. of Indiana.	1024	1013
Whitewater Valley	Richmond Power & Light	93	93
Culley	Southern Indiana Gas & Electric	414.93	414.93
Warrick #4	Southern Indiana Gas & Electric	161.5	161.5
Jasper	Jasper Munic. Elec. Utilities	14.5	14.5
Washington	Washington City Lt. & Power	18	18

[1] Co-owner: Ohio Valley Electric Corp.

STATE: Michigan

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Cobb	Consumers Power Co.	510.6	510.6
Morrow	Consumers Power Co.	221	186
Karn	Consumers Power Co.	1135	1135
Campbell	Consumers Power Co.	670.6	650
Weadock	Consumers Power Co.	635.1	614.5
Whiting	Consumers Power Co.	345.6	325
Mistersky	City of Detroit Pub. Ltng.	204	80
DeYoung	Holland Brd. Pub. Wrks.	97.75	77.75
Eckert	Lansing Brd. of Water & Light	386	386
Erickson	Lansing Brd. of Water & Light	165	165
Ottawa	Lansing Brd. of Water & Light	82.7	81.5
Advance	Northern Michigan Elec. Co-op	40	40
Harbor Beach	Detroit Edison Co.	125	121
Marysville	Detroit Edison Co.	200	200
River Rouge	Detroit Edison Co.	852.5	260
St. Clair	Detroit Edison Co.	1798	1450
Monroe	Detroit Edison Co.	3024.35	3010.6
Conners Creek	Detroit Edison Co.	495.5	490
Pennsalt	Detroit Edison Co.	37	37
Trenton Channel	Detroit Edison Co.	713.75	700
Wyandotte North	Detroit Edison Co.	41.5	41.5
Port Huron	Detroit Edison Co.	11.75	6.25
Warden	Upper Peninsula Power Co.	17.7	17.7
Presque Isle	Upper Peninsula Power Co.[1]	338.87	338.87
Escanaba	Upper Peninsula Power Co.	28.88	28.88
Wyandotte	City of Wyandotte	64	49
Shiras (Marquette)	Marquette Brd. Lt. & Power	56.88	36.3

[1] Co-owner: Upper Peninsula Generating Co.

STATE: Minnesota

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Aurora (Syl Laskin)	Minn. Power & Lt. Co.	110	110
Boswell	Minn. Power & Lt. Co.	492.6	492.6
King	North. States Power Co.	560	560
Black Dog	North. States Power Co.	412	412
High Bridge	North. States Power Co.	339	339
Minnesota Valley	North. States Power Co.	47	47
Riverside	North. States Power Co.	334	334
Wilmarth	North. States Power Co.	25	25
Red Wing	North. States Power Co.	25	25
New Ulm	New Ulm Publ. Utilities Comm.	53.5	51
Silver Lake	Rochester Dept. Public Utilities	101	101
Elk River	United Power Assn.	68	45.5
Plant #1	Virginia Dept. Pub. Utils.	34.5	34.5
Willmar	Willmar Municipal Pub. Utils.	29.35	29.35
Hoot Lake	Otter Tail Power Co.	128	127
Ortonville Gener. Plant	Otter Tail Power Co.	15	15

STATE: Ohio

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Cardinal	Ohio Power Co.[1]	1190	1190
Muskingum River	Ohio Power Co.	1466.8	1466.8
Philo*	Ohio Power Co.	295	295
Tidd	Ohio Power Co.	222.22	222.22
Woodcock	Ohio Power Co.	42.5	42.5
Ashtabula	Cleveland Elec. Illuminating	640	440
Avon Lake	Cleveland Elec. Illuminating	1307	1117
East Lake	Cleveland Elec. Illuminating[2]	1289	1257
Lake Shore	Cleveland Elec. Illuminating	518	514
Conesville	Columbus & Southern Ohio Elec.	850.25	836.5
Picway	Columbus & Southern Ohio Elec.	422.02	170.75
Poston	Columbus & Southern Ohio Elec.	245.75	232
Edgewater	Ohio Edison Co.	242.97	192.87
Gorge	Ohio Edison Co.	87.5	87.5
Mad River	Ohio Edison Co.	134.92	75
Niles	Ohio Edison Co.	279.96	250
Burger	Ohio Edison Co.	550.42	544
Tornto	Ohio Edison Co.	175.75	175.75
Sammis	Ohio Edison Co.	1990.24	1979.54
Norwalk	Ohio Edison Co.	32.38	31.38
Gavin	Ohio Electric Co.	2600	2600
Kyger Creek	Ohio Valley Elec. Corp.	1075	1075
Piqua	Piqua Mun. Power System	59	39
Miami Fort	Cinncinnati Gas & Elec. Co.[3]	1051	844
Beckjord	Cinncinnati Gas & Elec. Co.[4]	1432.8	1188
Conesville #4	Cinncinnati Gas & Elec. Co.[4]	825.05	825.05
Stuart	Cinncinnati Gas & Elec. Co.[4]	2451.15	2440
Hutchings	Dayton Power & Light	447	414
Tait	Dayton Power & Light	460	449
Bayshore Station	Toledo Edison Co.	655.48	639.48
Acme	Toledo Edison Co.	307.5	307.5
Orrville	Orrville Municipal Util.	89.1	88.5
Painesville	Painesville Elec. Div.	56.5	56.5
Columbus	City of Columbus	52.6	39.5†
Hamilton	Hamilton Municipal	147.75	78

* In deactivated reserve - 12/76.
† Assumed based on multi-fuel from Elec. World Dir. of Utilities 1977-1978

[1] Co-owner: Buckeye Power, Inc.
[2] Co-owner: Duquesne Light Co.
[3] Co-owner: Dayton Power & Light
[4] Co-owner: Columbus & Southern Ohio Elec. Co.; Dayton Power & Light

STATE: Wisconsin

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Alma	Dairyland Power Co-op	208	208
Genoa #3	Dairyland Power Co-op	350	350
Stoneman	Dairyland Power Co-op	52	52
Bay Front	Lake Superior Dist. Power Co.	82.2	82.2
Manitowoc	Manitowoc Pub. Utilities	69	69
River Street	Menasha Elec. & Water Util.	33.49	32.64
North Oak Creek	Wisconsin Electric Power Co.	500	500
Port Washington	Wisconsin Electric Power Co.	420	400
South Oak Creek	Wisconsin Electric Power Co.	1190	1170
Valley	Wisconsin Electric Power Co.	272.42	269.67
East Wells B	Wisconsin Elec. Power Co.	13.7	13.7
Edge Water	Wisconsin Power & Light Co.[1]	449.94	449.94
Dewey	Wisconsin Power & Light Co.	200	200
Rock River	Wisconsin Power & Light Co.	333.9	150
Blackhawk	Wisconsin Power & Light Co.	50.4	50
Columbia #1	Wisconsin Power & Light Co.[2]	521	521
Blount Street	Madison Gas & Electric	195.5	195.5*
Wildwood	Marshfield Elec. & Water Dept.	41.5	29
Pulliam	Wisconsin Public Service Corp.	392.5	372.5
Weston	Wisconsin Public Service Corp.	135	135

* Assumed based on multi-fuel from Electrical World Dir. of Utilities 1977-1978

[1] Co-owner: Wisconsin Public Service Co.
[2] Co-owner: Wisconsin Public Service Co.; Madison Gas & Electric

EPA REGION VI
(Arkansas, Louisiana, New Mexico, Oklahoma, Texas)

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Four Corners	Arizona Public Service Company[1]	2208.2	2208.2
San Juan	Public Service Company of N.M.[2]	652	652
Raton	Raton Public Service Co.	12	12

[1]Co-owners: Tucson Gas & Electric
Southern California Edison
Public Service Company of New Mexico
Salt River Project
El Paso Electric

[2]Co-owners: Tucson Gas & Electric

STATE: Oklahoma

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity

STATE: Texas

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Big Brown	Texas Power & Light Company[1]	1150	1150
Monticello	Texas Power & Light Company[1]	1150	1150

[1]Co-owners: Dallas Power & Light
Texas Electric Service Company

EPA REGION VII

(Iowa, Kansas, Missouri, Nebraska)

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Lansing	Interstate Power Company	61.5	59.5
Kapp	Interstate Power Company	227.28	212.28
Dubuque	Interstate Power Company	77	58*
Bridgeport	Iowa Southern Utilities Company	71	71
Burlington	Iowa Southern Utilities Company	211.95	211.95
Muscatine	Muscatine Brd. of Wtr. & Lt.	108	108
Pella	Pella Munic. Pwr. & Lt.	43.5	43.5
Humboldt	Cornbelt Power Co-op	49	49
Wisdom	Cornbelt Power Co-op	39	39
Fair	Eastern Iowa Lt & Pwr Co-op	55	55
Boone	Iowa Electric Lt & Pwr	27	27
Sixth Street	Iowa Electric Lt & Pwr	105	105
Prairie Creek #4	Iowa Electric Lt & Pwr	140	140
Sutherland	Iowa Electric Lt & Pwr	149.5	149.5
Riverside	Iowa-Illinois Gas & Elec. Co.	283.41	201.65
Neal	Iowa Public Serv. Co.[1]	1022.94	1017.44
Maynard	Iowa Public Serv. Co.	77.4	77.4
Hawkeye	Iowa Public Serv. Co.	19	19
Carrol	Iowa Public Serv. Co.	10	10
Eagle Grove	Iowa Public Serv. Co.	7.5	7.5
Council Bluffs	Iowa Pwr. & Light Co.	138.7	138.7
Des Moines	Iowa Pwr. & Light Co.	276.9	276.9

*Single header unit - 3 of the 4 boilers on the header can burn coal.

[1]Co-owners: Iowa Power & Light
             Iowa-Illinois Gas & Elec. Co.
             Iowa Southern Utilities

STATE: Kansas

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Neosho	Kansas Gas & Electric	113.5	40*
La Cygne	Kansas City Pwr. & Lt.[1]	858.7	858.7
Riverton	Empire Dist. Elec. Company	157.5	80
Kaw	Kansas City Brd. of Pub. Utils.	156	156
Quindaro #2	Kansas City Brd. of Pub. Utils.	164	45
Quindaro #3	Kansas City Brd. of Pub. Utils.	228	228
Lawrence	Kansas Pwr & Lt. Company	575.7	529.7
Tecumseh	Kansas Pwr & Lt. Company	378	220

*Coal units on cold reserve.

[1]Co-owner: Kansas Gas & Electric

STATE: Missouri

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Asbury	Empire District Elec. Co.	200	200
Montrose	Kansas City Pwr & Lt.	546	546
Northeast	Kansas City Pwr & Lt.	468	135
Grand Ave	Kansas City Pwr & Lt.	99	99
Hawthorn	Kansas City Pwr & Lt.	836	836
Green	Missouri Public Serv. Co.	49.5	49.5
Sibley	Missouri Pub. Serv. Co.	523.5	523.5
Rush Island	Union Elec. Co.	555.04	555.04
Sioux	Union Elec. Co.	978	978
Labadie	Union Elec. Co.	2220	2220
Meramec	Union Elec. Co.	800	800
Fulton #2	Fulton Brd. Pub. Wrks.	44.15	11.5
Thomas Hill	Associated Elec. Co-op	483	303
New Madrid	City of New Madrid[1]	600	600
Columbia	Columbia Water & Lt Dept	97.5	85
Lake Road (Lakeside)	St. Joseph Lt & Pwr Co.	235.5	150.5

[1]Co-owner - Associated Electric Co-op

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Kramer	Nebraska Public Power District	113	113
Sheldon	Nebraska Public Power District	225	225
North Omaha	Omaha Public Power District	600	600

EPA REGION VIII

(Colorado, Montana, North Dakota
South Dakota, Utah, Wyoming)

STATE: Colorado

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Hayden	Salt River Project[1]	465.4	465.4
Nucla	Colorado-Ute Electric Assoc.	34.5	34.5
Comanche	Public Service Co. of Colorado	700	700
Arapahoe	Public Service Co. of Colorado	232	232
Cameo	Public Service Co. of Colorado	66	66
Cherokee	Public Service Co. of Colorado	715.5	710.0
Canon City	Cent. Telephone & Utility Corp.	42.35	42.35
Drake	City of Colorado Springs	264	264

[1]Co-owner: Colorado UTE Electric Association

STATE: Montana

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Lewis and Clark	Montana-Dakota Utilities	50	50
Colstrip	Montana Power Company[1]	716.74	716.74
Corette	Montana Power Company[1]	172.8	172.8

[1]Co-owner: Puget Sound Power and Light Company

STATE: North Dakota

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Leland Olds	Basin Electric Power Co-op	650	650
Neal	Basin Electric Power Co-op	30	30
Young	Minnkota Power Co-op	240	240
Heskett	Montana-Dakota Utility Co.	100	100
Beulah	Montana-Dakota Utility Co.	13.5	13.5
Stanton	United Power Association	172	172

STATE: South Dakota

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Kirk	Black Hills Power & Light Co.	31.5	31.5
Ben French	Black Hills Power & Light Co.	32	22
Big Stone	Northwestern Public Serv. Co.[1]	419.4	419.4
Aberdeen Gen. Plt.	Northwestern Public Serv. Co.[1]	7.5	7.5
Mitchell Gen. Plt.	Northwestern Public Serv. Co.[1]	7.5	7.5
Lawrence	Northern States Power Company	45	45

[1]Co-owners: Otter Tail Power Company
Montana-Dakota Utilities Company

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Carbon	Utah Power & Light Company	166	166
Huntington #2	Utah Power & Light Company	446.4	446.4
Gadsby	Utah Power & Light Company	241	241
Hale	Utah Power & Light Company	62.75	62.75
Cedar Steam	Cal.-Pacific Utility Company	7.5	7.5

STATE: Wyoming

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Simpson	Black Hills Power & Light Co.	27.68	27.68
Osage	Black Hills Power & Light Co.	35.5	34.5
Jim Bridger	Pacific Power & Light Company[1]	1525.67	1525.67
Johnston	Pacific Power & Light Company	750.31	750.31
Naughton	Utah Power & Light Company	707.2	707.2

[1]Co-owner: Idaho Power and Light

EPA REGION IX

(Arizona, California, Nevada)

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Apache	Arizona Elec. Power Co-op	168	75
Cholla	Arizona Public Service Co.	116	116
Navajo	Salt River Project[1]	2250	2250

[1] Co-owner: Los Angeles Dept. of Water & Power; U. S. Bureau of Reclamation; Arizona Public Service; Nevada Power Co.; Tucson Gas & Electric

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity.

STATE: Nevada

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Gardner	Nevada Power Co.	330	330
Mohave	Nevada Power Co.[1]	1580	1580

[1] Co-owners: Southern California Edison Co.; Los Angeles Dept. of Water & Power; Salt River Project

EPA REGION X

(Idaho, Oregon, Washington)

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity.

STATE: Oregon

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)

No coal-fired capacity.

STATE: Washington

PLANT NAME	OWNER(S)	TOTAL CAPACITY (MW)	TOTAL COAL-FIRED CAPACITY (MW)
Centralia	Pacific Power & Light Co.[1]	1329.8	1329.8

[1] Co-owners: Washington Water Power Co.; Puget Sound Power & Light Co.; Portland General Electric Co.; City of Seattle; City of Tacoma; P.U.D. No. 1 of Snohomish County; P.U.D. No. 1 of Gray Harbor County

APPENDIX D

ELECTRIC UTILITY CAPITAL AND
MAINTENANCE EXPENSE DATA

1978 ELECTRIC UTILITY CAPITAL EXPENDITURES

Geographical Region	Total Capital Spending (millions of dollars)	% Investor-Owned	% Muni, State, and PPD's	% Co-op's	% Federal Agencies
New England	963.0	96.0	3.1	0.9	-
Middle Atlantic	3,318.0	98.7	0.1	1.2	-
East North Central	6,406.0	89.4	4.6	6.0	0
West North Central	4,052.9	46.0	20.2	33.5	0.3
South Atlantic	4,613.4	86.9	5.8	6.2	1.1
East South Central	2,670.1	28.1	4.2	17.3	50.4
West South Central	3,939.0	72.8	10.3	16.5	0.4
Mountain	2,183.1	67.4	20.3	9.9	2.4
Pacific	4,542.2	50.8	31.2	0.5	17.5
U.S. Total	32,687.6	71.0	11.6	10.5	6.9

Source: "1978 Annual Statistical Report," *Electrical World*, Vol. 189, No. 6, March 15, 1978, p. 83.

1977 ELECTRIC UTILITY CAPITAL EXPENDITURES

Geographical Region	Total Capital Spending (millions of dollars)	% Investor-Owned	% Muni, State, and PPD's	% Co-op's	% Federal Agencies
New England	766.6	97.4	1.4	1.2	-
Middle Atlantic	3,106.2	91.8	7.3	0.9	-
East North Central	5,515.7	92.3	3.1	4.6	0
West North Central	3,071.9	42.9	25.0	31.2	0.9
South Atlantic	3,905.3	84.6	7.7	7.0	0.7
East South Central	2,466.7	32.9	3.5	18.8	44.8
West South Central	3,376.0	75.1	8.6	16.2	0.1
Mountain	1,821.1	64.8	20.5	12.3	2.4
Pacific	3,592.4	53.5	26.3	0.5	19.7
U.S. Total	27,621.8	71.5	11.6	10.0	6.9

Source: "1978 Annual Statistical Report," Electrical World, Vol. 189, No. 6, March 15, 1978, p. 82.

PERCENT GROWTH IN ELECTRIC UTILITY

CAPITAL EXPENDITURES 1977-1978

Geographical Region	Total	Investor-Owned	Muni, State, and PPD's	Co-op's	Federal Agencies
New England	+25.6%	+23.7%	+174.5%	+ 5.8%	-
Middle Atlantic	+ 6.8	+14.9	- 98.5	+50.6	-
East North Central	+16.1	+12.5	+ 74.6	+50.1	0.0%
West North Central	+31.9	+41.7	+ 6.1	+41.7	-53.5
South Atlantic	+18.1	+21.4	- 10.2	+ 4.5	+72.4
East South Central	+ 8.2	- 7.5	+ 28.1	- 0.1	+21.7
West South Central	+16.7	+13.0	+ 39.7	+19.1	+858.0
Mountain	+19.9	+24.6	+ 19.1	-3.8	+21.1
Pacific	+26.4	+20.2	+ 50.2	+10.9	+12.2
U.S. Total	+18.3	+17.4	+ 19.6	+23.5	+18.5

Source: "1978 Annual Statistical Report," Electrical World, Vol. 189, No. 6, March 15, 1978.

1977 ELECTRIC UTILITY MAINTENANCE COSTS

Geographic Region	Total Maintenance Costs (millions of dollars)	% Investor-Owned	% Muni, State, and PPD's	% Co-ops	% Federal Agencies
New England	200.0	89.9	8.9	1.2	-
Middle Atlantic	963.2	99.4	0.2	0.4	-
East North Central	920.7	90.1	5.9	4.0	0.0+
West North Central	357.7	66.7	15.9	12.7	4.6
South Atlantic	691.5	89.9	3.9	6.0	0.2
East South Central	365.4	34.1	10.1	8.2	47.6
West South Central	335.6	70.6	6.6	11.0	11.8
Mountain	197.9	66.6	13.9	7.6	11.9
Pacific	535.7	53.8	31.5	0.6	14.1
U.S. Total	4,567.7	79.0	9.1	4.7	7.2

Source: "1978 Annual Statistical Report," Electrical World, Vol. 189, No. 6, March 15, 1978.

1978 BUDGETED ELECTRIC UTILITY MAINTENANCE COSTS

Geographic Region	Total Budgeted Maintenance Costs (millions of dollars)	% Investor- Owned	% Muni, State, and PPD's	% Co-ops	% Federal Agencies
New England	213.6	90.1	8.7	1.2	-
Middle Atlantic	1,044.6	99.4	0.2	0.4	-
East North Central	1,020.1	91.4	4.5	3.9	0.0+
West North Central	388.7	68.0	13.7	13.9	4.4
South Atlantic	785.7	88.7	4.9	6.1	0.3
East South Central	429.6	34.4	10.0	7.9	47.7
West South Central	355.2	74.4	7.0	12.2	6.4
Mountain	224.9	68.0	13.3	7.7	11.0
Pacific	583.6	51.8	33.3	0.6	14.3
U.S. Total	5,046.0	79.1	8.9	4.9	7.1

Source: "1978 Annual Statistical Report," Electrical World, Vol. 189, No. 6, March 15, 1978.

PERCENT GROWTH IN ELECTRIC UTILITY

MAINTENANCE COSTS, 1977-1978

Geographic Region	Total	Investor-Owned	Muni, State, and PPD's	Co-op's	Federal Agencies
New England	+ 6.8	+ 7.0	+ 4.3	+10.1	-
Middle Atlantic	+ 8.5	+ 8.4	+ 8.2	+34.3	-
East North Central	+10.8	+12.4	-14.5	+11.6	+17.6
West North Central	+ 8.7	+10.7	- 6.4	+18.2	+ 4.2
South Atlantic	+13.6	+12.1	+41.7	+17.2	+30.9
East South Central	+17.6	+18.6	+17.3	+12.4	+17.8
West South Central	+ 5.8	+11.5	+11.3	+17.8	-42.3
Mountain	+13.7	+15.9	+ 8.8	+15.7	+ 2.4
Pacific	+ 8.9	+ 4.9	+15.0	+14.3	+10.5
U.S. Total	+10.5	+10.6	+ 9.0	+15.9	+ 7.4

Source: "1978 Annual Statistical Report," Electrical World, Vol. 189, No. 6, March 15, 1978.

Effects of Flue Gas Cleaning Waste on Groundwater Quality and Soil Characteristics

(U.S.) Army Engineer Waterways Experiment Station, Vicksburg, MS

Prepared for

Municipal Environmental Research Lab, Cincinnati, OH

Aug 79

&EPA

Effects of Flue Gas Cleaning Waste on Groundwater Quality and Soil Characteristics

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1. REPORT NO. EPA-600/2-79-164	2.	3. RECIPIENT'S ACCESSION NO.

4. TITLE AND SUBTITLE EFFECTS OF FLUE GAS CLEANING WASTE ON GROUNDWATER QUALITY AND SOIL CHARACTERISTICS	5. REPORT DATE August 1979 (Issuing Date)
	6. PERFORMING ORGANIZATION CODE

7. AUTHOR(S) U.S. Army Engineer Waterways Experiment Station (WES)	8. PERFORMING ORGANIZATION REPORT NO.

9. PERFORMING ORGANIZATION NAME AND ADDRESS U.S. Army Engineer Waterways Experiment Station (WES) Environmental Laboratory Vicksburg, Mississippi 39180	10. PROGRAM ELEMENT NO. 1DC818 SOS 1 Task 27
	11. CONTRACT/GRANT NO. EPA-IAG-D4-0596

12. SPONSORING AGENCY NAME AND ADDRESS Municipal Environmental Research Laboratory--Cin, OH Office of Research and Development U.S. Environmental Protection Agency Cincinnati, Ohio 45268	13. TYPE OF REPORT AND PERIOD COVERED Final July 1976 thru Dec.1978
	14. SPONSORING AGENCY CODE EPA/600/14

15. SUPPLEMENTARY NOTES

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16. ABSTRACT

Soil and water samples from several test borings and hydrological data were collected and analyzed for three flue gas cleaning sludge disposal sites in order to assess the extent of migration of pollutants into the local groundwater and the effects on surrounding soils. Physical testing of soils indicated that two major types of sites were included: one site was underlain by impermeable materials such as clay and shale; and two other sites underlain by relatively permeable silty sands and gravel with discontinuously distributed finer materials.

At the site underlain by impermeable substrata, no change in permeability or other physical properties of the soils could be related to the presence of the disposal site. At the two sites underlain by permeable substrata, only at one could variations in permeability, dry density, water content, and percent fines be related to the presence of the disposal site. Irregular occurrences of fine-grained materials (clays and silty sands) at the other site obscured any variations in these parameters which might have been caused by the disposal site.

Sludge/ash-derived constituents were found to have migrated out of the immediate area of the pit or pond at all three disposal sites degrading the quality of the local groundwater.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Wastes, Stabilization, Leaching, Sludge Groundwater, Permeability, Pollution Sulfates, Sulfites	Leachate, Solid Waste Management Flue Gas Cleaning, Chemical Stabilization (Fixation)	13B

18. DISTRIBUTION STATEMENT RELEASE TO PUBLIC	19. SECURITY CLASS *(This Report)* UNCLASSIFIED	21.
	20. SECURITY CLASS *(This page)* UNCLASSIFIED	22. PRICE

EPA Form 2220-1 (9-73)

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EFFECTS OF FLUE GAS CLEANING WASTE ON
GROUNDWATER QUALITY AND SOIL CHARACTERISTICS

by

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FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of that environment and the interplay between its components require a concentrated and integrated attack on the problems.

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This report presents results from the field investigation of three power plant waste disposal sites to determine the effects on surrounding soils and groundwater. It provides basic data on the potential pollution of waste from coal-fired power plants and will add to the knowledge required to determine the environmental consequences of conventional land disposal of these wastes.

Francis T. Mayo, Director
Municipal Environmental Research
Laboratory

ABSTRACT

Soil and water samples from several test borings and hydrological data were collected and analyzed for three flue gas cleaning sludge disposal sites in order to assess the extent of migration of pollutants into the local groundwater and the effects on surrounding soils. Physical testing of soils indicated that two major types of sites were included: one site was underlain by impermeable materials such as clay and shale; and two other sites underlain by relatively permeable silty sands and gravel with discontinously distributed finer materials.

At the site underlain by impermeable substrata, no change in permeability or other physical properties of the soils could be related to the presence of the disposal site. At the two sites underlain by permeable substrata, only at one could variations in permeability, dry density, water content, and percent fines be related to the presence of the disposal site. Irregular occurrences of fine-grained materials (clays and silty sands) at the other site obscured any variations in these parameters which might have been caused by the disposal site.

Sludge/ash-derived constituents were found to have migrated out of the immediate area of the pit or pond at all three disposal sites degrading the quality of the local groundwater. The subsurface migration of the sludge/ash-derived materials was least extensive at the site underlain by impermeable substrata. At the sites underlain by sands and gravels, evidence to a typical pollution plume under and down the groundwater gradient from the disposal site was found.

Analysis of distilled water extracts and nitric acid digests of soil samples from underneath and around the sludge/ash disposal sites indicated only slight changes in soil chemistry could be attributed to the presence of the disposal pit or pond. Evidently FGC sludge/ash leachates moved through the soils and sediments without appreciable interaction or attenuation of pollutants.

This report is submitted in partial fulfillment of Interagency Agreement No. EPA-IAG-D4-0569 between the U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory, Solid and Hazardous Waste Research Division (EPA, MERL, SHWRD) and the U.S. Army Engineer Waterways Experiment Station (WES). Work for this report was conducted during the period of July 1976 through December 1978.

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SECTION 1

INTRODUCTION

The growth of the electrical power industry coupled with the increasing use of coal as a primary fuel has resulted in a generally increased waste disposal problem for coal-fired power plants. The strict air pollution regulations regarding sulfur oxides (SO_x) emissions have caused many power plants to add stack scrubbing systems. These plants now produce a flue gas cleaning (FGC) sludge that must be disposed of along with flyash and bottom ash. Stack scrubbing is a necessary step due to the fact that sulfur dioxide (SO_2) particularly produces crop and plant damage, deterioration of many materials such as ferrous metals, marble and concrete, and increased incidence of bronchitis and lung cancer. Environmental Protection Agency (EPA) estimates have put the current total cost of SO_2 emission damage to property and people in the U.S. at \$22 billion per year (1).

The flue gas cleaning systems currently being installed, and those planned for the majority of installations through 1985, are "throw-away" or non-regenerative systems in which the product generated requires permanent disposal. The end product is a fine-grained slurry of high water content called either flue gas desulfurization (FGD) or flue gas cleaning (FGC) sludge. The term flue gas desulfurization sludge usually refers to only SO_x-reaction products, while flue gas cleaning sludge refers to a more general mixture of flyash and scrubber products (2). The twenty-one power plants now equipped with FGC systems are already producing around eight million metric tons of wet sludge per year (Table 1). By 1985, when power plants producing around 100,000 megawatts of power are projected to have installed FGC equipment, over 120 million metric tons of wet sludge will have to be disposed of annually.

Three major types of "throw-away" sludge producing FGC systems are currently being developed and installed on power plants in the U. S. One uses a wet slurry of limestone ($CaCO_3$); one a wet slurry of hydrated lime ($Ca(OH)_2$); and one--the double alkali--uses a clear Na_2SO_3 solution. Although the major reaction product of all three processes is calcium sulfite hemihydrate ($CaSO_3 \cdot \frac{1}{2}H_2O$), the constituents of the sludge produced will vary widely depending upon the impurities in the scrubbing materials, the type of coal being burned, the boiler configuration and the scrubbing method used. The overall reactions of these processes are (3,4):

Limestone:

$$CaCO_3 + SO_2 + \tfrac{1}{2}H_2O \rightarrow CaSO_3 \cdot \tfrac{1}{2}H_2O + CO_2$$

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TABLE 1. PROJECTED ANNUAL PRODUCTION OF FLUE GAS CLEANING
SLUDGE IN THE U. S. (3)

	Year		
	1977	1980	1985
Estimated on-line capacity (MW) with FGC	6500	35,000	100,000
Dry FGC sludge*	1.75	9.5	27.0
Dry ash*	2.15	11.5	33.0
Total Dry Sludge*	3.9	21.0	60.0
Water (sludge at 50% water)*	3.9	21.0	60.0
Total Wet Sludge*	7.8	42.0	120.0
Approximate total volume (m^3/yr)	4.9×10^6	2.5×10^7	7.4×10^7

* metric tons/year

<u>Lime (hydrated)</u>:

$$Ca(OH)_2 + SO_2 \rightarrow CaSO_3 \cdot \tfrac{1}{2}H_2O + \tfrac{1}{2}H_2O$$

<u>Double alkali</u>:

$$Na_2SO_3 + SO_2 + H_2O \rightarrow 2\,NaHSO_3$$

$$2NaHSO_3 + Ca(OH)_2 \rightarrow CaSO_3 \cdot \tfrac{1}{2}H_2O + 3/2\,H_2O + Na_2SO_3$$
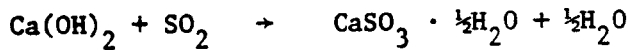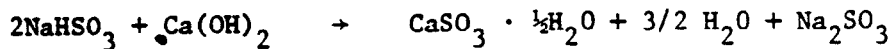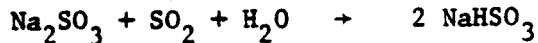
The calcium can also oxidize to calcium sulfate dihydrate (gypsum) by the reaction: $2[CaSO_3 \cdot \tfrac{1}{2}H_2O] + O_2 + 2H_2O \rightarrow 2[CaSO_4 \cdot 2H_2O]$. Therefore, the final product has variable proportions of calcium sulfate and sulfite, depending upon the amount of oxygen available during the scrubbing operation.

<u>Chemical Composition of FGC Sludges</u>

The composition of major solid components in several FGC sludges which have been analyzed are presented in Table 2. The major component of the sludge is seen to be variable amounts of calcium sulfite and calcium sulfate, depending upon the amount of oxidation which has taken place. Oxidation (and consequently the calcium sulfate-to-calcium sulfite ratio) is usually greater in systems burning low-sulfur western coal. In all three systems, operation of the burner and FGC system can be adjusted to produce almost pure calcium sulfite sludges; or intentional oxidation can bring about the production of almost pure calcium sulfate sludges.

Variable amounts of unreacted limestone ($CaCO_3$) will be found in the limestone and dual alkali sludges, and in some lime systems where it enters as an impurity in the lime or is produced by reaction with the large amount of CO_2 in the stack gas. The amount of fly ash, the other major component in the FGC sludge, will also vary widely depending upon the ash and sulfur content of the coal burned and whether electrostatic precipitators or collectors are run ahead of the FGC system. As new FGC equipment becomes operational, many sludges may incorporate variable amounts of fly ash as the FGC systems also are excellent fly ash collectors and separate fly ash removal equipment may not be employed.

A variety of trace elements are also found in FGC sludges; typical analyses are listed in Table 3. Note the wide range of concentrations found in different sludges make generalizations as to composition difficult. The original sources of these trace elements are the coal, the lime or limestone and the makeup water. Those elements in the fuel which are not highly volatile such as chromium, manganese and nickel, will be retained in the fly ash and bottom ash. Therefore, the relative ash content controls the concentration of these elements in the sludge. On the other hand, the concentration of the highly volatile elements such as arsenic, cadmium, fluorine, mercury and selenium in the sludge depends largely upon the efficiency of their capture from the flue gas by the scrubber (9). Mercury and selenium will probably be present in the flue gas as elemental vapors and be poorly scrubbed. Assuming that the coal is the major source of trace metals and that sludge and ash

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TABLE 2. COMPOSITION OF SOME TYPICAL FGC SLUDGE SOLIDS

Process	Type of coal utilized	$CaSO_3 \cdot \frac{1}{2}H_2O$ (% wt)	$CaSO_4 \cdot 2H_2O$ (% wt)	Ratio: $CaSO_4/CaSO_3$	$CaCO_3$ (% wt)	Fly ash (% wt)	Other (% wt)	(Ref.)
Limestone	Eastern	19–23	15–32	0.65–1.7	4–42	20–43		(5)
Limestone	Western	11	17	2.8	2.5	59	14% $CaS_2O_3 \cdot 6H_2O$	(5)
Lime	Eastern	13	19	2.2	0.2	60	9.8% CaS_3O_{10}	(5)
Lime	Eastern	50	6	0.12	3	41		(5)
Lime	Eastern	94	2	0.02	0	4		(6)
Dual Alkali	Western	0.2	64	400	11	9	18% $CaSO_4$	(5)
Dual Alkali	Eastern	14	52	5.1	8	7	20% $CaSO_4 \cdot \frac{1}{2}H_2O$	(7)

TABLE 3. TYPICAL CONCENTRATIONS OF TRACE ELEMENTS IN SEVERAL FGC SLUDGES (5) AND IN A VARIETY OF COAL SAMPLES (8)

Element	Conc. range in sludges (ppm)	Median conc. in sludges (ppm)	Conc. range in coal (ppm)
Arsenic	3.4-63	33.0	3-60
Beryllium	•0.62-11	3.2	0.08-20
Cadmium	0.7-350	4.0	--
Chromium	3.5-34	16.0	2.5-100
Copper	1.4-47	14.0	1-100
Lead	1.0-55	14.0	3-35
Manganese	11-120	63.0	--
Mercury	0.02-6.0	1.0	0.01-30
Nickel	6.7-27	17.0	--
Selenium	<0.2-19	7.0	0.5-30
Zinc	9.8-118	57.0	0.9-600

-- = no analysis available

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production equals from 5-40 percent of the coal burned on a weight basis, the trace elements will be concentrated in the sludge two to twenty times the level in the coal. The form and availability of these trace elements is also changed from that in the original coal where they are held in an organic matrix and/or as sulfides and carbonates. The trace elements appear in the sludge primarily as oxides (or in some cases in elemental form) which are more soluble and chemically reactive than sulfides or solid organic complexes. The trace elements, therefore, represent a potential pollution hazard since they can be leached from the sludge and contaminate surrounding surface water and groundwater.

Physical Properties of FGC Sludges

The physical properties of FGC sludges are of prime importance in their handling, transporting, dewatering, and leaching characteristics. The morphology and size of sludge particles varies widely as a function of the sulfur content of the coal, the way the boiler is operated, the type of particulate control employed, and the type of FGC system and the mode in which it is operated.

The most striking and troublesome physical characteristic of FGC sludges is the uniform size and form of the crystals of the calcium sulfite (10). Calcium sulfite crystals are in the form of thin platelets with 10-100 micron lateral dimensions and of 0.1 to 0.5 micron thickness. Single crystals are rare, most being found in loosely arrayed clusters. The preponderance of small, uniformly-sized crystal aggregates produces a thixotropic sludge with high moisture content and very poor settling characteristics. The high moisture content is due to .the highly open, porous or sponge-type configuration of the crystal clusters. FGC sludges are not easily dewatered. For example, twenty-five hours of centrifugation at 900 times gravity in a solid-bottom centrifuge tube caused an increase from 40 percent solids to only 50 percent solids for an eastern coal, lime-scrubbing sludge (9). Slight shaking or stirring will cause the centrifuged sludge to return to a liquid or plastic state (thixotrophy). FGC sludges can present serious handling and storage problems.

The permeability of unmodified FGC sludges also varies greatly depending upon their source and fly ash content. The permeability of several samples of untreated FGC sludges were found to vary between 5 x 10^{-4} to 5 x 10^{-5} cm/ sec if gravity settled, and from 1 x 10^{-4} to 1 x 10^{-5} if compacted by vibration or by the use of a plunger (4). These moderate permeability rates are comparable to a clay or silty clay soil.

FGC sludges exhibit low compactability. When confined to a mold, sludge samples exhibit significant resistance to the action of compaction hammers, but this resistance disappears when the mold is removed. Unconfined compressive strengths are quite low, ranging from nil to 1.5 kg/sq. cm (11).

Methods of FGC Sludge Disposal

As FGC sludges began being produced, they were commonly disposed of in a manner similar to that which had been used for fly and bottom ash. Most commonly, fly ash was collected as a slurry which was pumped to settling or decanting basins where the ash settled and the liquid was decanted to a river (12). The amount of pollutants from the decanted water as well as that leaching into the groundwater from these disposal ponds could have been significant, but water quality data related to these operations are not readily available.

Presently, lagooning of mixed ash and FGC sludges is the most common method of dealing with the disposal problem (13). The sludge is usually pumped with low solids content (20-40%) into a lagoon where the solids settle out; the liquor is then reused as make-up water for the FGC process. Two major problems with this method of sludge disposal are the high levels of Ca, SO_4, SO_3, Cl, and trace metals which potentially could be leached out of the sludge bed into the local groundwater, and the physical instability of the sludge which may preclude use of the deposited sludge beds for any other purposes for an indefinite period of time (14).

One alternative which deals directly with the leaching problems is that of using lagoons which have been lined with impervious materials such as polyethylene, butyl rubber, concrete, asphalt or pozzolan-stabilized soil (13). The liners prevent the leaching of material or seepage of liquors from the disposal ponds or lagoons into ground- or surface waters. The lining of lagoons is an effective technique over the lifetime of the liner. Long-term service data applicable to sulfate/sulfite sludge containment do not exist for any liner materials although short-term experimental data have been reported (15). Lifetime estimates for different liner materials and sludge types vary from about 20 to over 50 years normal life expectancy. The major problem in the use of pond liners is their impermanence. When their integrity eventually is lost by accident or deterioration, the original problem of permanent disposal reoccurs. The use of pond liners, therefore, appears to be an effective alternative for moderately long-time periods, but not an adequate permanent disposal scheme with the technology presently available (13).

The sludge disposal techniques currently receiving the widest interest and study are those that involve chemically stabilizing or encapsulating the FGC sludges. The aims of this sludge treatment are to produce a structurally sound product (a solid, or friable, soil-like waste) that can be disposed of so that the potential for surface or groundwater pollution is minimized or eliminated (16,17).

Scope of This Study

The disposal sites selected for this study include only unlined, unstabilized power plant waste disposal ponds containing FGC sludges. The unlined ponds are considered to present the worst risk for the release of pollutants to the environment. The water released from the sludge into the soil beneath the disposal pond will be saturated with the contaminants found in the FGC sludge/ash

mixture. This water is referred to as a leachate; and the capture or absorption of potentially contaminating materials from this leachate by soil under the disposal site is referred to as attenuation.

The objectives of this study are to examine three typical, unlined FGC sludge/ash ponding or disposal operations that are situated in different geological circumstances in order to:

a)   discover if changes have occurred in the chemical characteristics of the local groundwater because of the FGC sludge/ash disposal operation,

b)   determine the influence of any leachate from the ponded FGC sludge/ash on the chemical characteristics and physical properties of the geologic materials directly below the landfill,

c)   determine what chemical constituents present in the soil beneath the disposal site can be released into contacting water,

d)   establish if a relationship exists between the depth below the disposal site and the chemical properties of the earth materials, and

e)   discover if chemical characteristics of the material beneath the disposal site indicate contaminant attenuation is occurring.

To meet these objectives, a model or pattern (Figure 1) for leachate movement and attenuation was developed to provide a rationale for the sampling program. In this model precipitation falling on the disposal site saturates the sludge/ash and then percolates through the soil directly below. A variable portion of the filterable and exchangeable material in the leachate is deposited in the soil below the landfill and possibly selected constituents are released from the soil. The attenuated leachate then continues downward to the water table. Groundwater flowing under the landfill dilutes the leachate and carries the pollutants in a plume down the groundwater gradient. Based on this idealized model, borings were located in such a way as to produce:

a)   groundwater from wells beneath the disposal site and from wells located both up and down the groundwater flow gradient in the area of the disposal site,

b)   samples of soil from beneath the disposal site and from comparable depths outside the disposal site,

c)   soil samples collected at different levels down the boreholes both outside and beneath the disposal site, and

d)   samples collected near the top of the saturated zone (water table) beneath and outside the disposal site.

Physical testing of soil samples collected below the disposal site and at comparable depths outside the disposal site was undertaken to evaluate changes related to the deposition of FGC sludge. The physical characterization included

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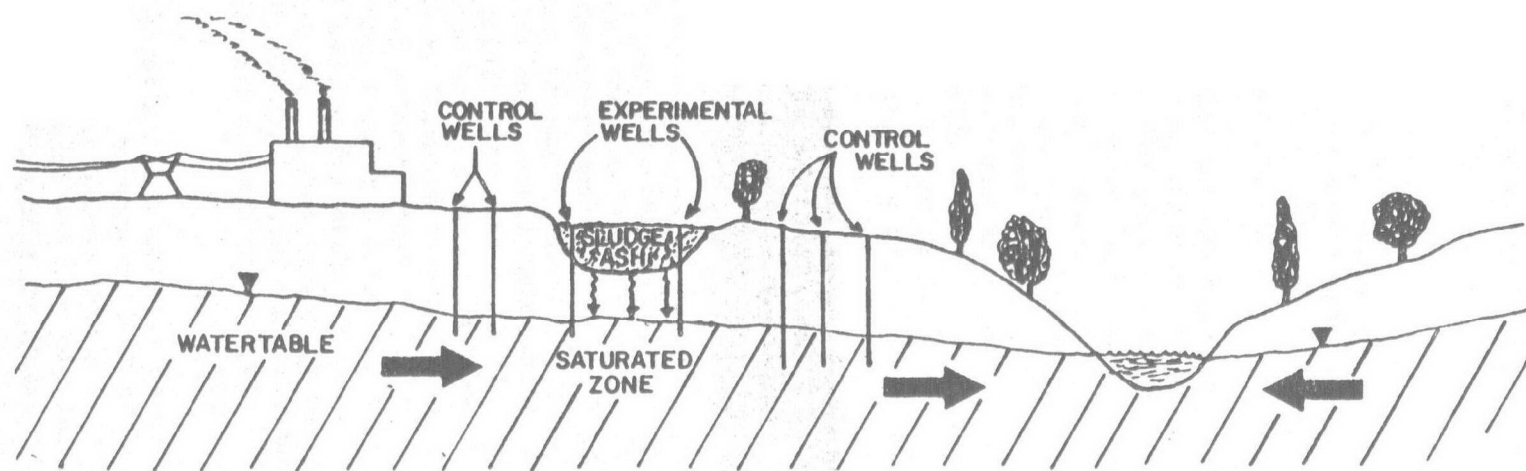
Figure 1. Sketch of a typical disposal area showing sampling plan.

percent moisture, dry density, grain-size distribution, permeability and soil classification. Randomization was used to test for significant differences in physical properties (18). Vertical variability in selected bore holes was also evaluated but the small sample sizes did not allow the use of statistical tests in this case.

The samples of groundwater collected in this study were used to indicate loss of contaminants from the sludge/ash or the soil beneath the disposal site into the local groundwater. If contaminants were moving to the water table, their concentrations should be higher beneath and downgradient from the disposal site. A list of analyses run is given in Table 4. A randomization technique was employed to assess the significance of changes in water quality.

Soil samples from beneath the sludge/ash and from comparable depths outside the disposal site were treated in two ways. One aliquot of soil was extracted with distilled water to remove all ions that could be dislodged by water alone. A list of analyses run on this extract is given in Table 4. The distilled water extract gives a rate of release of material from the soil into the surrounding water. The water extract is assumed to represent the concentration present in water contacting the soil, not the maximum, total amount bound or confined in the soil. The distilled water leach then indicates the mobility of various ions being held in the soil. The most effective attenuation occurs when the soil beneath the sludge/ash shows an ability to accumulate a contaminant and to release the contaminant at a very slow rate. A statistical randomization technique was used to test the significance of differences observed between the composition of the distilled water extracts of soil samples collected directly beneath the sludge/ash and the composition of extracts from samples collected at comparable depths outside the disposal site. The significant results of the randomization test point out those elements at each site whose mobility in aqueous solution is effected by material from the landfill.

A second aliquot of fresh soil was digested with hot, 8N nitric acid to bring all ions not bound into silicate lattices into solution. A list of analyses run is also given in Table 4. This digest represents the total of all materials that could potentially be leached from the soil under the most severe conditions. Since it is assumed that there is no significant lateral movement of leachate through the soil above the water table, differences in composition between digests of these samples beneath and outside the disposal area can be interpreted as the loss or gain of material in the soil due to the presence of the sludge/ash. A statistical randomization technique was used to test for significant differences in composition between acid digests of soil samples collected directly below the sludge/ash and samples collected at comparable depths (and above the water table) outside the disposal site. The significant results from the randomization tests point out those elements at each site that are being added to the soil or removed from the soil by the movement of leachate from the disposal site.

If the soil beneath the disposal site was being altered by leachate from the sludge/ash, any change should be most pronounced directly beneath the sludge/ash and the magnitude of this change should decrease with depth. Samples of soil were taken at intervals down the boreholes to determine if any correlation between the concentration of materials in the soil and depth (or

TABLE 4. CHEMICAL CONSTITUENTS ANALYTICALLY DETERMINED IN GROUNDWATER
          FILTRATES, DISTILLED WATER EXTRACTS AND NITRIC ACID DIGESTS

Constituent	Groundwater filtrate	Water extract	Nitric acid digests
SO_4	X	X	
SO_3	X	X	
Cl	X	X	
NO_3-N	X	X	
NO_2-N	X	X	
CN	X	X	
TOC	X	X	
Ca	X	X	
Fe	X	X	X
K		X	X
Mg	X	X	
Mn	X	X	X
Na	X	X	
As	X	X	X
B	X	X	X
Be	X	X	X
Cd	X	X	X
Cr	X	X	X
Cu	X	X	X
Hg	X	X	X
Ni	X	X	X
Pb	X	X	X
Se	X	X	X
Zn	X	X	X

sample elevation) could be observed. Correlation with sample elevation was only attempted with those elements that had shown a significant contrast in concentrations from samples under and outside the disposal site. A Spearman rank correlation technique was employed (18,19). The correlation technique made it possible to see if consistent relationships could be observed between sample elevation and sample composition in borings made inside and outside the disposal site.

Samples of soil collected near the top of the saturated zone both outside and inside the disposal site were examined to see if any effects of lateral movement of leachate below the water table could be observed. Distilled water leaches and nitric acid digests of these soil samples were analyzed. Plots of analyses were prepared to assess any changes in constituents that could be related to the presence of the sludge/ash. No attempt was made to evaluate these analyses statistically because of the small sample sizes involved.

SECTION 2

CONCLUSIONS

At all three FGC/ash disposal sites (K, L, and M) investigated, indications were found that FGC sludge/ash-related materials had moved into surrounding soils and groundwater. No consistent differences in physical properties (dry density, water content, soil permeability and grain size distribution) could be detected between the soil samples taken immediately below the disposal sites and at a comparable depth outside the disposal area. No conclusive evidence could be found that the untreated sludge/ash in the pits or ponds form an effective liner.

Analysis of groundwater samples collected at each of the three sites showed some evidence of movement of FGC sludge/ash-derived materials from the disposal pit or pond into the groundwater under the site. At all sites, increased levels of some constituents could be related to the presence of the disposal pit or pond. Increased lead and mercury levels were found under the disposal pond at site K. At site L, increased concentrations in the trace metals, iron, arsenic, chromium, and lead, could be found in groundwater under the disposal pit. At site M, groundwater from beneath the disposal pond showed significant increases in sodium, chloride, and sulfate. Distilled water extracts from soil samples under and outside the disposal sites showed very little contrast. The most consistent differences observed were increases in sodium and boron in the distilled water extracts from samples directly under the disposal pits or ponds. Examination of distilled water extracts taken from soil samples at or below the local water table showed that the maximum leachable levels of sodium, sulfate, and boron were consistently found under or down the groundwater gradient from the disposal areas.

Nitric acid digests prepared from soils below and away from the disposal sites showed no consistent differences at the three sites. This suggests that changes in soil composition cannot be easily related to the passage of leachate through the soil. The only elements that appeared to be readily fixed or exchanged into soil were calcium at site K and boron at site L.

In the site investigations reported here:

a)   there is no indication that FGC sludge/ash ponds or pits are self-sealing,

b)   there is evidence that FGC sludge/ash constituents move into surrounding soil and groundwater,

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c) there is no evidence that soils below the disposal sites are permanently retaining any FGC sludge/ash-derived materials with the exception of calcium and boron.

SECTION 3

RECOMMENDATIONS

FGC sludge/ash disposal sites can pollute surrounding groundwater and thus pose a significant threat to high-quality drinking water aquifers.

Ponds or pits for the disposal or storage of FGC sludge/ash should be engineered so as to prevent seepage from the pond or pits from moving into surrounding water and soil. There is no evidence that unaltered FGC sludge/ash in itself forms a suitable liner for a sludge and ash pond or pit.

Where the geologic and hydrologic conditions are such that contamination of usable groundwater is a possibility, plans for unsolidified sludge/ash disposal should include an artificial liner that will retain all water contacting the sludge materials. Soil attenuation is not adequate in most cases to prevent FGC sludge/ash-derived material from contaminating shallow aquifers.

An effective groundwater monitoring program should be included in plans for FGC sludge/ash disposal areas. Samples of water collected from wells adjacent to and down the groundwater gradient from the disposal site should be analyzed at regular intervals to insure the integrity of the containment system.

Additional research requirements exist particularly in the areas of evaluating the effectiveness and reliability of containment systems and designing adequate groundwater monitoring systems.

SECTION 4

MATERIALS AND METHODS

SITE SELECTION

Three electrical generating station disposal sites (containing mixed FGC sludge and ash) at different geographic areas in the central United States were selected for study. All sites were located in areas where precipitation and infiltration rates were sufficient to produce significant amounts of leachate. A brief summary of the important engineering and geologic charact-eristics of each site is presented in Table 5.

Some major factors effecting the character of the contaminants leaching from a disposal site are the type and amount of material placed in the site, the fossil fuel burned at the generating plant, boiler and scrubber operating conditions and the length of time the material has been in the site. Other factors effecting the character of sludge/ash leachate are oxidation-reduction conditions in the sludge and ash, and the temperatures in the disposal area. Ultimately, the concentration of pollutants in the groundwater is also related to the amount and chemical composition of local groundwater moving through the immediate area.

At site K (Figure 2), a 65-hectar pond has been receiving 31,750 metric tons per day of wet FGC sludge, fly ash and some bottom ash since the plant went on line in mid-1973. The pond can attain a maximum depth of 11 meters and has a life expectancy of 3 to 4 years as of the time of sampling. Imme-diately to the south of the disposal pond is a large exposed coal storage area. Runoff from the storage area also flows into the disposal pond.

The pit at site L covers 1.5 hectars with an average depth of approxi-mately 11 meters (Figure 3). Dumping of fly ash began in the southern portion of the pit in 1968. Beginning in mid-1973, FGC sludge and fly ash were dumped in the northern part of the pit. The middle third of the pit has not received any direct dumping of sludge or ash. Before the dumping of fly ash began, the pit was free draining. Shortly after dumping started, however, the pit began to retain water and now a pond exists in the pit throughout the year. The sludge disposal pit is approximately 2 kilometers from the generating plant. The FGC sludge disposed here is filter cake with a moisture content of approxi-mately 20%. Immediately west of the disposal area is a 40-hectar industrial tailings pond.

TABLE 5. SUMMARY OF THE CHARACTERISTICS OF THE THREE POWER
          GENERATION SITES SELECTED FOR STUDY

Characteristic	Site K	Site L	Site M
Unit size	820 Mw	70 Mw	130 Mw
Coal sulfur content	5.2%	3.0%	3.5%
Scrubber process type	Limestone	Lime	Limestone
Type of disposal operation	Settling pond	Pit	Settling pond
Geographic area within the U.S.	Central	Ohio Valley	Central
General geologic setting	Thin glacial outwash over bedrock	Glacial outwash (valley train deposits)	Alluvium
Mean annual precipitation	91 cm	105 cm	91 cm
Mean annual air temperature	13°C	14°C	13°C
Nature of waste	FGC sludge, fly and bottom ash	FGC sludge and fly ash	FGC sludge, fly and bottom ash
Liner used below waste material	None	None	None
Thickness of waste observed	2.49-5.49 m (avg. 3.99 m)	2.90-14.48 m (avg. 8.69 m)	2.29-4.36 m (avg. 3.33 m)
Thickness of unsaturated zone	2.44-9.00 m (avg. 6.91 m)	3.66-16.04 m (avg. 12.53 m)	4.36-7.86 m (avg. 5.72 m)
Nature of material in unsaturated zone	Clay	Clay, silty sand and gravel	Clay and silty sand
Average hydraulic conductivity below waste material	2.94×10^{-8} cm/sec	2.10×10^{-4} cm/sec	2.04×10^{-3} cm/sec
Dates of operation of site	1973 - present	1968 - present	1972 - present
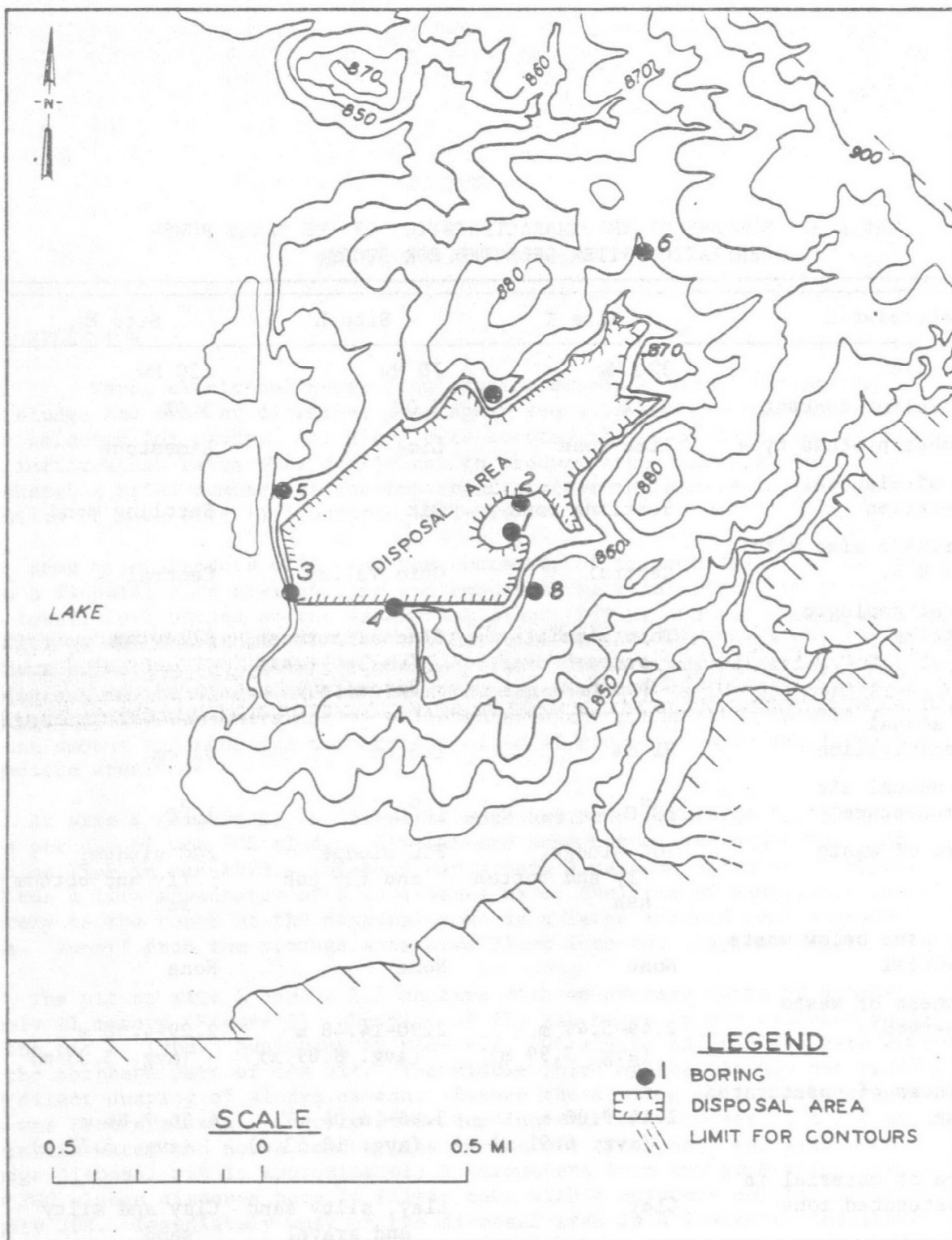
Figure 2.   Topographic map of site K.   1 foot = 0.305 meters.
Elevations are in ft above mean sea level.
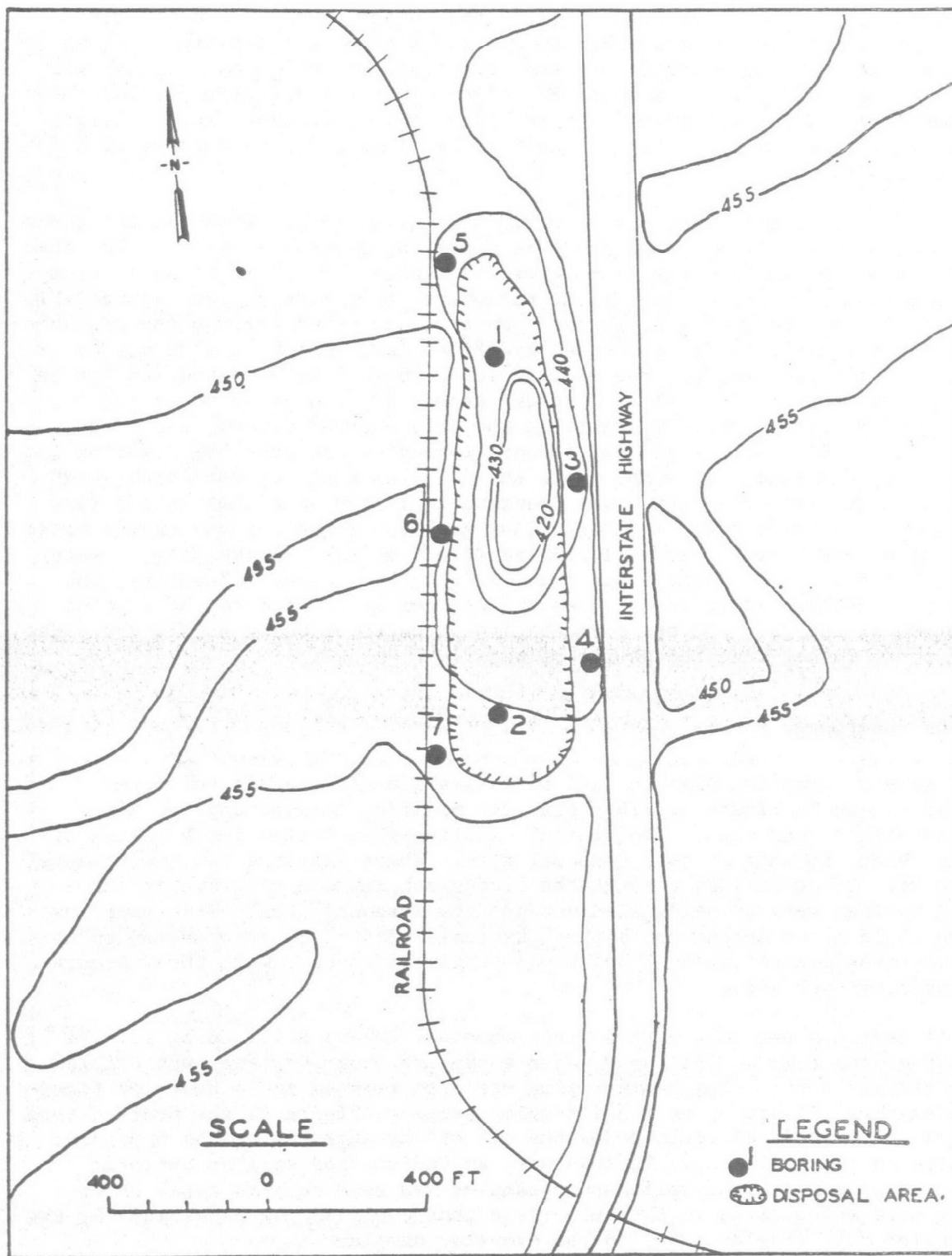
Figure 3. Topographic map of site L. 1 foot = 0.305 meters. Elevations are in ft. above mean sea level.

At site M, a 34-hectar disposal area began receiving FGC sludge, fly ash and bottom ash in 1972. Subsequent developments have resulted in off-site disposal of the majority of the bottom ash and some of the fly ash. Present practice is to discharge bottom ash into the central portion of the disposal area (see Figure 4) and fly ash and FGC sludge into the northern part of the disposal area. The southern section has received only bottom ash. A large coal storage area (approximately 30 hectars) is immediately southeast of the disposal basin.

The source of water that infiltrates the wastes is different at all three sites. At site K, the disposal pond was formed by damming a small valley that drained into the cooling lake. The water available for infiltration at this site is derived from operation of the scrubbers, plus rainfall and appreciable runoff from the surrounding hillsides. Water is recycled through the scrubber and any excess water beyond the capacity of the impoundment escapes over a spillway in the dam and into the cooling lake. Site L is an abandoned borrow pit and interrupts no natural surface drainage. The sludge as deposited contains very little water; rainfall is the only source of water available for infiltration. Site M is a series of ponds formed by constructing dikes on the floodplain. The sludge is pumped into the ponds as a slurry with high water content. After settling, the supernatant water is pumped either into a river or a sewage treatment plant. Future plans call for recycling the excess water to the scrubbers. Recycling will have no effect on the availability of water for infiltration. Permanent ponds exist at all three sites; therefore, the escape of contaminated water into the groundwater is related to the area of the bottom of the pond and the permeability of material below and at the sides of the pond, rather than the source of water.

SAMPLING PROCEDURES

A general sampling plan for all sites was generated using the model situation shown in Figure 1. This plan was modified to meet any specific requirements at each site. The general sampling plan called for a series of seven or eight borings at each disposal site. Where possible two experimental borings were to be drilled through the sludge/ash mixture and five or six control borings were to be drilled outside the disposal area. This sampling pattern would allow comparison between typical, uneffected groundwater and soil, and groundwater and soil which was in direct contact with the leachate draining from each site.

All sampling was done with a truck-mounted, rotary drill using 16.8 cm OD, hollow-stem auger. The auger, with a central plug in place, was drilled to the desired depth. The central plug was then removed and a Hvorslev fixed-piston sampler (Figure 5) or a split-spoon sampler (Figure 6) was pressed into the sediment or soil directly below the end of the auger using the hydraulic cylinders on the drill rig. In this way, an undisturbed soil or sediment sample was obtained. The split-spoon sampler was used only in cases where objects were encountered in the subsurface that could not be penetrated by the thin-walled tube (Shelby tube) on the Hvorslev sampler.
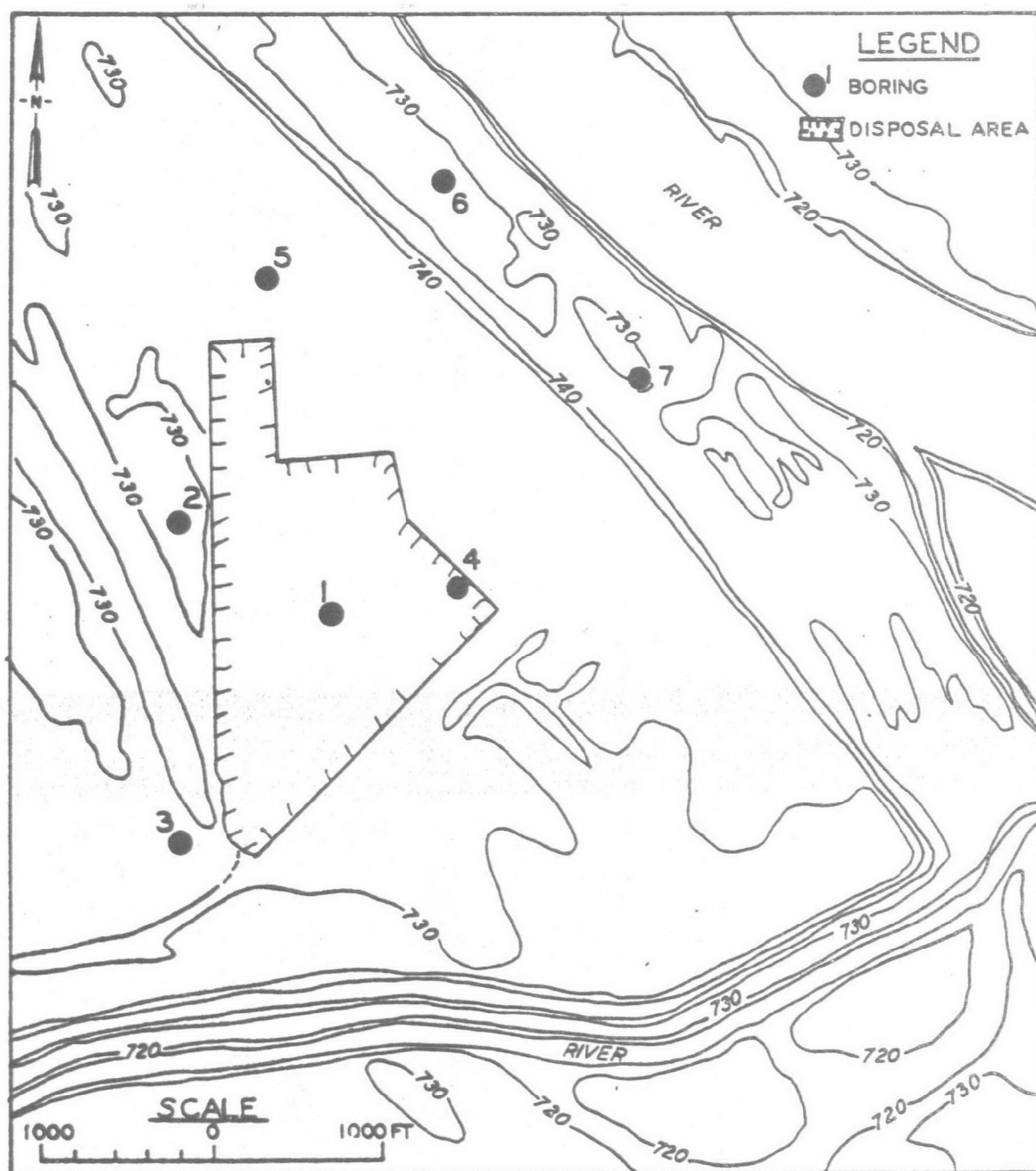
Figure 4. Topographic map of site M. 1 foot = 0.35 meters. Elevations are in ft. above mean sea level.

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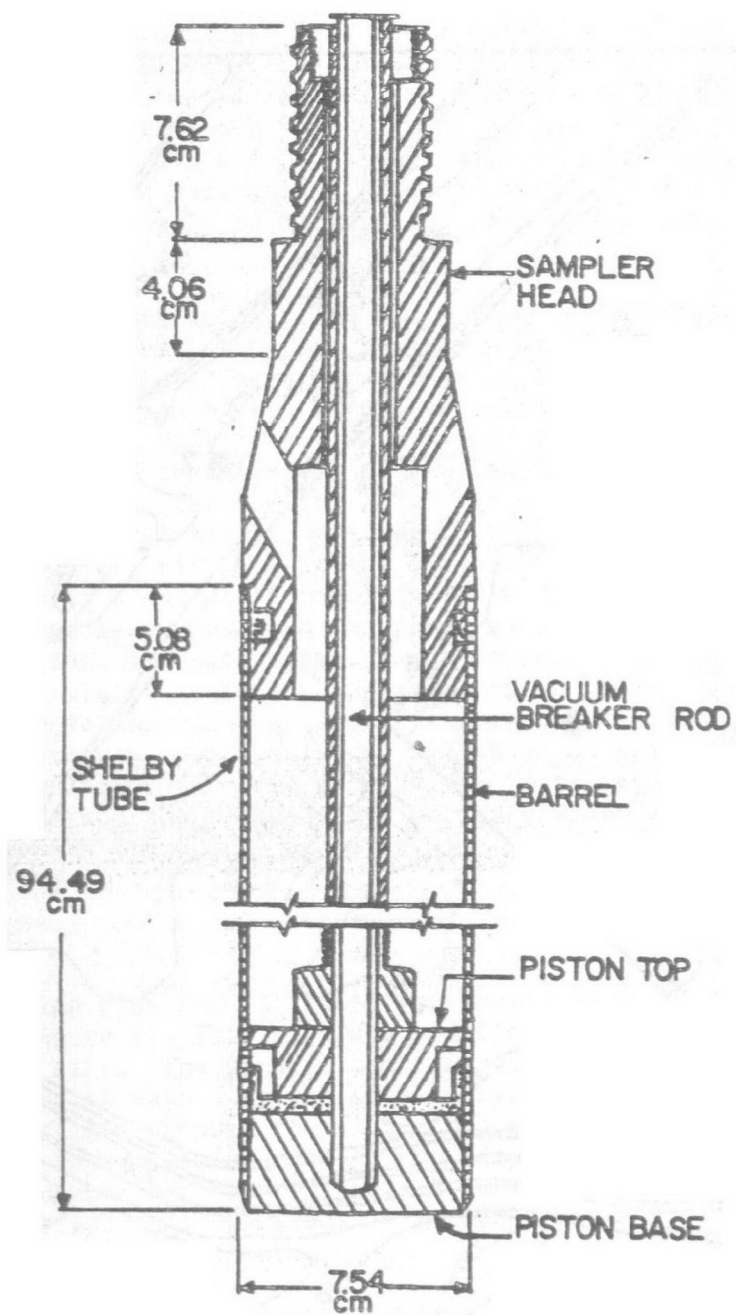
Figure 5. Sketch of Hvorslev fixed piston sampler.

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HEAD

SOLID OR SPLIT BARREL

45.72 to 91.44 cm

SHOE
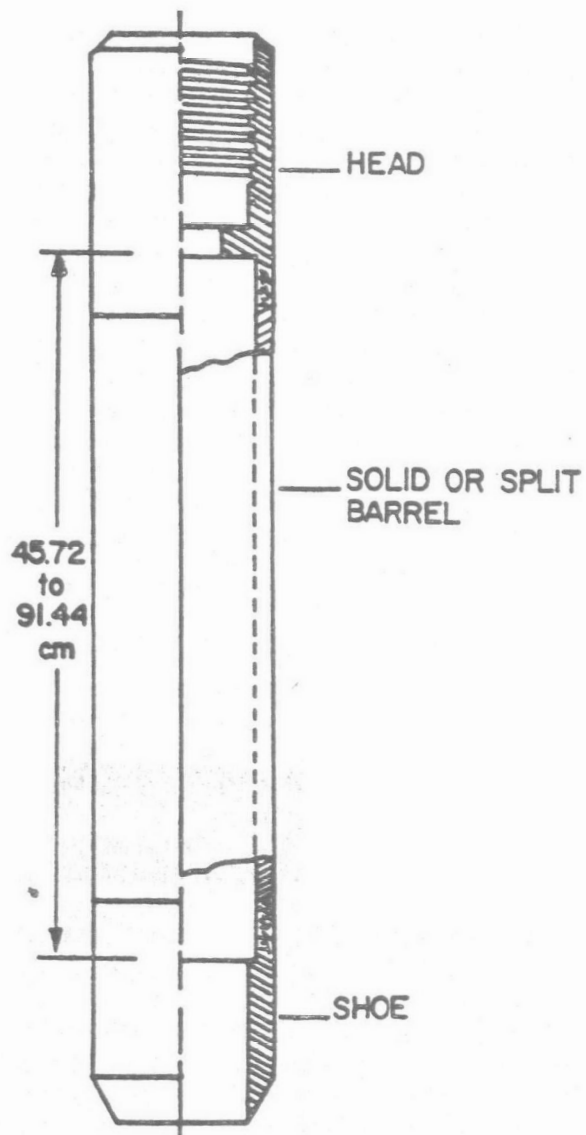
O.D. 5.08cm, 6.35cm, 7.62cm, or 8.89cm
I.D. 3.81cm, 5.08cm, 6.35cm, or 7.62cm

Figure 6.   Sketch of split spoon sampler.

The vertical distribution of soil/sediment samples collected down the hole was arranged in a way to maximize the probability of collecting samples at two critical points in the boring; the sludge/ash-soil interface and the top of the saturated zone. Since the strongest effects of leachate on the local material should occur directly below the wastes, a sample was always taken at the sludge/ash-soil interface. Sampling was then continued at closely spaced intervals down the hole. The top of the water-saturated zone was predicted from water table measurements that had been recorded for other wells in the area and a series of closely spaced samples was taken in this interval. The borings were allowed to remain open for two to three days following the actual drilling, with the augers left in place. The auger flights served as a temporary well casing to prevent seepage from the surface from entering the well. Depth to groundwater was measured with a chalked steel tape and groundwater samples were obtained from the temporary wells by lowering a bailer into the top of the hollow-stem augers. After a groundwater sample was obtained, the auger was removed and the hole was backfilled with grout and/or bentonite to a point well above the water table. The filling was then completed with well cuttings. This was done to assure that the well would not act as a conduit for the flow of polluted water to the water table.

The locations for all borings at each FGC sludge/ash disposal site are given in Figures 2 - 4. The most probable configuration of the water table at each site, as deduced from water level measurements in the borings, is given in Figures A-1, B-1 and C-1 (in appendices). The descriptive well logs are also presented in the appendices (Tables A-2--A-9, B-2--B-8, and C-2--C-8). Tables A-10, B-9 and C-9 list all soil/sediment samples examined from each boring, giving their elevations and other relevant data.

Minor variations in the general sampling plan were necessary at sites L and M. In three instances at site L, auger wrap was used for chemical testing. These were samples 1C1, 2C1 and 5C2. Auger wrap consisted of material removed from the outside of the auger bit. Although the physical properties of auger wrap samples were disturbed, the chemical properties should be consistent with an undisturbed Hvorslev or split-spoon sample. At site M, the bearing capacity of the recently disposed material was too low to support a drill rig. Consequently, boring through the newly deposited sludge/ash material was impossible. The drill rig was placed on older, firm FGC sludge and borings 1 and 4 were made near the margin of the settling pond.

SAMPLE HANDLING AND PREPARATION TECHNIQUES

Two different types of soil samples were collected in the boring program; samples for physical testing and samples for chemical analysis. Groundwater samples were also taken from each boring for chemical analysis. The set of samples obtained for physical testing was used to determine soil class under the unified soil classification system (20), dry density, grain-size distribution, water content and permeability. These physical parameters were determined using standard engineering test procedures. This sample set was collected without disturbing the soil more than necessary. The samples were carefully packaged and sealed in coring tubes to retain the original moisture content and sample texture.

The groundwater bailed from each boring was transferred to polyethylene bottles which were labelled and packed in an insulated chest filled with crushed ice. The samples were stored under refrigeration and kept tightly capped until they were prepared for chemical analysis. The preparation consisted of centrifuging each sample at 2200 rpm for 30 minutes. The resulting supernatant was membrane-filtered through a 0.45-micron filter and split into five subsamples which were preserved as shown in Table 6.

Samples of soil for chemical analysis were collected simultaneously with the samples for physical testing, but no attempt was made to maintain the soil in an undisturbed condition. Each sample removed from the sampler or collected from the auger, was placed in a wide-mouthed polyethylene bottle, labelled and packed in an ice-filled chest. These soil samples were refrigerated during all subsequent transportation and/or storage. Two extracts were made from each soil sample; one with distilled water and one with 8N nitric acid. The materials that could be easily extracted with distilled water were considered transient and would readily be leached from the soil by dissolution in rainwater. The nitric acid digest would contain the transient materials, and also all the materials that could be solubilized by a strong, oxidizing acid. Those elements present as carbonates or sulfides, or adsorbed to clay minerals, to iron oxide or to insoluble organic materials would be freed (21); while elements in non-clay silicate lattices would be solubilized only to a minor degree (22).

For distilled water extracts, the contents of each sample bottle were mixed to assure a homogeneous sample. A 200-gram subsample of moist soil was weighed out into a 1000-ml polycarbonate centrifuge bottle and six hundred ml of distilled-deionized water was added to each. The centrifuge bottles were shaken on a rotary shaker for one hour, and then centrifuged at 2200 rpm for 30 minutes. The supernatant was filtered through a 0.45-micron membrane filter. The filtrate was split into five subsamples for chemical analysis. The subsamples were preserved as outlined in Table 6.

A second subsample consisting of 50 grams of moist soil was taken from each sample bottle for nitric acid digestion. In each digestion, the soil was weighed into a 250-ml fluorocarbon beaker and 60 ml of 8N reagent-grade nitric acid was added. The soil-acid suspension was heated to 95°C for 45 minutes and stirred every fifteen minutes. After cooling to room temperature, the suspension was filtered through a 0.45-micron membrane filter. The digested soil was washed in the filter three times with 20-ml portions of 8N nitric acid. The filtrate was quantitatively transferred to a 250-ml volumetric flask and brought up to volume with 8N nitric acid and then stored in a polyethylene bottle. No preservation procedure was necessary.

A third subsample was taken from each sample bottle to determine the moisture content of the soil. These moisture contents were used to correct subsequent chemical analyses so that soil acid digests could be expressed in milligrams per kilogram dry weight of soil.

TABLE 6. METHODS OF PRESERVATION OF WATER EXTRACTS AND FILTERED GROUNDWATER
SUBSAMPLES FOR CHEMICAL ANALYSIS

Chemical species to be determined	Method of preservation
SO_4, SO_3, Cl, NO_3, NO_2.	Refrigeration to $4^\circ C$
CN	Samples brought to pH 11 with NaOH
Total organic carbon (TOC)	Refrigeration to $4^\circ C$
Ca, Fe, K, Mg, Mn, Na, As, B, Be, Cd, Cr, Cu, Ni, Pb, Se, Zn	Samples acidified with HCl to pH 1
Hg	$KMnO_4$ added and samples acidified with HNO_3 to pH 1

PHYSICAL TESTING METHODS

The physical tests run on these samples included water content, sample dry density, permeability, and grain-size analysis. Data gathered from these tests and visual examination of the samples were used to classify the materials into standard soil engineering categories. All testing was done using standard soil engineering methods (23).

To determine water content, a sample taken from the sealed coring tube was weighed into a tared sample dish, dried at 110°C and weighed periodically until a constant weight was obtained.

Sample dry density (or dry unit weight) is the weight of oven-dried soil per unit volume of soil. This measurement can be made in two different ways: by trimming the soil sample into a precisely measured regular shape and drying and weighing the trimmed sample; or, by sealing the surface of a soil specimen with wax and measuring its volume by water displacement, then removing the sealing material and drying and weighing the specimen. The water displacement procedure was used with samples containing gravel or other coarse material that prevented the sample from being trimmed accurately.

Grain-size analysis was performed by sieving the dried, disaggregated soil through a standard sieve series. Standard hydrometer density measurements were run on a suspension prepared from the fraction passing the 200-mesh sieve.

Permeability measurements were made using a constant-head test system with coarse-grained soils, and a falling-head test system with fine sands or clays. In all cases standard procedures and equipment were employed (23).

The major characteristics (especially grain-size analyses and characteristics of the fine fraction) of the samples were used to classify the soils. The USCS classification system is summarized and corresponding USDA classes are given in Table 7.

CHEMICAL ANALYTICAL METHODS

The techniques used in analyzing the filtered groundwater samples, distilled water extracts and nitric acid digests are summarized in Table 8. In all cases, the samples were run within the recommended time limits for the storage of samples (24).

The analyses of groundwater samples are given in milligrams per liter of filtered sample. The water extracts are also presented in milligrams per liter of filtered extractant. The water extract represents an equilibrium or near equilibrium solution with respect to the solid phases and the adsorbed phases in the soil; therefore, the analytical data are presented on a solution basis rather than a dry weight basis. The nitric acid digests are a determination of the total acid digestible fraction; therefore, the results are presented as milligrams extracted per kilogram dry weight of soil.

TABLE 7. DESCRIPTIONS OF USCS SOIL GROUPS (20)

Group symbol	Typical group description	Example of corresponding USDA soil textural description
GW	Well-graded (poorly-sorted) gravels, gravel-sand mixtures, little or no fines	Gravel, gravelly sand
GP	Poorly-graded (well-sorted) gravels, or gravel-sand mixtures, little or no fines	Same as above
GM	Silty gravels, gravel-sand-silt mixtures	Very gravelly sand or silt loam
GC	Clayey gravels, gravel-sand-clay mixtures	Very gravelly clay loam
SW	Well-graded (poorly-sorted) sands, gravelly sands, little or no fines	Same as above
SP	Poorly-graded (well-sorted) sands, gravelly sands, little or no fines	Coarse to fine sand
SM	Silty sands, sand-silt mixtures	Loamy sand or sandy loam
SC	Clayey sands, sand-clay mixtures	Sandy clay loam or sandy clay
ML	Inorganic silts, very fine sands, clayey silts, low plasticity	Silt or silt loam
CL	Inorganic clays, low to medium plasticity, lean clays	Silty clay loam or clay loam
OL	Organic silts and organic silty clays of low plasticity	Mucky silt loam
MH	Inorganic silts, micaceous or diatomaceous fine, sandy or silty soils, elastic silts	Micaceous or diatomaceous silt

TABLE 7. DESCRIPTIONS OF USCS SOIL GROUPS (20) (continued)

Group symbol	Typical group description	Example of corresponding USDA soil textural description
CH	Inorganic clays of high plasticity, fat clays	Silty clay
OH	Organic clays of medium to high plasticity, organic silts	Mucky silty clay
Pt	Peat and other highly organic soils	Mucks and peots

TABLE 8. TECHNIQUES USED IN THE ANALYSIS OF DISTILLED WATER EXTRACTS,
NITRIC ACID DIGESTS AND GROUNDWATER FILTRATES

Chemical species	Procedures and/or instrumentation*	Lowest reporting concentration (ppm)
SO_4	Standard Turbidimetric Method[+] in combination with a Varian Model 635 Spectrophotometer	8
SO_3	Standard Potassium Iodide-Iodate Titration method[+]	1
Cl	Standard Mercuric Nitrate Titration method[+]	5
NO_3-N	Technicon II Auto Analyzer, Industrial Method no. 100-70W±	0.01
NO_2-N	Same as above	0.01
CN	Technicon II Auto-Analyzer, Industrial Method no. 315-74W±	0.01
TOC	Determined with Envirotech Model No. DC 50 TOC Analyzer	1
Ca	Determined with a Spectrametrics Argon Plasma Emission Spectrophotometer Model II	0.03
Fe	Determined with Perkin-Elmer Heated Graphite Atomizer Atomic Absorption Unit	0.003
K	Determined with a Spectrametrics Argon Plasma Emission Spectrophotometer Model II	0.05
Mg	Same as above	0.03
Mn	Determined with Perkin-Elmer Heated Graphite Atomizer Atomic Absorption Unit	0.001
Na	Determined with Spectrametrics Argon Plasma Emission Spectrophotometer Model II	0.03

(continued)

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TABLE 8 (continued)

Chemical species	Procedures and/or instrumentation*	Lowest reporting concentration (ppm)
As	Determined with a Gaseous Hydride System, Perkin-Elmer Atomic Absorption Unit	0.001
B	Determined with a Spectrametrics Argon Plasma Emission Spectrophotometer Model II	0.02
Be	Determined with a Perkin-Elmer Heated Graphite Atomizer Atomic Absorption Unit	0.0005
Cd	Same as above	0.0003
Cr	Same as above	0.003
Cu	Same as above	0.003
Hg	Determined with a Nisseisangyo Zeeman Shift Atomic Absorption Spectrophotometer	0.0002
Ni	Determined with a Perkin-Elmer Heated Graphite Atomizer Atomic Absorption Unit	0.005
Pb	Same as above	0.002
Se	Same as above	0.005
Zn	Same as above	0.014

* Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

† Standard Methods for the Examination of Water and Wastewater, American Public Health Association, New York, 13th Edition, 1971.

± Technicon Industrial Systems, Tarrytown, New York.

SECTION 5

RESULTS AND DISCUSSION

PHYSICAL TESTING

The geologic materials under a FGC disposal site are subjected to several different effects due to the presence of the waste materials. Any changes observed in the soil are probably brought about by contact with leachate saturated with respect to calcium, sulfate and sulfite. FGC sludge leachate typically has a pH between 9 and 11 and contains high concentrations of sodium and chloride. The goal of the physical testing program is to detect any changes in the soil engineering parameters which could be related to the presence of the FGC sludge/ash disposal site. Data for physical testing of soil samples from all three sites are given in Tables 9-11. The most pronounced effects should occur directly below the sludge/soil interface . For this reason Table 12 compares physical properties of the topmost soil samples taken below the disposal area with soil samples taken at comparable depths outside the disposal area. Interaction between the sludge (and its leachate) and the underlying soil would be expected to:

a) increase the dry density of the sediment (or soil) because the calcium sulfate/sulfite sludge would be filling intergranular spaces in the sediment under the disposal site;

b) increase water holding capacity in coarse-grained sediments due to the increased surface area brought about by the addition of fine-grained material;

c) decrease the permeability due to obstruction of interpore connections in the sediment; and,

d) increase the percent fines in grain-size analyses due to the infiltration of small sludge crystals or crystal aggregates.

At site K, there was no consistent influence of the disposal site on the physical characteristics measured in soil below the site. Only one sample was tested from under the disposal site and it showed a very slightly decreased dry density, increased water content and a slightly higher permeability. The percent fine-grained material was approximately the same under the site and outside the site. The usual low permeability observed in shales and clays found at this site minimizes any infiltration and therefore its effects on physical properties. At site L, a pattern of changes in physical characteristics closer to that predicted was observed. The most obvious change was the

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TABLE 9. PHYSICAL TESTING DATA FOR SAMPLES FROM SITE K

Boring no.	Sample no.	Depth (m)	Dry density (g/cc)	Water content (%)	Permeability or hydraulic cond. (cm/sec)	Classification
1	P3	5.76-5.91	--	--	--	Lean clay (CL) with sand, light brown
2	P1	5.76-6.10	1.49	29.2	2.94×10^{-8}	Plastic clay (CH) with trace sand, brown
	P3	8.70-8.92	--	--	--	Silty clay (CL), dark gray
3	P1	1.74-2.15	1.72	18.5	2.55×10^{-8}	Lean clay (CL) with sand, dark brown
	P3	4.79-5.29	1.67	19.5	6.30×10^{-9}	Plastic clay (CH), brown
	P5	7.83-8.32	--	--	--	Lean clay (CL) with trace sand, dark gray
4	P1	3.26-3.75	1.73	17.7	6.20×10^{-9}	Lean clay (CL) with sand, dark brown
	P4	7.83-8.29	1.60	23.4	23.0×10^{-8}	Plastic clay (CH) with trace sand, dark gray
5	P1	1.74-2.20	1.58	24.9	4.70×10^{-9}	Plastic clay (CH) with sand, dark brown
	P2	2.65-3.12	1.71	20.0	1.03×10^{-8}	Lean clay (CL) with sand, dark brown
6	P1	2.96-3.41	--	--	--	Lean clay (CL) with trace sand, light brown
	P2	3.87-4.19	1.81	18.0	6.80×10^{-9}	Lean clay (CL) with sand, dark brown
7	P1	2.96-3.31	--	--	--	Sandy clay (CH), brown

Note: -- indicates no data available.

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TABLE 10. PHYSICAL TESTING DATA FOR SAMPLES FROM SITE L

Boring no.	Sample no.	Depth (m)	Dry density (g/cc)	Water content (%)	Permeability or hydraulic cond. (cm/sec)	Classification
1	P1	3.14- 3.81	1.65	20.2	1.82×10^{-6}	Lean clay (CL) with sand, brown
2	P1	14.72-14.78	--	--	--	Silty sand (SM), light brown
	P2	15.67-15.97	1.84	9.5	4.20×10^{-4}	Gravelly sand (SP-SM), brown
3	P1	6.64- 6.95	1.78	5.7	1.18×10^{-3}	Gravelly sand (SP-SM), brown
4	P1	3.87- 4.30	1.51	9.4	1.92×10^{-3}	Silty sand (SM), light brown
	P2	8.20- 8.41	1.87	4.1	2.76×10^{-3}	Gravelly, silty sand (SM), dark brown
6	P1	4.51- 4.69	1.61	8.3	1.02×10^{-3}	Silty sand (SM), gray
	P2	12.71-12.80	--	--	--	Gravelly sand (SP), gray
7	P1	4.51- 4.91	1.53	13.2	1.13×10^{-3}	Silty sand (SM), brown

Note: -- indicates no data available.

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TABLE 11. PHYSICAL TESTING DATA FOR SAMPLES FROM SITE M

Boring no.	Sample no.	Depth (m)	Dry density (g/cc)	Water content (%)	Permeability or hydraulic cond. (cm/sec)	Classification
1	P1	4.01-4.27	1.55	12.7	2.44×10^{-3}	Sand (SP), gray
	P2	5.85-6.25	1.48	13.5	7.92×10^{-4}	Silty sand (SM), gray
	P3	7.10-7.62	1.58	23.5	2.89×10^{-3}	Sand (SP-SM), gray
2	P1	0.55-1.07	1.51	8.2	2.41×10^{-3}	Sand (SP-SM), gray
	P4	4.36-4.82	1.62	22.2	2.94×10^{-3}	Sand (SP), light brown
3	P1	0.55-1.01	1.39	8.5	1.74×10^{-4}	Silt (ML), gray
	P2	1.46-1.77	1.49	13.3	7.27×10^{-5}	Sandy silt (ML), brown
	P3	3.63-4.08	1.42	29.0	4.94×10^{-4}	Silty sand (SM), brown
	P4	4.36-4.88	1.59	25.2	1.80×10^{-3}	Silty sand (SM), gray
4	P1	2.53-3.05	1.11	44.4	1.54×10^{-5}	Plastic clay (CH) with sand, dark gray
	P2	3.44-3.87	1.30	39.2	2.17×10^{-8}	Plastic clay (CH), gray
	P3	5.58-5.88	1.49	4.5	1.75×10^{-3}	Silty sand (SM), brown
	P5	7.86-8.38	1.62	22.3	2.42×10^{-3}	Sand (SP-SM), gray
5	P4	5.12-5.64	1.64	20.1	2.42×10^{-3}	Sand (SP), brown
6	P1	0.55-1.07	1.60	19.3	1.96×10^{-6}	Plastic clay (CH), gray
	P2	1.46-1.92	1.49	3.1	1.95×10^{-3}	Silty sand (SM), gray
	P3	3.57-3.96	1.60	12.2	2.29×10^{-3}	Sand (SP-SM), gray
7	P1	0.55-0.76	1.29	30.4	4.42×10^{-3}	Plastic clay (CH), gray
	P2	1.46-1.98	1.33	36.2	1.79×10^{-6}	Plastic clay (CH), gray
	P3	3.60-4.11	1.21	45.0	1.86×10^{-7}	Plastic clay (CH) with trace sand, dark gray
	P4	5.12-5.64	1.56	26.6	7.09×10^{-5}	Silty sand (SM), gray

Note: -- indicates no data available.

TABLE 12. COMPARISON OF THE PHYSICAL PROPERTIES OF THE UPPERMOST SOIL/SEDIMENT SAMPLES COLLECTED WITHIN AND OUTSIDE THE DISPOSAL SITE

Sample	Location (inside/outside)	Dry density (gm/cc)	Water content (%)	Permeability or hydraulic cond. (cm/sec)	Weight % finer than 200 mesh
K2P1	inside	1.49	29.2	2.94×10^{-8}	95
K3P1	outside	1.72	18.5	2.55×10^{-8}	89
K4P1	outside	1.73	17.7	6.20×10^{-9}	89
K5P1	outside	1.58	24.9	4.70×10^{-9}	95
K6P1	oustide	--	--	--	95
K7P1	outside	--	--	--	84
L1P1	inside	1.65	20.2	1.82×10^{-6}	91
L2P2	inside	1.84	9.5	4.20×10^{-4}	--
L3P1	outside	1.78	5.7	1.18×10^{-3}	8
L4P1	outside	1.51	9.4	1.92×10^{-3}	15
L6P1	outside	1.61	8.3	1.02×10^{-3}	15
L7P1	oustide	1.53	13.2	1.13×10^{-3}	27
M1P1	inside	1.55	12.7	2.44×10^{-3}	5
M4P1	inside	1.11	44.4	2.00×10^{-5}	93
M2P1	outside	1.51	8.2	2.41×10^{-3}	6
M3P1	outside	1.39	8.5	1.70×10^{-4}	99
M6P1	outside	1.60	19.3	2.00×10^{-6}	98
M7P1	outside	1.29	30.4	4.40×10^{-4}	98

Note: -- indicates no data available.

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decreased permeability found in samples from beneath the disposal pit. At least one soil sample under the disposal area showed increased dry density, increased water content and a higher percentage of fines. At site M, great variability in soil type was observed at the disposal site (see Table 11) and this masked the effects that might be produced by the disposal pond. If homogenous coarse-grained sediments underlie the disposal area, it is possible to detect physical changes that can be related to the presence of the disposal site; but these effects are easily concealed by natural variations in sediment types. Although there is some suggestion of decreased permeability at the sludge-soil interface at sites L and M there is no conclusive evidence of self-sealing under the sludge pit or pond.

CHEMICAL ANALYSIS OF GROUNDWATER

The goal on the groundwater investigation is to determine if changes in chemical parameters observed in different borings at each site could be related to the position of the boring underneath or outside the disposal area. Data for chemical analysis of the groundwater samples are given in Tables 13, 14, and 15.

Published analyses of FGC sludge liquors and elutriates indicate that high and variable levels of many chemical constituents can be released to contacting waters (Table 16). As would be expected, calcium and sulfate are found at extremely high levels -- >700 and >2000 ppm respectively in typical sludge liquor samples. Calcium levels in high quality water supplies are normally around 10 ppm, and the calcium limit for water of good potability is about 200 ppm, producing a very hard water. Water quality standards (25) recommend sulfate levels of less than 250 ppm due to taste and laxative effects; ideal drinking water having none or a trace. Sludge liquors also contain trace metals which are contributed mainly by ash co-disposed with the FGC sludge. Many of these trace metals occur in quantities which are well above the levels permitted in public drinking water supplies. The most frequent problems are excessive amounts of boron, cadmium, chromium, iron, lead, manganese, and selenium (16). Chloride typically runs about 10 times (median of 2300 ppm) the drinking water standards and thus constitutes a major problem as it is always present in soluble forms which are easily leached into contacting waters.

The randomization test (Table 17) did indicate significant contrasts between groundwater samples taken underneath and outside the disposal sites. Significant increases in means between samples under the site as contrasted to outside the site could be found in mercury and lead at site K; iron, arsenic, chromium and lead at site L; and in sulfate, chloride and sodium at site M.

The experimental borings at site K were made through pads of bottom ash dumped into several feet of standing water in the pond. At this site, one of the holes under the disposal site (boring 2) showed indications of being badly contaminated by sludge pond liquor. Sulfate, iron, manganese, boron and chromium all were found at higher concentrations than are acceptable for drinking water. In contrast, boring 1 which is also within the pond and only about 100 meters from boring 2, showed no evidence of infiltrating pond liquor

TABLE 13. CHEMICAL COMPOSITION OF GROUNDWATER OBTAINED FROM BORINGS AT SITE K

Parameters	Up groundwater gradient Boring 6	Under site Boring 1	Under site Boring 2	Down groundwater gradient Boring 3
SO_4	900	180	1400	42
SO_3	1	<1	1	<1
Cl	10	5	5	5
NO_3-N	0.02	0.06	0.03	0.11
NO_2-N	<0.01	0.01	<0.01	<0.01
CN	<0.01	<0.01	<0.01	<0.01
TOC	10	5	12	6
Ca	95.00	49.00	59.00	65.00
Fe	0.100	0.055	0.534	0.079
Mg	70.00	18.00	39.00	20.00
Mn	0.117	0.009	4.430	0.123
Na	310.00	82.00	23.00	95.00
As	ND	ND	ND	ND
B	0.34	0.22	1.07	0.03
Be	0.0390	0.1130	0.0280	0.0390
Cd	0.0004	<0.0003	0.0003	<0.0003
Cr	<0.003	<0.003	0.076	<0.003
Cu	<0.003	<0.003	<0.003	<0.003
Hg	<0.0002	0.0017	0.0004	<0.0002
Ni	0.456	0.251	1.360	0.365
Pb	0.002	0.002	0.003	<0.002
Se	ND	ND	ND	ND
Zn	0.170	0.082	0.090	0.170

Note: All values are in mg/ℓ.

ND = Not determined.

TABLE 14. CHEMICAL COMPOSITION OF GROUNDWATER OBTAINED FROM BORINGS AT SITE L

Parameters	Up groundwater gradient Boring 3	Boring 4	Under site Boring 1	Boring 2	Down groundwater gradient Boring 5	Boring 6	Boring 7
SO_4	249	139	ND	ND	ND	1399	299
SO_3	<1	<1	ND	ND	ND	<1	<1
Cl	35	30	ND	ND	ND	50	35
NO_3-N	5.08	6.60	ND	ND	ND	7.25	3.42
NO_2-N	0.05	0.06	ND	ND	ND	0.04	0.04
CN	0.01	0.01	<0.01	<0.01	<0.01	0.07	ND
TOC	14	12	19	10	29	22	ND
Ça	215.00	169.10	325.00	272.00	235.00	432.00	212.4
Fe	<0.003	<0.003	0.117	0.103	<0.003	<0.003	<0.003
Mg	71.90	68.80	64.60	50.90	93.10	160.00	64.80
Mn	9.230	6.390	1.290	2.81	3.780	12.000	2.070
Na	22.40	18.90	18.60	60.00	22.70	30.20	31.90
As	<0.005	<0.005	0.008	0.006	0.008	<0.005	<0.005
B	0.76	1.56	1.99	3.48	1.93	4.22	4.71
Be	0.0050	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cd	0.0003	<0.0003	0.0008	<0.0003	<0.0003	0.0007	0.0003
Cr	<0.003	<0.003	0.003	0.003	<0.003	<0.003	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Hg	ND	ND	<0.0002	<0.0002	ND	ND	<0.0002
Ni	0.082	0.054	0.024	0.021	0.048	0.047	0.029
Pb	<0.002	<0.002	0.008	0.005	<0.002	<0.002	<0.002
Se	0.005	<0.005	<0.005	0.005	<0.005	0.005	0.007
Zn	<0.014	<0.014	0.014	0.019	0.020	<0.014	<0.014

Note: All values are in mg/l.

ND = Not determined.

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TABLE 15. CHEMICAL COMPOSITION OF GROUNDWATER OBTAINED FROM BORINGS AT SITE M

| Parameters | Up groundwater gradient | | Under site | | Down groundwater gradient | | |
	Boring 2	Boring 3	Boring 1	Boring 4	Boring 5	Boring 6	Boring 7
SO_4	124	69	259	499	54	99	219
SO_3	<1	<1	<1	<1	2	<1	<1
Cl	15	10	45	30	<5	15	15
NO_3-N	9.24	4.10	0.61	0.12	0.68	0.49	0.14
NO_2-N	0.09	0.13	0.06	0.03	0.05	0.04	0.04
CN	<0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	11	10	18	13	13	18	21
Ca	158.30	177.80	121.60	221.00	148.70	151.90	225.00
Fe	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Mg	34.00	47.40	2.80	11.50	43.50	41.90	94.10
Mn	0.716	1.740	<0.002	0.566	1.130	1.350	2.34
Na	8.20	10.60	87.70	81.40	11.70	21.50	61.70
As	<0.005	0.005	<0.005	0.006	0.006	0.005	0.005
B	0.32	0.22	0.64	4.40	0.25	0.84	0.82
Be	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cd	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003
Cr	<0.003	<0.003	0.014	<0.003	<0.003	<0.003	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Hg	0.0004	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002
Ni	0.008	0.011	0.014	<0.005	0.011	<0.005	<0.005
Pb	0.009	0.002	0.006	<0.002	0.002	0.002	<0.002
Se	0.009	0.035	<0.005	0.011	<0.005	0.011	0.008
Zn	0.014	<0.014	<0.014	0.041	0.090	0.018	<0.014

Note: All values are in mg/ℓ.

ND = Not determined.

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TABLE 16. TYPICAL CONCENTRATIONS OF SELECTED CONSTITUENTS IN FGC SLUDGE
POND LIQUOR AND ELUTRIATES (3) AND SURFACE WATER CRITERIA FOR
PUBLIC WATER SUPPLIES (25,26)

Constituent	Eastern coal median conc. (ppm)		Western coal median conc. (ppm)		Maximum permissible level (ppm)
Arsenic	0.020	(15)*	0.009	(7)*	0.05
Beryllium	0.014	(6)	0.013	(7)	—
Boron	41.0	(1)	8.0	(1)	1.0
Cadmium	0.023	(11)	0.032	(11)	0.01
Calcium	700	(15)	720	(6)	—
Chromium	0.020	(15)	0.08	(7)	0.05
Cobalt	0.35	(3)	0.14	(2)	—
Copper	0.015	(15)	0.20	(7)	—
Iron	0.026	(5)	4.3	(2)	0.3
Lead	0.12	(15)	0.016	(7)	0.05
Manganese	0.17	(8)	0.74	(6)	0.05
Mercury	0.001	(10)	<0.01	(7)	—
Molybdenum	5.3	(1)	0.91	(1)	—
Nickel	0.13	(11)	0.09	(6)	—
Selenium	0.11	(14)	0.14	(7)	0.01
Sodium	118	(6)	—		—
Zinc	0.046	(15)	0.18	(7)	—
Chloride	2,300	(9)	—		250
Fluoride	3.2	(9)	1.5	(3)	1.0
Sulfate	2,100	(13)	3,700	(7)	250
Total dissolved solids	7,000	—	12,000	(3)	500

* Total number of observations recorded.

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TABLE 17. RESULTS OF RANDOMIZATION TESTS FOR CHEMICAL ANALYSIS OF
GROUNDWATER SAMPLES FROM SITES K, L AND M

Parameters	Site K	Site L	Site M
SO_4	NS	ND	S(increases)
SO_3	NS	ND	BDL
Cl	S(decreases)*	ND	S(increases)
NO_3-N	NS	ND	NS
NO_2-N	BDL	ND	NS
CN	NS	S(decreases)	S(decreases)
TOC	NS	NS	NS
Ca	S(decreases)	NS	NS
Fe	NS	S(increases)	NS
Mg	NS	S(decreases)	S(decreases)
Mn	NS	S(decreases)	S(decreases)
Na	S(decreases)	NS	S(increases)
As	NS	S(increases)	NS
B	NS	NS	NS
Be	NS	BDL	NS
Cd	NS	NS	NS
Cr	BDL	S(increases)	BDL
Cu	BDL	BDL	NS
Hg	S(increases)	ND	BDL
Ni	NS	S(decreases)	NS
Pb	S(increases)	S(increases)	NS
Se	ND	NS	NS
Zn	S	NS	NS

NS = Not significant at 80% confidence level.

S = Significant at 80% confidence level.

BDL = Below detection limits.

ND = Not determined.

*Refers to increase or decrease of constituent under disposal site relative
to outside.

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and was below the maximum permissible level for public water supplies for all constituents measured (25,26).

The groundwater sampling program at site K was complicated by impervious rock units and a low water table associated with the Pennsylvanian shales and limestones in the area. Five of the holes drilled failed to reach the saturated zone before encountering rock units that could not be penetrated by the auger. The well bored as a control up the postulated groundwater gradient from the disposal pond (boring 6) appears to have encountered a local, saturated zone created by infiltration of pond liquor into the colluvium and weathered shale forming the valley wall. The water level (elevation 260.06 m) measured in the well is 2.37 meters below the elevation of the surface of the disposal pond, suggesting the ponding has caused invasion (for distance of at least 200 meters) into the local colluvial materials.

In boring 3, which is down the apparent groundwater gradient from the disposal pond, levels for most chemical constituents are present in lower concentrations than that observed for boring 6 (the upgradient control hole). Boring 3 is in close proximity (approximately 10 meters distance) to the margin of a 1052-hectar cooling lake. Uncontaminated water from the lake could easily infiltrate the boring and bring about the low concentrations found in this groundwater sample. The elevation of water in the well is less than a meter below the level of the lake surface suggesting an hydraulic connection.

Of the two experimental borings through the disposal pond, one (boring 2) yielded a groundwater sample that approaches pond liquor in composition (see Table 16); while groundwater from the other (boring 1) appears to be much less effected by the surrounding waste. In fact, groundwater from boring 1 is (with the exception of sulfate content) within the range of composition observed for groundwater from other wells in the county (Table 18). The sulfate level was 190% higher than the highest value obtained from local water wells. The difference in water levels observed in experimental borings 1 and 2 (approximately 7 m) suggests no hydraulic connection exists between the two wells. The materials in the disposal pond include ash and FGC sludges. Sludge was noted, mixed with ash, in the hole during the drilling of boring 2. Boring 1, on the other hand, penetrated only ash and clay. The differences in water samples may be related to this inhomogeneous distribution of FGC sludge and ash in the disposal pond. The only trace metals that the randomization test indicated were sigificantly increased in groundwater below the disposal pond are lead and mercury; two elements probably associated with ash, present in both experimental borings.

In summary, at site K, only the groundwater in borings 6 (control boring) and 2 (experimental boring) show the effects of contamination from disposal pond liquor. The lack of wider contamination is probably due to the low permeability of the ash, clay and shale at and around boring 1, and the lack of permeability in the clay and shales under the disposal pond and between the pond and boring 3.

At site L, the experimental borings were made directly through the surface of the solid sludge that had been dumped into the pit. The material had

TABLE 18.   CHEMICAL COMPOSITION OF GROUNDWATER FROM WELLS NEAR SITE K

Local well number	31	36	3	19	32	29	Range
Conc. (mg/l)							
SO_4	62.0	52.0	4.1	30.0	41.0	30.0	4.1-62.0
Cl	5.0	14.0	1030.0	6.0	49.0	8.0	5.0-1030.0
F	0.3	0.1	0.1	0.1	0.8	0.4	0.1-0.8
CO_3	0	0	0	0	0	0	0
HCO_3	127	254	405	310	566	239	127-566
NO_3	12.0	8.9	8.0	44.0	0.4	6.2	0.4-44.0
SiO_2	12.0	17.0	17.0	7.5	12.0	12.0	7.5-17.0
Ca	43	99	315	114	28	65	28-315
Fe	0.18	0.05	0.04	0.24	2.30	0.15	0.04-2.30
K	ND	ND	ND	ND	ND	ND	
Mg	13.0	5.1	69.0	5.7	13.0	15.0	5.1-69.0
Mn	0.00	0.00	0.00	0.00	ND	0.00	0
Na	12.0	9.7	330.0	9.9	209.0	9.4	9.4-330.0

ND = Not determined.

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sufficient bearing capacity to support the drill rig. No standing water was present. At this site, all of the wells show some effects of pollutants. Even groundwater from upgradient borings show high levels of nitrate, manganese and boron. In the case of nitrate and manganese most of the groundwater samples analyzed in this study exceeded the levels obtained from other wells in the general area that intercept the same surface aquifer (Table 19). Groundwater from all the borings made in this investigation exceeded the concentration limits recommended for public water supplies for manganese and all except boring 3 exceeded the limits for boron. These high background levels are probably due to materials added to the groundwater by other industrial disposal pits in the area.

The most severe groundwater contamination at site L was not observed in the borings directly through the disposal pit (borings 1 and 2), but rather in the borings made down the groundwater gradient from the pit (borings 5, 7 and expecially 6). The randomization test points out significant differences between groundwater from the borings inside and outside pit; therefore in this case, the results are not as helpful in pointing out the materials leaching from the pit as they might be if the maximum pollutant concentrations had occurred (as would be expected) in borings through the waste. The randomization test did show significantly increased concentrations of iron, arsenic, chromium and lead in groundwater directly under the disposal pit. With the exception of calcium, magnesium and manganese the concentrations of all elements measured directly under the disposal area were within the range observed for water from wells drilled into this same aquifer (Table 19). Calcium in water from directly under the disposal pit was only 30% higher than the highest values obtained from local water wells. Concentrations of magnesium increased by about 40% under the disposal area and manganese increased by 3%. In the down gradient holes, calcium levels increased by 73% over the highest values for local water wells. Concentrations of magnesium increased by 208% and manganese by 344%. Sulfate levels, where measured, exceeded limits for public water supply and were up to 191% above highest level in local water wells.

The results of groundwater analyses at site L were unexpected in that the contaminants reached maximum levels in wells down the groundwater gradient from the disposal pit. These high levels may be due to the flow pattern involved in movement of groundwater through and under the disposal site. The borings in the pit are approximately centered so that the half of the disposal pit up the groundwater flow gradient is the only part contributing pollutants to the groundwater collected from the experimental borings. The down gradient control holes, on the other hand, (especially boring 6) are located on the edge of the disposal pit and receive water contaminated during travel under the entire width of the pit. In addition, water washing across the surface of the sludge and infiltrating at the edge of the pit may be a source of some of the contaminants appearing in the downdip borings.

At site M, the sludge is placed in the pond as a slurry and in some places has such low bearing capacity that it will not support the drill rig. At this site, the results of the randomization test indicate that sulfate, chloride and sodium levels are significantly increased in the groundwater under the disposal pond. This is as would be expected if typical sludge pond liquor were moving into the groundwater. Groundwater samples from borings

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TABLE 19.   CHEMICAL COMPOSITION OF GROUNDWATER FROM WELLS NEAR SITE L

Local well number	1	3	4	5	6	7	8	10	Range
Conc. (mg/ℓ)									
SO_4	56	480	97	96	64	130	290	74	56-480
Cl	24	13	82	140	42	15	190	18	13-190
F	0.5	0.2	0.2	0.2	0.3	0.2	0.2	0.2	0.2-0.5
CO_3	0	0	0	0	0	0	0	0	0
HCO_3	122	473	418	363	435	424	450	440	122-473
NO_3	0.00	0.00	2.70	0.40	1.30	0.00	0.60	0.00	0.00-2.70
SiO_2	7.6	17.0	16.0	16.0	20.0	19.0	17.0	19.0	7.6-20.0
Ca	40	250	120	140	100	120	220	120	40-250
Fe	0.10	6.30	1.50	14.000	0.30	1.30	3.60	6.90	0.10-14.00
K	3.0	3.9	2.9	2.4	3.0	2.6	7.5	1.4	1.4-7.5
Mg	10	52	38	40	35	36	49	35	10-52
Mn	0.18	1.60	0.90	1.20	0.00	0.50	2.70	0.21	0-2.70
Na	18	10	46	26	35	14	65	11	10-65

ND = Not determined.

under the disposal site contain concentrations of sulfate, manganese, boron and selenium that exceed the levels recommended for public water supply systems. Concentrations of sulfate, chloride, calcium and sodium in groundwater under the site are above the maximum concentrations found in published well water analyses in the same gravel aquifer (Table 20). Sulfate levels are up to 177% higher, chloride 32% higher, calcium 16% higher and sodium 47% higher.

At site M, the location of the control holes and the flow pattern in the aquifer allowed dilution to be observed in down gradient control borings 5 and 6. These two holes may be on the margin of the pollution plume. Control boring 7, however, has the highest levels of total organic carbon, calcium, magnesium and manganese observed in any groundwater sample from this site. Boring 7 may be showing the maximum effect of the plume from the disposal pond with possible added effects of pollution from coal storage pile drainage.

From the groundwater analyses of all three sites sampled, it can be concluded that FGC sludge (and ash) disposal degrades groundwater quality if contaminated water from the site is allowed to escape into the water table. At site K contaminants are found only in borings penetrating directly through FGC sludge or through a local, perched water table associated with the disposal pond. The lack of permeable geologic materials around the pond appears to be responsible for the high degree of pollutant containment that could be observed. At site L, the surrounding materials are permeable sands and gravels, but relatively dry sludge is being placed in a pit not in a settling pond and little water is maintained above the sludge and ash. Greatest contamination is observed in borings down the groundwater gradient rather than under the disposal pit. At site M, the settling/disposal pond is also situated on permeable sands and gravel. Degradation of groundwater quality was detected both beneath and down the groundwater gradient from the disposal pond.

CHEMICAL ANALYSIS OF DISTILLED WATER EXTRACTS .

The goal of the distilled water extraction procedure was to determine the availability of chemical constituents to water contacting the soils. The content of this soil extract varies depending on the following:

a) the original components of the soil and their solubilities in distilled water,

b) the way in which these components have interacted with leachate from the FGC sludge/ash mixture,

c) the extent to which water-soluble and leachate-soluble components of the soil have been removed through solution,

d) the solubilities of materials that are precipitated, filtered or · absorbed from the leachate, and

e) the amount and content of the interstitial water present in the samples.

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TABLE 20.   CHEMICAL COMPOSITION OF GROUNDWATER FROM WELLS NEAR SITE M

Local well number	1	16	15	14	Range
Conc. (mg/ℓ)					
SO_4	133	180	52	130	52-180
Cl	34	21	16	16	16-34·
F	0.5	0.4	0.2	0.3	0.2-0.5
CO_3	ND	ND	ND	ND	
HCO_3	390	219	790	337	219-790
NO_3	1.1	0.1	0.2	0.2	0.1-1.1
SiO_2	20	16	30	18	16-30
Ca	105	70	190	85	70-190
Fe	3.30	0.22	3.80	0.59	0.22-3.80
K	5.4	6.1	9.1	5.7	5.4-9.1
Mg	34	20	45	23	20-45
Mn	1.9	0.30	5.60	0.44	0.30-5.60
Na	46	60	23	55	23-60

ND = Not determined.

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Examination of pond liquor and elutriate from FGC sludges (Table 16) indicates that leachate from FGC disposal areas will be saturated with respect to calcium sulfate, will have a high pH, and will contain appreciable amounts of sodium, chloride, boron, cadmium, chromium, iron, lead, manganese and selenium. In passing through the soil/sediment, this solution will undergo ion exchange with clay minerals encountered, bring about increased solubilization of silica or aluminum, and cause some precipitation of metals dissolved in interstitial water but the major portion of material in solution in the leachate will remain in solution and will be carried into the groundwater. It is expected that attenuation by filtration, adsorption or ion exchange will reduce the pollution potential of the leachate only slightly.

Comparison of Distilled Water Extracts Beneath and Outside the Disposal Sites

The results of the chemical analyses of the distilled water extracts of the soil samples are given in Tables 21-26. The results of the randomization test are given in Table 27. At site K, significant differences in the composition of the distilled water extracts were observed only for nitrate and mercury. Nitrate showed a small increase in water extracts of sub-site soils. This may have been due to the presence of nitrates scrubbed from the flue gas. The small decrease in mercury observed in the distilled water extract from the sub-site soil may be related to the increased alkalinity (high pH) of the leachate from the sludge pond. Most metals have low solubility under moderately alkaline conditions.

At site L, the randomization test showed significant increases in sulfate, sodium and boron in distilled water extracts from soil directly beneath the disposal site as compared to soil samples taken at comparable depths outside the disposal site. These were the only significant contrasts noted at this site. High sodium and sulfate levels would be expected from a FGC sludge leachate because the interstitial water in the sludge commonly contains both of these constituents. Elevated levels of boron are usually associated with ash, not FGC sludge. Therefore it is likely that the boron is derived from ash co-disposed with the air cleaning sludge.

At site M, sulfate, boron, potassium, arsenic and selenium showed significantly increased levels in the distilled water extracts from under the disposal site as contrasted with the soil/sediment samples collected at similar elevations outside the disposal site. The latter four elements are associated with ash more often than with FGC sludges, therefore the increases detected in these elements can probably be related to the ash co-disposed with the FGC sludge. Significant decreases in nitrite, iron, magnesium and manganese were detected in the distilled water extracts from under the disposal site. The lower nitrite level was probably related to low levels of nitrite in the FGC/ash leachate and the lack of vegetation that releases nitrogen compounds in the disposal pit as compared to the surrounding area. The lower iron, · magnesium and manganese levels were probably related to the higher pH that would lower the solubility of these metals under the disposal site.

In general very little contrast in concentration of distilled-water extractable materials was detected under the disposal sites. The most con-

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TABLE 21. ANALYSES OF DISTILLED WATER EXTRACTS OF SOIL SAMPLES FROM EXPERIMENTAL BORINGS AT SITE K

Boring and sample	1C1	1C2	1C3	1C4	2C1	2C2	2C3	2C4
Elevation (m)	261.23	260.42	258.30	256.32	260.01	258.04	257.17	255.07
Depth below sludge/soil interface (m)	0.23	1.14	3.26	5.24	-1.83	0.14	1.01	3.11
Ht. above water table (m)	-0.23	-1.14	-3.26	-5.24	· 5.00	3.17	2.16 ·	.0.06
Conc. (mg/ℓ)								
SO_4	42	24	8	20	1400	20	20	16
SO_3	1	<1	<1	<1	16	<1	<1	<1
Cl	<5	<5	<5	<5	10	<5	<5	<5
NO_3-N	0.11	ND	0.29	0.18	0.03	0.11	0.06	0.10
NO_2-N	0.02	ND	0.02	<0.01	<0.01	<0.01	<0.01	<0.01
CN	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	14	12	2	2	1	8	<1	<1
Ca	7.50	5.00	11.00	7.00	2.70	650.00	9.50	14.00
Fe	26.750	6.500	0.332	0.099	2.350	0.340	1.560	0.117
K	ND	ND	ND	ND	ND	ND	ND	ND
Mg	12.00	3.20	3.00	2.90	2.80	2.50	2.20	3.40
Mn	0.158	0.043	0.004	<0.002	0.362	0.019	0.009	<0.002
Na	23.00	12.00	8.50	12.00	4.90	5.10	7.90	6.20
As	ND	ND	ND	ND	ND	ND	ND	ND
B	0.12	0.05	0.04	0.08	<0.02	<0.02	0.94	<0.02
Be	0.0070	0.0040	<0.0005	<0.0005	0.0005	0.1330	<0.0005	<0.0005
Cd	0.0450	0.0005	0.0003	0.0003	<0.0003	<0.0003	<0.0003	0.0003
Cr	0.141	0.032	0.036	<0.003	<0.003	0.078	<0.003	<0.003
Cu	0.041	0.003	<0.003	<0.003	0.008	<0.003	<0.003	<0.003
Hg	<0.0002	<0.0002	<0.0002	0.0010	<0.0002	<0.0002	0.0007	<0.0002
Ni	0.106	0.030	0.073	0.034	<0.005	1.560	0.038	0.075
Pb	0.010	0.002	<0.002	<0.002	0.002	<0.002	<0.002	<0.002
Se	ND	ND	ND	ND	ND	ND	ND	ND
Zn	0.231	0.080	0.043	0.079	0.108	0.246	0.159	0.079

ND = Not determined.

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TABLE 22. ANALYSES OF DISTILLED WATER EXTRACTS OF SOIL SAMPLES FROM CONTROL BORINGS AT SITE K

Boring and sample	3C5	5C1	5C2	6C1	6C2	6C3	7C1	7C2	7C3	7C4
Elevation (m)	251.45	257.30	256.38	262.42	261.51	257.89	262.42	261.20	259.40	257.84
Ht. above water table (m)	7.65	(dry)	(dry)	2.36	1.45	-2.17	(dry)	(dry)	(dry)	(dry)
Position in groundwater gradient	Downdip	Downdip	Downdip	Updip	Updip	Updip	Updip	Updip	Updip	Updip
Conc. (mg/l)										
SO_4	120	180	46	68	18	40	28	24	16	16
SO_3	1	1	<1	<1	<1	1	<1	1	<1	<1
Cl	<5	<5	<5	<5	<5	<5	<5	<5	<5	<5
NO_3-N	0.06	0.05	0.02	<0.01	<0.01	0.02	0.01	0.03	0.04	0.04
NO_2-N	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01	0.01	<0.01	0.01
CN	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	38	42	14	2	10	8	7	11	5	16
Ca	8.00	13.00	9.00	6.50	6.00	2.50	7.00	6.00	8.00	4.50
Fe	35.000	180.000	34.500	0.690	2.500	1.210	0.623	10.500	1.130	8.000
K	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Mg	8.00	34.00	16.00	3.50	3.10	2.80	4.20	3.20	3.20	4.40
Mn	0.402	0.577	0.227	<0.002	0.018	0.011	0.010	0.176	0.004	0.074
Na	4.30	25.00	25.00	36.00	54.00	14.00	16.00	9.00	15.00	11.00
As	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
B	0.06	0.24	0.10	0.04	0.05	0.06	0.10	<0.02	0.046	0.043
Be	0.0060	0.0180	0.0080	<0.0005	0.0010	<0.0005	<0.0005	0.0010	<0.0005	0.0020
Cd	<0.0003	0.0011	0.0011	0.0006	<0.0003	0.0007	0.0005	0.0006	0.0004	0.0003
Cr	0.081	0.401	0.138	<0.003	0.004	<0.003	<0.003	0.099	<0.003	0.003
Cu	0.025	0.110	0.030	<0.003	<0.003	<0.003	<0.003	0.055	<0.003	0.003
Hg	0.0004	0.0015	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	0.0012	0.0004	<0.0002
Ni	0.106	0.283	0.134	0.035	0.035	<0.005	<0.005	0.067	<0.005	0.006
Pb	0.012	0.090	0.026	0.002	<0.002	0.087	0.006	0.016	<0.002	0.012
Se	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Zn	0.317	0.764	0.326	0.066	0.033	0.050	0.058	0.135	0.102	0.130

ND = Not determined.

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TABLE 23. ANALYSES OF DISTILLED WATER EXTRACTS OF SOIL SAMPLES FROM EXPERIMENTAL BORINGS AT SITE L

Boring and sample	1C1	1C2	1C3	1C4	2C1	2C2	2C3	2C4
Elevation (m)	132.02	130.48	129.56	127.48	130.25	123.39	122.91	121.98
Depth below sludge/soil interface (m)	-1.45	0.09	1.01	3.10	-7.24	-0.38	0.10	1.03
Ht. above water table (m)	10.75	9.21	8.29	6.21	8.49	1.63	1.15	0.22
Conc. (mg/ℓ)								
SO_4	1721	226	46	<8	316	496	146	66
SO_3	<1	<1	<1	<1	<1	<1	<1	<1
Cl	<5	<5	15	<5	<5	<5	<5	<5
NO_3-N	<0.01	0.21	0.04	0.24	0.50	0.58	0.02	0.34
NO_2-N	0.01	<0.01	<0.01	<0.01	0.01	0.02	<0.01	<0.01
CN	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	6	4	4	<1	<1	<1	<1	16
Ca	478.30	24.70	17.60	12.00	149.30	254.30	13.30	20.30
Fe	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
K	ND	ND	ND	ND	ND	ND	ND	ND
Mg	0.940	11.600	7.300	2.400	1.000	0.400	0.250	1.600
Mn	<0.002	<0.002	4.720	<0.002	<0.002	<0.002	0.038	<0.002
Na	12.20	5.40	1.40	0.59	5.80	3.10	21.20	18.60
As	0.058	<0.005	<0.005	<0.005	0.025	0.025	<0.005	<0.005
B	8.10	2.19	0.15	<0.02	11.25	3.95	0.51	0.54
Be	<0.0005	<0.0005	<0.0005	<0.0005	0.0100	0.0100	<0.0005	<0.0005
Cd	<0.0003	0.0220	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003
Cr	0.003	<0.003	<0.003	<0.003	0.035	0.036	<0.003	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Hg	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002
Ni	<0.005	<0.005	0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Pb	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Se	0.058	<0.005	<0.005	<0.005	0.035	0.041	<0.005	<0.005
Zn	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014

ND = Not determined.

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TABLE 24. ANALYSES OF DISTILLED WATER EXTRACTS OF SOIL SAMPLES FROM CONTROL BORINGS AT SITE L

Boring and sample	3C5	4C1	4C2	4C3	4C4	4C5	5C1	5C2	5C3
Elevation (m)	121.30	132.97	128.70	124.60	123.55	121.42	130.60	129.69	127.59
Ht. above water table (m)	-0.41	10.93	6.66	2.56	1.51	-0.62	9.22	8.31	6.21
Position in groundwater gradient	Updip	Updip	Updip	Updip	Updip	Updip	Downdip	Downdip	Downdip
Conc. (mg/l)									
SO_4	8	14	<8	<8	<8	<8	21	56	14
SO_3	<1	<1	<1	<1	<1	<1	<1	<1	<1
Cl	35	<5	15	<5	100	25	<5	<5	<5
NO_3-N	0.50	0.11	0.01	0.09	0.02	0.19	0.67	1.14	1.60
NO_2-N	<0.01	<0.01	<0.01	<0.01	<0.01	0.02	<0.01	<0.01	<0.01
CN	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.01	<0.01
TOC	2	<1	17	18	<1	10	<1	<1	<1
Ca	38.30	4.30	15.10	8.50	69.10	33.50	16.10	27.50	13.80
Fe	<0.003	<0.003	<0.003	0.290	<0.003	<0.003	0.100	<0.003	<0.003
K	ND	ND	ND	ND	ND	ND	ND	ND	ND
Mg	6.60	3.10	6.00	2.20	7.90	6.20	6.50	11.50	5.00
Mn	0.020	0.003	<0.002	0.016	0.044	0.120	<0.002	<0.002	<0.002
Na	1.00	0.97	0.51	0.38	0.38	1.10	1.40	4.10	1.20
As	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
B	0.04	<0.02	<0.02	<0.02	<0.02	<0.02	0.10	0.30	0.17
Be	<0.0005	<0.0005	0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cd	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003
Cr	0.011	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	0.004	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Hg	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002
Ni	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Pb	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Se	0.008	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Zn	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014

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TABLE 24 (CONTINUED)

Boring and sample	5C4	5C5	5C6	6C5	7C1	7C2	7C3	7C4	7C5
Elevation (m)	125.45	123.44	121.30	121.64	133.22	128.89	124.98	123.94	121.90
Ht. above water table (m)	4.07	2.06	-0.08	0.33	11.37	7.04	3.13	2.09	0.05
Position in groundwater gradient	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip
Conc. (mg/l)									
SO_4	15	<8	<8	11	<8	<8	<8	<8	<8
SO_3	<1	<1	<1	<1	<1	<1	<1	<1	<1
Cl	<5	15	10	<5	25	<5	<5	<5	<5
NO_3-N	2.90	0.16	0.13	0.19	0.07	<0.01	0.01	0.04	0.12
NO_2-N	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
CN	<0.01	<0.01	<0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	8	6	6	<1	2	5	<1	<1	3
Ca	18.50	19.10	14.50	12.10	11.50	10.30	8.80	9.50	11.80
Fe	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
K	ND	ND	ND	ND	ND	ND	ND	ND	ND
Mg	5.60	3.90	3.60	2.60	7.90	3.60	2.10	2.40	2.00
Mn	<0.002	0.003	<0.002	<0.002	0.107	<0.002	<0.002	0.002	0.002
Na	1.20	0.34	0.42	0.47	0.55	0.38	0.30	0.51	1.40
As	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
B	0.18	<0.02	<0.02	0.10	<0.02	<0.02	<0.02	<0.02	0.30
Be	<0.0005	<0.0005	0.0100	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cd	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	0.0007	<0.0003	<0.0003	<0.0003
Cr	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Hg	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002
Ni	<0.005	<0.005	<0.005	<0.005	0.020	0.009	<0.005	<0.005	<0.005
Pb	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Se	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Zn	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014

ND = Not determined.

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TABLE 25. ANALYSES OF DISTILLED WATER EXTRACTS OF SOIL SAMPLES FROM EXPERIMENTAL BORINGS AT SITE M

Boring and sample	1C1	1C2	1C3	1C4	4C1	4C2	4C3	4C4	4C5
Elevation (m)	222.77	220.81	218.96	217.73	221.98	221.07	218.93	217.41	216.65
Depth below sludge/soil interface (m)	-1.89	0.07	1.92	3.15	0.09	1.00	3.14	4.66	5.42
Ht. above water table (m)	5.96	4.00	2.15	0.92	5.03	4.12	1.98	0.46	-0.30
Conc. (mg/ℓ)									
SO_4	76	<8	28	39	150	11	<8	41	39
SO_3	190	<1	<1	<1	<1	<1	<1	<1	<1
Cl	10	<5	<5	<5	10	<5	<5	<5	<5
NO_3-N	<0.01	<0.01	0.12	0.09	0.10	0.02	0.04	0.02	0.01
NO_2-N	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
CN	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	5	3	2	2	11	18	2	<1	9
Ca	56.20	5.10	10.10	10.50	73.20	17.80	6.20	14.10	13.70
Fe	<0.003	<0.003	0.320	<0.003	<0.003	0.470	0.320	0.530	0.380
K	35.00	14.80	3.00	1.30	18.20	9.70	1.80	3.70	3.10
Mg	<0.03	<0.03	1.10	2.00	0.22	0.76	0.52	1.40	0.67
Mn	<0.002	0.002	0.007	0.002	0.003	0.030	0.010	0.011	0.006
Na	20.30	8.20	3.90	4.30	24.50	16.00	2.60	6.10	4.80
As	0.011	0.031	0.007	<0.005	0.018	<0.005	<0.005	<0.005	<0.005
B	0.04	0.13	0.05	0.04	2.15	1.01	0.11	0.97	0.36
Be	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cd	0.0019	<0.0003	<0.0003	<0.0003	0.0005	0.0005	<0.0003	<0.0003	<0.0003
Cr	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Hg	<0.0002	0.0005	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002
Ni	0.024	<0.005	<0.005	<0.005	<0.005	0.009	<0.005	<0.005	<0.005
Pb	<0.002	<0.002	0.009	0.005	<0.002	0.004	0.007	<0.002	0.002
Se	0.014	0.009	0.005	<0.005	0.151	0.005	<0.005	<0.005	<0.005
Zn	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014

ND = Not determined.

TABLE 26. ANALYSES OF DISTILLED WATER EXTRACTS OF SOIL SAMPLES FROM CONTROL BORINGS AT SITE M

Boring and sample	2C1	2C2	2C3	2C4	3C1	3C2	3C3
Elevation (m)	221.78	220.87	218.72	217.97	221.70	220.79	218.64
Ht. above water table (m)	3.96	3.05	0.90	0.15	4.60	3.69	1.54
Position in groundwater gradient	Updip	Updip	Updip	Updip	Updip	Updip	Updip
Conc. (mg/ℓ)							
SO_4	12	<8	16	<8	16	21	8
SO_3	<1	<1	<1	<1	<1	<1	<1
Cl	<5	<5	<5	<5	<5	<5	<5
NO_3-N	0.92	0.16	0.30	0.24	ND	0.01	0.07
NO_2-N	0.02	<0.01	<0.01	<0.01	0.02	0.04	<0.01
CN	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	7	3	5	2	6	4	4
Ca	12.80	11.60	19.00	17.10	30.10	17.10	19.90
Fe	0.620	0.410	1.100	<0.003	0.620	3.000	<0.003
K	0.77	0.53	1.80	1.70	1.60	0.95	5.00
Mg	3.00	5.10	4.30	3.70	4.70	3.60	7.00
Mn	0.007	0.053	0.018	0.002	0.014	0.039	0.012
Na	0.83	0.96	1.20	1.20	0.83	0.74	4.50
As	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
B	0.03	<0.02	<0.02	<0.02	0.04	0.03	0.05
Be	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cd	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003	<0.0003
Cr	<0.003	<0.003	<0.003	<0.003	0.003	<0.003	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Hg	<0.0002	0.0005	<0.0002	0.0003	0.0004	<0.0002	<0.0002
Ni	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Pb	0.003	0.006	0.002	<0.002	0.002	0.005	<0.002
Se	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Zn	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014	<0.014

(continued)

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TABLE 26 (CONTINUED)

Boring and sample	3C4	5C4	6C4	7C1	7C2	7C3	7C4
Elevation (m)	217.89	217.17	216.14	221.09	220.18	218.05	216.52
Ht. above water table (m)	0.79	0.03	-0.03	4.84	3.93	1.80	0.27
Position in groundwater gradient	Updip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip
Conc. (mg/ℓ)							
SO_4	8	<8	<8	45	37	45	35
SO_3	<1	<1	<1	<1	<1	<1	<1
Cl	<5	<5	<5	<5	<5	<5	<5
NO_3-N	0.12	0.04	0.01	2.72	1.02	0.62	0.01
NO_2-N	<0.01	<0.01	<0.01	<0.01	<0.01	0.13	<0.01
CN	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
TOC	3	<1	<1	9	4	8	<1
Ca	12.70	7.20	6.20	31.70	33.50	38.50	20.80
Fe	0.290	0.650	0.710	0.440	0.230	0.120	<0.003
K	1.80	0.84	0.95	5.10	3.00	6.00	3.50
Mg	4.00	2.40	2.80	3.80	6.70	9.70	7.10
Mn	0.003	0.007	0.015	0.006	0.006	0.007	0.002
Na	1.10	0.61	1.00	1.60	2.10	8.30	6.70
As	<0.005	<0.005	<0.005	0.010	<0.005	<0.005	<0.005
B	0.02	<0.02	<0.02	0.07	0.05	0.03	0.03
Be	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cd	<0.0003	<0.0003	<0.0003	0.0007	<0.0003	<0.0003	<0.0003
Cr	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003
Cu	<0.003	<0.003	<0.003	<0.003	<0.003	0.434	<0.003
Hg	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002
Ni	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Pb	0.002	0.002	0.002	<0.002	<0.002	0.010	<0.002
Se	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Zn	<0.014	<0.014	<0.014	<0.014	<0.014	0.247	<0.014

ND = Not determined.

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TABLE 27. RESULTS OF RANDOMIZATION TEST ON DISTILLED WATER EXTRACTS OF
SOIL SAMPLES DIRECTLY UNDER THE FGC DISPOSAL SITES AND AT
COMPARABLE DEPTHS OUTSIDE THE SITES

Parameters	Site K	Site L	Site M
SO_4	NS	S(increase)	NS
SO_3	NS	BDL	BDL
Cl	BDL	BDL	BDL
NO_3-N	S(increase)*	NS	NS
NO_2-N	NS	BDL	S(decrease)
CN	BDL	BDL	BDL
TOC	NS	NS	NS
Ca	NS	NS	NS
Fe	NS	BDL	S(decrease)
K	ND	ND	S(increase)
Mg	NS	NS	S(decrease)
Mn	NS	NS	S(decrease)
Na	NS	S(increase)	S(increase)
As	ND	ND	S(increase)
B	NS	S(increase)	S(increase)
Be	NS	BDL	BDL
Cd	NS	BDL	NS
Cr	NS	BDL	BDL
Cu	NS	BDL	BDL
Hg	S(decrease)	BDL	NS
Ni	NS	BDL	BDL
Pb	NS	BDL	NS
Se	ND	BDL	S(increase)
Zn	NS	BDL	BDL

NS = Not significant at 80% confidence level.

S = Significant at 80% confidence level.

BDL = Below detection limits.

ND = Not determined.

* Refers to increase or decrease of constituent under disposal site
relative to outside.

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sistent changes found were increased levels of sodium and boron. Elevated concentrations of sulfate were detected at site L. The immobilization of some metals, probably due to high pH levels, was detected at sites K and M.

Vertical Variations of Concentrations in
Distilled Water Extracts of Soil Samples

For those elements that did show a significant difference between control (outside disposal site) samples and experimental (inside disposal site) samples, a test was made for a significant relationship between the available concentration of a particular constituent and sample elevation. As suggested by the model (Figure 1), those materials present in the sludge liquor should show a positive correlation with elevation in experimental borings (these below the disposal area). A significant negative correlation would be predicted by the model for those soil constituents that are being dissolved by the sludge liquor and moved down out of the soil and into the groundwater. In control borings the distribution of available soil constituents depends on weathering processes and the concentration and solubility of the particular material, and could therefore have a significant positive or negative correlation with elevation or no significant correlation at all.

A non-parametric test of association, the Spearman rank correlation coefficient, was used to assess the strength of association between the concentration of a particular soil constituent and sample elevation. This technique is suited especially for use with small sample numbers where the statistical distribution is not known. In several cases, the small number of samples having detectable quantities of a particular constituent made it impossible to judge the significance of the correlation coefficients obtained. The results of the statistical tests are given in Tables 28-30. Plots of concentration versus sample elevation for all constituents in experimental borings that showed statistically significant relationships with depth are presented in Figures 7-11. Plots of significant relationships in control borings are included for contrast. At site K, no soil/sediment constituents, as tested above, showed any significant relationship with sample elevation. This was not unexpected, as only nitrate and mercury showed any contrast under and outside the disposal pond. The pond itself is underlain by impervious Pennsylvanian shales and limestones which decreases the likelihood of vertical migration of sludge constituents.

Site L (especially boring 1) comes closest to giving results predicted by the model for pollutant migration. The pattern of leachable constituents observed under the disposal pit (a significant positive correlation with elevation) indicates that the sludge/ash in the pit is contributing boron, sodium, and sulfate to the soil below the pit in a water-extractable form. The sands and gravels below the pit in this hole have low cation exchange capacities and most of the material in these samples is probably reflecting the concentration of these constituents in the infiltrating water.

At site M, many soil constituents showed significant contrasts beneath and outside the disposal pond; but, only potassium and selenium (in boring 1) showed a significant correlation of concentration in distilled water extracts versus sample elevation under the site. The most striking aspect of this data

TABLE 28. CORRELATION OF CHEMICAL ANALYSES OF DISTILLED WATER EXTRACTS OF SOILS WITH SAMPLE ELEVATION AT SITE K

Boring	1	2	6	7
NO_3-N	NS(0.80)*	NS(-0.40)	NS(-0.50)*	NS(-0.80)
Hg	NS(-0.20)	NS(0.40)	**	SP(1.00)

SP = Significant positive correlation at 95% level.

SN = Significant negative correlation at 95% level.

NS = No significant correlation.

* = Significance level reduced to 83% because of small sample size for this constituent in this boring.

** = Too few samples above detection limits.

Number in parentheses is the calculated value of r_s, the Spearman rank correlation coefficient.

TABLE 29. CORRELATION OF CHEMICAL ANALYSES OF DISTILLED WATER EXTRACTS OF SOILS WITH SAMPLE ELEVATION AT SITE L

Boring	1	2	4	5	7
SO_4	SP(1.00)	NS(0.80)	**	SP(0.88)	**
Na	SP(1.00)	NS(-0.60)	NS(0.00)	SP(0.88)	NS(-0.30)
B	SP(1.00)	NS(0.80)	**	NS(0.60)	**

SP - Significant positive correlation at 95% level.

SN - Significant negative correlation at 95% level.

NS - No significant correlation.

** - Too few samples above detection limits.

Number in parentheses is the calculated value of r_s, Spearman rank correlation coefficient.

TABLE 30. CORRELATION OF CHEMICAL ANALYSES OF DISTILLED WATER EXTRACTS OF SOILS WITH SAMPLE ELEVATION AT SITE M

Boring	1	4	2	3	7
NO$_2$-N	**	**	**	NS(0.80)	**
Fe	**	NS(-0.50)	NS(0.40)	NS(0.60)	SP(1.00)
K	SP(1.00)	NS(0.70)	NS(-0.60)	NS(-0.60)	NS(0.00)
Mg	NS(-0.80)	NS(-0.50)	NS(-0.20)	NS(0.00)	NS(-0.80)
Mn	NS(-0.40)	NS(-0.10)	NS(0.40)	NS(0.80)	NS(0.40)
Na	NS(0.80)	NS(0.70)	NS(-0.80)	NS(-0.60)	NS(-0.80)
As	NS(0.80)	**	**	**	**
B	NS(0.40)	NS(0.70)	**	NS(0.40)	SP(1.00)
Se	SP(1.00)	**	**	**	**

SP = Significant positive correlation at 95% level.

SN = Significant negative correlation at 95% level.

NS = No significant correlation.

** = Too few samples above detection limits.

Number in parentheses is the calculated value of r$_s$, the Spearman rank correlation coefficient.
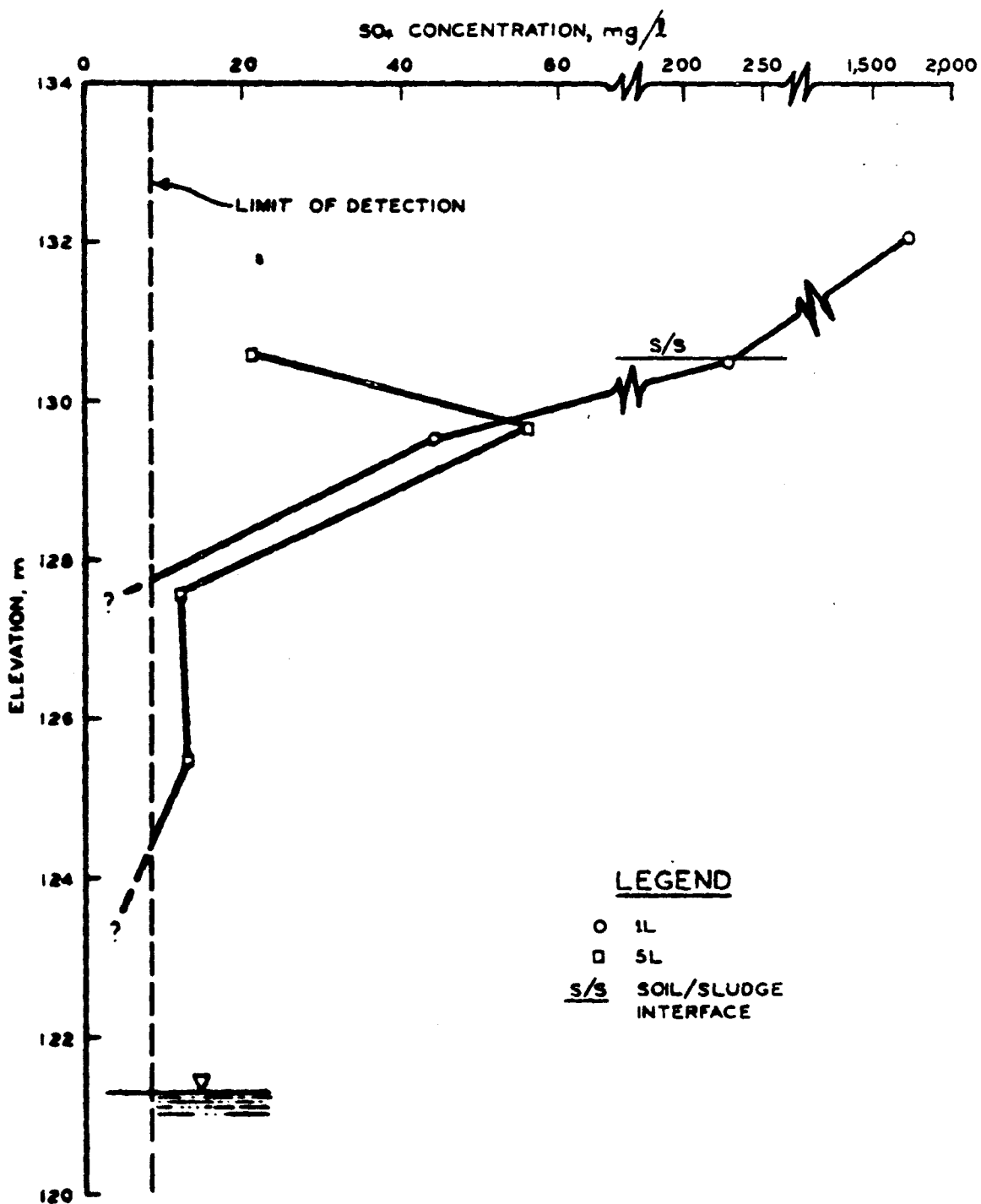
Figure 7. Variation of sulfate concentration in distilled water extracts of soil/sediment samples with elevation in borings 1 and 5 at site L.

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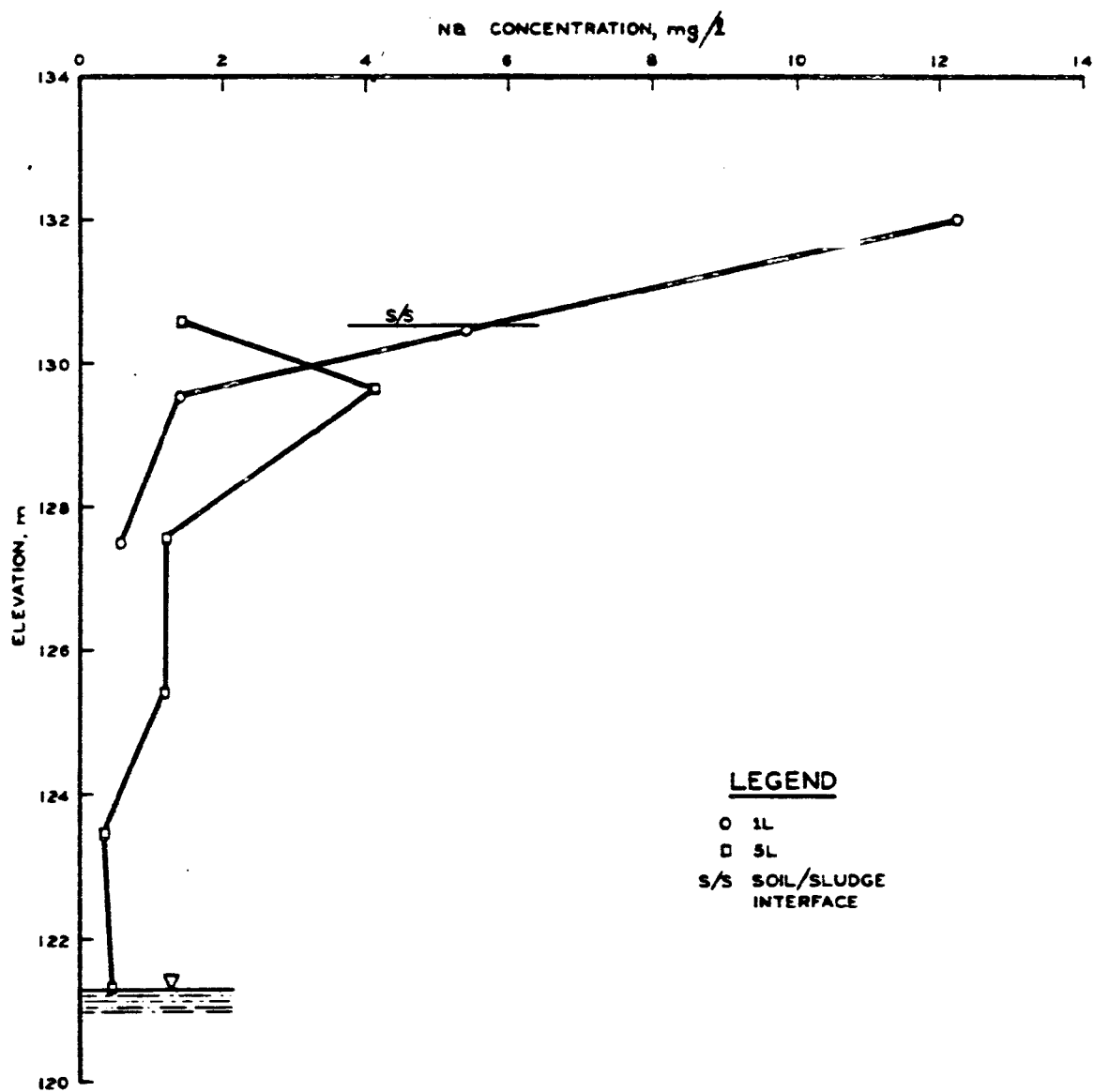
Figure 8. Variation of sodium concentration in distilled water extracts of soil/sediment samples with elevation in borings 1 and 5 at site L.
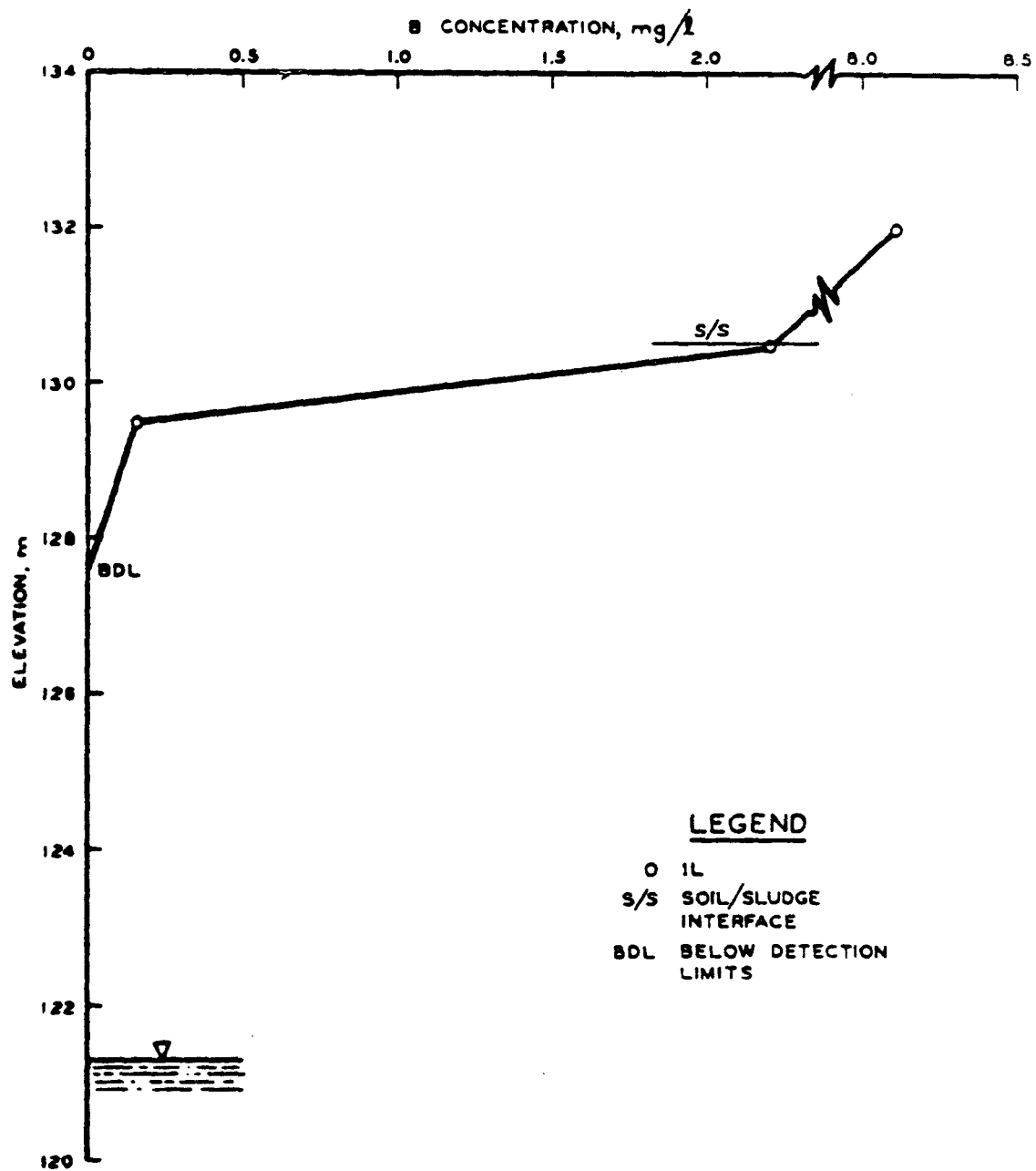
Figure 9. Variation of boron concentration in distilled water extracts of soil/sediment samples with elevation in boring 1 at site L.
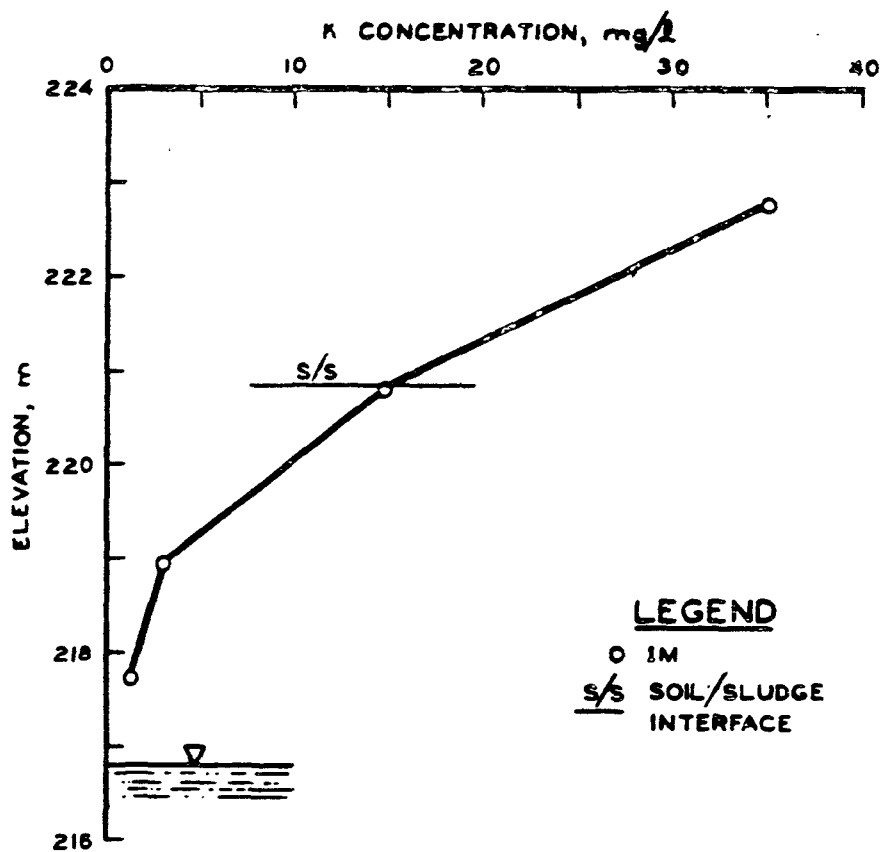
Figure 10. Variation of potassium concentration in distilled water extracts of soil/sediment samples with elevation in boring 1 at site M.
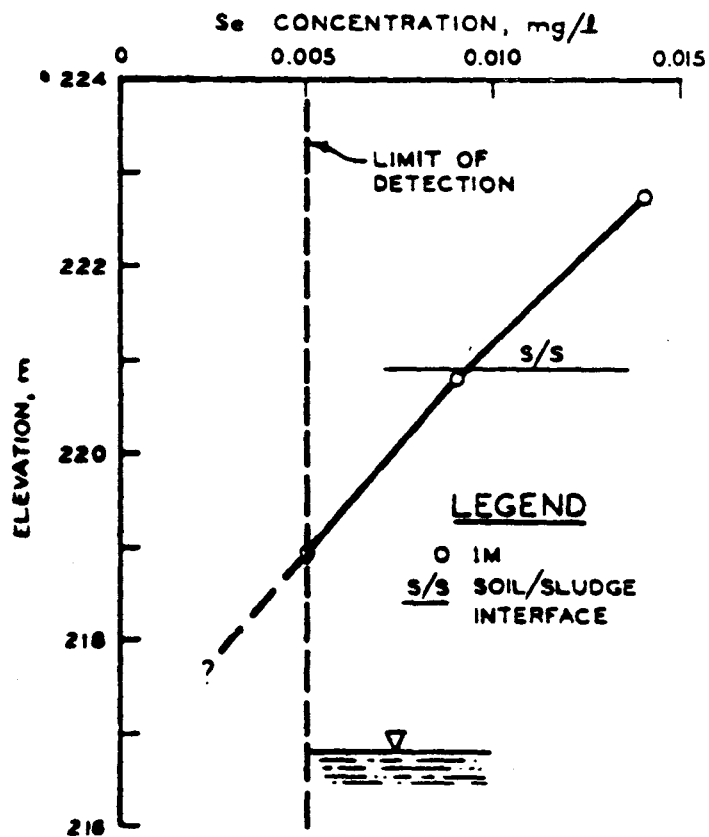
Figure 11. Variation of selenium concentration in distilled water extracts of soil/sediment samples with elevation in boring 1 at site M.

set is that most of the soil constituents were so uniformly distributed through the soil/sediment column. Possible explanations of this uniformity are that the interstitial water is the major source of the materials measured and that this water moves unchanged through the soil/sediment column, or that the removal capacity of the soil has been exhausted.

As expected, the soils beneath the disposal sites did not hold any appreciable quantities of water-extractable materials that could be related to the pollutants from the FGC sludge/ash. The high levels of contamination observed in the groundwater indicate that pollutants have passed through the soil, but the low levels of contaminants found in the distilled water extracts indicate the polluting material does not remain in the soil in a water soluble condition.

Horizontal Variation in Distilled Water Extracts of Soil/Sediment Samples Below the Water Table

Analyses of distilled water extracts of soil/sediment samples collected below the water table were examined in order to determine if sludge-derived materials were accumulating below this horizon in a water-extractable form. Plots of cross-sections through the site versus concentration are shown in Figures 12-15. The model of groundwater movement assumes all significant lateral migration of pollutants occurs below the water table. Two factors should effect the concentration of contaminants in distilled water extracts; the concentration of sludge-derived materials in infiltrating water and the character of the soil/sediment.

At site K, the highest values for all constituents measured were found under the disposal pond or downdip from the pond as predicted from the model situation. Sulfite, nitrate, nitrite, cyanide, calcium, magnesium, sodium, boron, beryllium, cadmium, chromium, copper, and lead were found in their maximum amounts in water extracts from directly under the disposal pond. Sulfate, total organic carbon, iron, manganese, mercury, nickel and zinc were found in their maximum concentrations down gradient from the disposal site. With the exception of boring 1, the level of contamination in the groundwater is not reflected by the level of constituents in the distilled water extract from soils. The low correlation with groundwater chemistry may reflect the strong influence of the original composition of the material that was extracted.

At site L, maximum concentrations in distilled water extracts were observed in borings under the disposal pit for sulfate, total organic carbon, sodium and boron. Maximum concentrations for chloride, nitrite, calcium, magnesium, manganese, chromium and selenium were found in upgradient borings. Maximum concentrations for cyanide and beryllium were found in down gradient borings. These results agree with the groundwater analyses in that elevated sodium and boron levels were noted under the disposal pit. For other constituents there seems to be no consistent pattern and all were found in low concentrations.

At site M, maxima for sulfate, total organic carbon, boron and lead were found under the disposal area. Maxima for nitrate and mercury were found in upgradient borings. Maximum levels for calcium, iron, potassium, magnesium, manganese and sodium occurred in the down gradient borings. At both sites M and L where the substrate is sand and gravel many consistuents were below

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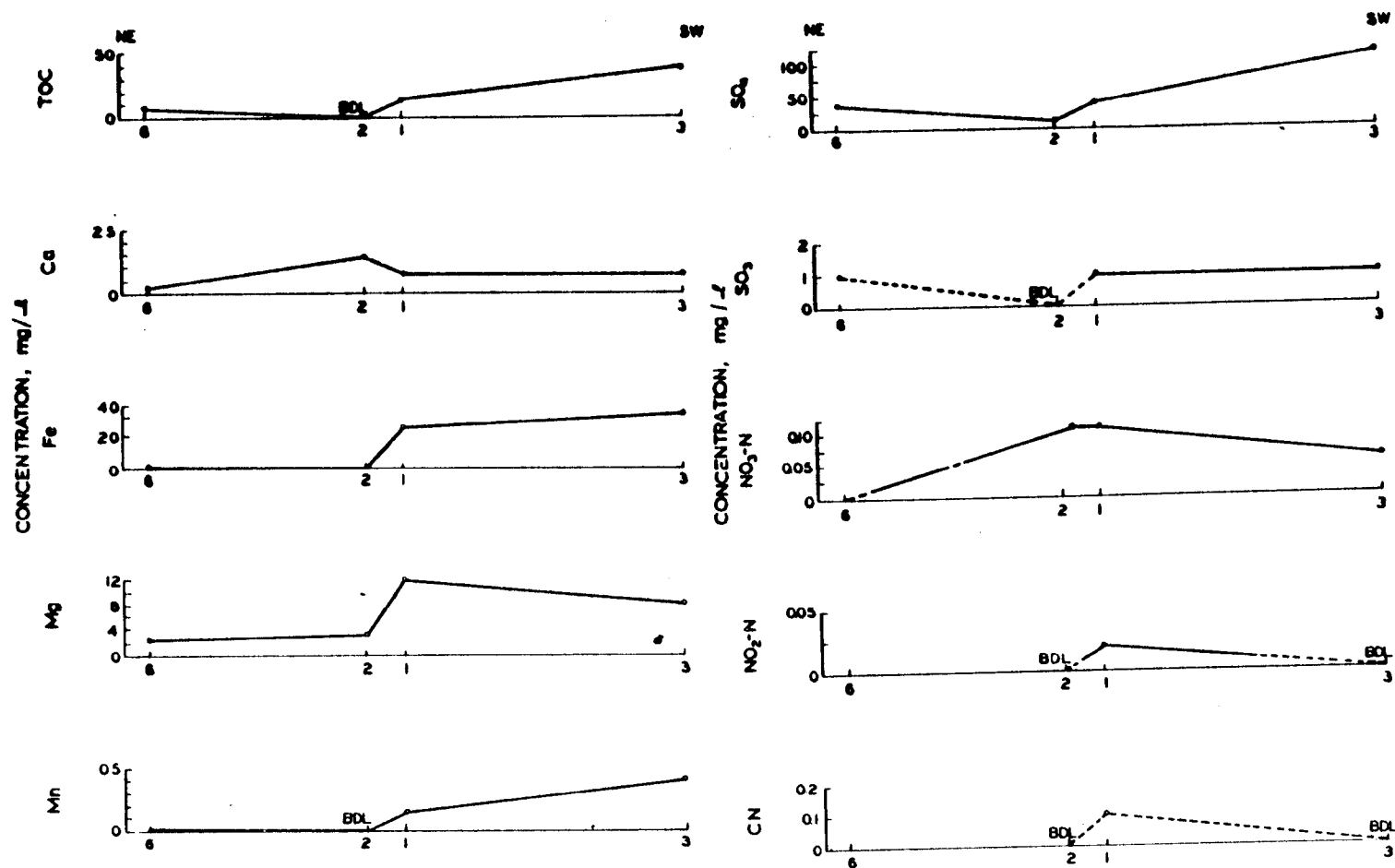
Figure 12. Horizontal variation in chemical composition of distilled water extracts at site K. BDL indicates below detection limits.
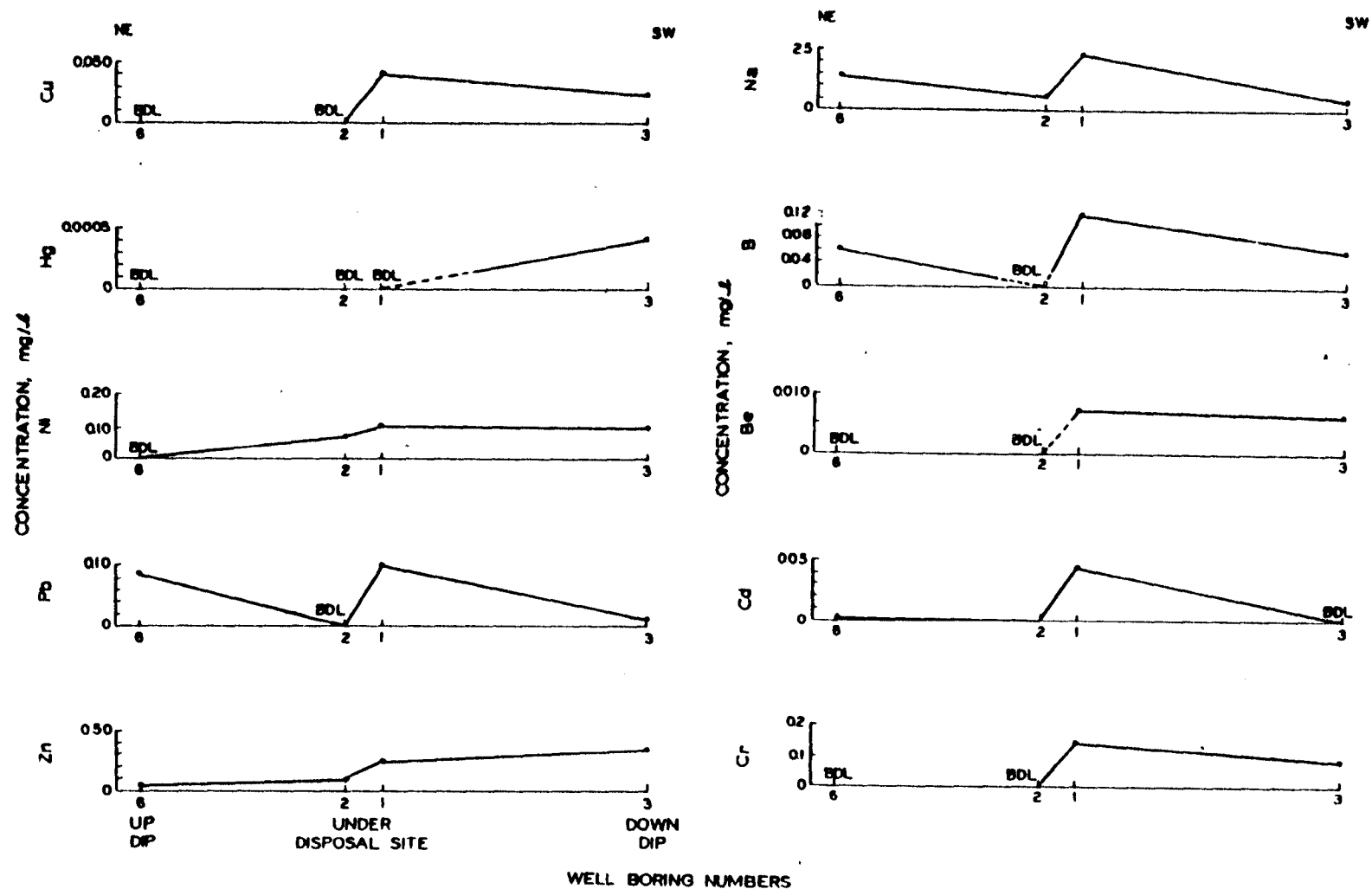
Figure 13. Horizontal variation in chemical composition of distilled water extracts at site K, continued. BDL indicates below detection limits.
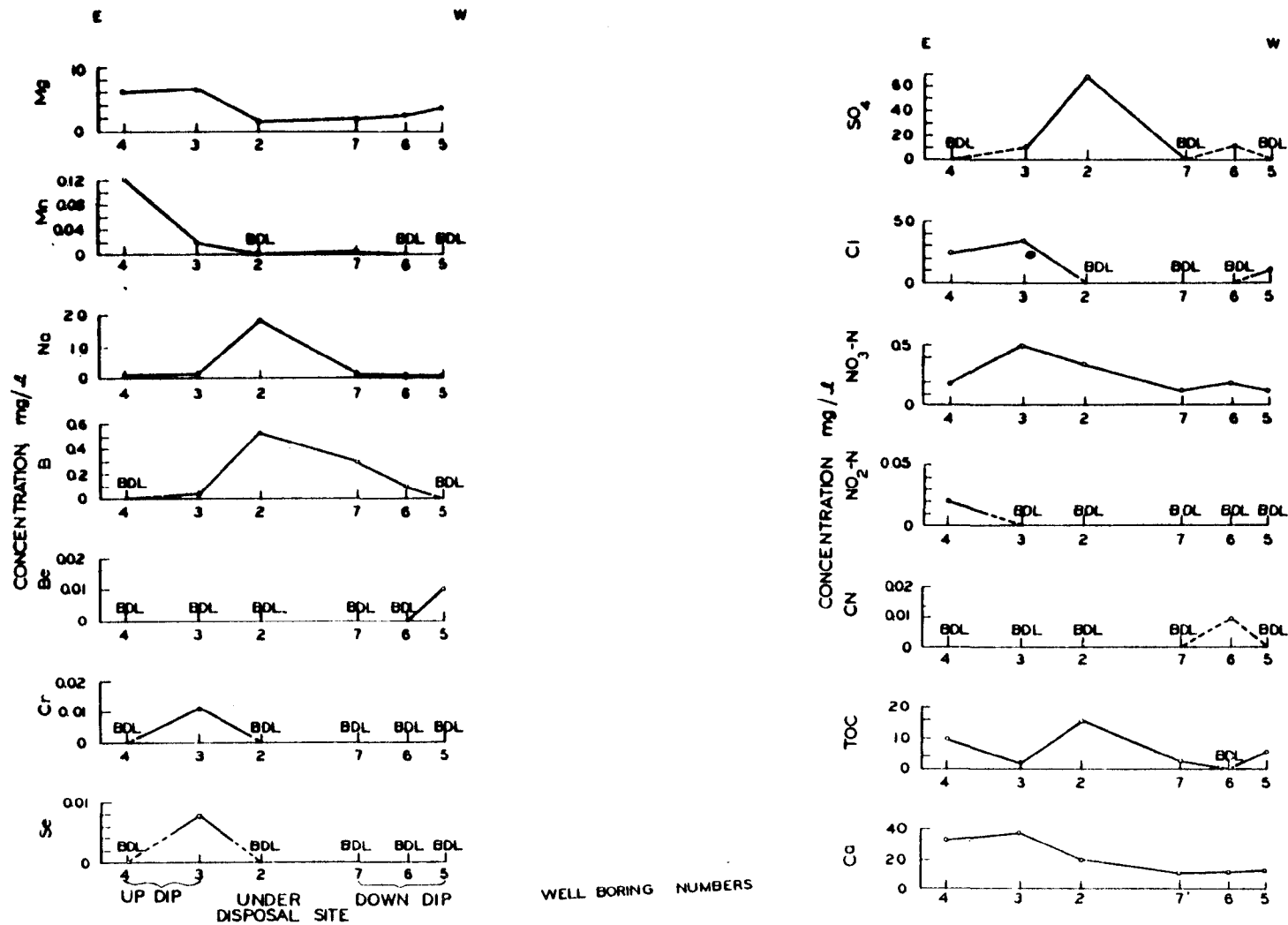
Figure 14. Horizontal variation in chemical composition of distilled water extracts at site L. BDL indicates below detection limits.

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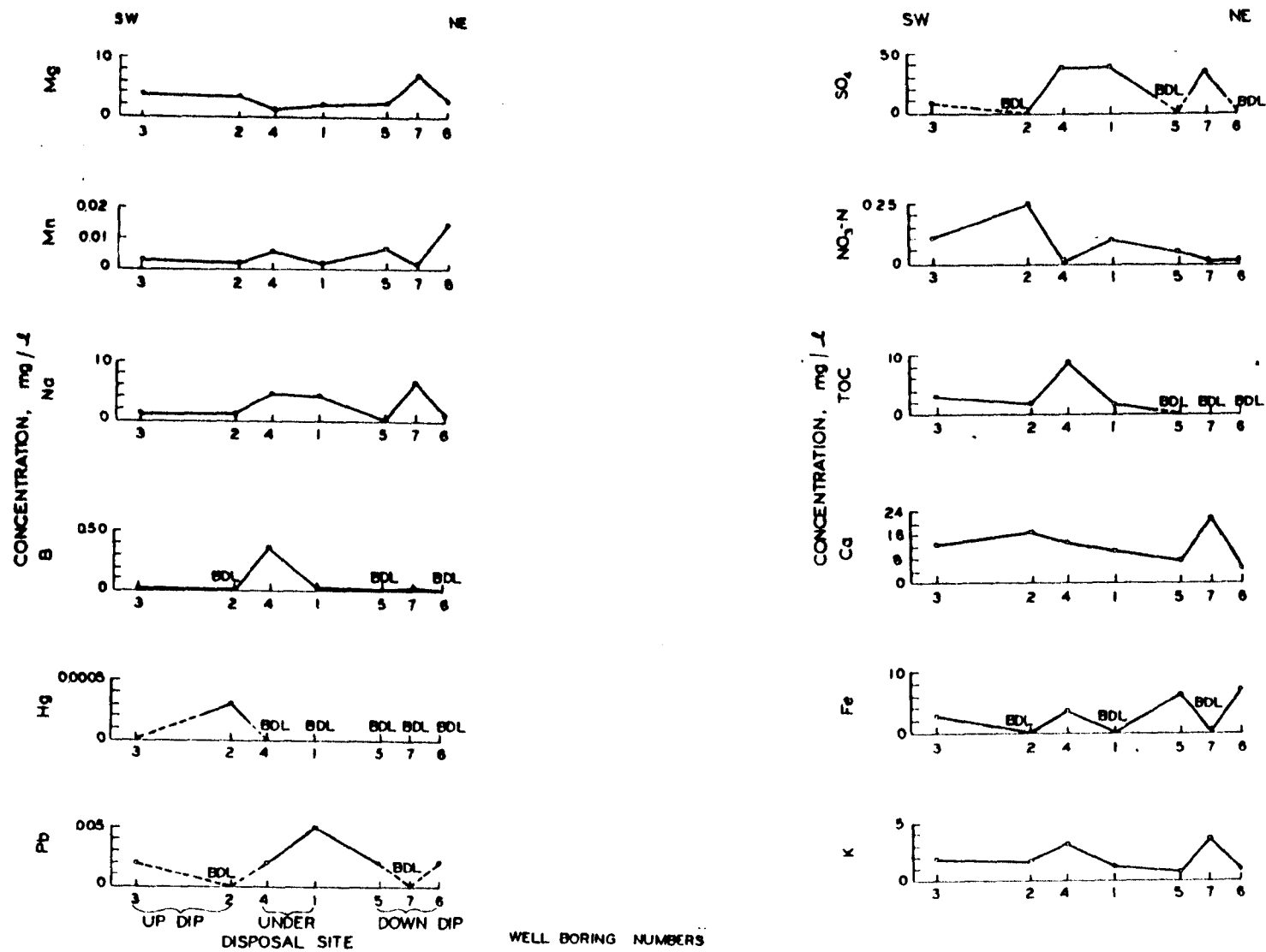
Figure 15. Horizontal variation in chemical composition of distilled water extracts at site M.
BDL indicates below detection limits.

detection limits in all borings. In this situation where the underlying material is relatively uniform, the highest levels of sulfate, total organic carbon and boron are consistently associated with borings under the disposal area.

The use of analytical data from distilled water extracts to indicate the presence of loosely bound pollutant materials is limited because of the large differences produced by the changing nature of the geologic materials underneath the disposal areas, the background levels of exchangeable constituents that are likely to be present under an industrial area, and the limited capacity of many materials (expecially sand and gravel) to exchange or absorb incoming materials. Several major constituents (sulfate, sodium and boron) associated with FGC pond liquors did show a consistent distribution with maxima occurring under or down the groundwater gradient from the disposal areas.

CHEMICAL ANALYSES OF NITRIC ACID DIGESTS

The goal of the nitric acid digestion procedure was to determine the total amount of material that could be removed from the soil by rigorous treatment with a strong, oxidizing acid. This digest brings into solution all materials that are not tightly bound in a silicate lattice. Contaminants leached from the FGC disposal area and deposited in the soil should be released in the nitric acid digest. The evidence that attenuation is occurring would be the higher concentration of the attenuated materials under the disposal site as compared to similar samples outside the site and decreasing concentration in nitric acid digests of samples taken at decreasing elevations (increasing depths) below the disposal site. Evidence that mobilization of material from soil under the site is occurring would be the lower concentrations of material under the disposal site as compared to similar samples outside the site. In this case, concentrations of mobilized constituents would increase with decreasing elevations (increasing depth) below the disposal site. The absence of any significant difference between the concentrations of constituents in the nitric acid digest from the soils would indicate either no leachate is passing through the soil, or leachate passing through the soil is not interacting with the soil. Analyses of groundwater obtained from borings under and down the groundwater gradient from the disposal sites can indicate if sludge-derived constituents get through the soil into the groundwater.

Published analyses of pond liquor or elutriates (Table 16) indicate any leachate escaping from the disposal areas is saturated with calcium and sulfate, and is high in sodium and chloride. Common pH's are between 8 and 10. Leachate with a composition similar to pond liquor would be expected to pass through the soil with little interaction except possibly the displacement of exchangeable cations with calcium and loss of boron and potassium into clays in the soil. Any calcium, boron and potassium fixed in the soil should be. brought into solution by the nitric acid digestion procedure.

Comparison of Nitric Acid Digests Beneath and Outside the Disposal Area

The chemical analyses for all the nitric acid digests are given in Tables 31-36. The results of the randomization test on nitric acid digests of soil

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TABLE 31. ANALYSIS OF NITRIC ACID DIGESTS OF SOIL SAMPLES FROM EXPERIMENTAL BORINGS AT SITE K

Boring and sample	1C1	1C2	1C3	1C4	2C1	2C2	2C3	2C4
Elevation (m)	261.33	260.42	258.30	236.32	260.01	258.04	257.17	255.07
Depth below sludge/soil interface (m)	0.23	1.14	3.26	5.24	-1.83	0.14	1.01	3.11
Ht. above water table (m)	-0.23	-1.14	-3.26	-5.24	5.00	3.17	2.16	0.06
Conc. (mg/kg dry wt.)								
Ca	3416.96	4607.68	12240.04	31579.70	299450.35	2138.04	3458.55	14404.11
Fe	41649.47	83009.96	57120.17	25078.00	14518.80	30543.44	45595.02	51582.27
K	ND	ND	ND	ND	ND	ND	ND	ND
Mg	4334.94	11474.38	9384.03	7244.75	4083.41	2660.24	4781.92	8515.94
Mn	60.52	89.64	63.97	27.86	32.67	40.69	57.83	44.96
Na	86.70	699.22	261.12	380.81	204.17	128.09	211.29	107.06
As	ND	ND	ND	ND	ND	ND	ND	ND
B	ND	ND	ND	ND	ND	ND	ND	ND
Be	3.13	3.80	2.57	3.66	4.72	1.99	3.14	2.71
Cd	5.10	0.70	0.03	0.03	28.92	0.04	0.51	0.08
Cr	24.14	37.29	23.66	23.68	19.74	19.41	27.36	23.16
Cu	13.16	20.80	57.28	7.52	19.51	9.85	15.35	30.37
Hg	ND	ND	ND	ND	ND	ND	ND	ND
Ni	41.14	78.17	56.06	60.65	79.17	23.65	44.04	64.72
Pb	15.64	10.22	2.86	14.86	86.21	13.79	17.79	5.06
Se	ND	ND	ND	ND	ND	ND	ND	ND
Zn	39.61	85.88	57.28	24.24	969.81	34.88	56.05	57.03

ND = Not determined.

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TABLE 32. ANALYSES OF NITRIC ACID DIGESTS OF SOIL SAMPLES FROM CONTROL BORINGS AT SITE K

Boring and sample	3C5	5C1	6C1	6C2	6C3	7C1	7C2	7C3	7C4
Elevation (m)	251.45	257.30	262.42	261.51	257.89	262.42	261.20	259.40	257.84
Ht. above water table (m)	-7.65	(dry)	2.36	1.45	-2.17	(dry)	(dry)	(dry)	(dry)
Position in groundwater gradient	Downdip	Downdip	Updip	Updip	Updip	Updip	Updip	Updip	Updip
Conc. (mg/l)									
Ca	2568.73	2386.07	2527.65	14934.53	30.76	2918.62	1804.83	62724.06	56444.98
Fe	28966.50	56749.82	53564.75	42750.09	487.26	44530.42	56499.06	38116.93	22003.98
K	ND	ND	ND	ND	ND	ND	ND	ND	ND
Mg	2732.69	3408.67	5593.11	7560.61	80.90	4560.34	3040.75	13220.30	11958.68
Mn	64.93	59.70	93.58	29.12	0.53	90.99	100.05	40.14	47.26
Na	98.38	690.95	516.29	774.73	2.73	386.29	196.18	299.15	344.41
As	ND	ND	ND	ND	ND	ND	ND	ND	ND
B	ND	ND	ND	ND	ND	ND	ND	ND	ND
Be	2.14	2.51	2.20	3.96	0.02	2.37	1.78	3.66	3.46
Cd	0.13	0.10	0.32	0.04	ND	0.50	0.54	0.08	0.07
Cr	17.71	24.14	23.66	26.14	0.21	22.96	18.44	21.52	20.57
Cu	12.24	15.02	23.45	19.04	0.08	13.31	14.71	8.99	5.93
Hg	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ni	28.86	30.95	49.91	64.87	0.42	44.75	38.75	58.86	64.58
Pb	14.21	22.11	12.91	4.48	0.03	16.10	15.69	4.05	7.46
Se	ND	ND	ND	ND	ND	ND	ND	ND	ND
Zn	36.84	49.75	13.66	42.47	0.50	60.20	63.76	50.37	43.91

ND = Not determined.

TABLE 33. ANALYSES OF NITRIC ACID DIGESTS OF SOIL SAMPLES FROM EXPERIMENTAL BORINGS AT SITE L

Boring and sample	1C1	1C2	1C3	1C4	2C1	2C2	2C3	2C4
Elevation (m)	132.02	130.48	129.56	127.48	130.25	123.39	122.91	121.98
Depth below sludge/soil interface (m)	-1.45	0.09	1.01	3.10	-7.24	-0.38	0.10	1.03
Ht. above water table (m)	10.75	9.21	8.29	6.21	8.49	1.63	1.15	0.22
Conc. (mg/kg dry wt.)								
Ca	6949.27	BDL	BDL	97063.43	10023.27	14768.90	38920.79	60524.81
Fe	14792.03	15800.24	27879.87	5545.00	16277.46	19654.31	10056.00	8166.02
K	2134.42	547.46	580.23	186.22	1811.31	1572.66	351.60	253.00
Mg	9331.88	12811.01	21604.33	220551.42	9752.59	8048.80	85932.42	143099.23
Mn	75.79	374.03	659.38	377.58	90.16	88.01	572.14	485.39
Na	368.31	50.39	19.55	85.31	332.75	320.58	129.90	103.08
As	CI	CI	CI	CI	CI	CI	CI	CI
B	384.20	11.44	8.74	4.59	431.41	416.30	8.74	6.94
Be	3.20	1.31	0.60	0.28	2.34	2.53	0.22	0.23
Cd	9.13	1.15	2.13	BDL	9.77	3.51	BDL	BDL
Cr	23.85	11.39	12.49	5.48	22.35	19.72	4.85	5.22
Cu	19.56	8.54	16.67	4.12	16.99	13.51	5.92	5.64
Hg	ND	ND	ND	ND	ND	ND	ND	ND
Ni	20.58	17.79	15.67	14.18	19.19	17.30	12.46	13.26
Pb	29.78	17.08	9.26	3.64	19.33	28.23	4.38	2.91
Se	1.08	0.88	BDL	0.18	0.99	0.55	0.52	0.44
Zn	363.20	47.19	56.32	15.97	152.64	104.84	22.56	20.36

ND = Not determined.

CI = Chemical interference.

BDL = Below detection limits.

TABLE 34. ANALYSES OF NITRIC ACID DIGESTS OF SOIL SAMPLES FROM CONTROL BORINGS AT SITE L

Boring and sample	3C3	4C1	4C2	4C3	4C4	4C5	5C1	5C2	5C3
Elevation (m)	121.30	132.97	128.70	124.60	123.55	121.42	130.60	129.69	127.59
Ht. above water table (m)	-0.41	10.93	6.66	2.56	1.51	-0.62	9.22	8.31	6.21
Position in groundwater gradient	Updip	Updip	Updip	Updip	Updip	Updip	Downdip	Downdip	Downdip
Conc. (mg/kg dry wt.)									
Ca	99471.09	BDL	87854.64	60889.13	48323.21	118154.98	64425.96	107457.12	92233.30
Fe	2457.31	20183.24	5900.45	6006.69	5968.15	5753.95	7869.52	10247.30	6132.68
K	169.94	468.40	180.37	147.95	100.46	147.87	183.01	276.67	128.25
Mg	261724.70	19211.41	380088.13	129438.18	111607.54	199605.68	182993.14	513523.15	206783.56
Mn	168.76	465.60	4390.36	225.43	214.69	263.20	357.30	490.53	275.27
Na	83.99	28.84	101.18	61.03	53.95	94.26	82.99	139.33	85.50
As	CI	CI	CI	CI	CI	CI	CI	CI	CI
B	5.30	5.93	8.25	4.74	3.10	5.58	4.72	1.21	7.01
Be	0.21	0.44	0.49	0.32	0.28	0.42	0.18	0.34	0.27
Cd	BDL	1.60	1.51	1.04	1.52	1.16	BDL	1.63	0.99
Cr	5.87	9.32	5.96	4.62	4.49	6.36	4.08	11.43	6.63
Cu	3.63	15.90	6.23	3.13	3.47	3.56	7.62	10.05	5.27
Hg	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ni	8.73	18.49	32.43	9.39	7.68	6.99	13.79	15.88	9.77
Pb	2.02	69.91	4.40	4.81	4.74	3.33	1.97	14.74	4.78
Se	0.48	0.82	0.43	0.43	0.50	0.52	BDL	0.43	0.22
Zn	13.77	55.23	22.61	15.72	16.00	15.34	21.92	39.01	17.10

(continued)

TABLE 34 (CONTINUED)

Boring and sample	5C4	5C5	5C6	6C5	7C1	7C2	7C3	7C4	7C5
Elevation (m)	125.45	123.44	121.30	121.64	133.22	128.89	124.98	123.94	121.90
Ht. above water table (m)	4.07	2.06	-0.08	0.33	11.37	7.04	3.13	2.09	0.05
Position in groundwater gradient	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip
Conc. (mg/kg dry wt.)									
Ca	59016.09	75206.65	113848.83	83497.61	BDL	118910.67	51865.90	70760.66	61700.25
Fe	5846.68	8301.94	5710.91	6697.04	11710.84	5426.97	5507.47	6913.72	11039.06
K	116.77	84.52	128.68	183.00	220.34	146.91	96.65	161.54	159.30
Mg	201778.52	141170.26	237000.00	139865.94	7085.34	364173.30	242151.06	200775.64	109450.09
Mn	372.63	291.53	194.31	114.23	BDL	452.12	140.20	148.76	115.32
Na	71.00	61.30	87.02	74.60	19.34	104.50	15.61	73.01	60.59
As	CI	CI	CI	CI	CI	CI	CI	CI	CI
B	5.80	3.17	4.50	4.72	2.77	6.80	2.90	4.96	4.76
Be	0.23	0.22	0.31	0.27	0.28	0.28	0.19	0.27	0.25
Cd	1.02	BDL	0.96	BDL	1.12	BDL	BDL	0.95	BDL
Cr	5.66	4.87	6.58	7.08	5.28	7.31	4.94	6.36	6.01
Cu	5.86	3.84	9.16	6.11	8.38	5.72	4.85	5.77	5.66
Hg	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ni	14.54	12.22	9.50	10.51	10.55	12.70	10.07	9.20	9.66
Pb	2.04	2.29	2.91	5.83	5.37	5.17	BDL	4.56	4.89
Se	0.13	0.16	0.33	0.26	BDL	0.23	BDL	0.16	BDL
Zn	18.03	15.05	17.31	20.98	31.60	16.76	BDL	20.17	17.40

ND = Not determined.

CI = Chemical interference.

BDL = Below detection limits.

TABLE 35. ANALYSES OF NITRIC ACID DIGESTS OF SOIL SAMPLES FROM EXPERIMENTAL BORINGS AT SITE M

Boring and sample	1C1	1C2	1C3	1C4	4C1	4C2	4C3	4C4	4C5
Elevation (m)	222.77	220.81	218.96	217.73	221.98	221.07	218.93	217.41	216.65
Depth below sludge/soil interface (m)	-1.89	0.07	1.92	3.15	0.09	1.00	3.14	4.66	5.42
Ht. above water table (m)	5.96	4.00	2.15	0.92	5.03	4.12	1.98	0.46	-0.30
Conc. (mg/kg dry wt.)									
Ca	122338.27	6844.07	8719.09	3444.40	46703.96	3446.29	11764.94	21432.61	3014.33
Fe	19277.54	5520.36	8001.76	2726.38	25481.03	33521.89	5164.61	7976.06	2458.31
K	1167.77	330.44	418.27	123.83	1464.54	4352.11	264.21	546.09	126.10
Mg	66729.97	21571.57	31724.15	9365.43	31826.49	52473.91	27916.81	47758.52	8290.52
Mn	250.18	93.29	143.40	40.94	314.74	523.64	100.64	150.65	23.34
Na	840.61	90.21	102.04	49.95	366.71	267.58	70.79	130.16	48.37
As	CI	CI	CI	CI	CI	CI	CI	CI	CI
B	225.06	3.21	3.92	0.72	118.48	18.89	3.03	8.63	1.93
Be	4.21	0.22	0.29	0.10	2.08	1.61	0.12	0.20	BDL
Cd	2.09	BDL	1.01	BDL	7.52	4.55	BDL	BDL	BDL
Cr	29.54	3.74	6.02	1.66	24.98	26.40	5.35	7.25	1.93
Cu	29.66	1.97	5.59	BDL	38.40	24.68	1.77	3.31	BDL
Hg	ND	ND	ND	ND	ND	ND	ND	ND	ND
Ni	31.57	7.13	9.24	3.53	37.64	31.07	7.19	9.07	3.31
Pb	53.75	2.21	2.94	2.47	38.06	15.62	2.84	4.89	1.38
Se	0.33	0.10	0.11	0.06	2.42	0.94	0.05	BDL	0.16
Zn	95.46	17.84	23.74	6.88	351.72	98.70	19.14	24.96	8.64

ND = Not determined.

CI = Chemical interference.

BDL = Below detection limits.

TABLE 36. ANALYSES OF NITRIC ACID DIGESTS OF SOIL SAMPLES FROM CONTROL BORINGS AT SITE M

Boring and sample	2C1	2C2	2C3	2C4	3C1	3C2	3C3	3C4
Elevation (m)	221.78	220.87	218.72	217.97	221.70	220.79	218.64	217.89
Ht. above water table (m)	3.96	3.05	0.90	0.15	4.60	3.69	1.54	0.79
Position in groundwater gradient	Updip	Updip	Updip	Updip	Updip	Updip	Updip	Updip
Conc. (mg/kg dry wt.)								
Ca	4900.45	7084.11	23589.01	12068.92	7891.82	12529.70	22318.88	27196.65
Fe	23119.05	4880.78	13138.22	6123.80	20179.09	11676.24	10213.72	8269.48
K	2588.54	290.99	1031.18	328.09	1997.93	1178.64	856.82	616.17
Mg	61255.61	26588.65	68294.45	36791.23	59538.29	48418.57	69226.36	61192.46
Mn	615.46	102.21	261.65	109.83	520.40	269.63	263.80	134.23
Na	92.87	63.22	124.68	70.30	99.90	114.68	125.78	108.79
As	CI	CI	CI	CI	CI	CI	CI	CI
B	8.89	2.34	6.00	2.18	9.93	6.20	6.45	4.92
Be	1.24	0.23	0.48	BDL	0.74	0.40	0.40	0.32
Cd	2.66	0.95	1.67	BDL	2.11	1.89	1.29	1.07
Cr	18.05	4.49	10.76	5.07	17.75	11.21	9.81	7.93
Cu	19.56	1.62	14.94	2.57	24.27	9.86	7.20	4.81
Hg	ND	ND	ND	ND	ND	ND	ND	ND
Ni	22.58	8.42	20.38	7.62	22.34	13.21	11.12	11.78
Pb	14.82	2.32	8.99	2.21	9.99	5.31	4.73	4.25
Se	0.06	0.07	0.25	0.20	0.13	0.17	0.07	0.13
Zn	77.56	27.15	46.84	18.86	67.43	41.62	32.44	31.96

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TABLE 36 (CONTINUED)

Boring and sample	5C4	6C1	6C4	7C1	7C2	7C3	7C4
Elevation (m)	217.17	220.77	216.14	221.09	220.18	218.05	216.52
Ht. above water table (m)	0.03	4.60	-0.03	4.84	3.93	1.80	0.27
Position in groundwater gradient	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip	Downdip
Conc. (mg/kg dry wt.)							
Ca	4795.95	11083.92	9614.80	6826.47	2758.40	4481.85	19333.84
Fe	3386.38	17960.39	4814.58	11194.91	22964.75	22405.08	8726.38
K	167.78	2164.50	240.37	1415.46	2822.80	2573.74	487.33
Mg	15440.91	48485.67	28060.94	31354.10	48941.54	46148.41	47835.09
Mn	17.62	511.25	59.26	222.11	395.98	483.61	170.35
Na	38.63	95.15	60.96	79.73	89.08	126.47	112.61
As	CI	CI	CI	CI	CI	CI	CI
B	1.33	6.73	1.34	4.87	7.87	7.53	2.70
Be	0.10	0.68	BDL	0.48	1.17	0.98	0.13
Cd	BDL	2.98	1.17	2.13	3.49	3.15	1.62
Cr	2.11	14.07	3.94	9.91	20.24	18.92	5.87
Cu	1.15	15.48	1.70	9.05	17.71	17.08	5.21
Hg	ND	ND	ND	ND	ND	ND	ND
Ni	4.45	19.49	7.28	12.13	22.80	22.68	10.40
Pb	2.35	13.59	2.24	10.75	11.81	13.31	4.98
Se	0.18	0.21	0.17	0.12	0.17	0.19	0.10
Zn	12.92	71.94	17.37	72.65	72.77	71.22	30.10

ND = Not determined.
CI = Chemical interference.
BDL = Below detection limits.

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samples beneath and from comparable depths outside the disposal area are given in Table 37. At site K, significant differences in concentrations in the nitric acid digests were observed only for iron, sodium and copper. All three metals showed a decrease in concentration in soil below the disposal site. The reduction in sodium noted below the landfill is very likely related to the replacement of sodium by calcium in clays beneath the disposal pond. Mobilization and ion exchange may also account for the slightly smaller amounts of iron and copper reported in the soil samples from below the pond. The only other significant difference between samples inside and outside any of the other disposal areas was an increase in boron concentration in soil samples below the disposal pit at site L. Boron is a common contaminant associated with leachate from ash. Ash was co-disposed with sludge at site L; therefore the occurrence of boron was not unexpected. Boron also was found in significantly larger quantities in the distilled water extracts from soil under site L.

The lack of significant increases in sludge-derived constituents in soil beneath the disposal areas indicates very little of the contaminating material is being trapped and removed as the leachate passes through into the ground-water. The major materials derived from the FGC sludge/ash are in solution at high concentrations. Typical soils below the disposal sites showed few changes in composition that can be related to the passage of leachate through them. At only one site (site K) was there evidence that calcium was displacing other ions from the available exchange positions, and becoming fixed in the soil.

Vertical Variations of Concentrations in the Nitric Acid Digests of Soil Samples

For elements that showed a significant difference between experimental and control samples, a test was made for a significant relationship between the concentration of a particular constituent and sample elevation in the boring. As suggested by the model (Figure 1), those materials attenuated from the sludge leachate should show a positive correlation with elevation in experimental borings (those below the disposal area). A significant negative correlation would be predicted for those soil constituents that are being mobilized by the sludge leachate and moved down into the groundwater. In control borings the distribution of soil constituents depends upon the weathering processes; therefore, the concentration of any particular material could have a positive or negative correlation or have no correlation at all. The Spearman rank correlation coefficient was used to assess the strength of the association between concentration of a particular constituent and sample elevation. The results of these statistical tests are given in Tables 38 and 39. Plots of concentration versus sample elevation for all constituents in experimental borings that showed significant relationships with depth are shown in Figures 16 and 17. Significant trends in control borings are shown for contrast.

At site K, only iron in boring 2 showed a significant correlation with sample elevation. The amount of iron in the samples increased with increasing elevation. This is the effect which would be expected if iron were being added to the soil/sediment. At site L, boron showed a positive correlation in both borings one and two under the disposal pit. This increase is what would

TABLE 37. RESULTS OF RANDOMIZATION TEST ON NITRIC ACID DIGESTS OF SOIL
SAMPLES DIRECTLY UNDER THE FGC DISPOSAL SITES AND AT COMPARABLE
DEPTHS OUTSIDE THE SITES

Parameters	Site K	Site L	Site M
Ca	NS	NS	NS
Fe	S(decrease)*	NS	NS
K	ND	NS	NS
Mg	NS	NS	NS
Mn	NS	NS	NS
Na	S(decrease)	NS	NS
As	ND	ND	ND
B	ND	S(increase)	NS
Be	NS	NS	NS
Cd	NS	NS	NS
Cr	NS	NS	NS
Cu	S(decrease)	NS	NS
Hg	ND	ND	ND
Ni	NS	NS	NS
Pb	NS	NS	NS
Se	ND	NS	NS
Zn	NS	NS	NS

NS = Not significant at 80% confidence level.

S = Significant at 80% confidence level.

ND = Not determined.

* Refers to increase or decrease of constituent under disposal site relative
to outside.

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TABLE 38. CORRELATION OF CHEMICAL ANALYSES OF NITRIC ACID DIGESTS OF SOILS WITH SAMPLE ELEVATION AT SITE K

Boring	1	2	6	7
Fe	NS(0.40)	SN(-1.00)	SP(1.00)*	NS(0.80)
Na	NS(-0.40)	NS(0.40)	NS(0.50)*	NS(0.20)
Cu	NS(0.20)	NS(-0.40)	SP(1.00)*	NS(0.80)

SP = Significant positive correlation at 95% level.

SN = Significant negative correlation at 95% level.

NS = No significant correlation.

* = Significance level reduced to 83% because of small sample size for this constituent in this boring.

Number in parentheses is the calculated value of r_s, the Spearman rank correlation coefficient.

TABLE 39. CORRELATION OF CHEMICAL ANALYSES OF NITRIC ACID DIGESTS OF SOILS WITH SAMPLE ELEVATION AT SITE L

Boring	1	2	4	5	7
B	SP(1.00)	SP(1.00)	NS(0.60)	NS(0.08)	NS(-0.30)

SP = Significant positive correlation at 95% level.

SN = Significant negative correlation at 95% level.

NS = No significant correlation.

Number in parentheses is the calculated value of r_s, the Spearman rank correlation coefficient.

Figure 16. Variation of iron concentration in nitric acid digests of soil/sediment samples with elevation in borings 2 and 6 at site K.
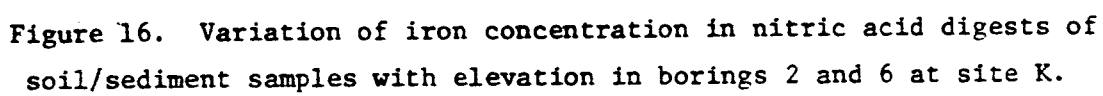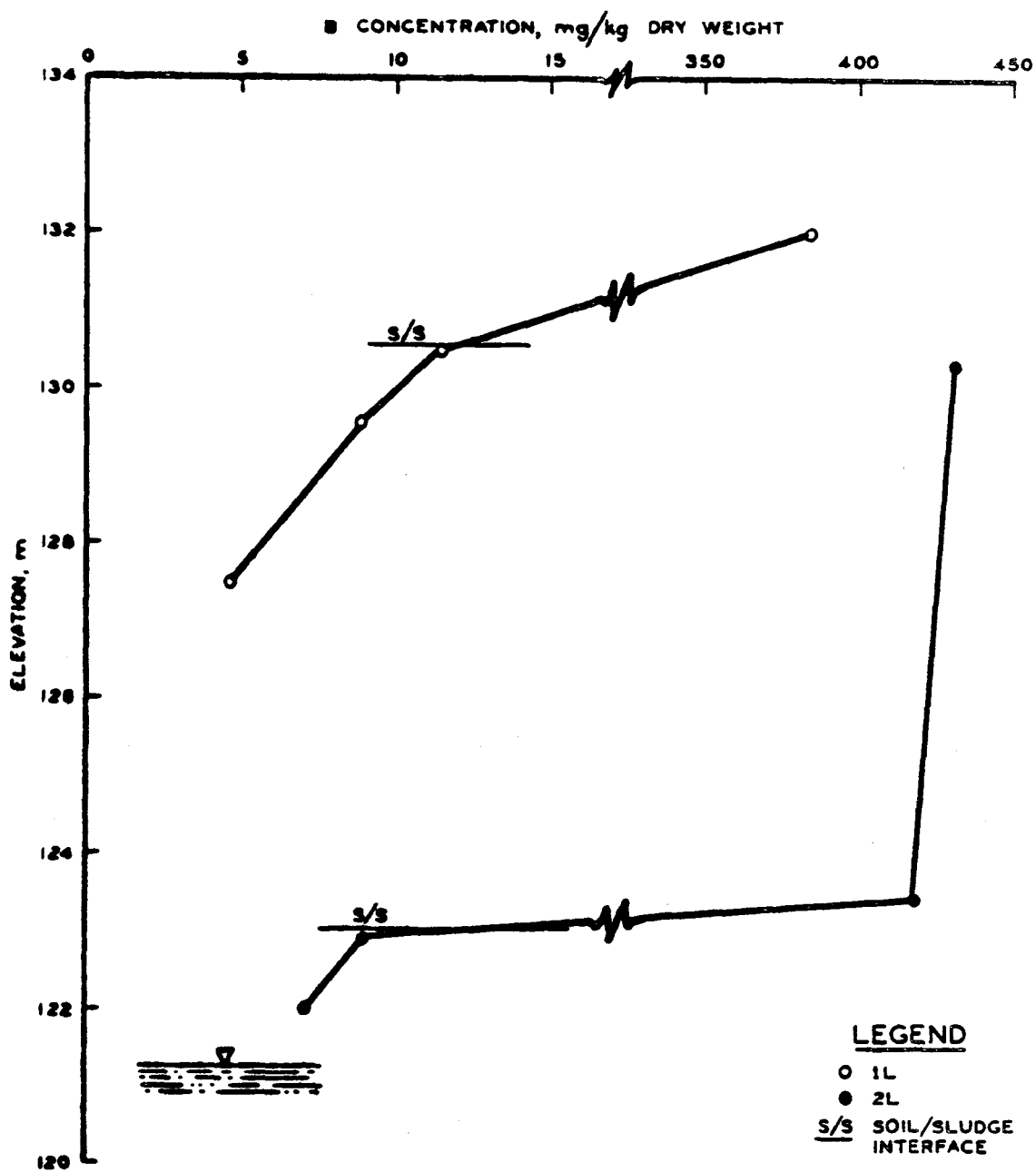
Figure 17. Variation of boron concentration of nitric acid digests of
soil/sediment samples with elevation in borings 1 and 2 at site L.

be expected if boron was being added from the disposal pit.

As noted above, there was only slight evidence of attenuation. Only iron at site K and boron at site L showed the patterns indicating they were being removed by the soil. The most striking feature of the data was the lack of other demonstrable interaction of leachate with the soil.

Horizontal Variation in Nitric Acid Digests of Soil/Sediment Below the Water Table

Analyses of nitric acid digests of soil/sediment samples collected below the water table were examined in order to determine if contaminants were migrating with the groundwater flow below the disposal site. The model for groundwater movement assumes all significant lateral migration of contaminants takes place below the water table. The increased concentrations of contaminants in the nitric acid digests should be a measure of the attenuation occurring during lateral migration of the pollutants. Plots of metal concentrations in the nitric acid digests versus the positions of the borings are given in Figures 18-20.

At site K, all of the constituents measured in nitric acid digests of soil/sediment, with the exception of manganese, showed maximum levels underneath the disposal pond. This distribution pattern suggests that the materials being leached from the sludge (FGC wastes and ash) are being attenuated or contained in soils under the landfill. The sediments at this site are largely clays, shales and limestone that are impervious and could prevent dispersal of incoming constituents down the groundwater gradient.

Sites L and M are both underlain by porous sands and gravels. No consistent patterns could be found that related concentrations of various materials in the nitric acid digests to the position of the soil samples with respect to the disposal areas. There was no increase in contaminants under the disposal areas that could be interpreted as indicating that attenuation or containment of the pollutants had taken place.

SUMMARY

The physical testing data indicate two major types of sites were included in this study; one type underlain by impermeable materials, clay and shale, etc. (site K), and a second type underlain by relatively permeable, silty, sands and gravel with discontinuously distributed finer material included (sites L and M). At the site underlain by clay and shale the typical permeabilities or hydraulic conductivities were very low ($\sim 2 \times 10^{-8}$ cm/sec) and no change in permeability could be related to the presence of the sludge/ash disposal site. At site M, changes in permeability could be noted, but these changes appeared to be more related to the irregular occurrences of fine-grained materials (clays and silty sands) than to the presence of the disposal facility. Only at site L could variations in physical properties (permeability, dry density, water content, percent fines) measured in soil samples from test borings be related to the disposal of FGC sludge and ash.
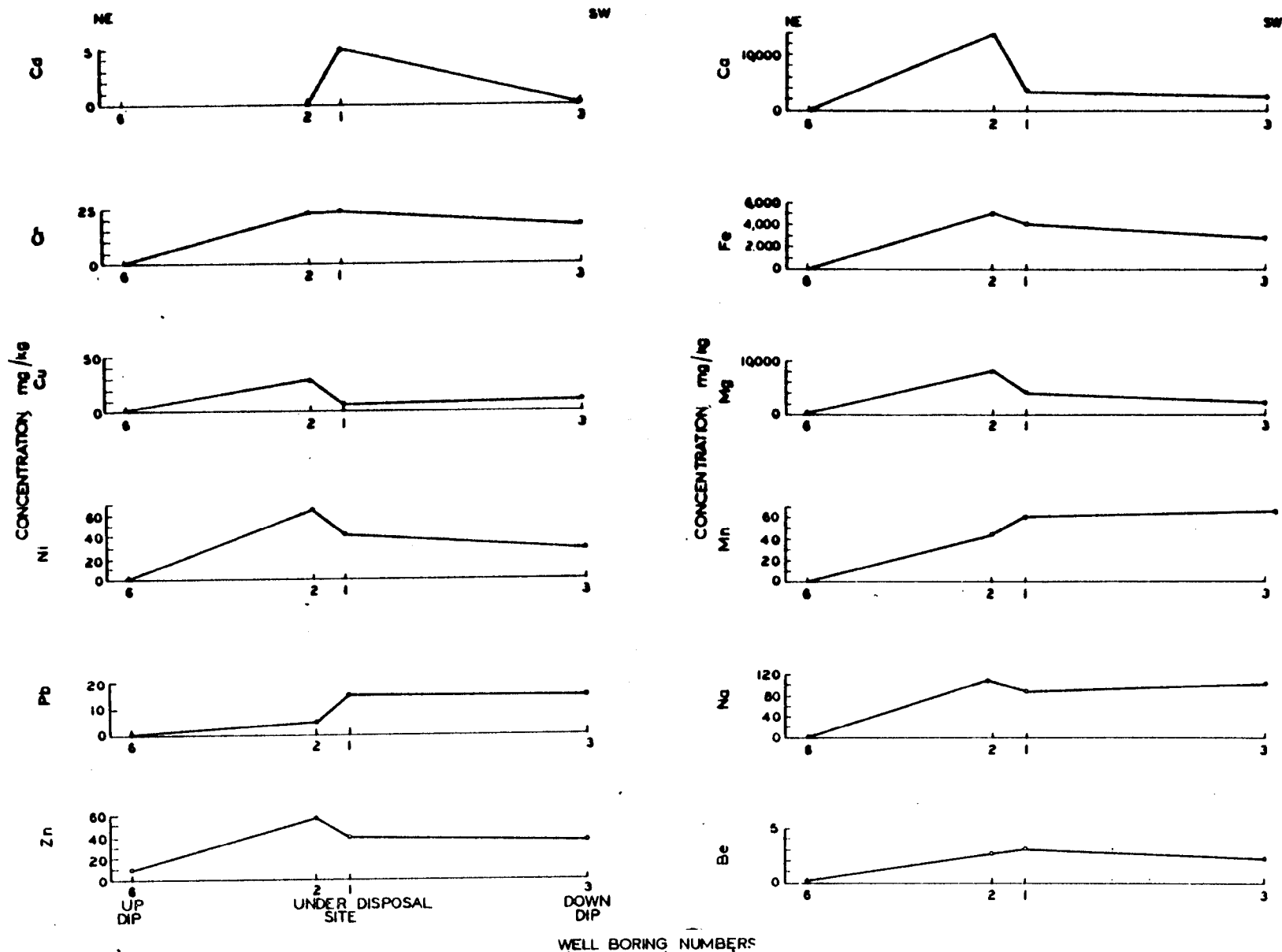
Figure 18.   Horizontal variation in chemical composition of nitric acid digests at site K.
BDL indicates below detection limits.

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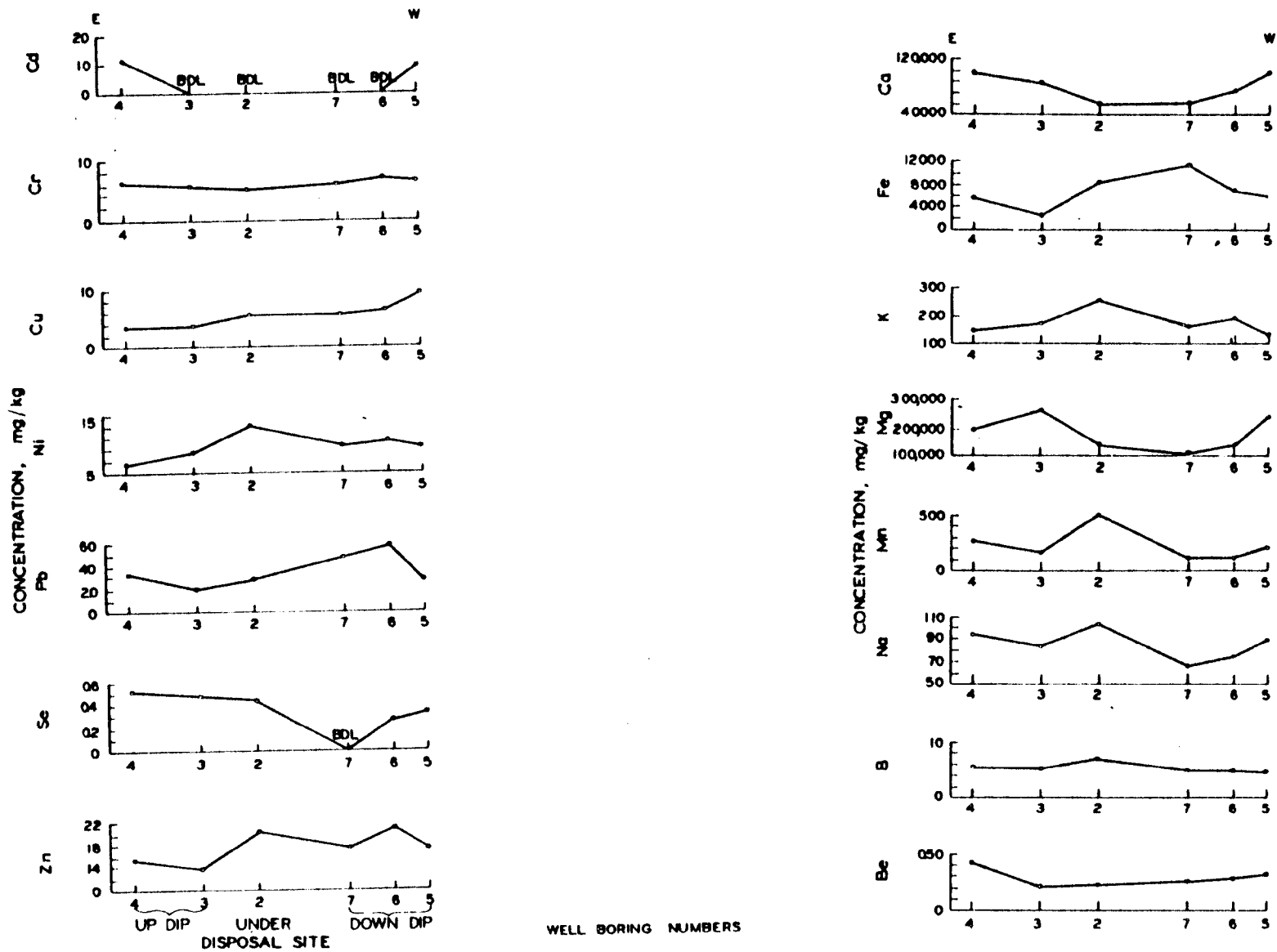
Figure 19. Horizontal variation in chemical composition of nitric acid digests at site L. BDL indicates below detection limits.
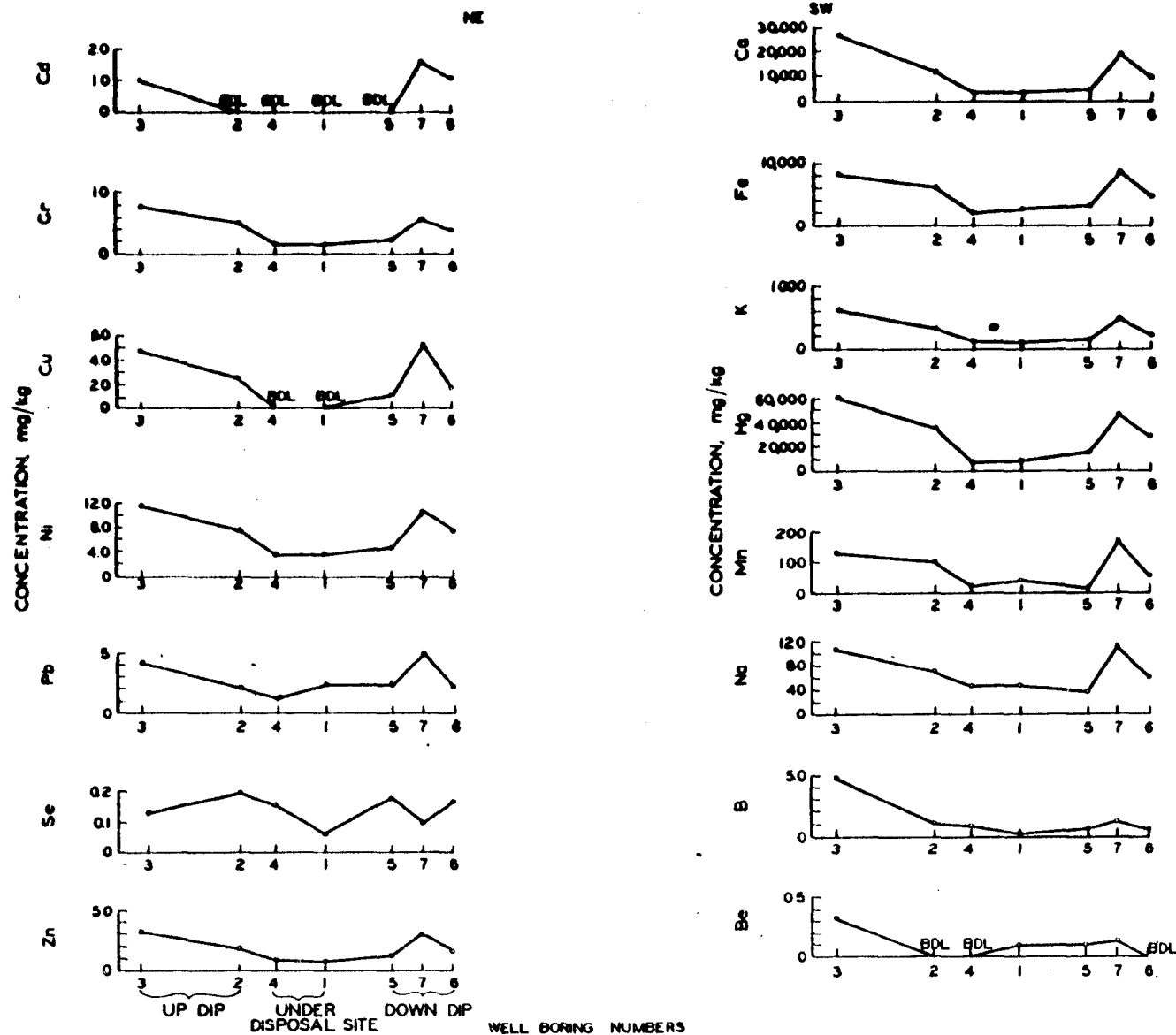
Figure 20. Horizontal variation in chemical composition of nitric acid digests at site M. BDL indicates below detection limits.

Although the potential of FGC sludge and ash for pollution of local groundwater has been noted, (13) no field evidence of such pollution occurring has been reported. At all three sites in this study, it could be shown that sludge/ash-derived constituents had migrated out of the immediate area of the disposal site and were found in local groundwater. The subsurface migration of FGC/ash-derived materials seemed to be most limited at the site where the pond was underlain by impermeable strata (site K). Although one boring outside the pond was severely contaminated, additional borings around the pond showed no groundwater when drilled to comparable depths. The only other boring from which a groundwater sample was obtained at this site was down the apparent groundwater gradient from the pond and near a large cooling lake. No contaminants from the pond were detected in this boring. At the other sites (L and M) which were underlain by sands and gravels, evidence of a typical pollution plume under and down the groundwater gradient from the disposal site was found.

The investigation of distilled water extracts and nitric acid digests of soil samples from underneath and around sludge/ash disposal sites indicates only slight changes in soil chemistry can be attributed to the presence of the disposal site. Evidently FGC sludge/ash leachates can move through the soils and sediments studied without appreciable interaction.

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Figure A-1. Water table map of site K. Elevations are in feet above mean sea level. 1 foot = 0.305 meters.

TABLE A-1. LOG OF BORING 1 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
264.00 - 261.56	0.00 - 2.44	Bottom ash
261.00 - 260.49	2.44 - 3.51	Clay, brown, wet
260.49 - 258.36	3.51 - 5.64	Clay, brown, moist, hard
258.36 - 256.26	5.64 - 7.74	Shale, green, hard

* MSL = Mean sea level.

Water table elevation above MSL = 261.56 m

TABLE A-2. LOG OF BORING 2 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
263.67 - 261.84	0.00 - 1.83	Bottom ash
261.84 - 258.18	1.83 - 5.49	FGC sludge, wet
258.18 - 257.88	5.49 - 5.79	Clay, black
257.88 - 257.27	5.79 - 6.40	Clay, brown
257.27 - 255.13	6.40 - 8.54	Clay, brown, wet
255.13 - 254.52	8.54 - 9.15	Clay, green, dry, hard
254.52 - 253.61	9.15 - 10.06	Clay, green
253.61 - 253.46	10.06 - 10.21	Bedrock

* MSL = Mean sea level.

Water table elevation above MSL = 255.01 m

TABLE A-3.   LOG OF BORING 3 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
259.15 - 251.53	0.00 - 7.62	Clay, dark brown with trace of black clay in lower portion
251.53 - 250.80	7.62 - 8.35	Shale, dark, weathered

* MSL = Mean sea level.

Water table elevation above MSL = 259.10 m

TABLE A-4.   LOG OF BORING 4 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
263.53 - 262.92	0.00 - 0.61	Road bed
262.92 - 255.91	0.61 - 7.62	Clay, brown
255.91 - 254.54	7.62 - 8.99	Clay, black, wet

* MSL = Mean sea level.

Water table elevation above MSL = (dry hole)

TABLE A-5.   LOG OF BORING 5 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
258.90 - 256.46	0.00 - 2.44	Clay, brown
256.46 - 252.99	2.44 - 5.91	Limestone, brown, weathered with some interbedded silty layers

* MSL = Mean sea level.

Water table elevation above MSL = (dry hole)

TABLE A-6.   LOG OF BORING 6 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
265.24 - 264.33	0.00 - 0.91	Clay, black, wet
264.33 - 261.58	0.91 - 3.66	Clay, gray-brown, wet
261.58 - 260.06	3.66 - 5.18	Clay, brown with weathered limestone colluvial material
260.06 - 258.08	5.18 - 7.16	Shale, gray
258.08 - 256.86	7.16 - 8.38	Shale, gray, wet

* MSL = Mean sea level.

Water table elevation above MSL = 260.06 m

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TABLE A-7. LOG OF BORING 7 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
265.24 - 264.33	0.00 - 0.91	Clay, black
264.33 - 262.50	0.91 - 2.74	Clay, brown
262.50 - 260.06	2.74 - 5.18	Clay, brown with decomposed limestone material
260.06 - 259.94	5.18 - 5.30	Competent layer
259.94 - 258.84	5.30 - 6.40	Shale, green, hard
258.84 - 257.70	6.40 - 7.54	Shale, gray, hard

* MSL = Mean sea level.

Water table elevation above MSL = (dry hole)

TABLE A-8.   LOG OF BORING 8 AT SITE K

Elevation above MSL* (m)	Depth (m)	Description
265.24 - 264.63	0.00 - 0.61	Roadbed (bottom ash)
264.63 - 264.02	0.61 - 1.22	Clay, black, hard
264.02 - 263.41	1.22 - 1.83	Clay, brown, hard
263.41 - 262.50	1.83 - 2.74	Clay, brown, hard, moist
262.50 - 262.19	2.74 - 3.05	Clay, brown, with limestone pebbles
262.19 - 261.73	3.05 - 3.51	Clay, brown with shale chips
261.73 - 261.58	3.51 - 3.66	Competent layer
261.58 - 260.36	3.66 - 4.88	Clay, brown with shale chips
260.36 - 257.31	4.88 - 7.93	Shale, brown, hard
257.31 - 257.07	7.93 - 8.17	Shale, gray, hard

* MSL - Mean sea level.

Water table elevation above MSL = (dry hole)·

TABLE A-9. LIST OF SAMPLES EXAMINED FROM SITE K

Boring	Elevation of top of hole (m)	Elevation of water table (m)	Total depth (m)	Thickness of cover (m)	Thickness of fill (m)	Elevation sludge/soil interface (m)	Sampled depth interval (m) From	To	Elevation of sampled intervals (m) From	To	Type of sample	Sample number
1	264.00	261.56	7.74	0	2.44	261.56	2.59	2.74	261.41	261.26	Chemical	1C1
							3.51	3.66	260.49	260.34	Chemical	1C2
							5.64	5.76	258.36	258.24	Chemical	1C3
							5.76	5.91	258.24	258.09	Physical	1P3
							7.62	7.74	256.38	256.26	Chemical	1C4
2	263.67	255.01	10.21	0	5.49	258.18	3.61	3.73	260.06	259.94	Chemical	2C1
							5.55	5.70	258.12	257.97	Chemical	2C2
							5.76	6.10	257.91	257.57	Physical	2P1
							6.40	6.59	257.27	257.08	Chemical	2C3
							8.53	8.66	255.14	255.01	Chemical	2C4
							8.72	8.93	254.95	254.74	Physical	2P3
3	259.15	259.10	8.35	NA	NA	NA	1.74	2.15	257.41	257.00	Physical	3P1
							4.79	5.29	254.36	253.86	Physical	3P3
							7.62	7.77	251.53	251.38	Chemical	3C5
							7.83	8.32	251.32	250.83	Physical	3P5
4	263.53	(dry)	8.99	NA	NA	NA	3.26	3.75	260.27	259.78	Physical	4P1
							7.83	8.29	255.70	255.24	Physical	4P4
5	258.90	(dry)	5.91	NA	NA	NA	1.52	1.68	257.38	257.22	Chemical	5C1
							1.74	2.20	257.16	256.70	Physical	5P1
							2.44	2.59	256.46	256.31	Chemical	5C2
							2.65	3.12	256.25	255.78	Physical	5P2
6	265.24	260.06	8.38	NA	NA	NA	2.74	2.90	262.50	262.34	Chemical	6C1
							2.96	3.41	262.28	261.83	Physical	6P1
							3.66	3.81	261.58	261.43	Chemical	6C2
							3.87	4.91	261.37	260.33	Physical	6P2
							7.32	7.39	257.92	257.85	Chemical	6C3
7	265.24	(dry)	7.54	NA	NA	NA	2.74	2.90	262.50	262.34	Chemical	7C1
							2.96	3.31	262.28	261.93	Physical	7P1
							3.96	4.11	261.28	261.13	Chemical	7C2
							5.79	5.91	259.45	259.33	Chemical	7C3
							7.32	7.48	257.92	257.76	Chemical	7C4

NA = Not applicable

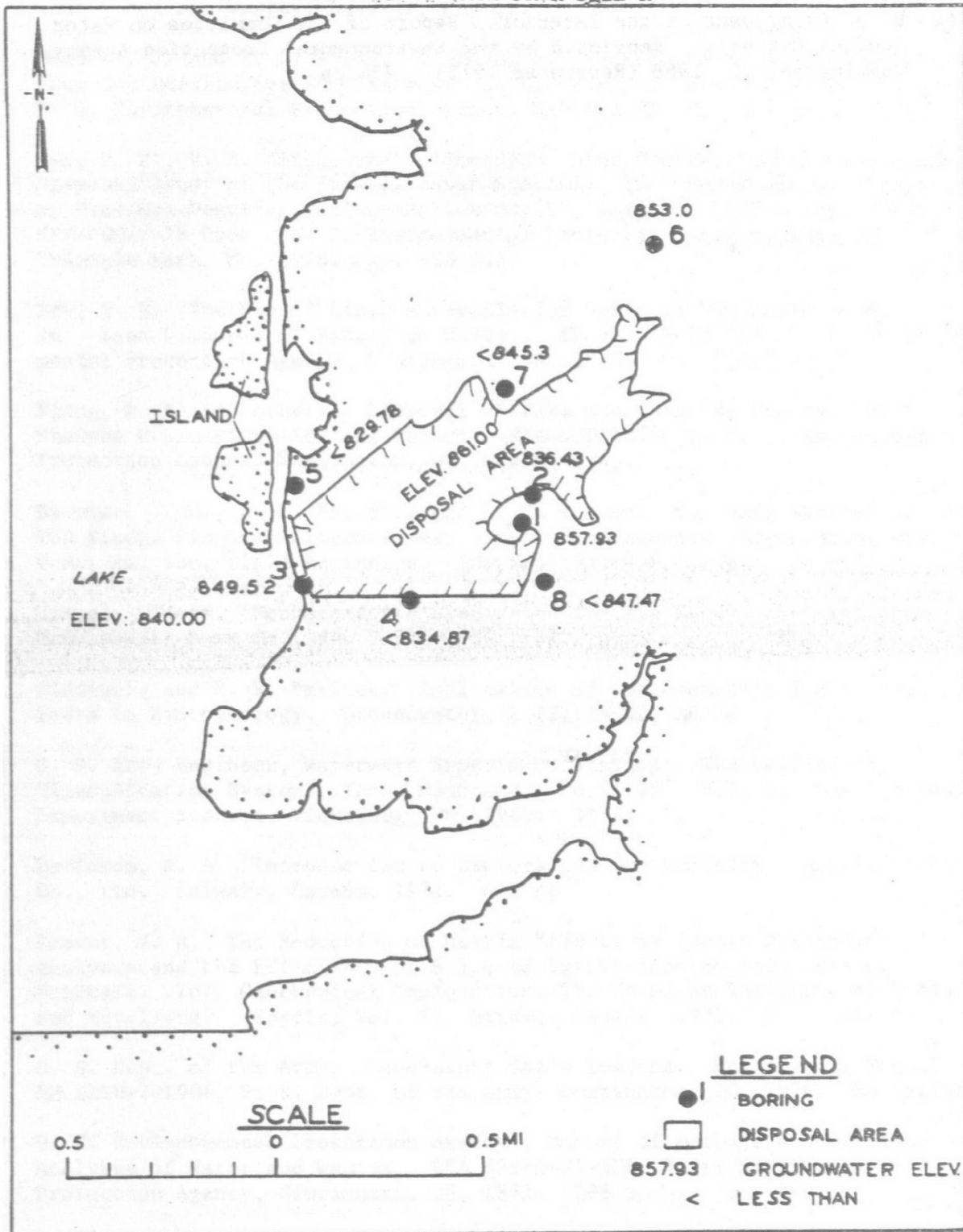Note: All elevations are given with respect to mean sea level.
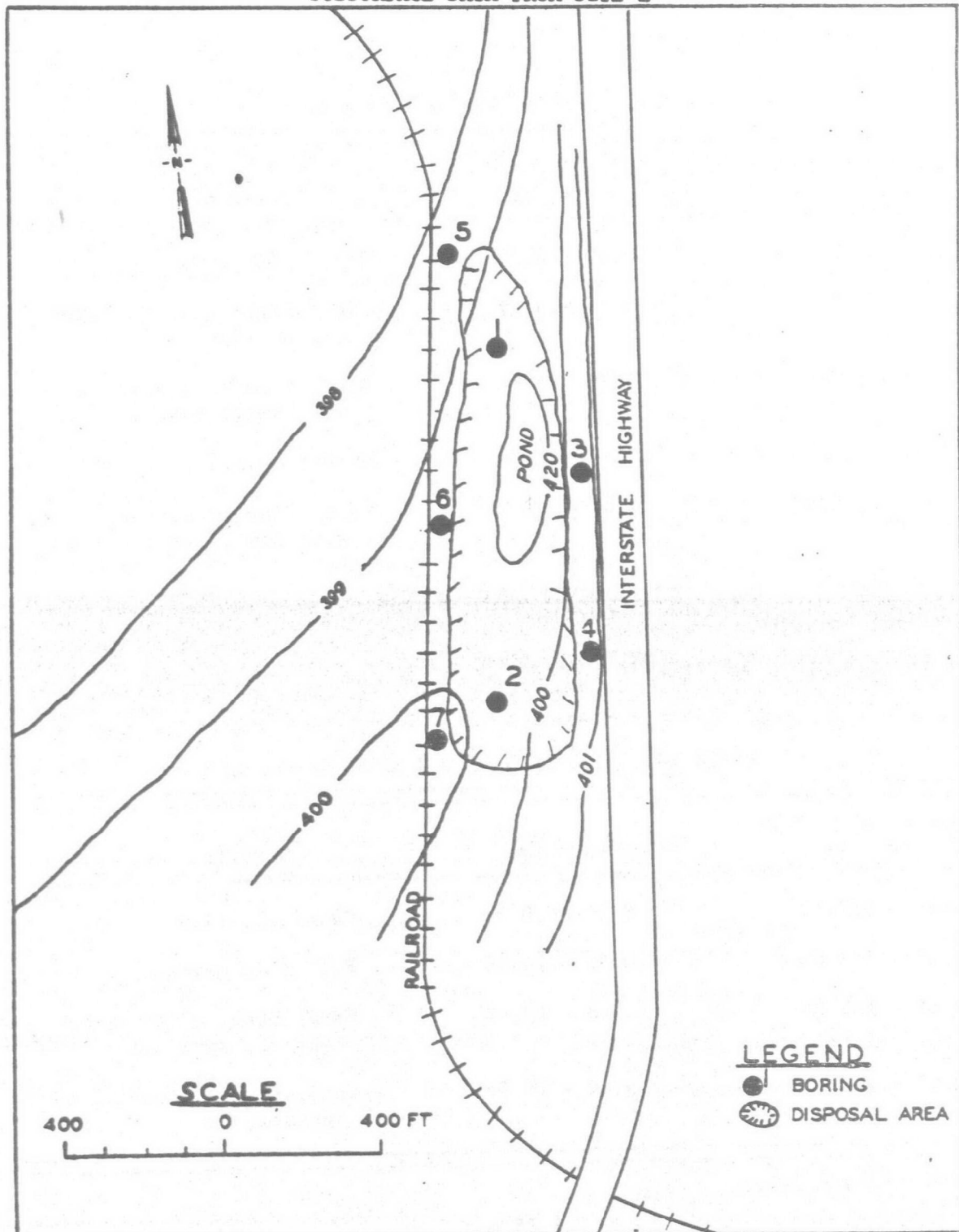
Figure B-1. Water table map of site L. Elevations are in feet above mean sea level. 1 foot = 0.305 meters.

TABLE B-1. LOG OF BORING 1 AT SITE L

Elevation above MSL* (m)	Depth (m)	Description
133.47 - 130.57	0.00 - 2.90	Fill (FGC sludge)
130.57 - 127.68	2.90 - 5.79	Clay, light gray to light brown, silty
127.68 - 127.53	5.79 - 5.94	Sand, fine to coarse, silty with small gravel
127.53 - 126.00	5.94 - 7.47	Gravel, small to large
126.00 - 119.90	7.47 - 13.57	Sand, fine to coarse, silty, with small gravel, light tan

*MSL = Mean sea level.
Water table elevation above MSL = 121.27 m

TABLE B-2. LOG OF BORING 2 AT SITE L

Elevation above MSL* (m)	Depth (m)	Description
137.49 - 137.19	0.00 - 0.30	Backfill (clay)
137.19 - 123.01	0.30 - 14.48	Fill (FGC sludge)
123.01 - 122.09	14.48 - 15.40	Sand, fine, silty with gravel, dark tan
122.09 - 120.87	15.40 - 16.62	Sand, fine to coarse, with gravel, wet

* MSL = Mean sea level.
Water table elevation above MSL = 121.85 m

TABLE B-3. LOG OF BORING 3 AT SITE L

Elevation above MSL* (m)	Depth (m)	Description
133.70 - 131.26	0.00 - 2.44	Clay, brown
131.26 - 129.74	2.44 - 3.96	Sand, wet, dark brown
129.74 - 125.16	3.96 - 8.54	Sand, fine to coarse with some gravel, damp, dark brown
125.16 - 123.34	8.54 - 10.36	Sand, fine to coarse, some gravel, damp, light tan
123.34 - 121.50	10.36 - 12.20	Sand, fine to coarse, gravely, moist, light tan
121.50 - 119.68	12.20 - 14.02	Sand, fine to medium, some gravel, wet, light tan

* MSL = Mean sea level.

Water table elevation above MSL = 121.58 m

TABLE B-4. LOG OF BORING 4 AT SITE L

Elevation above MSL* (m)	Depth (m)	Description
136.71 - 133.05	0.00 - 3.66	Clay, brown
133.05 - 131.47	3.66 - 5.24	Sand, fine, silty, light brown
131.47 - 124.67	5.24 - 12.04	Sand, fine to coarse with small to large gravel
124.67 - 121.16	12.04 - 15.55	Sand, fine to coarse with small gravel

* MSL = Mean sea level.

Water table elevation above MSL = 122.05 m

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TABLE B-5. LOG OF BORING 5 AT SITE L

Elevation above MSL* (m)	Depth (m)	Description
133.73 - 132.51	0.00 - 1.22	Silt, sandy with gravel
132.51 - 131.90	1.22 - 1.83	Clay, silty, light brown
131.90 - 130.83	1.83 - 2.90	Gravel, clayey
130.83 - 130.76	2.90 - 2.97	Gravel, sandy
130.76 - 128.24	2.97 - 5.49	Sand, fine to coarse, silty with small to large gravel dark tan
128.24 - 127.79	5.49 - 5.94	Gravel, small to large
127.79 - 125.68	5.94 - 8.05	Sand, fine to coarse with small to large gravel
125.68 - 121.53	8.05 - 12.20	Sand, fine to coarse with gravel
121.53 - 119.43	12.20 - 14.30	Sand, fine to coarse, with some gravel, moist

* MSL = Mean sea level.

Water table elevation above MSL = 121.38 m

TABLE B-6. LOG OF BORING 6 AT SITE L

Elevation above MSL* (m)	Depth (m)	Description
137.34 - 133.07	0.00 - 4.27	Clay, light brown
133.07 - 131.85	4.27 - 5.49	Sand with gravel
131.85 - 124.84	5.49 - 12.50	Sand, fine to coarse with small to large gravel
124.84 - 123.93	12.50 - 13.41	Sand, fine to coarse with gravel
123.93 - 121.79	13.41 - 15.55	Sand, fine to coarse, with gravel, damp
121.79 - 120.57	15.55 - 16.77	Sand, fine to coarse with some gravel, damp

* MSL = Mean sea level.

Water table elevation above MSL = 121.30 m

TABLE B-7. LOG OF BORING 7 AT SITE L

Elevation above MSL* (m)	Depth (m)	Description
137.58 - 133.31	0.00 - 4.27	Clay
133.31 - 130.87	4.27 - 6.71	Sand, fine, silty, light brown
130.87 - 128.74	6.71 - 8.84	Sand, fine to coarse, with small to large gravel
128.74 - 126.00	8.84 - 11.58	Gravel, small to large
126.00 - 125.08	11.58 - 12.50	Sand, fine to coarse, with gravel
125.08 - 120.81	12.5 - 16.77	Sand, fine to coarse with some gravel

* MSL = Mean sea level.

Water table elevation above MSL = 121.85 m

TABLE B-8. LIST OF SAMPLES EXAMINED FROM SITE L

Boring	Elevation of top of hole (m)	Elevation of water table (m)	Total depth (m)	Thickness of cover (m)	Thickness of fill (m)	Elevation sludge/soil interface (m)	Sampled depth interval (m) From	To	Elevation of sampled intervals (m) From	To	Type of sample	Sample number
1	133.47	121.27	13.57	0	2.90	130.57	0.00	2.90	133.47	130.57	Chemical	1C1
							2.90	3.08	130.57	130.39	Chemical	1C2
							3.14	3.81	130.33	129.66	Physical	1P1
							3.81	3.99	129.66	129.48	Chemical	1C3
							5.94	6.04	127.53	127.43	Chemical	1C4
2	137.49	121.85	16.62	0.30	14.18	123.01	7.18	7.30	130.31	130.19	Chemical	2C1
							13.80	14.40	123.69	123.09	Chemical	2C2
							14.48	14.66	123.01	122.83	Chemical	2C3
							14.72	14.78	122.77	122.71	Physical	2P1
							15.39	15.61	122.10	121.88	Chemical	2C4
							15.67	14.97	121.82	121.52	Physical	2P2
3	133.70	121.58	14.02	NA	NA	NA	6.64	6.95	127.06	126.75	Physical	3P1
							12.20	12.59	121.50	121.11	Chemical	3C5
4	136.71	122.05	15.55	NA	NA	NA	3.66	3.81	133.05	132.90	Chemical	4C1
							3.87	4.30	132.84	132.41	Physical	4P1
							7.92	8.08	128.79	128.63	Chemical	4C2
							8.20	8.41	128.51	128.30	Physical	4P2
							12.04	12.16	124.67	124.55	Chemical	4C3
							12.95	13.35	123.76	123.36	Chemical	4C4
							15.09	15.48	121.62	121.23	Chemical	4C5
5	133.73	121.38	14.30	NA	NA	NA	2.97	3.28	130.76	130.45	Chemical	5C1
							3.81	4.27	129.92	129.46	Chemical	5C2
							5.94	6.19	127.79	127.54	Chemical	5C3
							8.05	8.50	125.68	125.23	Chemical	5C4
							10.06	10.52	123.67	123.21	Chemical	5C5
							12.19	12.65	121.54	121.08	Chemical	5C6
6	137.34	121.30	16.77	NA	NA	NA	4.51	4.69	132.83	132.65	Physical	6P1
							12.71	12.80	124.63	124.54	Physical	6P2
							15.54	15.85	121.80	121.49	Chemical	6C5
7	137.58	121.85	16.77	NA	NA	NA	4.27	4.45	133.31	133.13	Chemical	7C1
							4.51	4.91	133.07	132.67	Physical	7P1
							8.53	8.84	129.05	128.74	Chemical	7C2
							12.50	12.71	125.08	124.87	Chemical	7C3
							13.41	13.87	124.17	123.71	Chemical	7C4
							15.55	15.82	122.03	121.76	Chemical	7C5

NA = Not applicable.

Note: All elevations are given with respect to mean sea level.
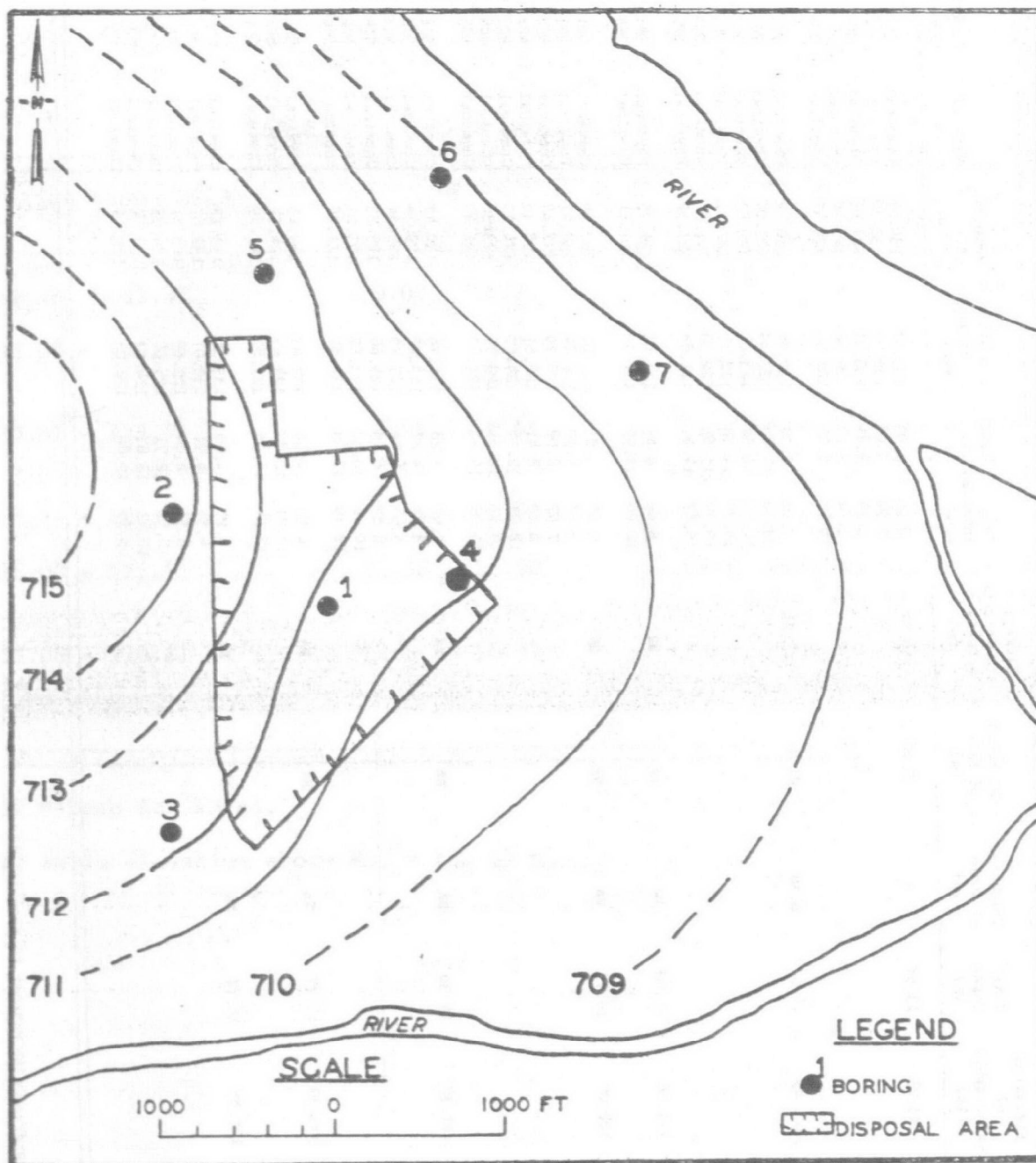
Figure C-1. Water table map of site M.
Elevations are in feet above mean sea level.
1 foot = 0.305 meters.

TABLE C-1.   LOG OF BORING 1 AT SITE M

Elevation above MSL* (m)	Depth (m)	Description
224.68 - 220.88	0.00 - 3.80	Fill (FGC sludge)
220.88 - 217.82	3.80 - 6.86	Sand, fine, silty, light tan
217.82 - 216.14	6.86 - 8.54	Sand, fine, silty, light gray, wet

* MSL = Mean sea level.

Water table elevation above MSL = 216.82 m

TABLE C-2.   LOG OF BORING 2 AT SITE M

Elevation above MSL* (m)	Depth (m)	Description
222.18 - 221.88	0.00 - 0.30	Top soil
221.88 - 221.42	0.30 - 0.76	Silt, light tan
221.42 - 218.83	0.76 - 3.35	Sand, fine, silty, light tan
218.83 - 216.39	3.35 - 5.79	Sand, fine, silty, wet, with trace of organic matter

* MSL = Mean sea level.

Water table elevation above MSL = 217.82 m

TABLE C-3. LOG OF BORING 3 AT SITE M

Elevation above MSL* (m)	Depth (m)	Description
222.10 - 221.80	0.00 - 0.30	Topsoil
221.80 - 220.88	0.30 - 1.22	Clay, silty, dark gray
220.88 - 220.58	1.22 - 1.52	Silt, light tan to dark gray
220.58 - 218.75	1.52 - 3.35	Sand, fine, silty, light tan
218.75 - 218.44	3.35 - 3.66	Clay with silt and sand, soft, dark gray
218.44 - 216.31	3.66 - 5.79	Sand, fine, wet

* MSL = Mean sea level.

Water table elevation above MSL = 217.10 m

TABLE C-4. LOG OF BORING 4 AT SITE M

Elevation above MSL* (m)	Depth (m)	Description
224.36 - 222.07	0.00 - 2.29	Fill (FGC sludge and soil)
222.07 - 219.02	2.29 - 5.34	Clay, silty, wet, dark gray
219.02 - 217.50	5.34 - 6.86	Sand, fine, silty, light tan
217.50 - 215.21	6.86 - 9.15	Sand, fine, silty, wet

* MSL = Mean sea level.

Water table elevation above MSL = 216.77 m

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TABLE C-5. LOG OF BORING 5 AT SITE M

Elevation above MSL* (m)	Depth (m)	Description
222.14 - 221.84	0.00 - 0.30	Topsoil
221.84 - 221.23	0.30 - 0.91	Clay, silty, dark gray
221.23 - 217.26	0.91 - 4.88	Sand, fine, silty, light tan
217.26 - 216.19	4.88 - 5.95	Sand, fine, silty, wet

* MSL = Mean sea level.

Water table elevation above MSL = 217.14 m

TABLE C-6. LOG OF BORING 6 AT SITE M

Elevation above MSL* (m)	Depth (m)	Description
221.17 - 220.87	0.00 - 0.30	Topsoil
220.87 - 219.34	0.30 - 1.83	Clay, silty, dark to light brown
219.34 - 215.32	1.83 - 5.85	Sand, fine, silty, light tan

* MSL = Mean sea level.

Water table elevation above MSL = 216.17 m

TABLE C-7. LOG OF BORING 7 AT SITE M

Elevation above MSL* (m)	Depth (m)	Description
221.49 - 221.19	0.00 - 0.30	Topsoil
221.19 - 218.14	0.30 - 3.35	Clay, silty, dark gray
218.14 - 216.61	3.35 - 4.88	Clay, silty, dark gray, soft
216.61 - 215.33	4.88 - 6.16	Sand, fine, silty, wet, gray

* MSL = Mean sea level.

Water table elevation above MSL = 216.25 m

TABLE C-8. LIST OF SAMPLES EXAMINED FROM SITE M

Boring	Elevation of top of hole (m)	Elevation of water table (m)	Total depth (m)	Thickness of cover (m)	Thickness of fill (m)	Elevation sludge/soil interface (m)	Sampled depth interval (m) From	To	Elevation of sampled intervals (m) From	To	Type of sample	Sample number
1	224.68	216.82	8.54	0	3.80	220.88	1.52	2.13	223.16	222.55	Chemical	1C1
							3.79	3.95	220.89	220.73	Chemical	1C2
							4.01	4.27	220.67	220.41	Physical	1P1
							5.64	5.79	219.04	218.89	Chemical	1C3
							5.85	6.25	218.83	218.43	Physical	1P2
							6.86	7.04	217.82	217.64	Chemical	1C4
							7.10	7.62	217.52	217.06	Physical	1P3
2	222.18	217.82	5.79	NA	NA	NA	0.30	0.49	221.88	221.69	Chemical	2C1
							0.55	1.07	221.63	221.11	Physical	2P1
							1.22	1.40	220.96	220.78	Chemical	2C2
							3.35	3.57	218.83	218.61	Chemical	2C3
							4.11	4.30	218.07	217.88	Chemical	2C4
							4.36	4.82	217.82	217.36	Physical	2P4
3	222.10	217.10	5.79	NA	NA	NA	0.30	0.49	221.80	221.61	Chemical	3C1
							0.55	1.01	221.55	221.09	Physical	3P1
							1.22	1.40	220.88	220.70	Chemical	3C2
							1.46	1.77	220.64	220.33	Physical	3P2
							3.35	3.57	218.75	218.53	Chemical	3C3
							3.63	4.08	218.47	218.02	Physical	3P3
							4.11	4.30	217.99	217.80	Chemical	3C4
							4.36	4.88	217.74	217.22	Physical	3P4
4	224.36	216.77	9.15	0	2.29	222.07	2.29	2.47	222.07	221.89	Chemical	4C1
							2.53	3.05	221.83	221.31	Physical	4P1
							3.20	3.38	221.16	220.98	Chemical	4C2
							3.44	3.87	220.92	220.49	Physical	4P2

(continued)

TABLE C-8 (CONTINUED)

Boring	Elevation of top of hole (m)	Elevation of water table (m)	Total depth (m)	Thickness of cover (m)	Thickness of fill (m)	Elevation sludge/soil interface (m)	Sampled depth interval (m) From	To	Elevation of sampled intervals (m) From	To	Type of sample	Sample number
							5.33	5.52	219.03	218.84	Chemical	4C3
							5.58	5.88	218.78	218.48	Physical	4P3
							6.86	7.04	217.50	217.32	Chemical	4C4
							7.62	7.83	216.74	216.53	Chemical	4C5
							7.86	8.38	216.50	215.98	Physical	4P5
5	222.14	217.14	5.95	NA	NA	NA	4.88	5.06	217.26	217.08	Chemical	5C4
							5.12	5.64	217.02	216.50	Physical	5P4
6	221.17	216.17	5.85	NA	NA	NA	.30	0.49	220.87	220.68	Chemical	6C1
							0.55	1.07	220.62	220.10	Physical	6P1
							1.46	1.92	219.71	219.25	Physical	6P2
							3.57	3.96	217.60	217.21	Physical	6P3
							4.88	5.18	216.29	215.99	Chemical	6C4
7	221.49	216.25	6.16	NA	NA	NA	0.30	0.49	221.19	221.00	Chemical	7C1
							0.55	0.76	220.94	220.73	Physical	7P1
							1.22	1.40	220.27	220.09	Chemical	7C2
							1.46	1.98	220.03	219.51	Physical	7P2
							3.35	3.57	218.14	217.92	Chemical	7C3
							3.60	4.11	217.89	217.38	Physical	7P3
							4.88	5.09	216.61	216.40	Chemical	7C4
							5.12	5.64	216.37	215.85	Physical	7P4

NA = Not applicable.

Note: All elevations are given with respect to mean sea level.

Effects of Coal-ash Leachate on Ground Water Quality

I/A

Interagency Energy/Environment R&D Program Report

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EPA REVIEW NOTICE

Effects of Coal-ash Leachate
on Ground Water Quality

by

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Prepared for

U.S. ENVIRONMENTAL PROTECTION AGENCY
Office of Research and Development
Washington, DC 20460

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ABSTRACT

The four objectives of this research are: (1) to develop a methodology for the field collection of coal-ash leachate; (2) chemically characterize ash leachates from fossil-fueled power plants using different coal sources; (3) determine the characteristics of the hydrogeochemical environment in which the leachate occurs; and (4) determine the attenuation of coal-ash leachate by various soil types.

Groundwater monitoring wells were installed around the ash ponds at two TVA coal-fired steam plants. Continuous soil-core samples were collected and analyzed for physical and chemical parameters. .Groundwater samples were collected and analyzed periodically. Ash leachate was percolated through different clays and soil types in the laboratory to study attenuation rates. Results indicate that:

1. Coal-ash leachate is a highly variable solution, but characteristically is high in dissolved solids, boron, iron, calcium, aluminum, and sulfate. Ash leachate can be acidic, with pH values as low as 2 measured. Ash leachate is a chemically reducing solution.

2. The different coal sources associated with this study produced ash leachate with similar characteristics.

3. The use of an inert gas lift pump proved an effective means of collecting anoxic groundwater samples while minimizing oxidation.

4. Differences were found in the characteristics of leachate samples obtained by extracting the interstitial soil water and samples collected from monitoring wells. Interstitial water samples contained higher concentrations of metals and were more acidic than well samples.

5. The flux of metals from coal-ash leachate was found negligible when compared to the mass of metals discharged by the ash pond surface overflow (even though concentrations in the ground water were highest) because the surface discharge was much greater than the groundwater flow.

CONTENTS

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ACKNOWLEDGMENTS

SECTION 1

INTRODUCTION

The increasing use of coal for power generation will result in an increasing potential for adverse environmental impacts. Realizing the importance of knowing what impacts power generating facilities may have on the environment and anticipating the forthcoming Federal regulations applying to solid waste disposal (Resource Conservation and Recovery Act, P.L. 94-580), the Tennessee Valley Authority (TVA) in conjunction with the Environmental Protection Agency (EPA) initiated a study to characterize the various effluents associated with coal-fired generating facilities. As a part of that study, entitled "Characterization of Effluents from Coal-Fired Utility Boilers," the impact of coal-ash leachate on groundwater quality at two TVA fossil-fueled power plants was investigated.

TVA's coal-fired generating system produces approximately 650 tons of coal ash for every 1000 megawatts generated. During 1977, TVA's 12 coal-fired power plants produced approximately 6.7 million tons of ash. The ash consists of various proportions of fly ash and bottom ash depending on the methods of firing and ash collection systems used at the plants. Once collected, the ash is sluiced with raw river water to nearby settling ponds. After settling, overflow from the settling pond is discharged to adjacent receiving streams in compliance with the EPA National Pollutant Discharge Elimination System permit.

Ash leachate is generated by the infiltration of ponded sluicing water into the settled ash, and its subsequent percolation through the ash where it eventually acquires the characteristics of ash leachate. The dry disposal of coal-ash, or ash ponds that are no longer inundated, such as a filled pond, can generate ash leachate with water from direct precipitation and/or rainfall runoff. Once the leachate is generated and has entered the subsurface environment below the deposited ash, its chemical characteristics can be affected by various attenuation phenomena. This report presents the results of a field and laboratory project performed to characterize coal-ash leachate and its attenuation by selected soil types at two TVA power plants. The major objectives of the project were to: (1) develop a methodology for the field collection of coal-ash leachate, (2) chemically characterize ash leachate from fossil-fueled power plants using different coal sources, (3) determine the characteristics of the hydrogeochemical environment in which the leachate occurs, and (4) determine the attenuation of coal-ash leachate by various soil types.

SECTION 2

CONCLUSIONS AND RECOMMENDATIONS

Results of the ash pond leachate investigation indicate:

1. Coal-ash leachate is a highly variable solution, but characteristically is high in dissolved solids, boron, iron, calcium, aluminum, and sulfate.

2. Several constituents in ash leachate were found to exceed EPA's criteria for drinking water. These include cadmium, chromium, iron, manganese, and lead. Dissolved solids and pH also did not meet these criterion.

3. Ash leachate can be acidic, with pH values as low as 2.0 measured. This acidity was found at two steam plants even though one generated an alkaline ash.

4. Ash leachate is a chemically reduced solution.

5. The different coal sources associated with this study produced ash leachate with similar characteristics. However, the acidic ash at one plant produced higher concentrations of metals in the leachate than the alkaline ash at another plant.

6. The use of an inert gas lift pump provides a means of collecting anoxic groundwater samples while minimizing oxidation.

7. Differences found in the characteristics of leachate samples obtained by extracting the interstitial soil water and those of samples from monitoring wells include: (a) interstitial water samples were more acidic than well samples, (b) interstitial water samples contained higher concentrations of metals than well samples and, (c) the collection of interstitial water samples by compressing soil samples is a time consuming and costly technique relative to the collection of water samples via monitoring wells. Further investigation into groundwater and leachate sampling techniques and their effects on sample integrity is needed to ensure accurate evaluations of leachate impacts.

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8. Coal-ash leachate migration and/or attenuation in subsoils cannot be accurately determined by analyzing total soil samples for leachate constituents. Fractionization and analysis of specific particle sizes may be necessary to reduce the large variation associated with total soil sample analysis.

9. The mass of metals in the ash pond leachate entering adjacent surface water was found negligible when compared to the mass discharged by the ash pond surface overflow (even though concentrations in the ground water were higher) because surface flows were much larger.

10. Soils containing a large percentage of clay provide a better medium for attenuating metals from ash leachate than soils with more sand.

11. The anoxic leachate attenuation system, developed during this investigation, is a viable approach for studying leachate attenuation by soils under anoxic conditions.

12. Further work on the speciation chemistry of coal-ash leachate needs to be performed to determine if toxic metal species are present.

SECTION 3

LITERATURE REVIEW

The solubilization of ions from coal ash during the percolation of
water through ash disposal areas and their potential for groundwater
contamination, have long been recognized. In a study conducted during
1951 and 1952 by Merz and Snead,[1] the leaching of soluble salts and
alkaline compounds from incinerator ash dumps was investigated. Their
studies indicated that salts and alkaline compounds would leach from ash
during the percolation of water from direct precipitation and/or ground-
water movement through the ash; however, the leaching rate would be very
low. The study also found that chlorides, nitrates, and sulfates were
the most readily leached anions, and sodium and potassium were the most
readily leached cations. Calcium and magnesium were found to leach, but
very slowly. A limited number of leachate samples were also analyzed
for copper, aluminum, lithium, manganese, vanadium, barium, boron, and
strontium. Only boron and strontium were found in quantities above one
milligram per liter.

In a much later report, Rohrman[2] discussed a comprehensive ash
study which involved 12 fossil-fueled power plants. One of the major
findings of this study was that all the ponded ash sluice waters con-
tained boron and phosphorus, and data was presented showing that many of
the elements in coal ash were present in the oxide form. In one of the
first studies to determine the major chemical elements of coal-ash
leachate, O'Connor, et al.,[3] mixed 50 grams of fly ash of different ages
with 500 milliliters of demineralized water and analyzed the liquid for
extracted solutes. This study indicated that the coal-ash extract could
be characterized as an alkaline solution of calcium sulfate. The pH of
the extracts ranged from 7.5 to 11.1, calcium ranged from 27 to 288
mg/l, and sulfate from 45 to 600 mg/l. An additional finding was that
the pH of the extract decreased with increasing age of the ash, but a
reason for this is unknown. As a further part of this study, coal ash
was placed into columns and subjected to repeated elutions with demin-
eralized water. In this experiment, the highest concentrations of
solids were observed during the first elutions and most of the readily
soluble constituents were eluted with the first three liters of elutant.
Calcium and sulfate were found to be the most abundant ions in all the
elutants. Between pH 10.0 and 11.5, calcium was equal to the total
hardness. At a pH below 10.0, calcium accounted for approximately half
the hardness, indicating the solubilization of other hardness-producing
ions.

In a study by Weeter, et al.,[4] 500 grams each of fly ash and bottom ash were mixed with two liters of distilled water for 48 hours. The liquid of this mixture was then analyzed for the extracted solutes. The results of their analysis is presented in Table 1. The calcium and sulfate concentrations in fly ash samples ranged from 400 to 600 mg/l and from 1300 to 2000 mg/l, respectively, somewhat higher concentrations than those found by O'Connor, et al. However, in the O'Connor study a smaller concentration of ash was used in the mixing experiment. The concentration of solutes in a water-ash mixture has been shown by Weeter to be dependent on ash concentration. Batch mixing studies performed by Weeter indicated that the supernatant concentrations of sulfate, alkalinity, calcium, and iron increase with increasing fly ash loading, and steady-state concentrations for these constituents were found to occur within a mixing time of one hour. This indicates that the most readily soluble constituents in coal ash are loosely bound to sites on the surface of the ash particle.

TABLE 1. CHEMICAL CHARACTERISTICS OF COAL-ASH SHAKER TEST SUPERNATANT[a]
(after Weeter, et al.)

Parameter	Fly ash[b]	Bottom ash[b]
Iron	0.1-1.8	0.05-0.15
Potassium	33-112	0.4-6.6
Calcium	400-600	8-135
Magnesium	1-19	0.8-7.1
Titanium	Trace	0.1
Arsenic	0.01	0.01-0.8
Boron	3-10	0.1-0.2
Aluminum	1.5-6.8	0.05-0.5
Sodium	15-90	0.8-7.8
Sulfate	1300-2000	12-60
Phosphate	0.1-0.6	0.1-0.5
Silica	3-40	1-2

[a]500 grams of ash with 2 liters of distilled water; shaken for 48 hours.
[b]Values in mg/l.

In batch shake tests similar to those performed by Weeter, Theis and Wirth[5] mixed various weights of fly ash that were collected from different plants with one liter of distilled water and determined the equilibrium pH of the supernatant. Their experiments indicated that equilibrium pH levels were achieved with an ash concentration of 1 to 2 grams per liter. They also discovered that some ashes induced alkaline conditions in the supernatant, while others produced acidic conditions. An analysis of the supernatant for trace metals showed that the highest concentrations were present in the acidic ash mixture, indicating increased solubilization as would be expected with the depressed pH.

In addition to changes in the pH and metals concentrations, changes in the dissolved oxygen concentration resulting from the addition of fly ash, have been found. In Figures 1 and 2, from Theis,[6] it can readily be seen that after mixing fly ash with water the original dissolved oxygen concentration is depressed, and the pH either increased or decreased. The decreased dissolved oxygen concentration may be a result of oxygen depletion by a sulfite ion oxidation reaction, and may be expressed by:

$$SO_3^{-2} + 1/2 \ O_2 \rightarrow SO_4^{-2}$$
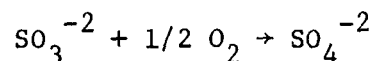
The rate of this reaction is normally very slow:[7] however, the catalytic effects of metal ions in this reaction are well known, and the reaction rate may also be affected by certain organic compounds. Increases in pH are likely a result of the solubilization of calcium oxide (lime) present on the surface of the ash particles. Upon mixing with water, calcium oxide becomes hydrated as follows:

$$CaO + H_2O \rightarrow Ca(OH)_2$$
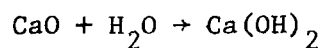
In distilled water partial dissociation of the hydrated lime would cause an increase in pH. In an actual coal ash disposal area, water mixed with ash would contain alkalinity and hardness. Under these conditions, an increase in pH would result from the establishment of the following equilibriums:

$$CaO + H_2O \rightarrow Ca(OH)_2 \quad \text{hydrated lime formation} \tag{1}$$

$$Ca(HCO_3)_2 \rightarrow Ca^{+2} + 2HCO_3^{-} \tag{2}$$

$$HCO_3^{-} \rightleftarrows CO_3^{-2} + H^{+} \tag{3}$$

$$CO_3^{-2} + H_2O \rightleftarrows HCO_3^{-} + OH^{-} \tag{4}$$
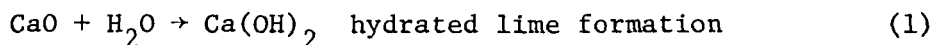
Equations 3 and 4 effectuate the increase in pH by removing hydrogen ions in equation 3 and increasing hydroxide ions in equation 4. These reactions, along with others, were utilized by Chu[8] in a study to demonstrate the efficacy of lime-soda ash softening of ash pond water for closed-cycle water reuse in power plants.
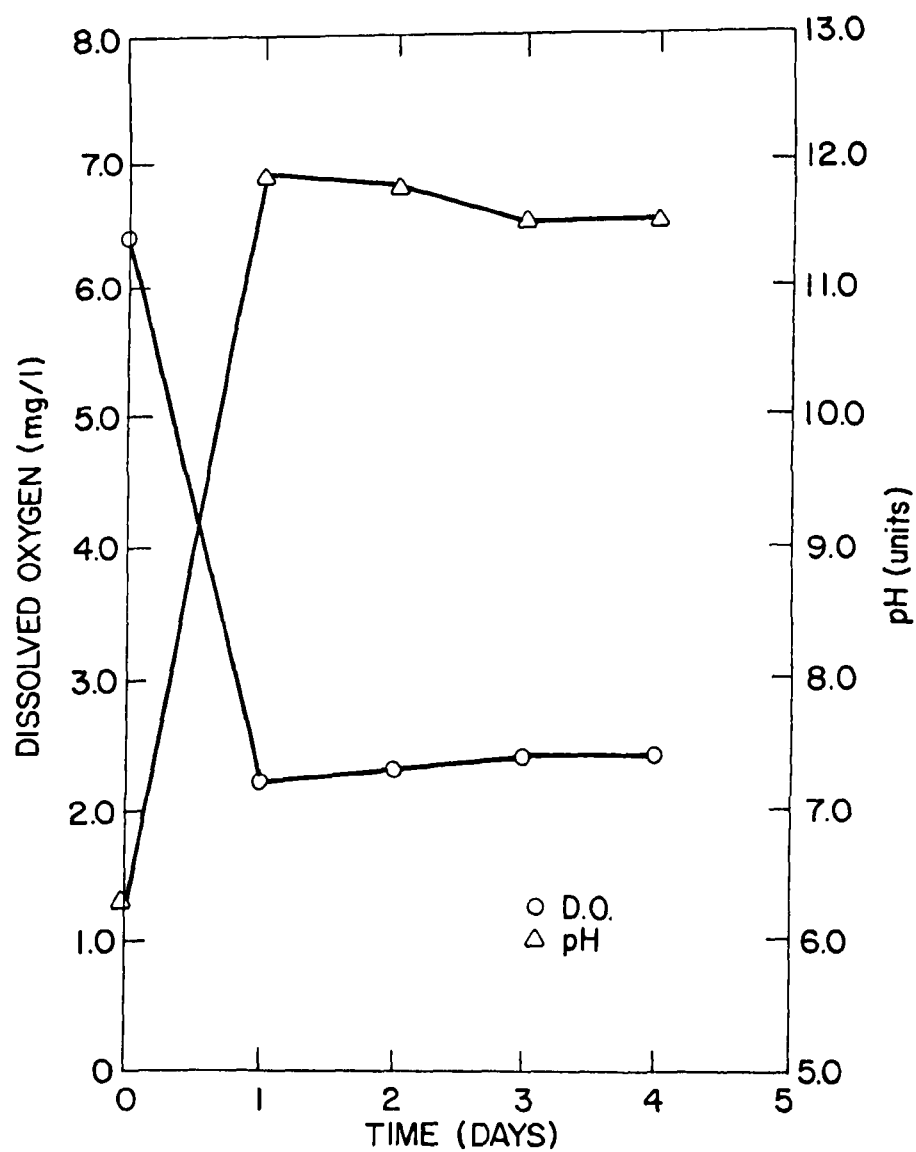
Figure 1. Effects of fly ash with lime on pH
and dissolved oxygen of distilled
water (after Theis).
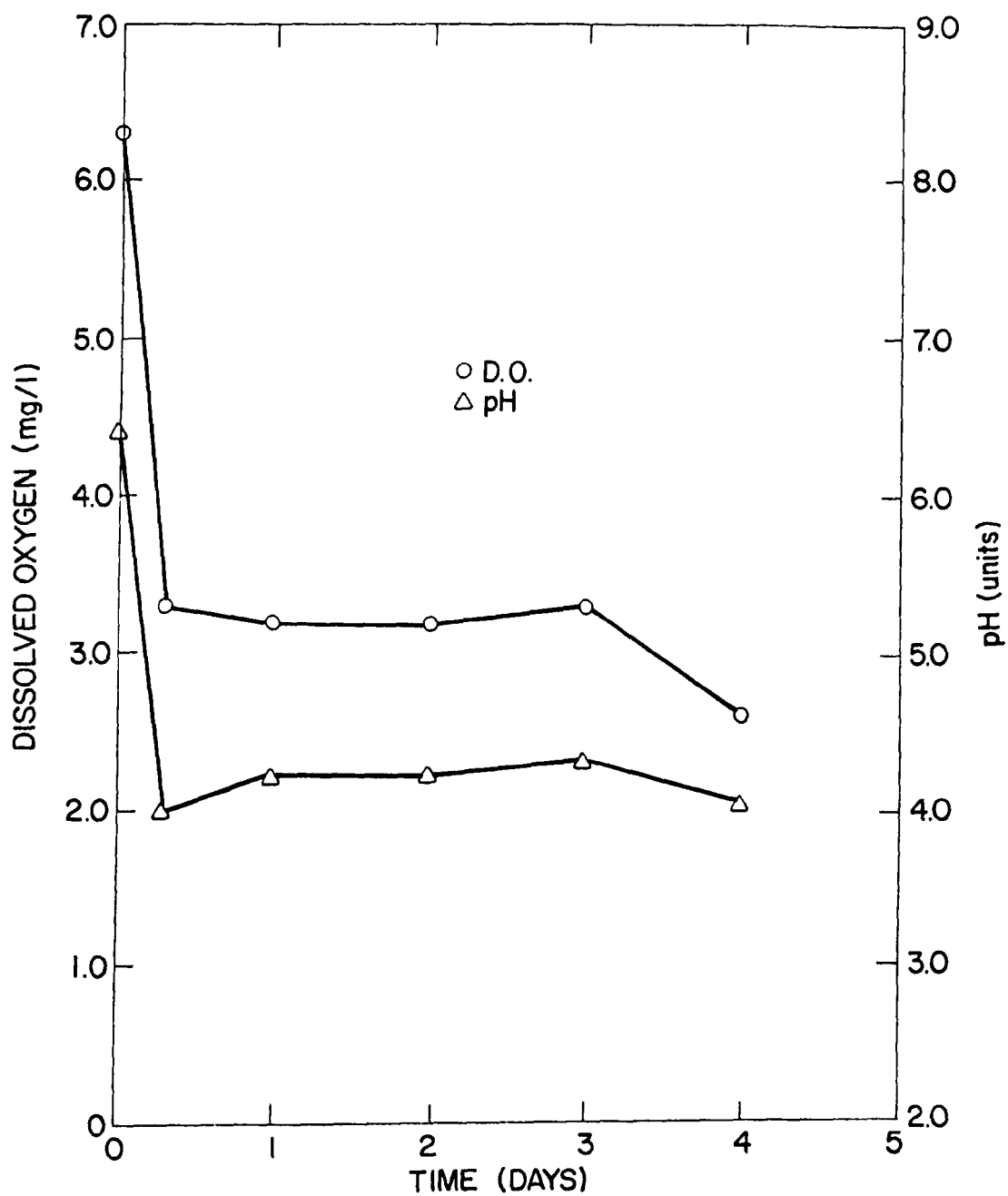
Figure 2. Effects of fly ash with lime on pH and dissolved
oxygen of distilled water (after Theis).

Certain ashes may contain high concentrations of transition metal oxides, concurrent with low quantities of lime. Under these circumstances the acidic character of the transition metals, especially iron, can cause decreases in solution pH. In addition, salts of metal ions show varying degrees of acidity when dissolved in water; hydrolysis of these ions effectuates acidic conditions. For example:

$$Fe(H_2O)_5 \, OH^{+2} + H_2O \rightleftharpoons Fe(H_2O)_4(OH)_2^+ + H_3O^+ \tag{7}$$
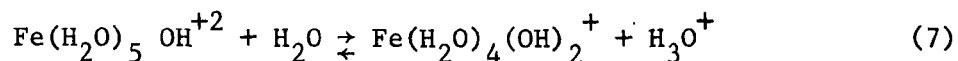
Decreased solution pH could present a potential water quality problem if a portion of the metals associated with coal ash were present on the surface of the ash particle where they could be easily solubilized. The sorption properties of fly ash were also investigated by Theis, and his findings indicated that surface coatings of amorphous iron, manganese, or aluminum oxides could provide a sorptive medium for trace metals.

Reed, et al.,[9] performed mixing studies using various concentrations of fly ash. From these studies he concluded that equilibrium concentrations were established within 20 minutes of contact. He also noted that the equilibrium concentrations achieved under his batch study were considerably lower than concentrations found by Burnett,[10] during a column percolation study in which fly ash from the same location was used.

In another column study performed by Brown,[11] distilled water was percolated through plastic columns filled with coal ash. Successive volumes of the percolate were then analyzed for various elements. The concentrations found in the percolate by Brown were considerably higher than concentrations found by previous investigators using batch mixing techniques. Column percolation studies tend to allow greater contact time and higher ash-to-water ratios than batch tests. These investigations indicate that column studies may be a more accurate method of predicting actual coal-ash leachate quality.

Harriger, et al.,[12] in one of the few recent studies to characterize coal-ash leachate by analyzing groundwater samples, reported highly variable concentrations of calcium, sulfate, alkalinity, iron, and magnesium in groundwater samples collected in an ash disposal area, and much higher values overall as compared to samples collected away from the ash disposal area. Some of their values are presented in Table 2 for comparison with values obtained in laboratory studies using batch and column techniques.

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TABLE 2. COMPARISON OF COAL ASH LEACHATE OBTAINED BY DIFFERENT TECHNIQUES

Parameter	Mixed batch extraction[a]	Column elution extraction[b]	Leachate well sample[c]
Calcium, mg/l	20-280	490	91-660
Sulfate, mg/l	18-740	39,000	345-4000
Alkalinity, mg/l as $CaCO_3$	42-190	390	56,000-985,500
Iron, mg/l	0.03-0.1	0.1	0.04-10.4
Magnesium, mg/l	1-19	80	2.6-16.8
Arsenic, mg/l	0.01	0.13	0.01-0.19

[a]Weeter (see references).

[b]Brown (see references).

[c]Harriger (see references).

SECTION 4

DESCRIPTION OF ASH DISPOSAL AREAS

This section describes the coal-ash disposal areas at plants
J and L, and provides some detailed information regarding coal use and
ash generation. In addition, the geological stratigraphy beneath both
disposal areas is described.

PLANT J

Plant J is a coal-fired power generating facility located on a
peninsula formed by two reaches of the Clinch and Emory Rivers in eastern
Tennessee. The facility was put into full operation in December 1955,
and has a rated power generating capacity of 1700 megawatts. The plant
utilizes coal produced in eastern Tennessee and eastern Kentucky which
has an average of 2.0 percent sulfur and 19 percent ash. During 1972 the
plant consumed 3.9 million metric tons of coal, and generated 711,682
metric tons of ash. The ash consisted of 560,002 metric tons of fly ash,
and 151,680 metric tons of furnace bottom ash. The fly ash is collected
by mechanical collectors and electrostatic precipitators installed in
series, with an overall efficiency of 98 percent. The bottom ash is
collected in hoppers located at the bottom of the furnace. The ashes
required 30,112 million liters of raw river water to sluice it from the
collection systems at the plant to a nearby ash settling pond for disposal.

The original coal-ash disposal area at plant J (ash pond A) con-
sisted of a 242,800 m^2 settling pond. This pond was completely filled
with ash by 1972 and is no longer in use. The current disposal pond has
an area of approximately 594,900 m.^{2} The pond is situated adjacent to
the Emory River (see Figure 3) with the pond retainer dike separating the
pond and river. The pond overflow is discharged over a weir into a small
embayment of Watts Bar Reservoir.

The thickness of the ash in plant J's disposal area ranges from
0.5 meters near the pond's overflow weir to 14 meters in the original
242,800 m^2 pond. The whole disposal area is underlain by (1) a clay-silt
stratum immediately below the ash ranging in thickness from 2.4 to 6.1
meters, (2) a deeper alluvial sand stratum ranging in thickness from 1.5
to 3.6 meters, and (3) low permeability shale.[13] The thickness of the
shale is not accurately known.
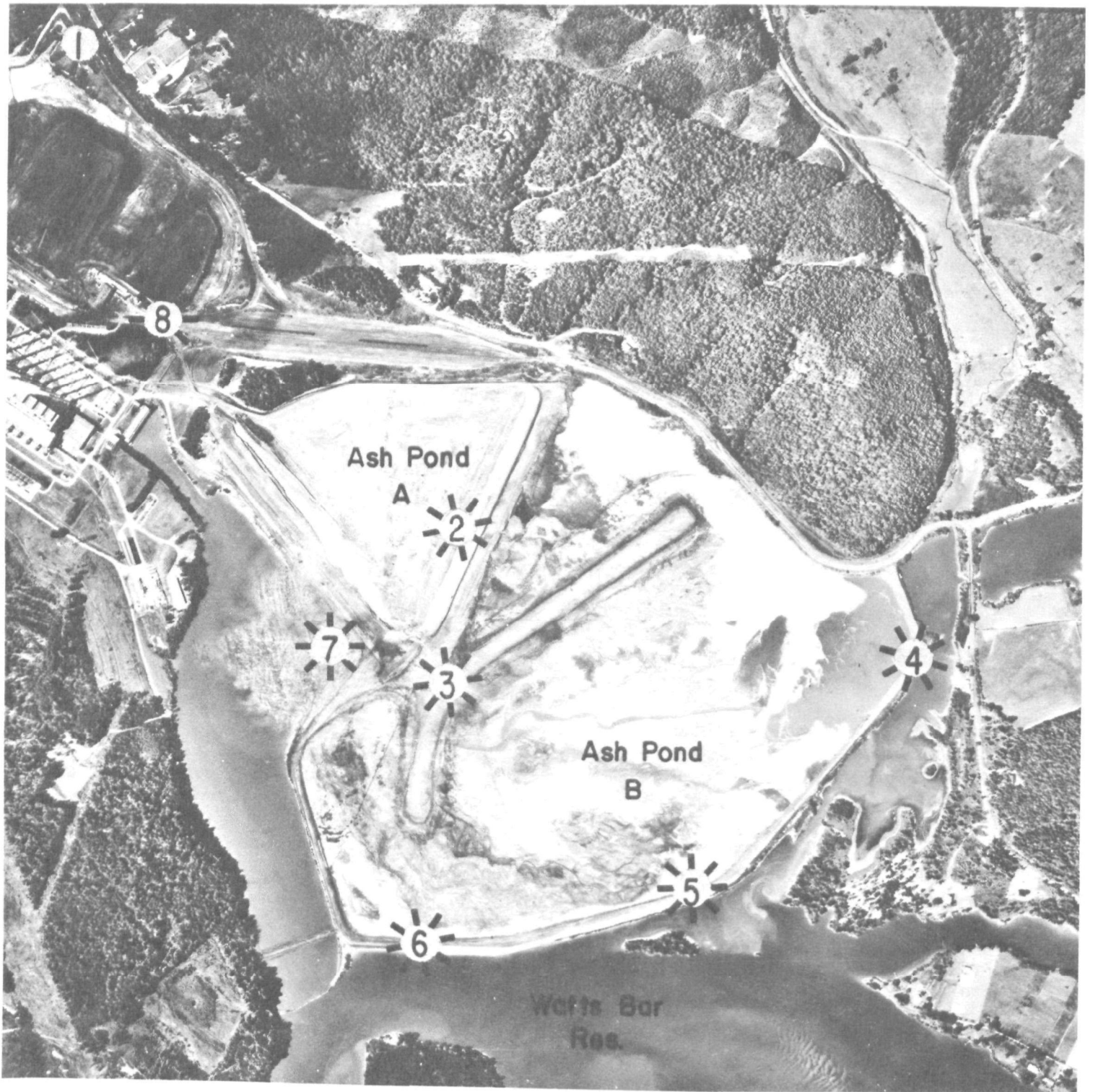
Figure 3. Plant J – ground water
sampling well locations.

The ash pond dike itself is composed of a variety of materials with varying compositions, mostly silty clay, but deposits of ash are present. The height of the dike from the original ground surface to the top of the dike was approximately 4.5 meters.

PLANT L

Plant L began commercial operation in 1952, with three coal-fired units generating about 420 megawatts. By 1975, five additional units had been added bringing the total full load generating capacity of the plant to about 1965 megawatts. The facility is located on the Tennessee River in northeastern Alabama. The plant burns coal obtained from areas in western Kentucky and north Alabama which averages about 2.8 percent sulfur and 16 percent ash. During 1972, the plant consumed 3.3 million metric tons of coal and produced 432,361 metric tons of ash consisting of 107,228 metric tons of bottom ash and 325,133 metric tons of fly ash. The fly ash is collected by electrostatic precipitators or mechanical collectors and the bottom ash is collected in hoppers located at the bottom of each furnace. The volume of raw river water required to sluice the ash from the collection systems to nearby ash settling ponds during 1972 was about 27,725 million liters.

Plant L originally had two principal disposal areas (see Figure 4), ponds 1 and 2. Pond 1 is no longer active or inundated and has begun to revegetate. Currently, all the ash discharged from the plant is received by pond 2. This pond is 95 percent full and acts merely as a conduit to transport the ash to pond 3, which is the latest ash pond constructed. Some settling occurs in pond 2, but it is mostly the very coarse, heavy material. Most of the ash is settled in pond 3. The total ash disposal area, including ponds 1, 2, and 3, is approximately 1,032,000 m^2.

The thickness of the ash in plant L's ash disposal area ranges up to 11 meters in depth. The greatest ash depths occur in the older ponds 1 and 2. Underlying the ash disposal areas, there are three geological formations present:[14] (1) recent deposits of river alluvium, (2) older river terrace deposits, and (3) limestone.

The material overlying the limestone bedrock in the disposal area was deposited by the Tennessee River. There is no clear-cut delineation between the alluvium and the terrace deposits. Both are composed mostly of clay and silt with some sand and gravel mixtures. In the ash disposal area, these deposits vary in thickness from 3.8 to 9.0 meters. The underlying bedrock in the area is Ordovician age limestone several hundred feet thick. Its composition is mainly shaly limestone with interbeds of purer limestone and zones of varicolored siltstone or argillite. There are numerous cavities in the limestone resulting from its dissolution by ground water.

Pond No. 3's dike material is composed of a highly compacted clay of low permeability. The thickness of the dike from its interface with the original ground surface to the top of the dike is approximately 10 meters.

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TENNESSEE RIVER

SO$_2$ SCRUBBER SLUDGE POND

ASH POND EFFLUENT

ASH POND # 2
(ACTIVE)

ASH POND # 1
(INACTIVE)

ASH POND # 3
(ACTIVE)

L1

L2

L3

L4

L5

L6

L7

L8

L9

L11

IDOWS CREEK EMBAYMENT

L10
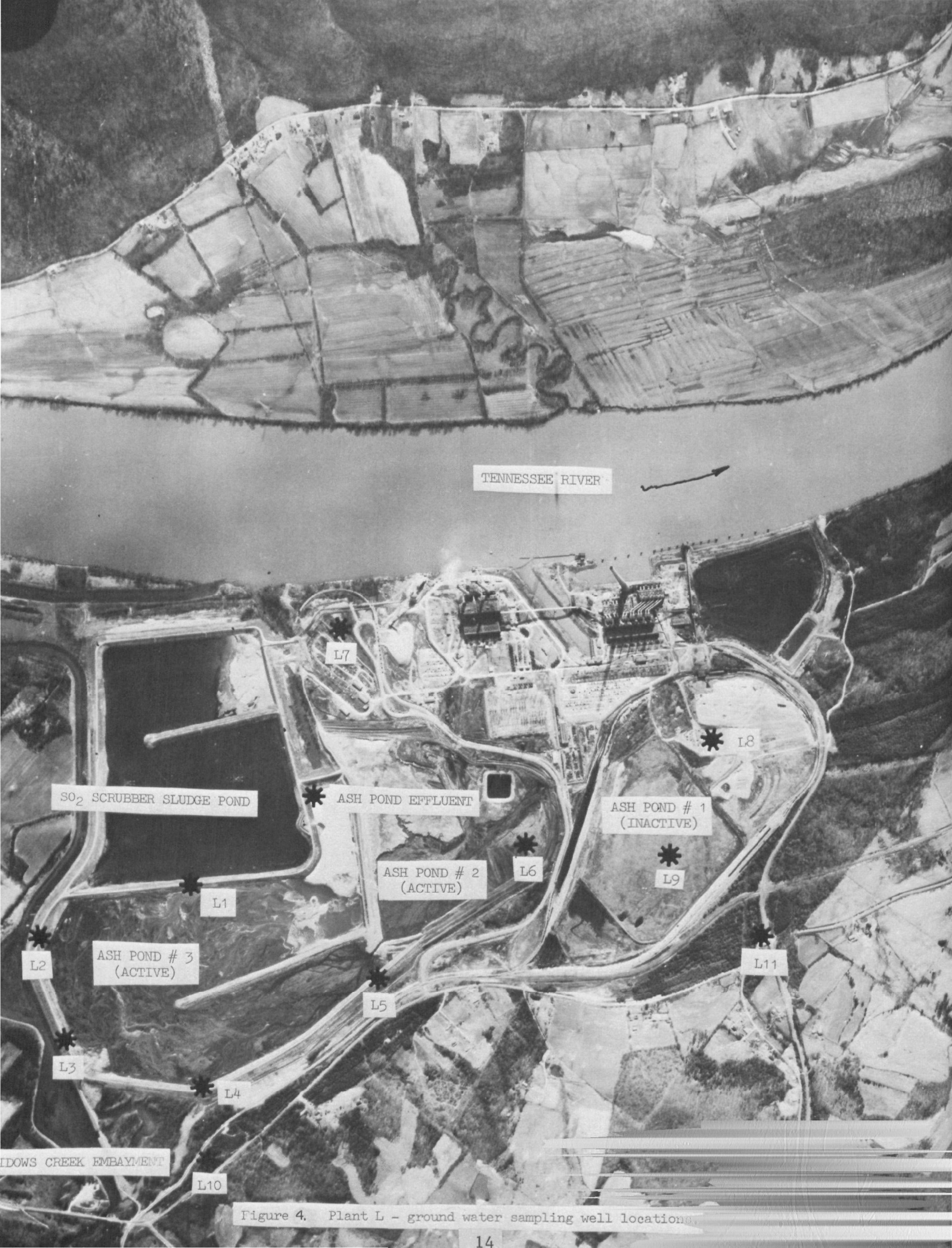
Figure 4.   Plant L - ground water sampling well locations

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SECTION 5

METHODOLOGY

This section describes the methods and procedures used to obtain data on ash leachates, groundwater contamination, leachate migration, and leachate attenuation by soils. The project involved field investigations at two power plants with different coal sources and subsurface soil conditions, and a laboratory simulation of leachate attenuation using leachate collected at one of the power plants.

FIELD INVESTIGATION

Plant J - Groundwater Sampling Well Design and Installation

During March and April 1976, groundwater sampling wells were installed at eight locations in and around the ash disposal pond at plant J (see Figure 3). Two of the eight sampling sites (J1 and J8) are located hydraulically upgradient from the ash disposal area and function as background locations. Site J2 was located in an older section of the disposal area, which is no longer inundated and is presently used as an equipment storage area. Site J3 was located in an area of the active disposal pond that had recently been filled and was stable enough to support drilling equipment. Sites J4, J5, J6, and J7 were located on the peripheral retainer dike downgradient from the active disposal area and were spaced in such a manner as to intercept any lateral flow of ash leachate from the ash disposal area.

To install a sampling well, a "split-spoon" soil sampler with a diameter of 5.1 cm was first used to obtain a soil sample and then a hollow-stem auger having a 30.5 cm outside diameter and powered by a hydraulic drill was used to drill the well hole. The soil samples were collected continuously by alternating the downward movement of soil sampler and auger until an impermeable substratum was encountered. After extracting the soil from the split-spoon sampling device, the outermost layers of the soil core were cut away, so as to eliminate any contamination from the sampling device, and a portion of the residual soil sample preserved in a plastic container for later laboratory analysis. The remainder of the residual soil sample was then
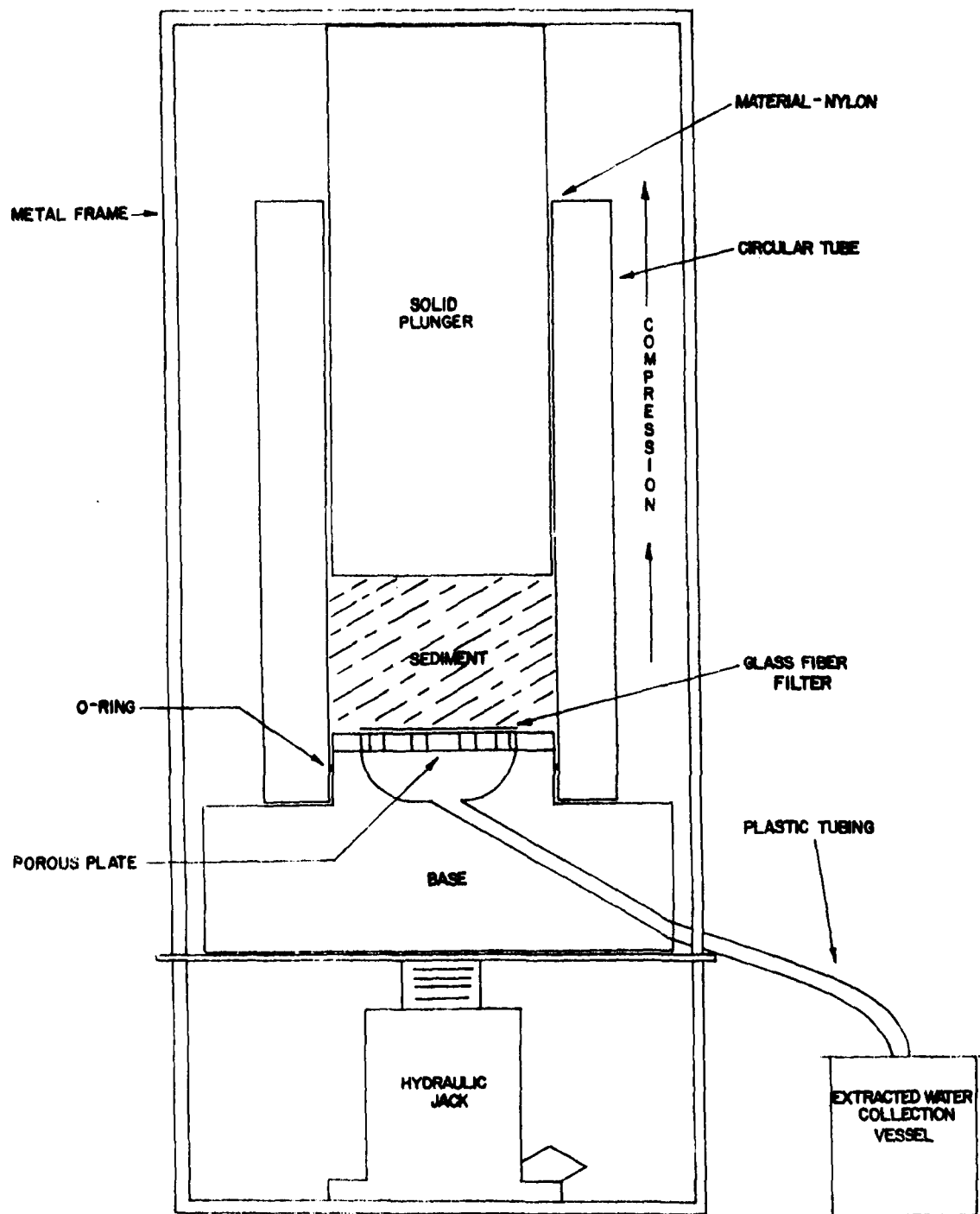
FIGURE 5. CROSS-SECTION OF
INTERSTITIAL WATER EXTRACTOR

Labels in figure:
- MATERIAL - NYLON
- CIRCULAR TUBE
- METAL FRAME
- SOLID PLUNGER
- COMPRESSION
- SEDIMENT
- GLASS FIBER FILTER
- O-RING
- POROUS PLATE
- BASE
- PLASTIC TUBING
- HYDRAULIC JACK
- EXTRACTED WATER COLLECTION VESSEL

hydraulically compressed with the nylon interstitial water extraction device illustrated in Figure 5. It was necessary to perform the extraction as soon as possible because any delay could alter the composition of the interstitial water, and no technique for preservation was available. A portion of the extracted interstitial water was then immediately analyzed for pH, conductivity, sulfate, alkalinity, and hardness. The remaining water was filtered through a 0.45 micron filter pad and acidified (HNO_3) for later laboratory analysis of metals.

The interstitial water extraction device was made of inert nylon. All parts were machined to fit with little or no water loss during operation. Actually, however, after many hours of use, extracted water began to leak out between the solid plunger and its sheath (the circular tube). This was a result of sand getting between the plunger and circular tube and the subsequent scoring of each during operation. A harder material, perhaps teflon, and more precise machine work would undoubtedly correct some of this problem. Operation of the extraction device was quite simple, but required a great deal of strength for optimum results. To operate the device, the solid plunger was removed from the circular tube and glass fiber filters (usually two) placed over the porous plate. A portion of subsoil sample collected within the zone of saturation was then placed inside the circular tube. The amount of subsoil compressed at one time varied depending on its composition. Clay material often required compression of several portions of a sample in order to obtain the needed volume of water, while a larger quantity of sand could be compressed with the same result. The amount of subsoil used per compression ranged from approximately 400 to 800 grams. After placing the subsoil in the circular tube, the solid plunger was inserted and the whole extraction device installed in the metal holding frame. A three-ton hydraulic jack was then used to force the base and plunger together, compressing the subsoil and forcing the water through the glass fiber filters to the collection vessel.

The volume of interstitial water obtained in this manner ranged from 6 to 30 milliliters per compression, or, by rough estimate, 4 to 15 percent of the subsoil's moisture content.

At all locations except J1 and J8, and at selected elevations (usually upon encountering a change in strata), undisturbed soil samples were collected for laboratory determinations of horizontal and vertical permeabilities, grain size classification, moisture content, and bulk density. (The results of these determinations are presented in Table 3, and the analytical methods in appendix A.) This was normally done by offsetting from the original sampling bore hole and drilling down to the predetermined sampling depth. The undisturbed samples were collected by hydraulically pushing a cylindrical tube (a Shelby tube) having a length of 76 cm and a diameter of 8.9 cm through the desired sampling area. The Shelby tube, with sample, was then extracted from the well hole and both ends sealed with paraffin wax to prevent moisture loss.

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TABLE 3. PERMEABILITY, DENSITY, GRAIN-SIZE DISTRIBUTION,[*] AND MOISTURE CONTENT
OF SUBSOILS COLLECTED FROM ASH DISPOSAL SITE - PLANT J

Sampling Location	Depth (m)	Vertical permeability (cm/s)	Horizontal permeability (cm/s)	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Density (g/cm^3)	Moisture (%)
J2	15.3	6.3×10^{-8}	7.4×10^{-8}	0	8	54	38	1.48	25.7
J3	14.1	1.3×10^{-6}	7.4×10^{-5}	3	25	63	9	1.42	24.6
J4	7.4	2.8×10^{-7}	6.6×10^{-8}	0	40	41	19	1.68	20.9
J4	9.7	3.1×10^{-6}	8.8×10^{-6}	0	80	14	6	1.60	22.6
J5	6.6	4.0×10^{-7}	2.8×10^{-7}	0	33	45	22	1.60	24.3
J6	7.4	4.4×10^{-7}	2.5×10^{-6}	0	29	51	20	1.48	26.7
J6	14.5	1.4×10^{-6}	1.3×10^{-6}	0	82	13	5	1.79	13.9
J7	12.9	6.1×10^{-6}	1.4×10^{-5}	0	69	24	7	1.45	20.7
J7	5.1	1.7×10^{-7}	2.6×10^{-7}	0	25	47	28	1.56	23.0

[*]Textural classification of soil fractions as per American Society for Testing and Materials designation,
where: clay <0.005 mm, silt 0.005-0.074 mm, sand 0.074-4.75 mm, and gravel >4.75 mm.

After the original split-spoon sampling was completed, the void that had been produced by the action of the hollow-stem auger was cleaned out and a groundwater monitoring well installed. Each well was constructed of schedule 80 (0.64 cm wall thickness) polyvinyl chloride (PVC) pipe that had an outside diameter of 11.4 cm. The bottom 45 cm of each PVC well was perforated with 0.95 cm drill holes to allow groundwater inflow. Each PVC well was lowered to the bottom of the well hole and the annular space created between the perforated PVC pipe and the side of the well hole filled with pea gravel to some approximate height above the perforations, usually about 0.6 meters; this served to filter the inflow of solids into the well and prevent clogging. A layer of fine sand (approximately 0.3 meters thick) was placed in the annular space above the pea gravel filter. This functioned as a barrier between the pea gravel filter and a bentonite slurry placed in the annular space above the sand. The bentonite slurry extended from the top of the fine sand upwards to the surface of the water table and served to prevent water from channeling down the side of the PVC pipe. The remaining annular space above the bentonite slurry was filled with soil extracted from the original hole. At the point where the well casing entered the ground, a 0.5 meter diameter concrete apron, sloping away from the well, was constructed. This was done to further inhibit water from channeling down the side of the casing.

Fourteen wells were installed at the eight locations at plant J. Multiple wells were installed at locations J2, J3, J4, J5, J6, and J7 to enable the sampling of leachate from different substrata. Table 4 gives the depth of each well, the water table elevation at time of installation, and the type of stratum sampled.

Groundwater Sample Collection

To ensure that contamination of monitoring well samples did not occur from lowering a sampling device into the wells, groundwater samples were collected with a gas lift pump as illustrated in Figure 6. Cylinders of commercial argon or nitrogen gas containing less than 0.5 ppm oxygen were used during sample collection to minimize oxidation of the samples.

The gas cylinders were equipped with a pressure regulator and hose. With the hose attached to the Shrader valve at the well head, the regulator was adjusted to deliver 4218 to 7031 kgs/sq meter (6 to 10 psi). The gas flowed down the quarter-inch flexible tubing and into the bottom of the half-inch rigid plastic pipe, and lifted the water in the pipe to the surface where it was collected. To avoid collecting a water sample that may have set in a well for several weeks (and no longer would be representative of the surrounding ground water), at least one water volume of each well was pumped out prior to collecting a sample for analysis. After collection, water samples were preserved and shipped to the laboratory for analysis (see section 5).
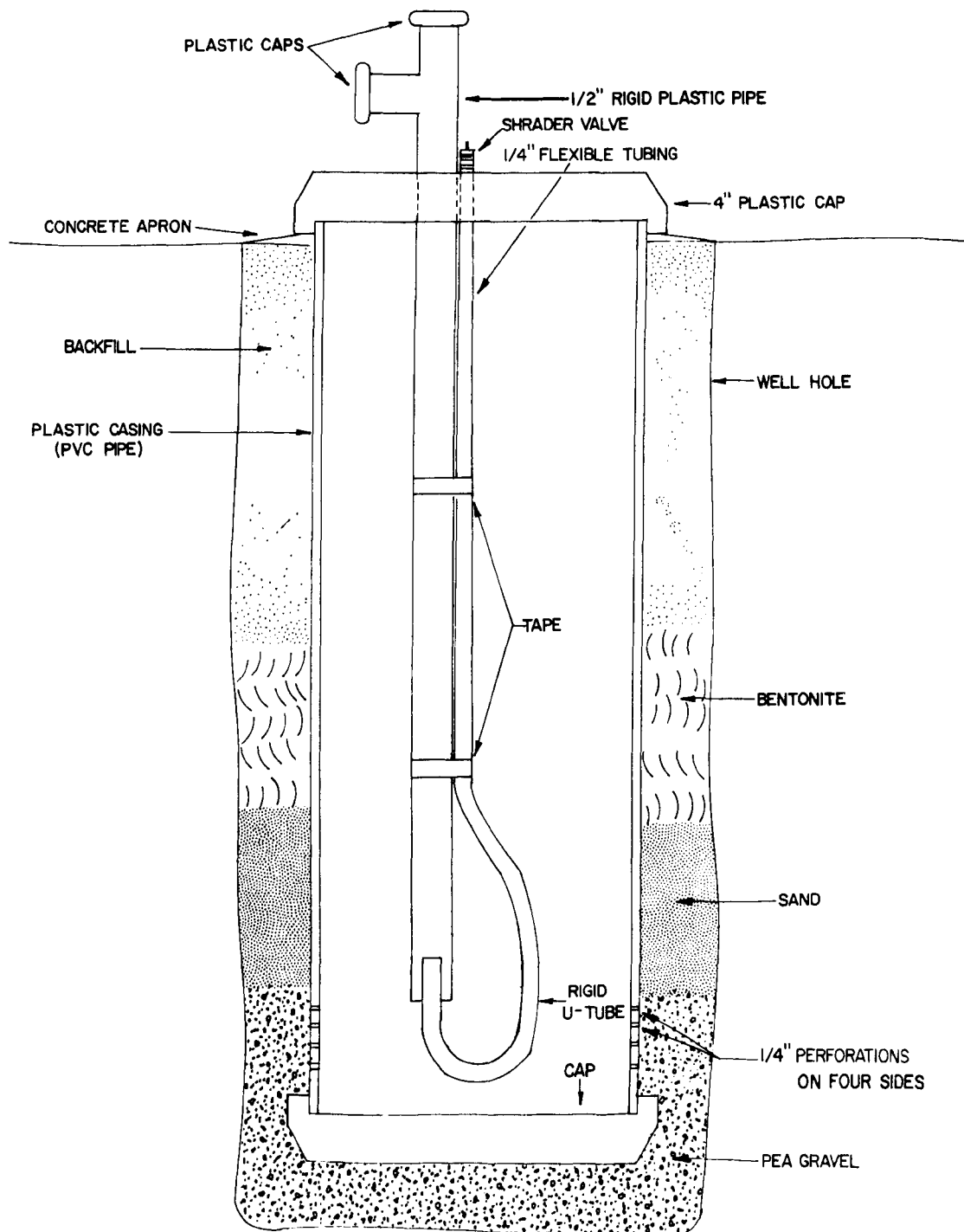
FIGURE 6. CROSS-SECTION OF
GROUNDWATER SAMPLING WELL

TABLE 4. GROUNDWATER SAMPLING WELL DEPTHS, WATER TABLE ELEVATIONS,
AND SUBSOIL STRATUM SAMPLED AT PLANT J

Well	Well depth (meters)	Elevation at well bottom[a]	Water table elevation[a]	Type of stratum sampled[b]
J1	8.5	224.8	231.28	Shale
J2	14.0	219.4	230.19	Silty clay
J3	12.8	216.7	227.75	Silty sand
J3A	8.2	221.4	227.81	Coal ash
J4	7.6	221.6	225.11	Sand
J4A	3.6	225.6	c	Clayey silt
J5	8.5	220.7	225.64	Sand
J5A	3.6	225.7	c	Clayey silt
J5B	8.2	221.1	c	Sand
J6	4.1	224.4	224.15	Sand
J6A	11.8	216.8	c	Clayey silt
J7	14.0	213.4	226.10	Clay
J7A	4.2	223.0	226.10	Sand
J8	7.5	226.6	230.90	Shale

[a]Elevation above mean sea level (meters).
[b]Monitoring well casing perforated within these strata.
[c]Not measured.

TABLE 5. COMPARISON OF DISSOLVED OXYGEN CONCENTRATIONS
IN GROUNDWATER SAMPLES COLLECTED BY PUMPING
WITH NITROGEN AND AIR AT PLANT J

Well	In situ[a] dissolved oxygen	Dissolved oxygen in sample	
		Pumped with nitrogen	Pumped with air
J2	0.5 mg/l	1.0 mg/l	20 mg/l
J3	0.2 mg/l	0.5 mg/l	8.3 mg/l
J3A	0.5 mg/l	0.2 mg/l	7.4 mg/l
J4	0.6 mg/l	0.2 mg/l	–
J4A	0.6 mg/l	0.7 mg/l	–

[a]In situ concentration after recovery from wasting one well water volume.

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The validity of using a gas, such as nitrogen, to collect ground-water samples is supported by the data in Table 5. These data indicate that less change will occur in the dissolved oxygen concentration in the ground water by using nitrogen gas to pump the wells rather than by using air, and that the difference is quite significant.

Plant L - Groundwater Sampling

Groundwater monitoring wells were installed at 11 locations in and around the coal-ash disposal area at plant L (Figure 4). Two of the eleven locations, L10 and L11, are located hydraulically upgradient from the ash disposal area and function as background locations. Sites L6, L8, and L9 are in successively older sections of the ash disposal area and for the most part are no longer inundated. Locations L8 and L9 have some vegetative cover, while L6 is barren and partially inundated. Location L7 is hydraulically downgradient from the disposal area and L1, L2, L3, L4, and L5, are situated along the active disposal area's peripheral retainer dike.

Eighteen groundwater monitoring wells were installed at plant L. Multiple wells were installed at seven locations in order to sample different substrata. The depth of each well, the water table elevation, and the type of substratum from which water samples were collected are presented in Table 6.

Plant L's monitoring well design, and procedures for their installation, are essentially the same as described for plant J. However, at plant L, several of the monitoring wells were constructed using schedule 40 PVC pipe (0.32 cm wall thickness) with an outer diameter of 8.8 cm (3-1/2 inches). This smaller pipe proved to be easier to handle and install with no observed loss in sampling performance. Split-spoon soil sample collections, interstitial water extractions, undisturbed Shelby tube samples, and groundwater sample collection procedures at plant L are the same as described for plant J. The results of the permeability, density, grain, size distribution, and moisture content determinations on the undisturbed soil samples collected at plant L are presented in Table 7.

LABORATORY ATTENUATION STUDIES

During the project design phase, several possibilities were considered for generating, or simulating, an ash leachate in the laboratory suitable for use in an attenuation study. However, after the groundwater sampling wells were installed and leachate samples analyzed, it became apparent that the most representative water to use, and the most easily obtainable, was the actual coal-ash leachate collected in situ. The coal-ash leachate used in the laboratory attenuation studies was collected from groundwater sampling well L6A, at plant L. This well is approximately 10.6 meters deep and

TABLE 6. GROUNDWATER SAMPLING WELL DEPTHS, WATER TABLE ELEVATIONS,
AND SUBSOIL STRATUM SAMPLES AT PLANT L

Well	Well depth (m)	Elevation at well bottom[a]	Water table elevation[a]	Type of stratum sampled
L1	10.67	179.72	183.22	Silt-clay
L1A	6.10	184.28	–	Silt-clay
L2	13.72	177.81	182.22	Silt-clay
L2A	6.10	185.76	186.50	Silt-clay
L3	13.41	178.13	181.51	Silt-clay
L3A	12.50	178.98	181.48	Silt-clay
L4	15.85	176.03	182.52	Silt-clay
L4A	6.10	185.44	182.60	Silt-clay
L5	15.24	175.89	183.03	Silt-clay
L5A	8.53	182.77	186.58	Silt-clay
L6	14.33	179.23	186.85	Silt-clay
L6A	10.67	182.54	186.81	Silt-clay
L7	8.84	175.04	182.10	Silt-clay
L8	11.89	179.51	187.55	Silt-clay
L9	8.53	183.13	189.08	Silt-clay
L9A	5.49	186.04	189.58	Silt-clay
L10	19.51	177.06	183.98	Silt-clay
L11	17.68	174.97	183.50	Silt-clay

[a]Elevation above mean sea level (m).

TABLE 7. PERMEABILITY, DENSITY, GRAIN-SIZE DISTRIBUTION,[*] AND MOISTURE CONTENT
OF SUBSOILS COLLECTED FROM ASH DISPOSAL SITE - PLANT L

Sampling location	Depth (m)	Vertical permeability (cm/s)	Horizontal permeability (cm/s)	Sand (%)	Silt (%)	Clay (%)	Density (g/cm^3)	Moisture (%)
L2	13.8	1.4×10^{-8}	0.2×10^{-8}	23	48	29	1.72	19.2
L2A	5.8	6.3×10^{-8}	1.7×10^{-7}	30	18	31	1.66	18.7
L3	13.4	6.7×10^{-8}	5.8×10^{-8}	15	31	50	1.03	56.1
L3A	6.6	6.9×10^{-9}	1.1×10^{-8}	27	24	36	1.77	17.5
L3A	11.0	2.7×10^{-8}	1.2×10^{-8}	10	15	68	1.43	31.6
L4	13.4	3.7×10^{-8}	5.5×10^{-8}	14	21	43	1.28	36.5
L4	13.9	4.4×10^{-8}	3.4×10^{-8}	4	54	42	1.23	38.0
L4A	10.1	1.9×10^{-8}	2.0×10^{-8}	23	28	41	1.72	18.0
L5	8.3	1.4×10^{-8}	1.7×10^{-8}	15	34	51	1.64	23.2
L5	9.0	4.3×10^{-8}	7.4×10^{-9}	29	26	40	1.60	22.0
L6	10.7	6.6×10^{-6}	5.6×10^{-6}	54	42	4	1.45	24.3
L6	12.7	1.2×10^{-8}	5.7×10^{-9}	13	31	56	1.62	24.2
L7	8.8	55.2×10^{-8}	45.0×10^{-8}	10	65	25	1.06	57.5
L8	3.0	5.6×10^{-8}	2.3×10^{-8}	14	41	45	1.57	24.6
L8	0.8	3.0×10^{-4}	1.5×10^{-4}	37	58	1	1.18	35.2
L8	5.1	1.9×10^{-8}	7.4×10^{-9}	1	22	77	1.44	32.7
L9A	5.9	2.0×10^{-8}	1.9×10^{-8}	9	36	55	1.55	26.5
L10	16.6	2.3×10^{-8}	4.7×10^{-8}	5	15	80	1.34	37.3
L11	5.9	6.3×10^{-8}	2.8×10^{-8}	30	28	42	1.64	22.9

[*]Textural classification of soil fractions as per American Society for Testing and Materials designation, where: clay <0.005 mm, silt 0.005-0.074 mm, sand 0.074-4.75 mm, and gravel >4.75 mm.

terminates at the interface of the ash and the original soil. Only the bottom 45 cm of the well is perforated to allow leachate inflow. The water collected at this well has thus percolated *downward* through approximately 10 meters of ash. Leachate from plant L was selected because it could be transported to the laboratory the quickest.

After deciding that field collection was the optimum method for obtaining a leachate sample, there were collection problems that needed to be surmounted.

Preliminary *in situ* measurements indicated that anoxic conditions existed in the leachate environment, and if these conditions were not maintained during collection, transportation, and storage, alterations in the chemical characteristics of the leachate would result (simply allowing the leachate to come in contact with air would cause precipitates to form). This problem was alleviated by using argon gas to pump the leachate to the well head. At the well head, tygon tubing was connected to the 1.27 cm pipe that expelled the sample, and the other end to a 20-liter plastic receiving carboy, which was closed to the atmosphere. After wasting one well volume, the collection apparatus was purged with argon gas and then connected to the well head for sample collection. By sampling in this manner it was possible to maintain anoxic conditions in the leachate. After sample collection was completed, an argon atmosphere was maintained over the leachate in the carboy during transportation to the laboratory. Argon gas was used in place of nitrogen in the attenuation study because it contained less oxygen (<0.1 mg/1). In addition, argon is heavier than nitrogen and air and tended to form a blanket over the leachate to aid in the maintenance of anoxic conditions.

In the laboratory, the carboy was placed into a controlled temperature compartment maintained at 20°C under an atmosphere of argon. Once temperature equilibrium was established (approximately two days), attenuation studies were performed with the following materials: (1) soil taken from the ash pond dike at plant L, (2) soil taken from the ash pond dike at plant J, and (3) kaolinite. Each of these materials was homogenized and mixed with equal volumes of silica and rehomogenized prior to packing into columns. It was necessary to mix these materials with silica so that flow through the columns could be sustained. The fine clay material tended to swell and clog upon wetting, inhibiting or completely stopping the flow of leachate. The silica increased the permeability of the mixture and allowed the maintenance of flow through the column. The effect of the silica was not determined in this study, but it was assumed to be negligible because of the nonreactive nature of the silica particles, and the fact that each column had the same amount of silica, so the relative effects would be nearly the same.

Into each column (a 25-ml buret with an inside diameter of 1.2 cm), 11.64 grams of soil-silica mixture (1:1 volume ratio) was packed, at a density approximating the field conditions of 1.6 grams
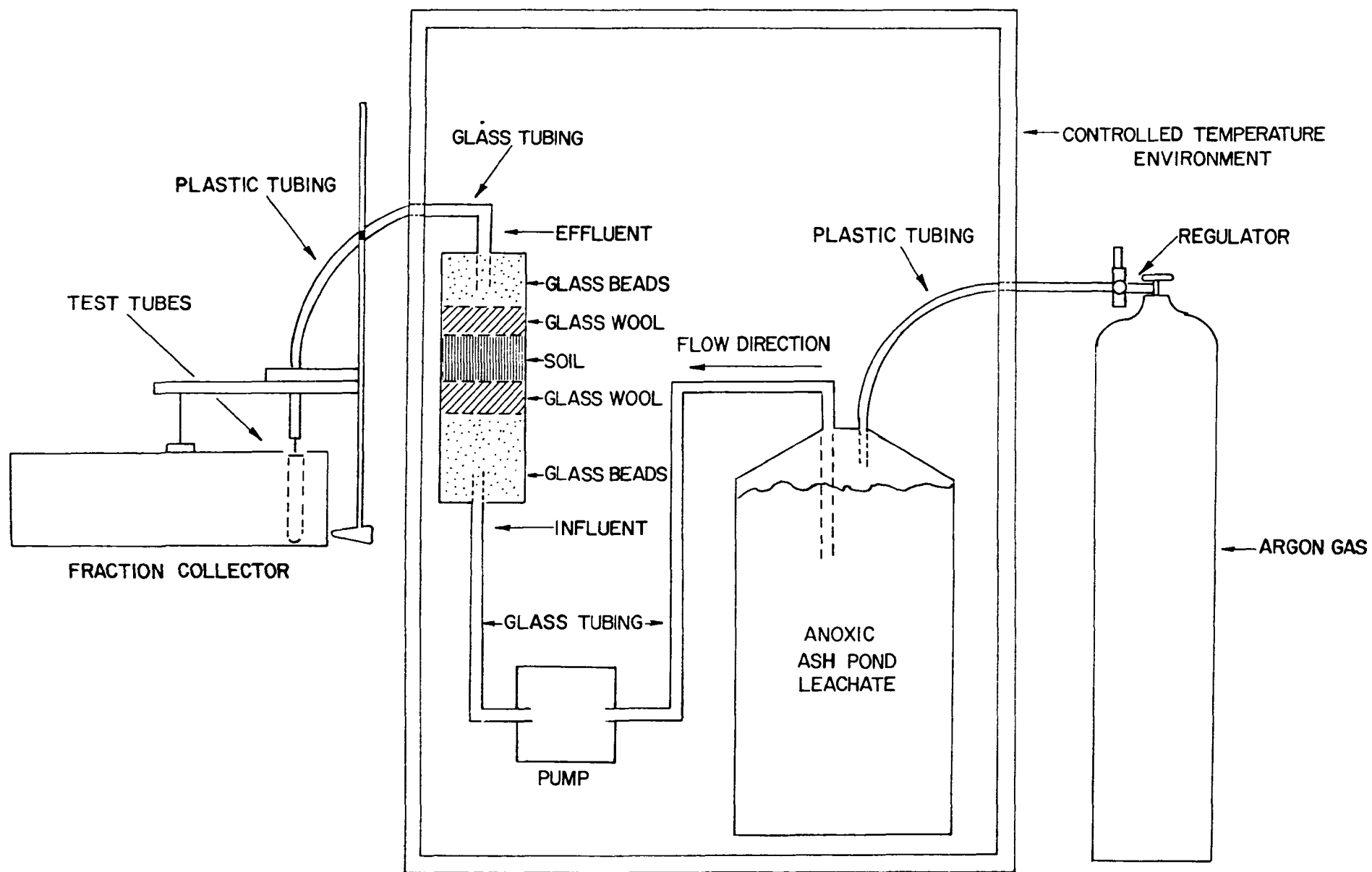
FIGURE 7. COLUMN ELUTION APPARATUS

per cubic centimeter. Both ends of the soil-silica mixture were
packed with glass wool and glass beads to maintain the integrity
of the soil column. The column was set up in a controlled tem-
perature chamber (see Figure 7) to maintain $20^{\circ}C$. The leachate
was pumped from the plastic carboy with a low flow, teflon-coated
pump, up through the soil column to allow air to escape and ensure
saturation. The effluent from the column was then collected in
glass test tubes predosed with 0.1 ml nitric acid. Effluent flow
was adjusted as needed to approximately 0.1 ml per minute.
Sample collection was accomplished by photoelectrical measurements of
the column effluent volume, coupled with a rotating automatic fraction
collector. Effluent samples were collected during the three attenuation
studies until the calcium exchange capacity of each soil-silica mixture
was theoretically exceeded (this was estimated from calculation). The
effluent from each column was collected in 30 ml test tubes which were
composited to make up 6 effluent samples per column. After compositing,
the samples were analyzed for the constituents listed in Table 8.
Procedures for these analyses are given in appendix A. In addition,
each of the soils and clays were subjected to powder X-ray diffraction
analysis for their relative amounts of montmorillite, kaolinite, illite,
and quartz. Each was also analyzed for the constituents listed in
Table 8 (see appendix A for description of soil procedure).

TABLE 8. CONSTITUENTS ANALYZED IN COLUMN
ATTENUATION STUDIES

Calcium	Copper
Magnesium	Chromium
Sodium	Zinc
Potassium	Nickel
Iron	Cadmium
Manganese	Lead
Sulfate	Aluminum
Barium	Beryllium
Mercury	Selenium

SECTION 6

COAL ASH LEACHATE FIELD INVESTIGATIONS

In this section, the results of analyses performed on soil core, interstitial water, and groundwater samples collected at plants J and L are presented.

SOIL CORE ANALYSES

The split-spoon soil cores collected at plants J and L, as described in the section on Methodology, were analyzed in the laboratory for the chemical constituents listed in Table 9. The analytical procedures used in these analysis are described in appendix B.

TABLE 9. CHEMICAL ANALYSIS PERFORMED ON SPLIT-SPOON SOIL CORES

Aluminum	Calcium	Iron	Lead
Arsenic	Cadmium	Magnesium	Selenium
Barium	Chromium	Mercury	Sulfate
Beryllium	Copper	Nickel	Zinc

At nearly all sampling locations at plants J and L, soil-core samples were taken from ground surface to bedrock at various vertical intervals depending on the type of substratum encountered. The purpose for analyzing these vertical strata profiles for the constituents in Table 9 was to determine the extent of leachate migration downward and away from the ash disposal area by comparing the vertical and horizontal distribution of constituents in the substrata. In addition, the concentration differentials between various strata were used to compare their relative capacities for attenuating ash leachate.

The data presented in Tables 10 and 11 are a result of the chemical analyses performed on the soil cores collected at plants J and L, respectively.

TABLE 10. SOIL-CORE ANALYSIS AT PLANT J[a]

Sampling location	Depth (m)	Strata	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
J1	0.7	Clay-silt	160	11	130	-	1,200	<1	30	20	30,000	<0.1	-	25	18	<2	81	57
	1.3	Clay-silt	140	-	130	-	1,600	<1	26	31	21,000	<0.1	-	28	14	-	49	55
	1.9	Clay-silt	160	11	240	-	4,100	<1	29	30	26,000	<0.1	-	39	18	-	130	60
	2.5	Clay-silt	150	-	180	-	4,200	<1	31	28	22,000	<0.1	-	39	21	<2	28	64
	3.2	Clay-silt	64	-	140	-	3,300	<1	9	16	14,000	<0.1	-	25	81	-	40	38
	3.7	Clay-silt	160	-	140	-	3,900	<1	37	28	27,000	<0.1	-	45	19	-	46	96
	4.0	Clay-silt	120	<4	120	-	4,800	<1	32	29	21,000	<0.1	-	40	22	<2	71	46
J2	1.3	Ash	60	48	130	-	2,200	<1	10	24	8,300	<0.1	-	15	<5	2	2,500	21
	3.3	Ash	56	-	100	-	3,700	<1	31	25	6,300	<0.1	-	15	7	-	500	23
	5.7	Ash	83	-	140	-	2,400	<1	20	30	13,000	<0.1	-	13	7	-	240	20
	8.1	Ash	470	-	110	-	2,000	<1	12	21	13,000	<0.1	-	11	9	-	230	13
	13.5	Clay-silt	83	-	130	-	1,200	<1	7	16	9,000	<0.1	-	13	22	<2	120	43
	14.0	Clay-silt	21,000	12	260	-	5,200	..	22	31	13,000	-	1,600	22	22	<1	67	27
	14.1	Clay-silt	620	-	130	-	1,000	<1	13	9	3,800	<0.1	-	<5	20	-	79	32
	14.6	Clay-silt	9,400	-	130	<1	970	<1	16	17	5,300	-	940	9	35	-	120	40
	15.0	Clay-silt	79	-	89	-	1,000	<1	17	16	30,000	<0.1	-	14	32	-	120	48
	15.5	Clay-silt	2,200	-	39	-	510	-	6	4	8,400	-	300	<5	14	-	320	12
	16.0	Sand	420	-	37	-	1,000	<1	11	11	10,000	<0.1	-	12	21	-	72	31
	16.4	Sand	3,900	-	74	<1	350	<1	10	8	4,700	-	590	6	38	-	110	83
	16.9	Sand	240	-	42	-	300	<1	7	11	25,000	<0.1	-	6	9	-	120	20
	17.3	Sand	1,000	-	-	-	460	-	9	<2	2,900	-	140	<5	7	<1	-	8
	17.8	Sand	360	-	47	-	400	<1	9	16	3,800	<0.1	-	5	<5	<2	130	140
J3	2.2	Ash	4,900	-	79	<1	1,200	<1	6	23	4,900	0.7	290	6	15	<2	3,600	5
	3.2	Ash	44,000	-	470	4	3,700	3[b]	47	72	30,000	0.52	3,000	56	69	-	590	90
	4.4	Ash	20,000	-	31	-	4,600	-[b]	23	37	14,000	<0.1	1,400	20	21	-	470	24
	10.2	Clay-silt	8,000	6	91	-	1,000	-[b]	11	14	13,000	<0.1	720	12	29	-	<4	39
	11.0	Sand	4,200	22	53	<0.1	550	<1	6	6	19,000	<0.1	340	<5	11	-	<4	12
	11.1	Sand	4,500	-	77	<1	520	<1	6	6	5,000	-	650	7	5	-	84	22
	11.5	Sand	-	-	-	-	-	-	-	-	-	<0.1	-	-	-	-	-	-
	12.1	Sand	2,300	-	30	-	770	-	<5	2	2,200	-	430	<5	7	-	<4	13
	13.4	Sand	2,700	-	45	-	800	-	<5	12	3,200	<0.1	650	<5	10	-	<4	19
	14.1	Sand	2,900	-	32	<1	300	<1	6	2	1,800	0.27	460	5	<5	<2	60	16
	14.5	Sand	-	-	-	-	290	-	-	-	-	-	360	-	-	-	-	-
	14.9	Sand	6,900	-	86	<1	870	1	13	8	8,900	0.28	2,400	16	8	<2	350	32
	15.0	Shale	16,000	-	180	2	16,000	2	35	16	20,000	0.20	8,600	39	13	<2	170	67

TABLE 10 (continued)[a]

Sampling location	Depth (m)	Strata	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
J4	1.7	Clay-silt	4,100	60	75	-	12,000	-	8	3	34,000	0.93	280	13	17	-	14,000	2
	3.5	Clay-silt	5,100	-	46	-	1,200	-	6	<2	8,300	<0.1	370	5	15	-	<4	10
	5.0	Clay-silt	11,000	-	54	<1	160	2	12	7	21,000	0.3	460	6	28	22	1,800	23
	6.5	Clay-silt	6,300	<4	23	-	440	-	11	<2	25,000	<0.1	420	5	13	-	800	13
	8.0	Sand	3,900	-	54	<1	160	<1	<5	<1	3,700	-	300	<5	7	-	130	12
	9.6	Sand	1,300	-	12	-	400	-	<5	<2	2,200	<0.1	190	<5	3	-	<4	2
	11.1	Shale	11,000	-	140	3	970	2	30	18	41,000	0.28	2,900	53	54	<2	180	86
J5	3.9	Clay-silt	12,000	-	32	1	330	1	10	8	14,000	0.23	600	8	26	<2	340	30
	7.6	Clay-silt	7,000	-	36	<1	160	<1	7	6	14,000	-	350	7	21	-	5,600	17
	10.0	Clay-silt	3,800	-	16	-	420	-	9	<2	13,000	<0.1	230	<5	11	-	<4	9
	11.8	Sand	4,300	-	32	<1	260	1	<5	4	1,900	0.55	380	5	<5	<2	150	16
	12.2	Shale	20,000	-	180	1	1,800	1	35	29	16,000	-	6,800	46	8	-	400	94
	12.3	Shale	17,000	-	190	1	1,800	1	34	24	23,000	-	8,700	58	11	-	130	86
J6	1.9	Ash	8,900	-	160	<1	1,400	2	16	32	38,000	1.2	410	18	25	<2	3,300	9
	4.8	Clay-silt	11,000	-	28	<1	480	1.0	13	10	18,000	-	520	8	19	-	100	26
	8.2	Clay-silt	5,900	-	80	-	600	-	14	<2	11,000	<0.1	800	10	12	-	<4	24
	9.6	Clay-silt	-	-	-	-	460	-	-	-	-	-	800	-	-	-	-	-
	11.5	Clay-silt	3,800	-	45	1	390	2	10	8	9,500	0.17	540	5	<5	<2	34	21
	12.8	Clay-silt	6,300	-	78	-	910	-	20	<2	12,000	<0.1	930	10	14	-	<4	26
	14.1	Sand	1,700	-	46	<1	1,500	<1	<5	2	1,800	0.48	390	5	<5	<2	170	15

[a] Values in μg/g.

[b] Not analyzed.

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TABLE 11. SOIL-CORE ANALYSES AT PLANT L[a]

Sampling location	Depth (m)	Strata	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
J7	0.7	Clay-silt	14,000	-	220	1	4,100	<1	31	22	21,000	0.37	6,400	42	15	<2	140	59
	1.6	Clay-silt	14,000	-	300	1	15,000	<1	31	22	19,000	0.37	6,600	42	15	<2	240	59
	2.5	Clay-silt	16,000	-	280	-	22,000	-	51	21	29,000	<0.1	6,700	43	24	-	490	71
	3.5	Clay-silt	12,000	-	190	-	14,000	-	33	17	26,000	<0.1	5,000	44	20	-	290	48
	4.5	Clay-silt	12,000	-	57	<1	1,000	<1	12	9	17,000	-	1,100	5	30	-	350	25
	5.0	Clay-silt	14,000	-	58	-	1,500	-	12	8	25,000	<0.1	110	12	17	-	470	35
	6.8	Shad	5,100	-	31	<1	150	<1	10	2	18,000	-	500	<5	24	-	120	19
	7.9	Sand	6,200	-	25	-	470	-	26	10	8,900	<0.1	120	9	16	-	<4	29
	8.8	Sand	5,900	-	29	-	550	-	11	6	8,400	<0.1	960	10	13	-	<4	18
	10.0	Sand	2,200	-	<10	<1	120	<1	<5	<1	22,000	0.23	250	<5	19	<2	37	18
	11.2	Sand	2,700	-	15	-	240	-	8	6	2,300	<0.1	370	<5	6	-	76	9
	11.8	Sand	1,300	-	<10	<1	160	<1	<5	2	1,800	0.15	250	<5	17	<2	93	8
	13.4	Shale	1,500	-	430	2	2,300	1	39	4	21,000	1.1	6,100	57	29	<2	7,000	77
J8	3.3	Shale	17,000	-	100	2	21,000	2	35	68	18,000	0.27	5,200	70	33	<2	7,000	70
	4.0	Shale	18,000	-	110	1	4,100	2	43	12	21,000	-	7,000	41	24	-	3,600	67
	4.8	Shale	15,000	-	200	-	4,600	-	42	4	27,000	<0.1	7,300	45	17	-	200	77
	6.3	Shale	18,000	-	310	1	11,000	1	46	42	22,000	-	8,500	43	28	-	160	76
	7.8	Shale	12,000	-	200	-	7,100	-	32	14	22,000	<0.1	6,000	34	10	1	180	51
	9.2	Shale	18,000	-	180	1	4,200	1	44	27	24,000	0.28	8,600	51	22	<2	93	74

TABLE 11. SOIL-CORE ANALYSES AT PLANT L[a]

Sampling location	Depth (m)	Strata	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
L1	2.4	Silty clay	13,000	<1	100	<1	82,000	<1	26	19	31,000	<0.1	1,800	13	28	<1	60	20
	4.3	Silty clay	17,000	<1	80	1	2,400	<1	23	17	28,000	0.2	820	7	15	<1	8	20
	6.7	Silty clay	15,000	<1	80	<1	1,200	<1	26	15	19,000	0.1	620	8	20	<1	50	25
	7.3	Silty clay	16,000	<1	70	2	2,000	<1	26	9	30,000	-	950	6	20	-	60	22
	9.1	Silty clay	16,000	3	110	1	89,000	<1	26	16	30,000	0.1	4,000	26	24	<1	60	48
	10.4	Silty clay	18,000	<1	80	1	5,400	<1	29	24	29,000	-	4,600	25	15	-	10	70
	11.6	Silty clay	10,000	<1	80	<1	93,000	<1	22	20	17,000	-	5,300	28	16	-	410	52
L2	2.1	Silty clay	10,000	<1	80	<1	2,200	<1	36	13	29,000	0.2	970	12	21	<1	100	38
	6.7	Silty clay	17,000	<1	100	2	1,200	<1	48	18	29,000	0.2	900	10	21	<1	46	42
	10.4	Silty clay	20,000	1	20	<1	730	<1	33	.13	29,000	0.1	440	8	14	<1	8	19
	11.0	Silty clay	15,000	<1	120	2	2,400	<1	28	10	16,000	-	2,000	16	20	-	80	42
	12.8	Silty clay	12,000	<1	80	1	2,000	<1	20	6	12,000	0.2	1,400	9	18	<1	60	29
	13.5	Silty clay	11,000	<1	50	<1	2,400	<1	18	4	6,200	-	990	8	14	-	30	39
	13.6	Silty clay	3,800	<1	190	<1	160,000	<1	16	18	39,000	-	3,200	32	25	-	20	28
L3	3.7	Silty clay	15,000	<1	170	1	7,900	<1	45	10	52,000	<0.1	2,300	28	24	<1	26	70
	6.7	Silty clay	10,000	<1	40	1	380	<1	25	6	11,000	0.2	710	8	10	<1	60	26
	9.5	Silty clay	14,000	<1	40	<1	350	<1	34	8	23,000	<0.1	760	8	24	<1	60	27
	10.1	Silty clay	19,000	<1	120	<1	2,400	<1	32	10	19,000	-	3,300	13	23	-	160	24
	11.9	Silty clay	19,000	5	100	3	2,600	<1	36	22	97,000	-	3,900	48	31	-	60	78
	13.7	Silty clay	14,000	<1	80	1	240,000	<1	27	13	8,600	-	15,000	34	22	-	34	42
	14.9	Silty clay	26,000	<1	110	3	4,600	<1	48	32	49,000	0.1	12,000	57	41	<1	28	90
L4	3.7	Silty clay	16,000	7	80	1	1,800	<1	38	6	22,000	0.2	2,000	10	14	<1	1,000	30
	6.7	Silty clay	13,000	<1	60	<1	610	<1	39	7	23,000	<0.1	1,600	10	14	<1	60	36
	9.7	Silty clay	11,000	<1	100	<1	2,300	<1	26	8	30,000	0.2	1,500	12	15	<1	90	35
	9.8	Silty clay	10,000	<1	60	1	800	<1	12	10	11,000	-	1,000	7	12	-	300	39
	10.8	Silty clay	11,000	<1	40	<1	650	<1	15	4	6,100	0.1	400	<5	7	<1	20	18
	12.8	Silty clay	11,000	<1	80	1	1,200	<1	22	20	18,000	-	1,400	32	31	-	8	97
	14.6	Silty clay	13,000	<1	60	2	12,000	<1	27	9	14,000	-	8,400	18	14	-	20	40
	15.9	Silty clay	17,000	<1	40	1	1,600	<1	26	8	29,000	<0.1	1,600	11	18	<1	8	25

TABLE 11 (continued)

Sampling location	Depth (m)	Strata	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
L5	2.1	Silty clay	12,000	9	140	2	3,800	<1	34	8	26,000	0.2	2,000	11	18	<1	180	34
	5.2	Silty clay	11,000	<1	80	<1	900	<1	33	9	25,000	<0.1	1,700	10	16	<1	10	42
	6.7	Silty clay	17,000	1	60	<1	620	<1	34	8	30,000	0.2	1,300	10	21	<1	60	26
	8.2	Silty clay	18,000	<1	40	<1	420	<1	30	7	25,000	-	1,600	7	17	-	26	24
	10.1	Silty clay	10,000	<1	50	1	350	<1	31	14	30,000	-	1,500	10	18	-	10	25
	11.9	Silty clay	10,000	<1	20	<1	570	<1	28	18	30,000	0.2	1,600	10	16	<1	12	26
	15.2	Silty clay	12,000	1.0	60	<1	32,000	<1	29	13	34,000	-	3,200	13	19	-	19	58
L6	0.9	Ash	11,000	5	140	2	14,000	<1	32	27	13,000	<0.1	1,300	42	25	2	3,700	91
	1.5	Ash	15,000	3	110	2	13,000	<1	24	34	11,000	<0.1	1,000	41	31	2	5,500	120
	2.1	Ash	14,000	<1	200	1	7,300	<1	26	34	65,000	0.2	1,100	31	11	<1	60	73
	3.7	Ash	10,000	2	220	1	5,500	<1	26	18	65,000	0.2	660	34	16	<1	200	57
	5.2	Ash	14,000	1	140	<1	3,800	<1	19	18	35,000	0.2	740	24	10	<1	360	34
	6.7	Ash	15,000	4	240	2	13,000	2	45	20	51,000	0.2	1,300	29	23	1	940	130
	8.2	Ash	12,000	5	140	2	8,700	<1	21	13	53,000	<0.1	680	24	8	<1	190	77
	8.8	Ash	15,000	4	220	1	10,000	2	42	35	48,000	<0.1	1,200	29	24	<1	190	140
	10.1	Silty clay	15,000	<1	140	2	4,800	<1	28	20	50,000	<0.01	640	28	8	<1	200	52
	10.7	Silty clay	22,000	<1	90	1	1,400	<1	36	9	40,000	0.1	980	11	17	<1	34	47
	11.3	Silty clay	18,000	<1	80	<1	610	<1	29	9	32,000	-	790	8	18	-	60	24
	11.9	Silty clay	14,000	<1	120	<1	1,800	<1	34	4	28,000	0.1	920	14	16	<1	8	19
	12.5	Silty clay	19,000	1	260	1	3,200	<1	42	14	39,000	-	160	13	32	-	64	33
	13.1	Silty clay	20,000	<1	80	2	3,800	<1	34	8	49,000	0.2	1,900	11	28	<1	60	32
	13.7	Silty clay	25,000	<1	100	2	3,800	<1	37	14	41,000	-	2,800	18	21	-	26	55
	14.2	Silty clay	17,000	<1	170	1	8,500	<1	40	13	44,000	0.1	2,700	20	31	<1	60	64
L7	0.9	Silty clay	17,000	<1	90	2	5,900	<1	31	15	27,000	<0.1	2,800	11	18	<1	80	56
	2.4	Silty clay	22,000	<1	190	2	5,000	<1	29	35	31,000	-	3,100	21	23	-	60	130
	3.7	Silty clay	22,000	<1	230	2	2,300	<1	32	13	44,000	0.1	3,000	23	23	<1	120	83
	5.2	Silty clay	18,000	<1	160	2	2,500	<1	32	20	34,000	-	2,800	18	25	-	26	74
	6.7	Silty clay	10,000	<1	90	1	1,200	<1	17	9	16,000	-	1,300	8	9	-	60	36
	8.2	Silty clay	18,000	<3	160	2	8,000	<1	28	14	36,000	0.1	6,500	37	37	<1	28	85

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TABLE 11 (continued)

Sampling location	Depth (m)	Strata	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
L8	0.9	Ash	12,000	<1	180	2	6,500	<1	37	21	60,000	<0.1	1,400	28	11	<1	700	47
	1.5	Ash	15,000	<1	320	2	7,600	<1	29	24	34,000	0.3	1,200	21	16	<1	3,400	45
	2.1	Ash	5,500	<1	180	<1	2,200	<1	21	12	46,000	0.1	640	16	13	<1	280	17
	2.7	Silty clay	13,000	<1	40	<1	830	<1	36	12	42,000	0.1	600	10	47	<1	360	49
	3.4	Silty clay	16,000	<1	40	1	740	<1	42	12	40,000	-	700	7	26	-	1,200	38
	5.2	Silty clay	15,000	14	40	1	1,000	<1	61	20	52,000	0.2	710	7	26	<1	100	79
	7.6	Silty clay	13,000	<1	160	2	53,000	<1	27	17	34,000	-	3,600	28	22	-	60	54
	10.1	Silty clay	12,000	<1	90	2	3,600	<1	29	25	33,000	-	1,600	16	27	-	100	67
	11.9	Silty clay	10,000	3	140	2	5,500	<1	19	5	63,000	0.1	2,800	18	14	<1	120	57
L9	0.9	Ash	10,000	<1	140	1	4,800	2	38	24	37,000	0.2	760	24	10	<1	50	93
	1.8	Ash	12,000	<1	200	2	5,200	<1	33	27	38,000	0.2	960	19	12	<1	36	42
	2.7	Silty clay	13,000	<1	300	2	8,500	<1	42	23	63,000	0.2	1,800	20	15	<1	60	31
	3.7	Silty clay	5,600	1	260	2	4,900	<1	21	28	21,000	0.2	860	10	14	<1	160	25
	4.6	Silty clay	7,000	20	270	3	5,800	<1	27	19	46,000	0.4	880	21	14	<1	240	39
	5.5	Silty clay	6,300	<1	90	1	1,300	<1	17	17	13,000	0.1	840	6	20	<1	180	49
	6.1	Silty clay	11,000	<1	50	2	430	<1	34	8	22,000	-	690	<5	15	-	60	18
	7.9	Silty clay	15,000	<1	120	3	4,200	<1	33	33	55,000	0.4	1,600	27	44	<1	80	48
	10.5	Silty clay	15,000	<1	160	3	8,000	<1	41	31	40,000	-	3,300	32	21	-	60	77
L10	2.1	Silty clay	11,000	11	40	<1	200	<1	45	12	37,000	0.1	330	7	15	<1	12	29
	5.2	Silty clay	4,500	<1	30	<1	170	<1	26	6	20,000	-	120	<5	14	-	60	13
	8.2	Silty clay	4,100	<1	20	<1	270	<1	17	9	13,000	-	160	<5	17	-	8	18
	12.8	Silty clay	11,000	24	20	1	330	<1	37	24	32,000	0.2	340	13	16	<1	60	66
	17.4	Silty clay	10,000	16	10	2	320	<1	39	26	41,000	0.2	400	23	22	<1	8	91
	20.4	Silty clay	13,000	<1	20	<1	360	<1	37	28	34,000	-	530	18	26	-	8	45
L11	3.7	Silty clay	9,900	<1	40	<1	380	<1	51	21	50,000	0.2	290	12	8	<1	8	24
	6.7	Silty clay	10,000	<1	40	<1	360	<1	37	16	40,000	-	760	9	23	-	60	22
	9.8	Silty clay	13,000	<1	40	1	650	<1	46	22	36,000	0.4	1,400	13	17	<1	60	48
	12.8	Silty clay	3,100	1	10	<1	490	<1	33	7	19,000	-	210	<5	14	-	8	16
	17.4	Silty clay	3,900	<1	80	1	830	<1	33	20	32,000	0.2	940	37	19	<1	10	41

[a] Values in $\mu g/g$.

The most notable characteristic about the data from plant J
(Table 10) is that the concentrations of most constituents are highly
variable, both within and between sampling locations. The concentration
of mercury, with two exceptions, was found to be below 1.0 µg/g. At
sampling locations J1 and J8, the least variability (in terms of absolute
differences) in concentrations was observed for most constituents. This
is not surprising since both these locations were hydraulically upgradient
from the ash disposal area and have homogeneous stratum types throughout
their vertical profiles, albeit different from one another. The variable
nature of the data is further illustrated in Table 12. This table gives
the ranges of concentrations in each soil type measured at locations J2
through J7 combined, and also the ranges within each location. These
data indicate that the more variable values are associated with the
clay-silt, ash, and shale materials, while the sand contained a narrower
range of concentrations. Locations J1 and J8 were omitted from the soil
type comparisons because of their previously mentioned homogeneity.
However, in the bottom portion of Table 12, where ranges of values are
given for each sampling location, the lower variability at locations J1
and J8 relative to other locations is again indicated. At sampling
locations J2 through J7, the magnitude of the variability is dependent
on the constituent under consideration. For example, location J2 has
the widest range of aluminum values, location J4 has the widest range of
calcium values, and location J7 has the widest range of chromium
concentrations.

The data from plant L (Table 11), although not nearly as variable
as plant J's, exhibits some variability, both within and between, samp-
ling locations. Beryllium, cadmium, and selenium values, like plant J's
data, are less than or near their analytical detection limits, while
concentrations of mercury were all measured below 0.5 µg/g. However,
unlike the data from plant J, the two hydraulically upgradient sampling
locations at plant L (L10 and L11) are no less variable than sampling
locations downgradient from the ash disposal area. It should be noted
that the strata types present at plant L consisted mostly of a homoge-
neous silty-clay material throughout the vertical profile. The varia-
tion in plant L's soil core concentrations is also illustrated in
Table 13, where the range of concentration in each stratum measured
and the ranges within each location are given.

The data in this table indicate that, except for selenium, sulfate,
and zinc, the highest constituent concentrations are associated with the
silty-clay material. At the bottom of Table 13, the range in concen-
tration within each sampling location indicates that, unlike the control
locations at plant J, plant L's upgradient control locations (L10 and
L11) are nearly as variable as the sampling locations downgradient from
the ash disposal area. Several parameters (iron, copper, chromium,
zinc, lead, and nickel) actually demonstrate higher maximum concen-
trations in the control locations than several of the hydraulically
downgradient locations.

TABLE 12. RANGES OF CONCENTRATIONS WITH EACH SOIL TYPE AND AT EACH SAMPLING LOCATION FOR THE SOIL CORE SAMPLES ANALYZED FROM PLANT J[a]

Soil type	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
Clay-silt	64-21,000	<4-60	16-300	<1-1.0	160-12,000	<1-2.0	6-37	<2-31	8300-34,000	<0.1-0.93	110-6700	<5-45	<5-81	<1-22	4-1400	2-96
Sand	240-6900	22	<10-86	<1-<0.1	120-1500	<1-1.0	<5-26	<1-16	1800-25,000	<0.1-0.55	120-2400	<5-16	<5-38	<2	<4-350	2-140
Shale	1500-20,000	-	100-430	1.0-3.0	970-21,000	1.0-2.0	30-46	4-68	16,000-41,000	<0.1-1.1	2900-8700	39-70	8-33	1.0-<2	93-7000	51-94
Ash	56-44,000	48	31-470	<1-4.0	1200-4600	<1-3.0	6-47	21-72	4900-38,000	<0.1-1.2	290-3000	6-56	<5-69	<2-2.0	230-3600	5-90
Sampling locations																
J1	64-160	<4-11	120-240	-	1200-4800	<1	9-37	16-31	14,000-30,000	<0.1	-	25-45	14-81	<2	28-130	38-96
J2	56-21,000	12-48	37-260	<1	300-5200	<1	6-31	<2-31	2900-30,000	<0.1	140-1600	<5-22	<5-38	<1-<2	67-2500	8-140
J3	2300-44,000	6-22	30-470	<0.1-4.0	290-16,000	<1-3.0	<5-47	2-72	1800-30,000	<0.1-0.7	290-8600	<5-56	<5-69	<2	<4-3600	5-90
J4	1300-11,000	<4-60	12-140	<1-3.0	160-12,000	<1-2.0	<5-30	<1-18	2200-41,000	<0.1-0.93	190-2900	<5-53	3-54	<2-22	<4-14,000	2-86
J5	3800-20,000	-	16-190	<1-1.0	160-1800	<1-1.0	<5-35	<2-29	1900-23,000	<0.1-0.55	230-8700	<5-58	<5-26	<2	<4-5600	9-94
J6	1700-11,000	-	28-160	<1-1.0	390-1500	<1-2.0	<5-20	<2-32	1800-38,000	<0.1-1.2	390-930	5-18	<5-25	<2	<4-3300	9-26
J7	1300-16,000	-	<10-430	<1-2.0	120-22,000	<1-1.0	<5-51	<1-22	1800-29,000	<0.1-1.1	110-6700	<5-57	6-30	<2	<4-7000	8-77
J8	12,000-18,000	-	100-310	1-2	4100-21,000	1.0-2.0	32-46	4-68	18,000-27,000	<0.1-0.28	5200-8600	34-70	10-33	1-<2	93-7000	51-77

[a]Values in µg/g.

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TABLE 13. RANGES OF CONCENTRATIONS WITHIN EACH STRATA AT EACH SAMPLING LOCATION FOR THE SOIL CORE SAMPLES ANALYZED FROM PLANT L[a]

Strata	Al	As	Ba	Be	Ca	Cd	Cr	Cu	Fe	Hg	Mg	Ni	Pb	Se	SO$_4$	Zn
Ash	5,500-15,000	<1-5	110-240	<1-2	2200-14,000	<1-2	19-45	13-35	11,000-65,000	<0.1-0.3	640-1400	16-42	8-31	<1-2	36-5500	17-140
Silty clay	3,100-26,000	<1-24	10-300	<1-3	320-240,000	<1	12-51	4-35	6,100-97,000	<0.1-0.4	120-15,000	<5-57	7-47	<1	8-3400	13-130
Sampling location																
L1	13,000-18,000	<1-3	70-110	<1-2	1200-93,000	<1	22-29	9-24	17,000-31,000	<1-0.2	620-5300	6-28	15-28	<1	8-410	20-70
L2	3,800-20,000	<1-1	20-190	<1-2	730-160,000	<1	18-48	4-18	6,200-39,000	0.1-0.2	440-3200	8-32	14-25	<1	8-100	19-42
L3	10,000-26,000	<1-5	40-170	<1-3	350-240,000	<1	25-48	6-22	8,600-97,000	<1-0.2	710-15,000	8-57	10-41	<1	26-260	26-90
L4	10,000-17,000	<1-7	40-100	<1-2	610-12,000	<1	12-39	4-20	6,100-30,000	<1-0.2	400-8400	<5-32	7-31	<1	8-1000	18-97
L5	10,000-18,000	<1-9	20-140	<1-2	350-32,000	<1	28-34	7-18	25,000-34,000	<1-0.2	1300-3200	7-13	16-21	<1	10-180	24-58
L6	10,000-25,000	<1-5	80-260	<1-2	610-14,000	<1-2	19-45	4-35	11,000-65,000	<0.1-0.2	160-2800	8-42	8-32	<1-2	8-5500	19-140
L7	10,000-22,000	<1-<3	90-230	<1-2	1200-8000	<1	17-32	9-35	16,000-44,000	<1	1300-6500	8-37	9-37	<1	26-120	36-130
L8	5,500-16,000	<1-14	40-320	<1-2	740-53,000	<1	19-42	5-25	33,000-63,000	<0.01-0.3	600-3600	7-28	11-47	<1	60-3400	17-79
L9	5,600-15,000	<1-20	50-300	1-3	430-8500	<1-2	17-42	8-33	13,000-63,000	0.2-0.4	690-3300	<5-32	10-44	<1	36-240	18-93
L10	4,100-13,000	<1-24	10-40	<1-2	170-360	<1	17-45	6-28	13,000-41,000	0.1-0.2	120-530	<5-23	14-26	<1	8-60	13-91
L11	3,100-13,000	<1-1	10-80	<1-1	360-830	<1	33-51	7-22	19,000-50,000	0.2-0.4	210-1400	<5-37	8-23	<1	8-60	16-48

[a]Values in ug/g.

TABLE 14. CONCENTRATIONS IN ASH SAMPLES COLLECTED FROM ASH DISPOSAL AREAS – PLANTS J AND L[a]

Constituent	Plant J		Plant L	
	Mean	Range	Mean	Range
Aluminum	9,938.4	56–44,000	12,346.2	5,500–15,000
Arsenic	48	48	2.3	<1.0–5
Barium	151.4	79–470	186.9	110–320
Beryllium	2.5	<1.0–4	1.5	<1.0–2
Calcium	2,828.6	1,200–4,600	7,815.4	2,200–14,000
Cadmium	1.3	<1.0–3	1.2	<1.0–2
Chromium	21.3	6–47	30.2	19–45
Copper	33.1	21–72	23.6	12–35
Iron	12,785.7	4,900–30,000	42,769.2	11,000–65,000
Magnesium	1,563.3	290–3,000	995.4	640–1,800
Mercury	0.3	<0.1–0.7	0.2	<0.1–0.3
Nickel	19.4	6–56	27.8	16–42
Lead	19.0	<5–69	16.2	8–31
Selenium	2.0	<2–2	1.2	<1.0–2
Sulfate	1,161.4	230–3,600	1,200.5	36–5,500
Zinc	28.0	5–90	74.3	17–140

[a]Values in mg/L.

The mean concentrations and range of values for the constituents listed in Table 9 were calculated for all ash samples collected at plants J and L. At plant J, ash samples were collected over various vertical intervals at sampling locations J2 and J3, and at plant L sampling locations L6, L8, and L9. The results of these calculations indicate (Table 14) that the mean concentrations of aluminum, barium, calcium, chromium, iron, nickel, sulfate, and zinc are highest in ash from plant L. Plant J, however, had the higher mean concentrations of arsenic, beryllium, cadmium, copper, mercury, lead, selenium, and magnesium. There are a multitude of reasons why the above elements would be partitioned as they are between the two plants; different coal sources, methods of firing the coal, and ash collection systems are the major reasons related to plant operations. The sample collection is undoubtedly another factor to consider. The age of the ash sample and its particle size characteristics may also affect concentrations within the sample. The longer coal ash has been exposed to the leaching process the more opportunity for the dissolution of ions, and at least one investigator[15] has shown that smaller ash particles contain higher concentrations of certain elements.

The variable nature of the constituents measured in the soil cores from plants J and L, precluded the use of this data for determining the extent of leachate migration. The natural variation of soils resulting from spatial differences in mineralology, organic content, and soil particle sizes makes the effects of leachate on constituent concentrations in the soil difficult or impossible to define. Determining differences in the attenuation capacity of the various substrata also suffers because the highly variable soil makes results ambiguous. In order to utilize soil core data to determine the magnitude of leachate migration, some means of eliminating or minimizing the natural variation will be needed. One approach may be to analyze only a certain particle size fraction of a sample for a contaminant. Another might be to separate the various components of a soil sample, such as organic and inorganic, and analyze separately. There are many approaches which might be considered, but the measurement of constituents in total soil samples, as this investigation indicates, does not appear to be a viable one.

INTERSTITIAL WATER ANALYSIS

During the collection of soil core samples from plants J and L, certain cores were selected for extraction and analysis of their interstitial water (the extraction methodology is given in section 4). The basis for selecting which cores were to be extracted was the apparent moisture content of the core and the density of the material under consideration. Both of these factors were evaluated in the field, based on field soil testing techniques, previous experience, and intuition. In all, 23 soil-core extractions were analyzed from plant J, and seven at plant L. The fewer number of samples extracted

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TABLE 15. ANALYSIS OF EXTRACTED INTERSTITIAL WATER FROM PLANT J

	Well locations											
	J3	J3	J4	J4	J4	J4	J4	J5	J5	J5	J6	J6
Depth, m	13.4	14.1	5.0	6.5	8.0	9.6	11.1	6.7	8.4	11.8	5.0	6.5
Strata	Sand	Sand	Clay-silt	Clay-silt	Clay-silt	Clay-silt	Sand	Clay-silt	Sand	Sand	Clay-silt	Clay-silt
Aluminum, mg/L	3.2	3.15	180	190	60	< 0.2	5.2	15	62	8.0	-	5.5
Barium, mg/L	< 0.1	-	< 0.1	< 0.1	< 0.1	< 0.1	1.3	-	-	-	-	-
Beryllium, mg/L	-	-	-	-	< 0.01	-	-	-	-	-	-	-
Calcium, mg/L	56	15.5	470	680	350	130	170	270	310	87	-	31
Cadmium, mg/L	-	< 0.001	-	-	-	-	-	0.002	0.005	0.003	-	-
Chromium, mg/L	< 0.01	0.008	0.04	0.05	0.06	< 0.01	< 0.01	0.02	0.7	0.02	< 0.001	0.011
Copper, mg/L	0.3	0.02	0.2	0.3	0.3	< 0.1	< 0.1	0.20	0.27	0.63	0.2	0.08
Iron, mg/L	15	0.82	3300	2400	1200	1.0	120	1000	1000	10	38	10
Magnesium, mg/L	14	-	200	220	91	33	31	-	-	-	-	-
Mercury, mg/L	-	< 0.01	< 0.02	< 0.02	< 0.02	< 0.02	< 0.02	< 0.02	< 0.01	< 0.01	< 0.02	-
Nickel, mg/L	< 0.5	< 0.05	0.9	< 0.5	0.8	< 0.5	< 0.5	0.35	0.5	0.2	< 0.5	< 0.5
Lead, mg/L	-	< 0.006	0.76	0.58	0.42	0.09	0.11	0.019	0.017	0.12	< 0.05	0.066
Selenium, mg/L	-	< 0.05	-	-	-	-	-	< 0.10	< 0.05	< 0.05	-	-
Sulfate, mg/L	-	-	6800	4800	2300	380	680	-	-	-	100	-
Zinc, mg/L	0.7	0.02	4.3	3.5	4.8	5.8	0.6	1.3	2.6	2.1	5.1	1.4
pH, std. units	7.3	8.0	4.3	4.2	4.3	6.0	6.8	2.6	1.8	1.9	2.4	2.4
Conductivity, μmhos/cm^2	-	-	-	-	-	-	-	4000	3700	6000	900	1700

TABLE 15 (continued)

			Well locations								
	J6	J6	J6	J7	J7	J7	J7	J7	J7	J7	J7
Depth, m	8.2	11.5	12.8	1.6	4.5	5.0	6.8	7.9	8.8	10	11.2
Strata	Clay-silt	Clay-silt	Clay-silt	Clay-silt	Clay-silt	Clay-silt	Clay-silt and sand	Clay-silt and sand	Clay-silt and sand	Clay-silt and sand	Clay-silt and sand
Aluminum, mg/L	2.5	6.4	9.1	1.15	3.5	2.8	3.6	8.4	8.1	23	11
Barium, mg/L	< 0.1	-	< 0.1	-	< 0.1	1.9	-	< 0.1	1.8	-	2.3
Beryllium, mg/L	-	-	-	-	-	-	-	-	-	-	-
Calcium, mg/L	62	78	130	420	660	510	37	51	58	59.5	73
Cadmium, mg/L	0.006	0.006	-	< 0.001	-	0.002	0.004	-	-	0.005	0.003
Chromium, mg/L	< 0.001	0.015	0.01	< 0.001	< 0.01	0.003	0.005	< 0.01	0.02	0.09	0.021
Copper, mg/L	< 0.1	0.07	< 0.1	0.01	< 0.1	0.3	0.06	0.2	0.2	0.03	< 0.1
Iron, mg/L	38	25	64	1.4	0.8	50	2.7	8.1	5.2	92	85
Magnesium, mg/L	12	-	19	-	63	41	-	20	32	-	15
Mercury, mg/L	< 0.02	< 0.01	< 0.02	< 0.01	< 0.02	< 0.02	-	< 0.02	< 0.02	< 0.01	< 0.01
Nickel, mg/L	< 0.5	< 0.05	< 0.5	0.11	< 0.5	< 0.5	0.09	< 0.05	< 0.05	0.12	< 0.5
Lead, mg/L	0.07	0.018	-	< 0.002	-	0.004	0.016	16	-	0.07	0.035
Selenium, mg/L	< 0.05	< 0.05	-	< 0.02	-	< 0.10	< 0.05	-	-	< 0.05	< 0.05
Sulfate, mg/L	260	-	< 100	-	1500	-	-	100	230	-	-
Zinc, mg/L	0.5	0.53	2.2	0.03	5.0	2.6	0.38	0.9	2.5	0.43	1.7
pH, std. units	7.1	2.3	2.7	7.2	6.8	1.7	1.7	1.5	1.4	2.1	2.2
Conductivity, μmhos/cm^2	575	2200	2100	2750	1950	2700	1200	4500	20,000	9500	3550

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TABLE 16. ANALYSIS OF EXTRACTED INTERSTITIAL WATER FROM PLANT L

Constituents	Sampling locations						
	L3	L6	L6	L6	L6	L7	L8
Depth, meters	15.0	8.0	8.8	10.0	10.6	8.2	0.9
Strata	Silt-clay	Ash	Ash	Ash	Ash	Silt-clay	Ash
Aluminum, mg/l	2.2	8.4	8.6	2.0	8.2	7.7	4.1
Barium, mg/l	0.400	0.600	0.600	0.500	0.600	0.600	0.200
Beryllium, mg/l	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
Calcium, mg/l	96	220	280	320	210	200	170
Cadmium, mg/l	0.010	0.080	0.040	0.020	0.010	0.020	0.020
Chromium, mg/l	<0.050	0.110	0.050	<0.050	0.060	<0.050	<0.050
Copper, mg/l	<0.010	0.110	0.090	0.020	0.060	0.070	0.030
Iron, mg/l	1.9	9.5	6.3	8.8	19	8.9	3.5
Mercury, mg/l	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004
Magnesium, mg/l	21	3.8	6.3	16	12	14	9.8
Manganese, mg/l	0.140	0.550	0.450	1.6	1.2	3.5	0.580
Nickel, mg/l	<0.050	0.100	0.070	0.120	0.090	0.080	0.090
Lead, mg/l	0.050	0.050	0.050	0.050	0.120	0.100	0.050
Zinc, mg/l	0.070	1.8	0.620	0.690	0.640	0.400	0.330
pH, standard units	7.5	3.5	3.6	6.5	4.1	2.9	3.4
Conductivity, μmhos/cm	880	1180	1130	1040	3000	6400	1650
Alkalinity, mg/l as $CaCO_3$	320	-	-	-	-	-	-
Sulfate, mg/l	100	900	900	920	600	500	860

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at plant L was because the compacted nature of the soils precluded the extraction of enough water for laboratory analysis. In Tables 15 and 16, the results of analyses on the interstitial waters from plants J and L, respectively, are presented. The dashes that appear in these tables indicate analyses were not performed because an insufficient amount of extracted water was available to accommodate the analytical evaluation. The minimum detection limits shown for the same elements are also a result of insufficient sample volume.

One of the most striking features about the data in Tables 15 and 16 is the low pH values observed. At plant J, 16 of the 23 pH measurements were less than 5.0, and 13 of the values were below 3.0. At plant L, five of the seven measurements for pH were below 5.0, and one less than 3.0. The data do not, however, indicate any trends in pH, such as with depth or stratum type. In addition, the constituent concentrations in these tables do not always track with variations in pH, as would be expected from solubility relationships. For example, iron concentrations at sampling location J7 (Table 15) are not inversely proportional to pH, and zinc concentrations at location J4 demonstrate similar behavior. The solubilities of the elements in Tables 15 and 16 in relation to other matrices in which they were measured are obviously a factor to consider and this subject is discussed later in the report.

GROUNDWATER ANALYSIS

In Tables 17 and 18, the results of the laboratory analysis performed on groundwater samples collected from the wells at plant J and L, respectively, are presented. Samples were collected from all sampling locations at plant J, except J1, on July 2, 1976, and from all locations except J3 and J3A on March 9, 1977. These sample omissions were a result of sampling well malfunctions. Groundwater samples were collected from all sampling locations at plant L, on February 22, 1977.

For comparative purposes, the EPA's domestic water supply criteria[16] are also listed in Tables 17 and 18. Although these criteria are not applicable to leachate from solid waste disposal sites, they are used here, and elsewhere in this report, as a screening process to identify water quality constituents that may deserve environmental consideration. Of the constituents measured in samples from plant J, five were found to exceed EPA's criteria for iron, manganese, lead, dissolved solids (as sulfate), and pH. Samples from all wells except J1, a background well, exceeded the criterion for iron; samples from all wells in the March collection exceeded the manganese criterion; and the criterion for dissolved solids was exceeded in March at locations J4, J4A, J5A, J7A, and J8. Lead exceeded the criterion in July at locations J6, and the pH was not within the criterion at wells J3, J4, J5B, J6, J6A, and J7.

At plant L, the data indicate that the criterion for iron was exceeded in samples analyzed from wells, L3A, L6A, L7, L8A, and L9A. Dissolved solids, as sulfate, exceeded the criterion in samples from

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TABLE 17. ANALYSIS OF GROUNDWATER SAMPLES COLLECTED FROM SAMPLING WELLS AT PLANT J - JULY 2, 1976

Parameter	J2	J3	J3A	J4	J4A	J5	J5A	J6	J6A	J7	J8	DWSC[a]
					July 2, 1976							
Aluminum, mg/L	0.8	2.0	520	5.2	170	2.3	40	2.3	85	7.2	5.4	b
Beryllium, mg/L	<0.01	<0.01	0.07	<0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	b
Cadmium, mg/L	<0.001	<0.001	<0.001	0.001	<0.001	<0.001	<0.001	0.001	0.003	<0.001	0.001	0.01
Calcium, mg/L	92	27	280	36	420	100	400	160	85	39	360	b
Chromium, mg/L	<0.005	<0.005	<0.005	0.01	<0.005	<0.005	<0.005	0.006	<0.005	0.012	0.012	0.05
Copper, mg/L	0.04	0.08	1.4	0.06	0.18	0.04	0.09	0.06	0.07	0.04	0.07	1.0
Iron, mg/L	10	9.3	2000	6.8	680	98	640	12	200	5.6	13	0.3
Lead, mg/L	<0.01	0.011	0.044	0.043	<0.01	<0.01	<0.01	0.042	0.089	0.011	0.026	0.05
Mercury, mg/L	<0.0002	<0.0002	0.0029	<0.0002	0.0006	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	<0.0002	0.002
Nickel, mg/L	<0.05	<0.05	0.82	<0.05	0.31	<0.05	0.18	0.06	0.07	<0.05	<0.05	b
Selenium, mg/L	<0.001	<0.001	0.005	0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.01
Dissolved solids, total, mg/L	450	150	680	210	-	590	8100	530	270	190	1800	250
Zinc, mg/L	0.04	0.09	1.3	0.2	2.1	0.08	0.41	0.05	0.19	0.03	0.08	5.0
pH, standard units	6.6	9.5	8.9	9.5	7.1	8.2	8.1	10	9.2	9.7	8.0	5.9

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TABLE 17 (continued)

Parameter	J1	J2	J4	J4A	J5	J5A	J5B	J6	J6A	J7A	J8	DWSC[a]
					March 9, 1977							
Aluminum, mg/L	0.3	<0.2	<0.2	<0.2	<0.2	<0.2	0.4	<0.2	<0.2	<0.2	<0.2	b
Arsenic, mg/L	<0.004	0.026	<0.004	<0.004	<0.004	0.006	<0.004	<0.004	<0.004	<0.002	<0.002	0.05
Boron, mg/L	0.16	5.6	0.2	0.26	0.09	0.61	0.13	0.2	0.2	2.0	0.16	b
Barium, mg/L	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	1.0
Beryllium, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	b
Calcium, mg/L	62	110	110	410	19	260	18	110	100	520	360	b
Cadmium, mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.01
Chromium, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	0.05
Copper, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.02	<0.01	<0.01	<0.01	<0.01	1.0
Iron, mg/L	<0.05	16	<0.05	420	0.57	2200	3.7	1.6	77.0	7.0	0.24	0.3
Mercury, mg/L	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002
Magnesium, mg/L	17	42	16	120	12	100	10	15	18	96	70	b
Manganese, mg/L	0.13	4.6	0.64	44	0.24	63	4.5	3.5	7.3	18	2.2	0.05
Nickel, mg/L	<0.05	<0.05	<0.05	0.08	<0.05	0.06	<0.05	<0.05	<0.05	0.08	<0.05	b
Lead, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.04	<0.01	<0.01	<0.01	<0.01	0.05
Selenium, mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.01
Sulfate, mg/L	68	65	310	2100	30	3400	130	120	150	1700	890	250
Sulfide, mg/L	0.08	<0.02	<0.02	0.07	<0.02	0.13	<0.02	<0.02	<0.02	0.03	<0.02	b
Dissolved solids, total, mg/L	410	1400	900	4000	210	4900	390	950	1200	3400	2700	250
Volatile solids, total, mg/L	10	70	40	570	10	500	50	50	70	250	120	b
Zinc, mg/L	<0.02	<0.01	<0.01	<0.01	<0.01	0.05	0.2	<0.01	0.02	0.01	0.02	5.0
pH, standard units	7.5	7.1	6.8	6.6	7.0	6.5	3.3	7.4	6.9	7.0	7.6	5.9
Oxidation reduction potential, MV	+130	-135	-70	-100	-130	-110	+200	-100	-160	-110	+150	b
Conductivity, μmhos/cm^2	470	790	510	2200	180	2900	360	520	680	1900	1460	b
Alkalinity, mg/L as CaCO$_3$	192	390	-	-	81	108	-	237	320	320	295	b
Water table elevation, m	230.48	228.55	224.79	224.03	224.70	223.75	221.16	220.19	224.36	225.13	228.22	b

[a] Domestic Water Supply Criteria, EPA, 1976.

[b] Not analyzed.

TABLE 18. ANALYSIS OF GROUNDWATER SAMPLES COLLECTED FEBRUARY 22, 1977, FROM SAMPLING WELLS AT PLANT L

Parameter	L1	L2	L2A	L3	L3A	L4	L5	L6	L6A	L7	L8	L8A	L9	L9A	L10	L11	DWSC[a]
Aluminum, mg/L	0.8	0.5	<0.2	0.3	0.2	0.4	<0.2	<0.2	<0.2	2.3	<0.2	11	<0.2	<0.2	<0.02	<0.02	b
Arsenic, mg/L	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	-	0.01	-	0.005	<0.002	0.019	<0.002	0.035	<0.002	<0.002	0.05
Boron, mg/L	0.15	0.35	0.23	0.2	0.21	0.16	0.24	0.44	8.1	0.31	2.3	20	0.76	6.1	0.16	0.08	b
Beryllium, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.04	<0.01	<0.01	<0.01	<0.01	<0.01	b
Calcium, mg/L	55	81	64	64	120	39	50	110	190	100	270	520	120	180	7.3	27	b
Cadmium, mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.01
Chromium, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	0.05
Copper, mg/L	<0.01	<0.01	0.04	0.04	0.01	0.02	<0.01	<0.01	0.02	<0.01	0.02	0.03	<0.01	<0.01	0.04	<0.01	1.0
Iron, mg/L	<0.05	<0.05	<0.05	0.06	0.34	0.26	0.13	0.23	13	0.44	0.22	380.0	<0.05	0.87	0.08	<0.05	0.3
Mercury, mg/L	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002
Magnesium, mg/L	4.4	7.1	6.1	15	29	9.0	7.9	25	11	14	40	91	30	20	1.3	12	b
Manganese, mg/L	0.75	0.45	2.9	0.69	1.9	0.47	0.29	0.85	5.6	7.1	2.2	4.7	0.87	1.0	0.22	0.74	0.05
Nickel, mg/L	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	b
Lead, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.03	<0.01	<0.01	0.05
Selenium, mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.01
Sulfate, mg/L	8.0	10	79	22	39	3.0	10	68	170	8.0	490	2500	21	450	47	13	250
Sulfide, mg/L	0.33	0.04	0.04	0.1	0.2	0.16	0.2	0.44	0.11	0.07	0.16	0.06	0.04	0.16	0.14	0.04	b
Dissolved solids, total, mg/L	190	180	290	180	250	150	150	330	400	450	1000	3700	270	920	160	150	250
Zinc, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	0.02	<0.01	<0.01	<0.01	<0.01	<0.01	0.21	<0.01	<0.01	<0.01	0.11	5.0
pH, standard units	7.7	7.8	7.7	8.0	7.9	7.4	8.0	7.4	7.5	8.5	7.2	4.4	7.6	7.5	6.0	7.2	5.9
Oxidation reduction potential, MV	+70	+390	+390	+120	+130	+130	+60	+130	+10	+110	+150	+340	+370	-	+180	+360	b
Conductivity, μmhos/cm2	280	325	530	350	680	280	350	720	930	790	1080	2450	740	250	220	250	b
Alkalinity, mg/L as CaCO3	198	230	246	235	425	132	165	350	416	185	225	-	490	149	57.2	98.5	b
Water table elevation, m	183.21	182.20	185.34	181.41	181.44	182.54	183.02	186.83	186.77	181.03	187.38	189.45	189.06	189.55	180.29	183.51	b

[a]Domestic Water Supply Criteria, EPA, 1976.

[b]Not analyzed.

wells L8, L8A, and L9A. The pH in well L8A was not within the criterion, and manganese exceeded the criterion in all wells sampled, including the background wells.

The data also indicate that pH values measured in samples collected from the sampling wells (Tables 17 and 18) were generally higher than values obtained from the extracted interstitial water samples (Tables 14 and 15). Samples from the wells were generally neutral to alkaline, with only two depressed pH values observed (wells L8A and J5B), while the pH of the interstitial water was generally very acidic (a pH of 1.4 was measured), but ranged to alkaline. Although the methods of collection differ between the interstitial water and the well water, it was possible to compare samples collected by both methods from the same location and depth. These comparisons are given in Table 19, along with the soil core sample concentrations for the same locations and depth. The table shows that in all cases where measurable quantities of a constituent were found the concentration in the interstitial water sample was greater than that found in the well sample. This comparison indicates that the measurement of interstitial water may be a more accurate method of determining groundwater contamination by leachates than utilizing groundwater monitoring wells of the design used in this study. However, more research is needed to economize and standardize this technique. Specific reactions and chemistry of selected constituents in Table 19 are discussed below.

Aluminum

In aqueous systems, aluminum is strongly influenced by its tendency to form soluble complexes with fluoride, hydroxide, sulfate, phosphate, and some organic ligands; its solubility is affected by the concentrations of these ligands.[17] The solubility of aluminum, as a function of pH, in a hydroxide equilibrium environment, is illustrated in Figure 8. On the alkaline side of the curve, the main species present is monovalent $AL(OH)_4^{-1}$, while under acidic conditions the main species is the trivalent form Al^{+3}. However, in aqueous solutions such as the interstitial water in Table 19, other complexing ligands in addition to hydroxide may alter the solubility of aluminum through complexation. For example, the concentration of sulfate in an interstitial water sample collected from location J4 was measured at 2300 mg/l, with a pH of 4.3. Roberson and Hem[18] studied the solubility of aluminum in the presence of different concentrations of sulfate and at different pH's, and Figure 9 illustrates the solubility of microcrystalline gibbsite $(Al(OH)_3)$ as a function of sulfate concentrations over the ranges of pH from their studies. Using the data from location J4, at 2300 mg/l sulfate and a pH of 4.3, the relationship presented in Figure 9 indicates that approximately 270 mg/l of aluminum could be soluble in the interstitial water sample. In addition, using values from location J4 at 310 mg/l sulfate and a pH of 6.8 for the well water sample, 0.04 mg/l of soluble aluminum could be present in the well water according

TABLE 19. COMPARISON OF CONCENTRATIONS MEASURED IN GROUNDWATER SAMPLING WELLS,
EXTRACTED INTERSTITIAL WATER, AND SOLID SUBSTRATUM MATERIAL,
PLANTS J AND L

| | Sampling location and site | | | | | | | | |
| | Location J3 (13.4 m) | | | Location J4 (8.0 m) | | | Location J5 (8.4 m) | | |
Constituent	Groundwater sampling wells	Extracted interstitial water	Solid substratum material	Groundwater sampling wells	Extracted interstitial water	Solid substratum material	Groundwater sampling wells	Extracted interstitial water	Solid substratum material
pH, standard units	9.5	7.3	-	6.8	4.3	-	7.0	1.8	-
Alkalinity, mg/L as $CaCO_3$	-	-	-	-	-	-	81	-	-
Oxidation reduction potential, MV	-	-	-	-70	-	-	-130	-	-
Dissolved solids, total, mg/L	150	-	-	900	-	-	210	-	-
Volatile solids, total, mg/L	-	-	-	40	-	-	10	-	-
Aluminum, mg/L	?.0	3.2	2700	< 0.2	60	3900	< 0.2	62	7000
Arsenic, mg/L	-	-	-	< 0.004	-	-	< 0.004	-	-
Boron, mg/L	-	-	-	0.2	-	-	0.09	-	-
Barium, mg/L	-	< 0.1	45	< 0.1	< 0.1	54	< 0.1	-	36
Beryllium, mg/L	< 0.01	-	-	< 0.01	< 0.01	< 1.0	< 0.01	-	< 1.0
Calcium, mg/L	27	56	800	110	350	160	19	310	160
Cadmium, mg/L	< 0.001	-	-	< 0.001	-	< 1.0	< 0.001	0.005	< 1.0
Chromium, mg/L	< 0.005	< 0.01	< 5.0	< 0.005	0.06	< 5.0	< 0.005	0.7	7.0
Copper, mg/L	0.08	0.3	12	< 0.01	0.3	< 1.0	< 0.01	0.27	6.0
Iron, mg/L	9.3	15	3200	< 0.05	1200	3700	0.57	1000	14,000
Mercury, mg/L	< 0.0002	-	< 0.1	< 0.002	< 0.02	-	< 0.002	< 0.01	-
Magnesium, mg/L	-	14	650	16	91	300	12	-	350
Manganese, mg/L	-	-	-	0.64	-	-	0.24	-	-
Nickel, mg/L	< 0.05	< 0.5	< 5.0	< 0.05	0.8	< 5.0	< 0.05	0.5	7.0
Lead, mg/L	0.011	-	10	< 0.01	0.42	7.0	< 0.01	0.017	21
Selenium, mg/L	< 0.001	-	-	< 0.001	-	-	< 0.001	< 0.05	-
Sulfate, mg/L	-	-	< 4.0	310	2300	130	30	-	5600
Sulfide, mg/L	-	-	-	< 0.01	-	-	< 0.02	-	-
Zinc, mg/L	0.09	0.7	19	< 0.01	4.8	12	< 0.01	1.3	17

TABLE 19 (continued)

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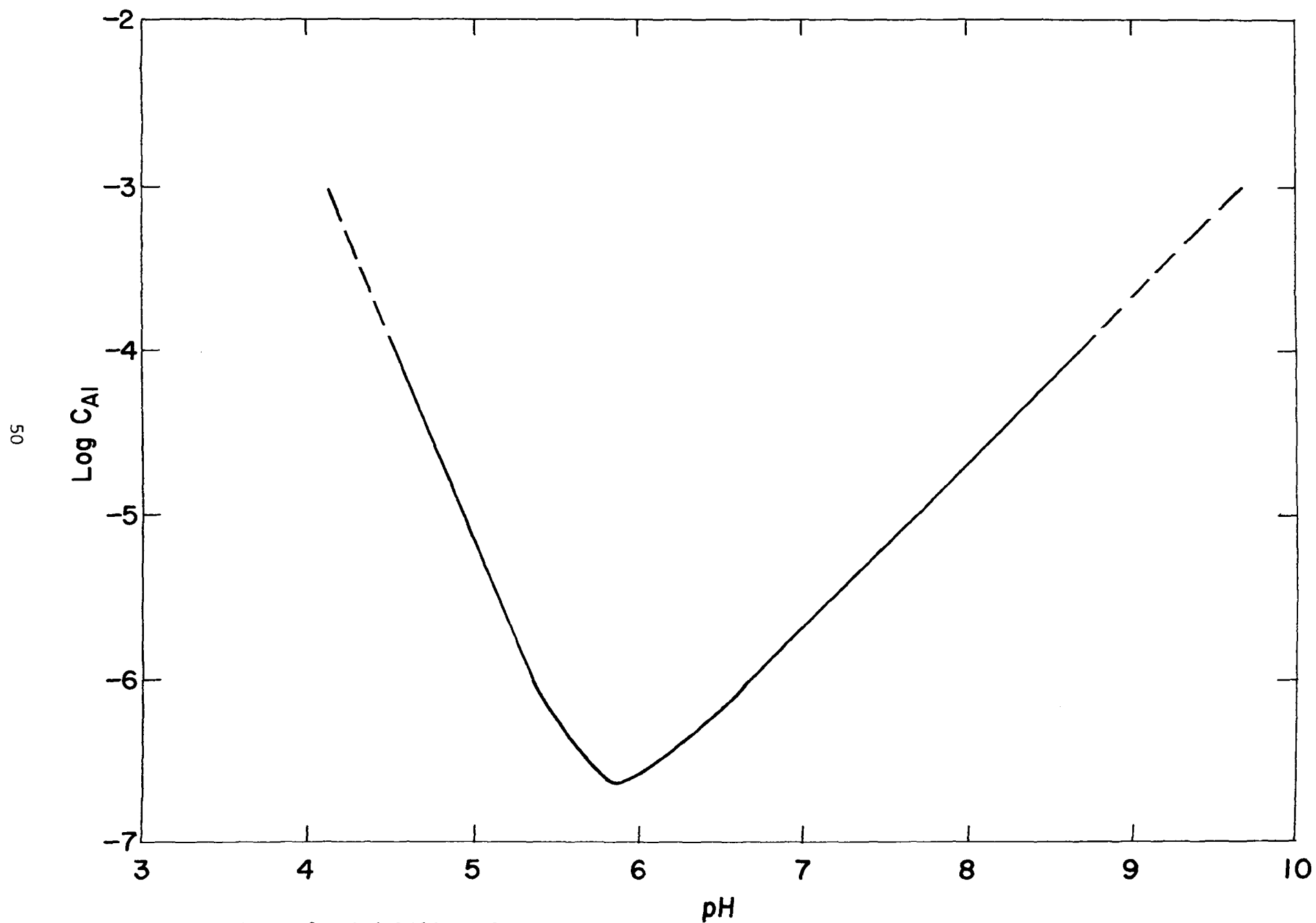| | Sampling location and site | | | | | | | | |
| | Location L3 (15.0 m) | | | Location L6 (10.6 m) | | | Location L7 (8.8 m) | | |
Constituent	Groundwater sampling wells	Extracted interstitial water	Solid substratum material	Groundwater sampling wells	Extracted interstitial water	Solid substratum material	Groundwater sampling wells	Extracted interstitial water	Solid substratum material
pH, standard units	8.0	7.5	-	7.5	4.1	-	8.5	2.9	-
Alkalinity, mg/L as $CaCO_3$	235	320	-	930	-	-	185	-	-
Oxidation reduction potential, MV	+120	-	-	+10	-	-	+110	-	-
Dissolved solids, total, mg/L	180	-	-	400	-	-	450	-	-
Volatile solids, total, mg/L	-	-	-	-	-	-	-	-	-
Aluminum, mg/L	0.3	2.2	26,000	<0.2	8.2	22,000	2.3	7.7	18,000
Arsenic, mg/L	<0.002	-	<1.0	-	-	<1.0	0.005	-	<3.0
Boron, mg/L	0.2	-	-	8.1	-	-	0.31	-	-
Barium, mg/L	-	0.4	110	-	0.6	90	-	0.6	160
Beryllium, mg/L	<0.01	<0.01	3.0	<0.01	<0.01	1.0	<0.01	<0.01	2.0
Calcium, mg/L	64	96	4600	190	210	1400	100	200	8000
Cadmium, mg/L	<0.001	0.01	<1.0	<0.001	0.01	<1.0	<0.001	0.02	<1.0
Chromium, mg/L	<0.005	<0.05	48	<0.005	0.06	36	<0.005	<0.05	28
Copper, mg/L	0.04	<0.01	32	0.02	0.06	9.0	<0.01	0.07	14
Iron, mg/L	0.06	1.9	49,000	13	19	40,000	0.44	8.9	36,000
Mercury, mg/L	<0.002	<0.004	0.1	<0.002	<0.004	0.1	<0.002	<0.004	0.1
Magnesium, mg/L	15	21	12,000	11	12	980	14	14	6500
Manganese, mg/L	0.69	0.14	-	5.6	1.2	-	7.1	3.5	-
Nickel, mg/L	<0.05	<0.05	57	<0.05	0.09	11	<0.05	0.08	37
Lead, mg/L	<0.01	<0.05	41	<0.01	0.12	17	<0.01	0.1	37
Selenium, mg/L	<0.001	-	<1.0	<0.001	-	<1.0	<0.001	-	<1.0
Sulfate, mg/L	22	100	28	170	600	34	8.0	-	28
Sulfide, mg/L	0.1	-	-	0.11	-	-	0.07	-	-
Zinc, mg/L	<0.01	0.07	90	<0.01	0.64	47	<0.01	0.4	85

Figure 8. Solubility of microcrystalle gibbsite as a function of pH, at 25° C., and 1 atmosphere total pressure (from Roberson and Hem, 1969).
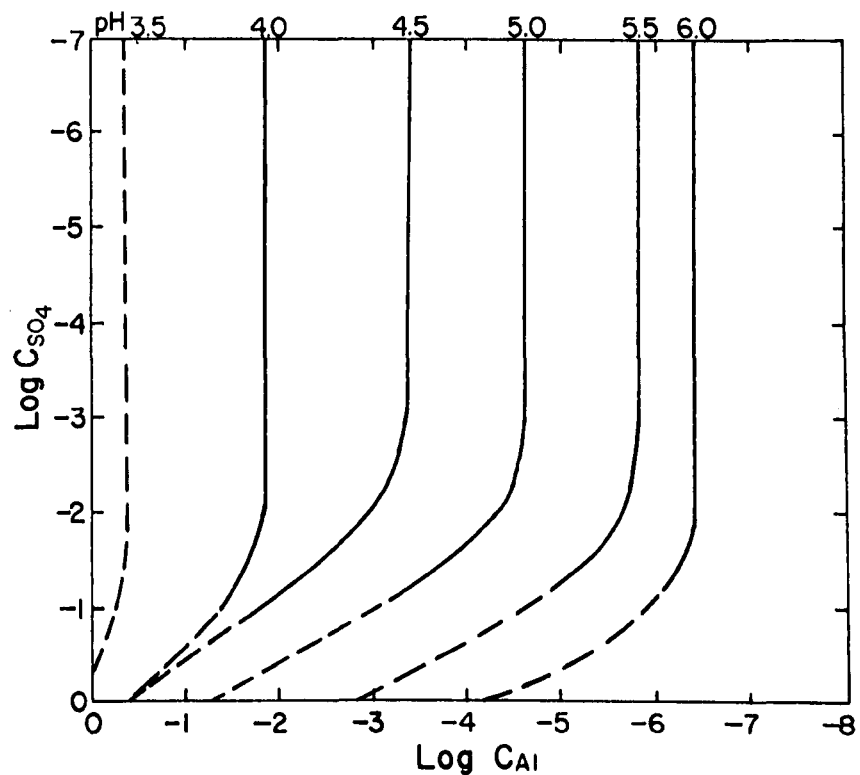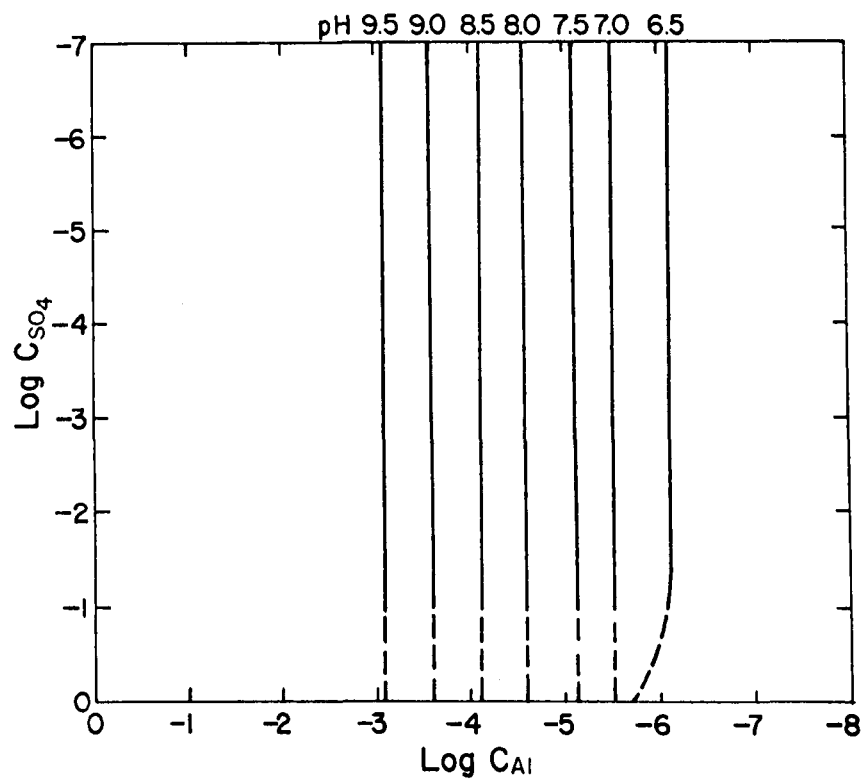
Figure 9. Solubility of microcrystalline gibbsite as a function of sulfate concentration. Ionic strength 0.10 for 25°C, and 1 atmosphere total pressure (from Roberson and Hem, 1969).

to the relationship expressed in Figure 9. From Table 19, the concentrations of aluminum actually measured in the interstitial and well water samples at location J4, are 60 and <0.20 mg/l respectively. The concentration of aluminum in the well water sample may agree with the value predicted from Figure 9, but it cannot be verified because the detectable limit is higher than the predicted value. The concentration of aluminum in the interstitial water sample (60 mg/l), however, is 4.5 times less than the predicted value (270 mg/l). It might be postulated that 60 mg/l was all the aluminum available for solubilization; however, a return to Table 19, location J4, shows that 3900 μg/g of aluminum was present in the (soil) substratum. The soil moisture content was approximately 20 percent (see Table 3), and if all the aluminum were to dissolve, calculations indicate that a 624 mg/l aluminum solution would result--more than enough to meet the predicted value.

Thus, it appears that other factors besides the sulfate concentrations affected the solubility of aluminum in the interstitial water samples from location J4. The type of matrix in which the aluminum is chemically bound, incomplete soil-water contact, and competition with other metal ions for the sulfate ligand are factors which might affect aluminum solubility. Further analysis of the data indicates that competition for the sulfate ligand may indeed be the limiting factor. The competition for the sulfate ligand by other metal ions, such as magnesium, calcium, barium, and lead, may force the hydroxide solubility to be limiting for aluminum. The following calculations tend to verify this point:

$$\{Al^{+3}\}\{OH^{-1}\}^3 = K_{so} = 10^{-32} \tag{1}$$

$$\{H^+\}\{OH^-\} = 10^{-14} \tag{2}$$

$$\{Al^{+3}\}_{apparent} = 2.2 \times 10^{-3} \text{ moles/liter} \tag{3}$$

$$pH = -\log \{H^+\} = 4.3 \tag{4}$$

$$\{OH^-\} = 10^{-14}/\{H^+\} \tag{5}$$

$$\{OH^-\} = 10^{-14}/5.0 \times 10^{-5}$$

$$\{OH^-\}_{apparent} = 1.99 \times 10^{-10}$$

Substitution of 3 and 5 into 1 yields:

$$\{Al^{+3}\}_{apparent} \{OH^-\}^3_{apparent} = 1.7 \times 10^{-32} \tag{6}$$

The calculated solubility (6), using the measured values of aluminum and pH from Table 19, is close to the theoretical solubility constant (1). These data suggest that the system is in equilibrium and it is the hydroxide solubility that is controlling the aluminum concentration in the interstitial water and, as stated earlier, the predominate form at a pH of 4.3 would be Al^{+3}.

It should be pointed out that the characteristics of the samples from location J4 are not necessarily indicative of all locations at plant J or L. For example, at location J3 (Table 19), the aluminum concentration in both the interstitial water and well water exceeds its hydroxide solubility ($K_{apparent}$ = 9.4 x 10^{-25} and $K_{apparent}$ = 2.3 x 10^{-18} respectively) indicating either the occurrence of complexation or the presence of a system not in equilibrium, while for location L6, calculations indicate the concentration of aluminum in the interstitial water has not reached the limit imposed by the hydroxide solubility constant ($K_{apparent}$ = 6.0 x 10^{-34}), even though the concentration is higher than at location J3. This suggests that the acidic conditions (pH 4.1) at location L6 are allowing more aluminum in solution, but not enough is present to attain maximum equilibrium concentrations.

It should be noted that the simplistic approach used in the equilibrium discussions herein minimizes the effects of ionic strength, temperature, and medium on K_{so}.

Iron

There are two oxidation states in which iron normally occurs, the ferrous form (Fe^{+2}), and the ferric form (Fe^{+3}). Under oxidizing conditions and near neutral pH (5 to 8), iron is precipitated as the highly insoluble (K_{so} = 6.0 x 10^{-38}) ferric hydroxide, $Fe(OH)_3$. The solubility of iron is dependent on pH and oxidation-reduction potential (Eh). Its chemical behavior is also influenced by its ability to form complex ions with sulfides, sulfates, oxides, hydroxides, chlorides, fluorides, carbonates, phosphates, and organic material. The ferrous form (Fe^{+3}) forms weaker complexes than the ferric, and forms few stable inorganic complexes. It is the predominate oxidation state in reducing and/or acidic environments.[19]

Figure 10 illustrates the relationships between pH and Eh that define the conditions under which specific ionic species of iron dominate. This is known as an Eh-pH, or stability field, diagram. Specific areas where species dominate are delineated by thermodynamic computations, in which the Nernst equation is the fundamental relationship that establishes the dividing line between oxidized and reduced species. A thorough discussion of Eh-pH diagrams is given by Garrels,[20] where this familiar form of the Nernst equation is utilized.
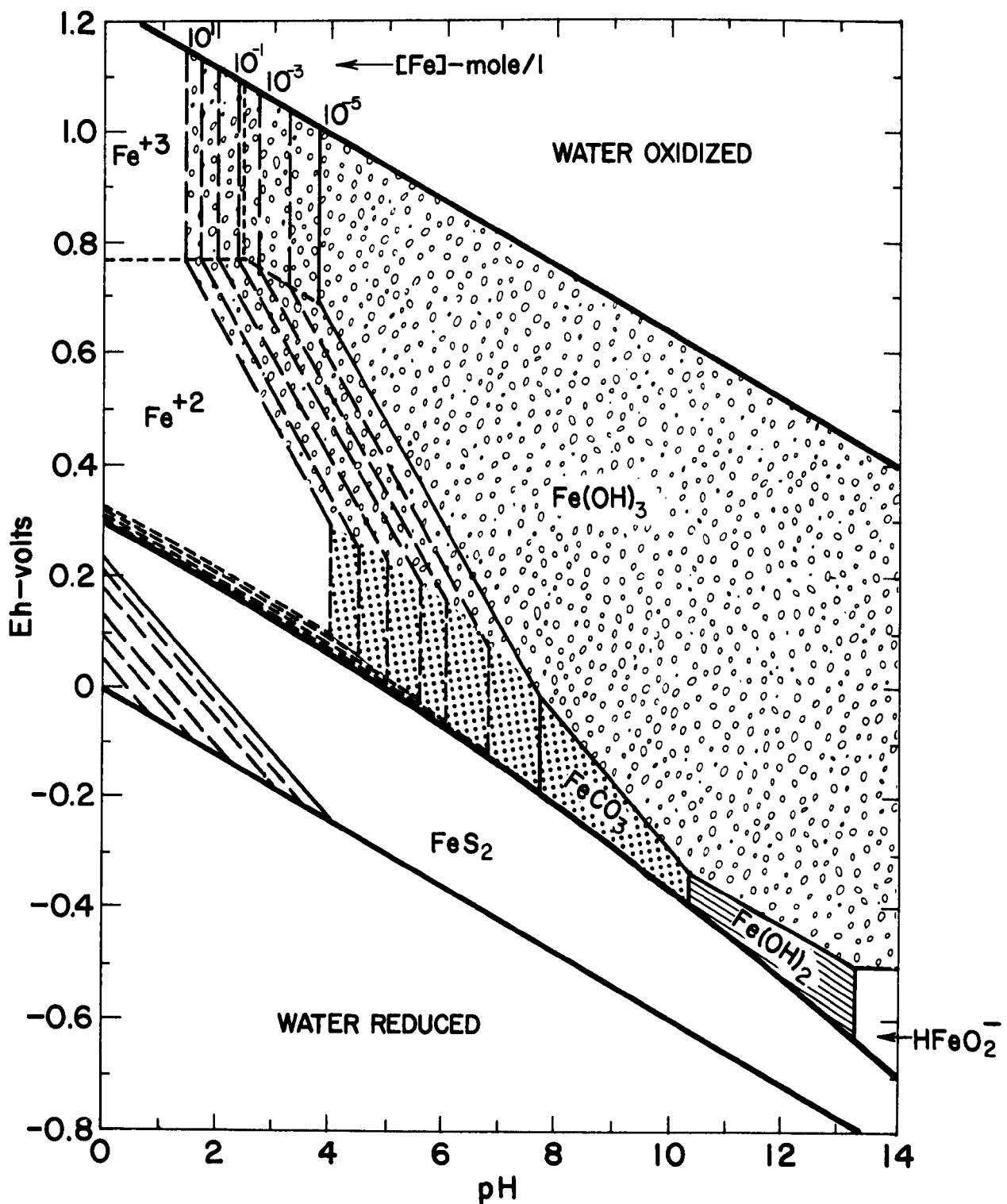
Figure 10. Solubility of iron in relation to pH and Eh at 25° C., and 1 atm. Total dissolved sulfur 10^{-4} M: bicarbonate species 10^{-2} M (Hem, 1960)

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$$Eh = E^O + \frac{RT}{nF} \ln \{\frac{ox}{red}\}$$

Where:

Eh   = oxidation reduction potential

E^O = standard oxidation potential for half-cell reaction

R   = universal gas constant (1.987 calories/degree mole)

T   = temperature in degrees Kelvin

n   = number of electrons involved in chemical reaction

F   = Faraday constant (96,484 coulombs)

{ox}   = activity of oxidized ions

{red}   = activity of reduced ions

This equation can be used with thermodynamic data to generate the stability field diagrams of the type illustrated in Figure 10, for any of the various metal equilibriums one would care to examine. They were not presented for all the metal ions measured during this study because the variable nature of the data, both between and within sampling locations, would require a voluminous exposition.

As Figure 10 indicates, ferrous iron (Fe^{+2}) can be produced by the oxidation of pyrite (FeS_2), reduction of ferric species (Fe^{+3}), dissolution of ferrous hydroxide ($Fe(OH)_2$), or dissolution of siderite ($FeCO_3$). Additionally, the kinetics and formation of intermediate species may affect the formation of ferrous iron. The FeS_2 equilibrium is a feasible condition because of the high concentration of sulfate present in several of the sampling wells, the low pH values of the interstitial water, and the presence of pyrite (FeS_2) in coal. In Figure 11, the stability fields for five different sulfur species are illustrated. The dotted line indicates the ferrous-ferric boundary and demonstrates that sulfate is the predominate form of sulfur stable in the ferric region.

At a pH less than about 8.0, and under reducing conditions, the H_2S and S^O forms predominate. Oxidation of H_2S will produce the S^O form, which upon further oxidation, forms SO_4^{-2}. The H_2S form has the notorious "rotten egg" odor, which was noticeable during the collection of water samples from several of the wells at both plants. This qualitative analysis indicates the potential presence of a reducing environment in the ground water at certain sampling locations. It was most noticeable in samples collected from wells downgradient from the ash disposal areas, and least, if at all, in the upgradient or background wells. In addition, the odoriferous wells upon inspection were found to have a black deposit (possibly FeS) on the 1.27 cm rigid PVC pipe installed inside the well. This deposit was most concentrated at the interface of the groundwater surface with the atmosphere and may be a result of
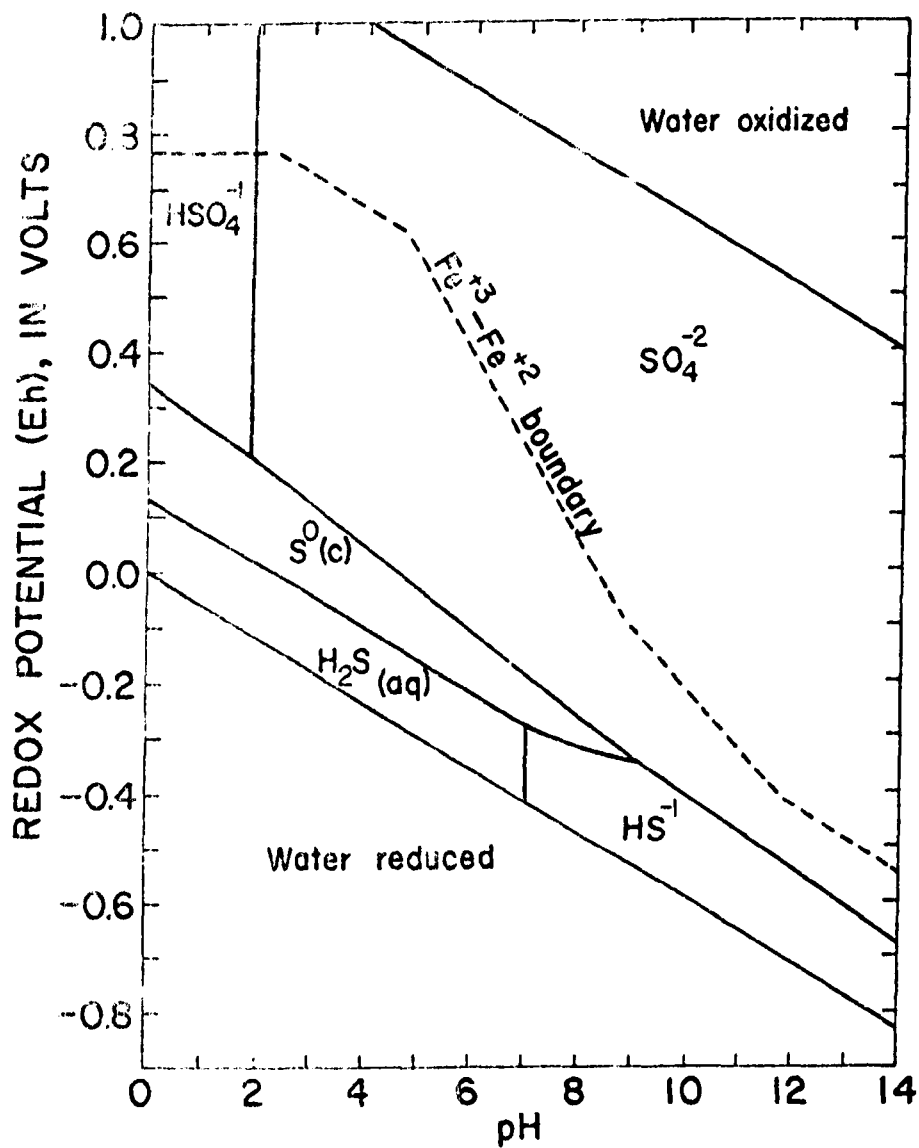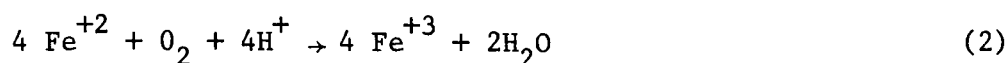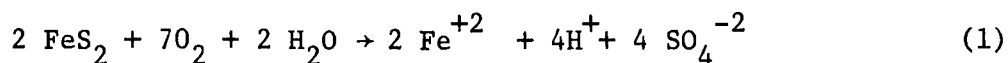
Figure 11. Fields of stability of sulfur species
likely to occur in natural water (Hem, 1960)

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oxidation within the well casing. Solubility product calculations were performed for the FeS, K_{so} = 2.6 x 10^{-22} equilibrium, using iron and sulfide data from Tables 17 and 18. The calculations indicated that concentrations of iron and sulfide in samples from three locations at plant J (J5A, J4A, and J7A) exceeded the solubility product for FeS, and at plant L, wells L3, L3A, L4, L5, L6, L6A, L7, L8, L8A, L9A, and L10 exceeded the FeS solubility. The water sampled from these wells was thus unstable (not at equilibrium) with regard to the ferrous sulfide equilibrium, and precipitation of FeS could be expected, which may account for the black deposits on the aforementioned PVC pipe.

A potentially important aspect of the speciation chemistry of iron-sulfate equilibrium, as it relates to coal-ash leachate, is the oxidation of pyrite present in the ash disposal area. In a study by Cox, et al,[21] to characterize the runoff from coal storage piles, the production of ferrous iron and acidity from pyrite oxidation was shown to be the major factor effectuating the solubilization of iron. The oxidation of pyrite results in the production of ferrous iron and acidity via the following reactions:

$$2\ FeS_2 + 7O_2 + 2\ H_2O \rightarrow 2\ Fe^{+2} + 4H^+ + 4\ SO_4^{-2} \tag{1}$$

$$4\ Fe^{+2} + O_2 + 4H^+ \rightarrow 4\ Fe^{+3} + 2H_2O \tag{2}$$

At this stage the ferric iron can either hydrolyze to form insoluble ferric hydroxide,

$$Fe^{+3} + 3\ H_2O \rightarrow Fe(OH)_3 + 3H^+, \tag{3}$$
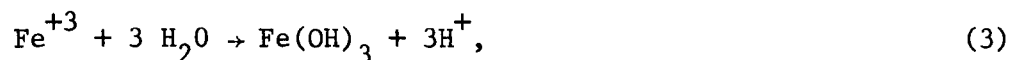
or oxidize pyrite directly producing more ferrous iron and acidity:

$$FeS_2 + 14\ Fe^{+3} + 8\ H_2O \rightarrow 15\ Fe^{+2} + 2\ SO_4^{-2} + 16\ H^+ \tag{4}$$
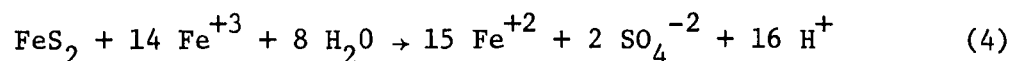
The speciation chemistry of the iron-sulfate equilibrium is a complex process, and may be worthy of further study in relation to ash pond leachate due to its potential for decreasing pH and subsequent solubilization of trace metals.

Copper, Lead, Zinc, and Other Metals

At pH values below 7.3, the predominate form of soluble copper is Cu^{+2}, while at higher values $CuOH^+$ is most prevalent. Copper is strongly complexed with inorganic ligands such as carbonates, hydroxides, and

chlorides and with organic matter. The low pH values observed in the interstitial water samples indicate the Cu^{+2} form would prevail, while at the higher pH values in the well samples it would precipitate as $Cu(OH)_2$ or $Cu_2(OH)_2CO_3$. In solution at the higher pH values, it could be in the form of $Cu(OH)_4^=$, or $CuO_2^=$.

Solubility calculations were performed for the copper equilibriums CuS, $CuCO_3$, and $Cu(OH)_2$ using data in Tables 17 and 18 and the stability constants listed in Table 20. The calculations indicated that all well water samples that contained measurable amounts of copper and sulfide were unstable (nonequilibrium) with regard to Cu^{+2}. The solubility limiting equilibrium was found to be CuS as the following calculation, using the data from location L6A, demonstrates:

$$\{Cu^{+2}\} = 0.02 \text{ mg/l} = 3.1 \times 10^{-7} \text{ mole/liter}$$

$$\{S^=\} = 0.11 \text{ mg/l} = 3.4 \times 10^{-6} \text{ mole/liter}$$

$$\{CO_3^=\} = 1.62 \text{ mg/l} = 2.7 \times 10^{-5} \text{ mole/liter}$$

$$\{OH^-\} = 3.1 \times 10^{-7} \text{ mole/liter}$$

Sulfide solubility:

$$\{Cu^{+2}\}\{S^=\} = 1.0 \times 10^{-12}$$
exceeds solubility product, system unstable, precipitation occurring

Hydroxide solubility:

$$\{Cu^{+2}\}\{OH^-\}^2 = 2.9 \times 10^{-20}$$
equal to solubility product, system stable, in equilibruim

Carbonate solubility:

$$\{Cu^{+2}\}\{CO_3^=\} = 8.3 \times 10^{-12}$$
less than solubility product, system unstable, available Cu^{+2}

The calculations indicate that the water sample from L6A is in equilibrium with regard to $Cu(OH)_2$ and not at equilibrium with regard to $CuCO_3$. Above a pH of approximately 7.5 such as at many of the sampling locations at plants J and L, the $Cu(OH)_2$ solubility is exceeded and precipitation occurs.

In waters with pH values below 6.0, the predominate ionic species of lead is the divalent Pb^{+2} form. The solubility of lead can be influenced by complexation with hydroxides, chlorides, and organic material.

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In association with sulfates, carbonates, or phosphate anions under alkaline conditions, soluble compounds will readily be formed resulting in the precipitation of lead. Solubility calculations for the PbS equilibrium, K_{so} = 7.0 x 10^{-29}, using data from Tables 17 and 18, indicate that the water sample collected from sampling well L9A at plant L is chemically unstable with regard to the PbS equilibrium, $K_{apparent}$ = 7.2 x 10^{-13}, and precipitation is occurring.

Where:

K_{so} = solubility product at equilibrium

$K_{apparent}$ = solubility product of the measured solution

TABLE 20. SOLUBILITY PRODUCT CONSTANTS FOR IRON, ALUMINUM, COPPER, LEAD, AND ZINC COMPOUNDS

Compound	Solubility product[a]
FeS	2.6 x 10^{-22}
Fe(OH)$_2$	5.0 x 10^{-15}
Fe(OH)$_3$	6.0 x 10^{-38}
FeCO$_3$	2.0 x 10^{-11}
Al(OH)$_3$	1.0 x 10^{-32}
CuS	7.9 x 10^{-37}
Cu(OH)$_2$	1.5 x 10^{-20}
CuCO$_3$	2.3 x 10^{-10}
PbS	7.0 x 10^{-29}
Pb(OH)$_2$	1.5 x 10^{-17}
PbCO$_3$	7.2 x 10^{-14}
ZnS	7.9 x 10^{-26}
Zn(OH)$_2$	5.0 x 10^{-17}
ZnCO$_3$	2.0 x 10^{-11}

[a]Adapted from: Sillen, L. G., and A. E. Martell, "Stability Constants of Metal-Ion Complexes." London: The Chemical Society, Burlington House, W. E. (1964).

Generally, the dominating ionic zinc species in solutions of low pH (<7.0) is Zn^{+2}, with the anionic forms $Zn(OH)_3^{-1}$ and $Zn(OH)_4^{-2}$ being the soluble species at high pH (11.0).[23] The carbonate is the most soluble form, $K_{so} = 2.0 \times 10^{-11}$, with the hydroxide following at $K_{so} = 5.0 \times 10^{-17}$. The least soluble zinc compound is the ZnS form with $K_{so} = 7.9 \times 10^{-26}$. Solubility calculations, again using the data in Tables 17 and 18, indicate that in the well samples where measurable zinc and sulfide concentrations were found (L4, L8A, L11, J5A, and J7A), unstable conditions ($K_{apparent} > K_{so}$) with regard to zinc solubility were present, and precipitation was occurring. With measured pH values less than 11, no hydroxo-zinc complexing would be expected; thus, any dissolved zinc species measured in the sampling wells would be predominately the divalent Zn^{+2}.

The concentrations of beryllium, cadmium, chromium, mercury, and selenium measured in the sampling wells at plants J and L, as Tables 17 and 18 indicate, were all less than or near their analytical detection limits. This contrasts with interstitial water analysis in Tables 15 and 16, where concentrations of cadmium, chromium, and mercury were present in measurable quantities.

Arsenic was present in measurable quantities in samples collected from wells J2, J5A, L6, L7, L8A, and L9A. Boron was present in all well samples collected from plants J and L, on March 9 and February 22, respectively. In addition, the concentration of boron in samples from wells J2, J7A, L6A, L8, L8A, L9, and L9A were found to exceed the EPA's quality criteria for water[16] of 0.750 mg/l for irrigation (at one location, L8A, by more than 26 times).

As a result of the comparisons between the data listed in Tables 15, 16, 17, 18, and 19, the solubility calculations that were made using these data, and observations made from the selected Eh-pH diagrams, several trends have become apparent. First, in general these data indicate that the concentrations of metals in the interstitial water samples were higher than concentrations measured for the same constituents in samples taken from the sampling wells; second, interstitial water samples tended to have lower pH values than well samples, perhaps indicating a more reducing environment in the former; third, the predominate form of iron, copper, zinc, and lead ions present in solution was as the divalent species, while aluminum was in the trivalent form; and fourth, the metal-sulfide equilibrium played a dominant role in controlling the solubility of metals (with the exception of aluminum, which was limited by the hydroxide solubility due to its +3 valence state).

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SECTION 7

HYDROLOGY

The ash disposal areas at plants J and L are located adjacent to reservoirs in which the water levels in winter are drawn down several feet for flood control. As a result, groundwater discharge to the river is greatest during this period. During the late spring and summer months the reservoir levels are higher, with minor daily variations in elevation. The higher reservoir levels during this time reduce the water table gradient toward the river, resulting in a decreased rate of groundwater discharge to the reservoir.

Although seasonal variations in groundwater flow beneath the ash disposal areas at both plants occur, the net groundwater movement is to the adjacent reservoir. In Figures 12 and 13, water table elevations in the sampling wells at plants J and L, measured respectively on March 9 and February 22, 1977, are plotted along with the average elevation of the associated reservoir levels observed in 1977 (far right horizontal axis). The sampling locations are plotted on the graph from left to right in order of decreasing distance from the river. An estimated line of best fit through the points (dashed line) indicates a riverward gradient and thus, the net direction of groundwater flow at the two disposal areas is toward the river.

Figure 14, is a geologic section and water table profile of the ash pond area at plant J, showing the water table gradient toward the river. The figure also shows a condition that may be common to ash disposal areas where the perimeter dike is constructed of a low permeable material. Subsurface water might be impounded behind the closed perimeter dike continuous with the pond's bottom, both of low permeability (which is the current design). After raising the dike several times to accommodate additional ash, a large bowl with low permeability sides would result. Water in the bottom of this bowl would tend to be passed around by ambient groundwater flows and mixing would be minimal. Static water in this zone would undoubtedly be of poorer quality than water merely passing through coal ash once, acquiring some characteristics of ash leachate, and then flowing on to mix with unaffected ground or surface waters or acted on by attenuation dynamics of the surrounding subsoils. The water in the dead space of the bowl would have a prolonged ash contact time, allowing for maximum dissolution of ions and the formation
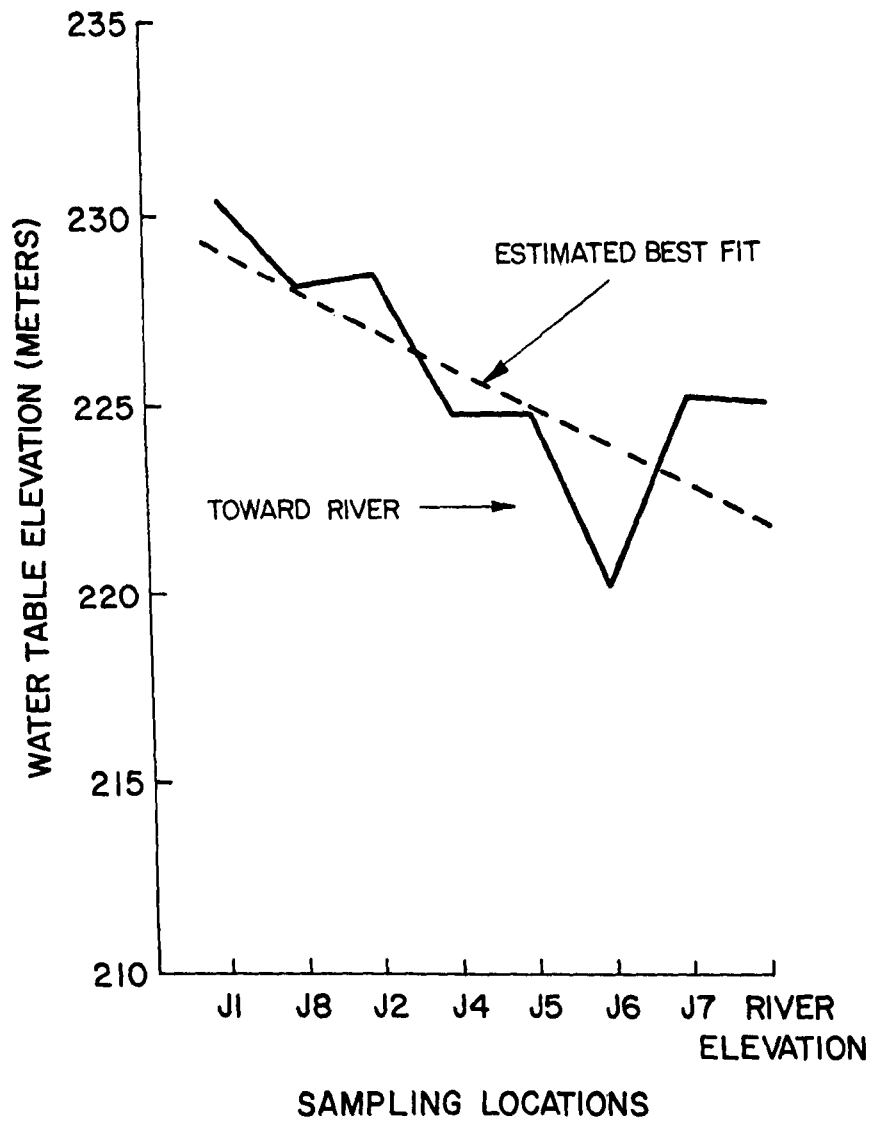
Figure 12.   Groundwater table elevation in sampling
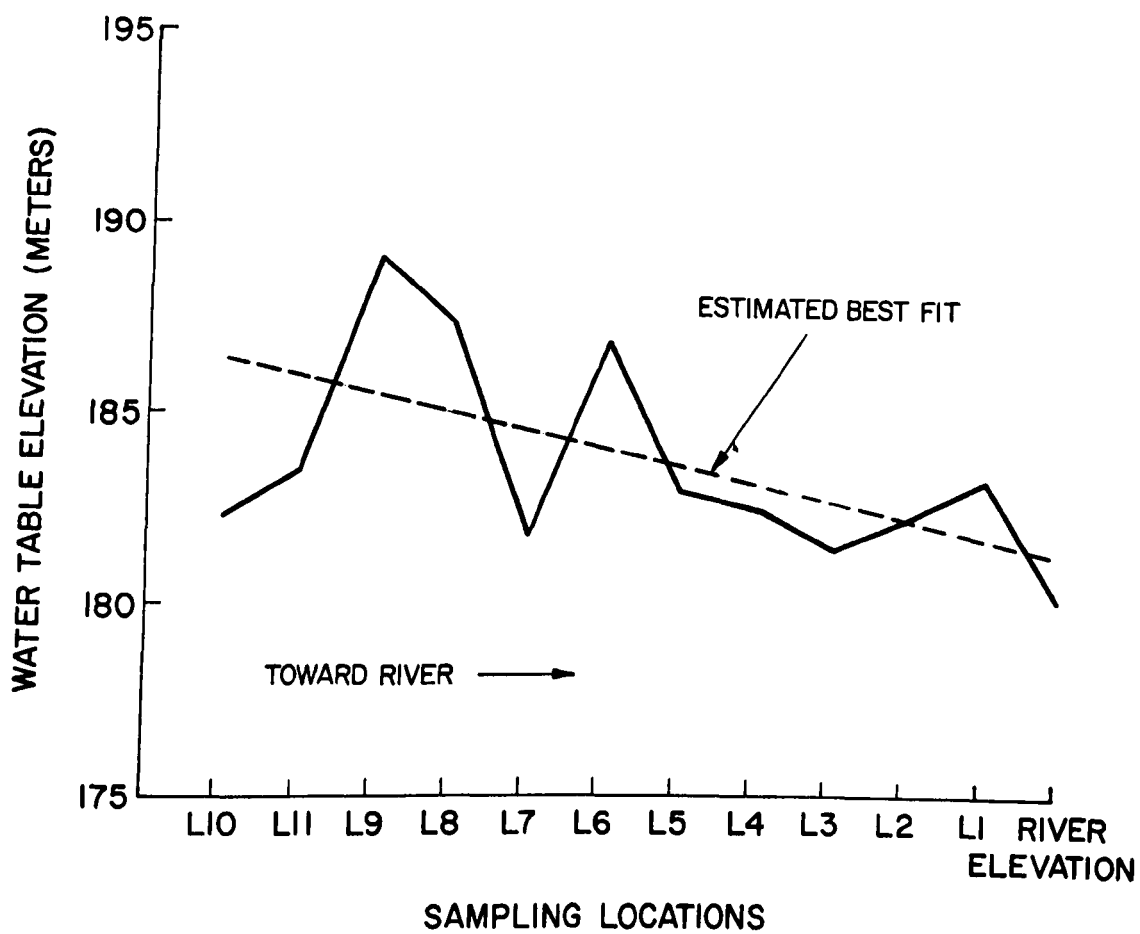   wells at plant J, March 9, 1977

Figure 13. Groundwater table elevation in sampling wells at plant L, February 22, 1977
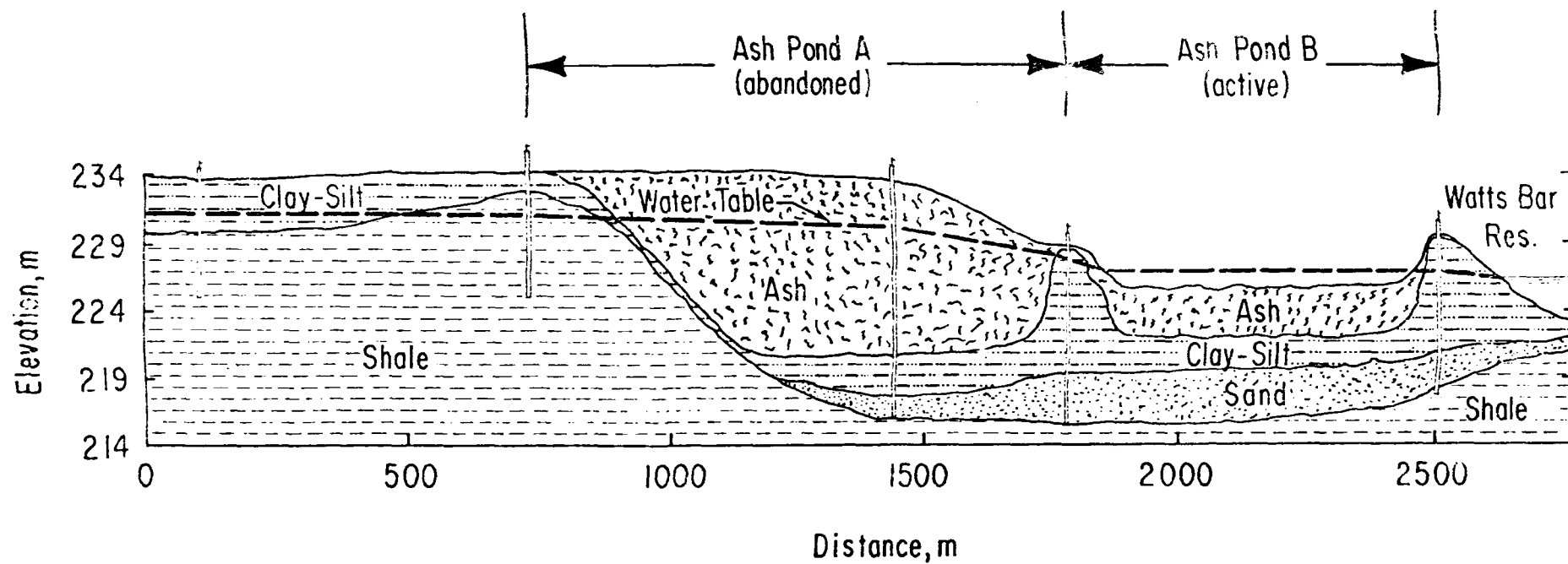
Figure 14. Cross section of ash disposal area at plant J.

of an acidic, highly reducing environment. The potential impact of a
hidden condition such as this on local groundwater quality may warrant
determination of whether the phenomenon exists.

In Figure 13, the water table elevations in the abandoned ash ponds
at plant L (locations L8 and L9), are higher than the surrounding water
table elevations, including the background locations. This is the type
of data that would indicate a leachate bowl. Inflows into the bowl
would not be able to equilibrate with the surrounding water table if
the bowl's sides and bottom were of low permeability (permeability of
the soil below the deposited ash at locations L8 and L9, averages 10^{-8}
cm/sec), and the water level inside the bowl would tend to rise above the
ambient groundwater table. The phenomena is speculative at this point;
more data would be required for confirmation.

The permeability of the ash pond dike and subsoil beneath the ash
at plant J, was measured in the laboratory in both the vertical and
horizontal directions. The vertical permeabilities ranged from 6.3
x 10^{-8} to 1.3 x 10^{-6} cm/sec (see Table 3), while horizontal permeabi-
lities ranged from 7.4 x 10^{-8} to 7.4 x 10^{-5} cm/sec. Subsoil samples
with lower permeabilities contained larger fractions of clay,[*] while
samples with higher permeabilities contained larger fractions of sand
(Table 3). The density of the soils ranged from 1.42 to 1.79 g/cm^3.
Sample density increased with sand content. Moisture content of plant J
soils ranged from 13.9 to 25.7 percent. Generally, the soil samples
from plant J, consisting mostly of sand with a moderate moisture
content, were of moderate permeability and medium to high density.

At plant L, vertical permeabilities ranged from 6.9 x 10^{-9} to 3.0 x
10^{-4} cm/sec, and horizontal permeabilities ranged from 5.7 x 10^{-9} to
1.5 x 10^{-4} cm/sec (see Table 7). The sample with 10^{-4} cm/sec permea-
bility was an ash sample collected at location L8. In general, soil
samples from plant L having the lowest permeability also contained
a large percentage of clay, while the higher permeable samples con-
tained more sand (with the exception of L8, which was ash at 0.8
meters). The densities of the soil material at plant L ranged from
1.03 to 1.77 g/cm^3; moisture ranged from 17.5 to 57.5 percent. Soil
samples from plant L, consisting mostly of highly compacted silty
clay, were generally of low permeability and high moisture content.

Figure 15 shows an idealized cross section of the ash pond dike
at plant J; the superimposed values are mean horizontal permeabilities
(K) as measured on samples of the clay-silt and sand strata collected
at locations J4, J5, and J6. The permeabilities of both strata were

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[*]Textural classification of soil fractions as per American Society for
 Testing and Materials designation, where: clay <0.005 mm, silt 0.005
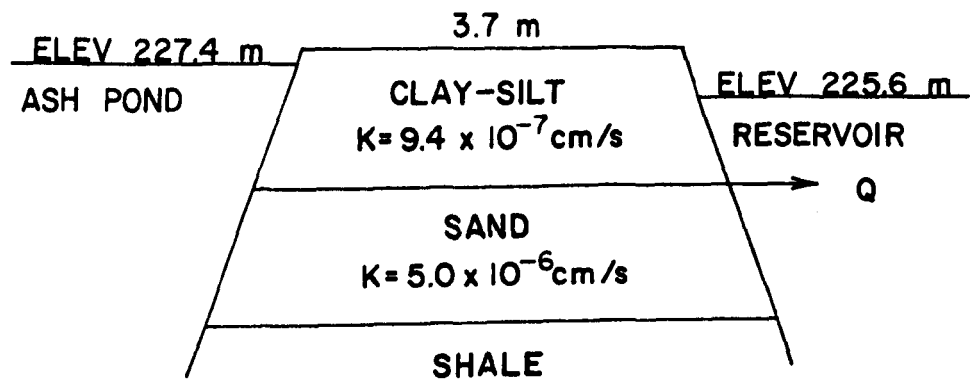 to 0.074 mm, sand 0.074 to 4.75 mm, and gravel >4.75.[24]

Figure 15. Cross section of the clay-silt and sand strata at plant J, showing mean horizontal permeabilities (K) for locations J4, J5, and J6.

very low, with the permeability of clay-silt stratum being the lower
of the two. Using a modification of Darcy's law, the flow through a
discrete saturated substratum, such as the clay-silt substratum, is:

$$Q = TIW \qquad (5)$$

Where: T = coefficient of transmissibility, volume per unit time
             per unit length of stratum width

          I = hydraulic gradient, unitless

          W = width of the vertical section through which the flow
             occurs, unit length

and, $T = Km$ (6)

Where: K = average coefficient of permeability from top to bottom of
           stratum, volume per unit time

        m = thickness of stratum, unit length

For example, in the clay-silt stratum at locations J4, J5, and J6,
the average horizontal coefficient of permeability (from Table 3) was
9.4×10^{-7} cm/sec, and the average thickness of the stratum was 9.8
meters. Conversion of the coefficient of permeability to liters per
day gives 0.83 liters per day, per square meter. Substitution into
equation 6 produces a coefficient of transmissibility of 7.9 liters
per meter of stratum width for the clay-silt stratum. The width of
the vertical section through which the flow occurs (W) corresponds to
the linear length of the ash pond dike adjacent to the reservoir, and
was measured to be approximately 1718 meters. The hydraulic gradient
(I) of the water table was 0.5. Substituting into equation 5:

$$Q = (7.95 \frac{\text{liters}}{\text{day-m}})(0.5)(1718 \text{ m})$$

$$Q = 6824 \text{ liters per day (1803 gallons per day)}$$

gives the total groundwater flow passing through this cross-sectional
area of the clay-silt stratum. A similar calculation for the sand
stratum, where the average permeability was 5.0×10^{-6} cm/sec and the
average stratum thickness for locations J4, J5, and J6 was 2.5 meters,
indicates a total flow through this stratum of 9265 liters per day.
Combining the flows through the clay-silt and sand strata gives a total
flow through both strata of 16,090 liters per day. This compares to a
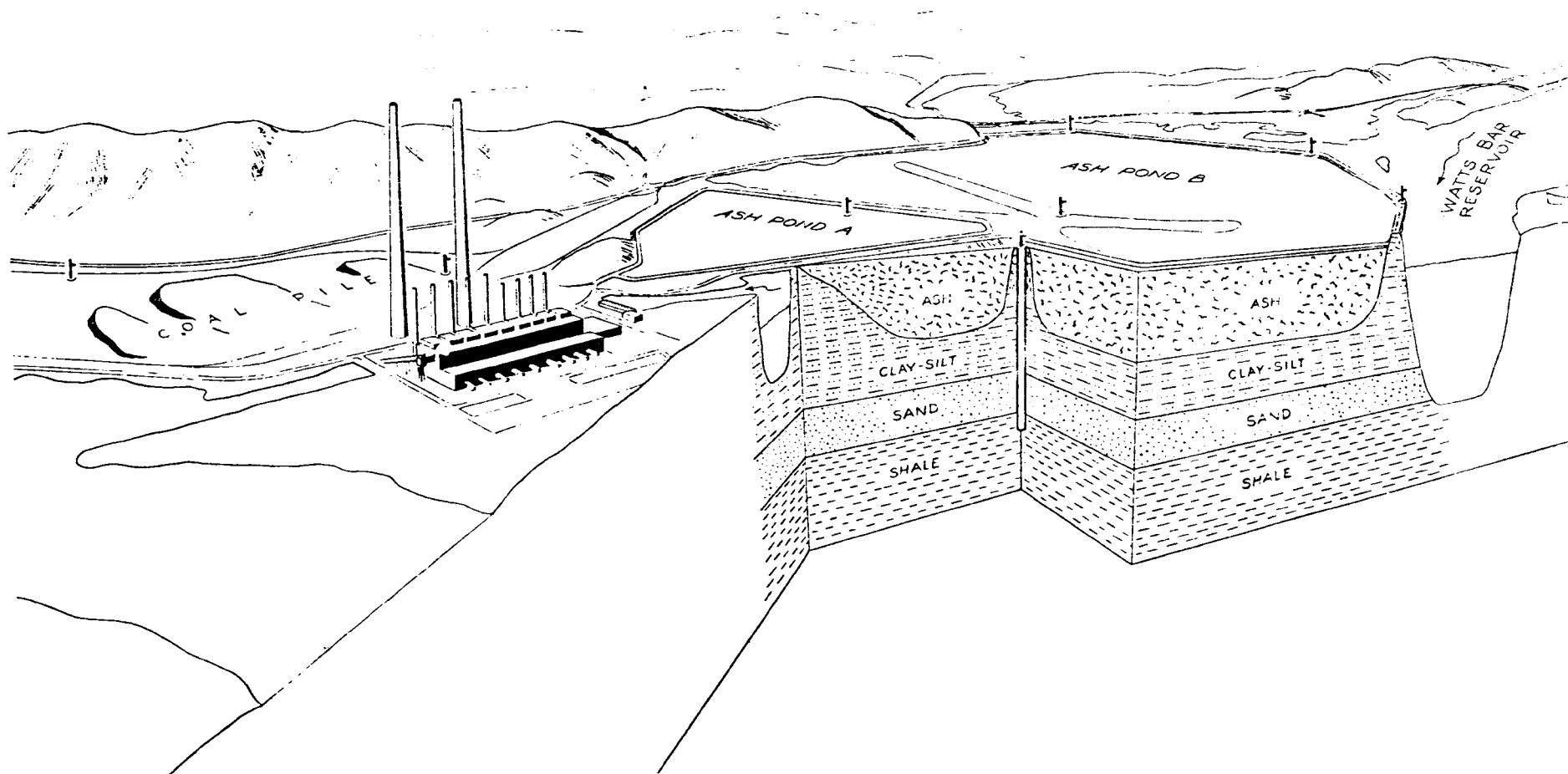surface discharge from the ash disposal pond at plant J, of from 59.4

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Figure 16. Cross section of substratum below plant J's ash disposal area.

TABLE 21. FLUX OF SELECTED CONSTITUENTS THROUGH SUBSTRATA AT PLANT J

Constituent	Concentration in clay-silt mg/l	Clay-silt flux g/24 hr.	Concentration in sand mg/l	Sand flux g/24 hr.	Total ground water flux g/24 hr.	Concentration in ash pond discharge mg/l	Ash pond discharge flux g/24 hr.
Aluminum	98	671	3.2	30	701	1.4	142,088
Calcium	301	2054	98	914	2968	22.5	2,283,567
Cadmium	<0.002	0.013	0.001	9.26	9.27	0.0015	152
Chromium	<0.005	0.034	0.007	0.064	0.098	<0.005	–
Copper	0.113	0.773	0.053	0.494	1.26	0.06	6089
Iron	506	3457	38.9	360	3817	2.35	238,505
Magnesium	79.3	541	14.3	132	673	3.1	314,624
Nickel	0.18	1.27	0.05	0.494	1.76	<0.05	–
Lead	0.036	0.247	0.032	0.293	0.540	<0.01	–
Sulfate	1883	12,851	153	1420	14,271	88.5	8,982,033
Zinc	0.90	6.14	0.110	1.02	7.16	0.05	5074

to 16.2 million liters per day during 1976. It should be noted that the flow (Q) is dependent on the hydraulic gradient (I), while transmissibility (T) and width (W) remain constant. The hydraulic gradient fluctuates with time according to the elevation of the adjacent reservoir, subsequently increasing or decreasing subsurface discharge.

Figure 16 shows a cross section of the ash pond and substrata at plant J and illustrates the spatial relationships between the ash, clay-silt, sand, and shale materials. The total flux of aluminum, calcium, cadmium, chromium, copper, iron, magnesium, nickel, lead, sulfate, and zinc were calculated for the combined flows in the clay-silt and sand strata illustrated in Figure 16 (detailed vertical profiles of the substratum at plants J and L are presented in appendix B). These fluxes are compared (see Table 21) to the total mass of the same constituents discharged from the ash pond's surface effluent. The fluxes in the groundwater were obtained by averaging the concentrations measured (Table 17) at locations J4A, J5A, and J6A for the clay-silt flux, and J4, J5, and J6, for the sand flux. The product of the average concentrations and the clay-silt and sand flows calculated above, gives the 24-hour flux in each strata. The summation of these two fluxes provides the total groundwater flux of each constituent. These calculations were made assuming concentration homogeneity within each stratum, and disregarding attenuation processes. The mass constituent discharged from the ash pond via the surface effluent was determined by averaging the concentrations measured in two samples collected during 1976, and multiplying this value by the average 1976 daily effluent discharge.

Comparison of the columns in Table 21 shows that the total flux of aluminum, calcium, copper, iron, magnesium, nickel, and zinc was greatest in the clay-silt stratum, while in the sand stratum cadmium, chromium, lead, and sulfate flux were greatest. Comparison of the total flux through both strata with the total mass from the ash pond surface discharge shows that the flux through both strata combined was a small percentage (less than 1 percent) of the total mass discharged from the ash pond surface effluent. Albeit, with the exception of copper, constituent concentrations were higher in the groundwater than in the ash pond discharge. The significant factor effectuating the total mass in the ash pond surface discharge as compared to the groundwater flux was flow. Flow from the ash pond during 1976 averaged 1.0×10^8 liters per day, whereas, the calculated total flow through the combined clay-silt and sand strata was 16,909 liters per day--only 0.016 percent of the ash pond discharge.

The physical measurements of the predominantly clay-silt substrata necessary to quantify groundwater flow at plant L, were not made because the large area involved would have required numerous and costly

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exploratory borings. The low permeabilities measured in the clay-silt substratum (less than 10^{-8} cm/sec) would, however, tend to minimize leachate movement if a generalization can be made from the calculations made with plant J's data.

As previously mentioned in the geological description (section 3), the primary material underlying the ash disposal area at plant L, is a mixture of alluvial and terrace deposits. In certain areas, such as location L4, these deposits consist of unconsolidated materials containing as much as 40 percent gravel. This type of material could not be collected in a Shelby tube for the determination of permeability. In order to determine the groundwater flow in this material, two wells were installed 1.5 meters apart with perforations transecting the pervious gravel material vertically. The difference in hydraulic head between the two wells was 3.1 meters. Nineteen liters of a saturated sodium chloride solution were then injected into the upgradient well. The downgradient well was then pumped and samples collected periodically for the measurement of conductance and chloride. Figure 17 shows a plot of chloride concentration and conductivity versus pumping time for the downgradient well. The figure indicates that the chloride concentration increased substantially within 24 minutes and reached a maximum at approximately 75 minutes. The plot of conductance versus time presents a shape nearly identical to the chloride plot, with a slight decrease in the beginning and a slower return to baseline being the only differences. The slow return to baseline of both parameters is thought to be a result of lateral dispersion of the solute.

Under the conditions of this test, if the time required to reach maximum concentration is assumed to be the flow time between the two wells, then 0.03·cm/sec would be the velocity of the ground water in this material. Using Darcy's law, this velocity converts to a permeability (K) of 0.14 cm/sec, substantially higher than values measured in the laboratory on other subsurface materials. At plant L, there are two locations (L4 and L5) where this highly permeable alluvial material was found. The water quality at both these locations, however, was not significantly different from the background wells 10CW and 11CW. This seemed to indicate that attenuation processes in the overlying clay-silt stratum were precluding ash leachate from infiltrating the porous alluvial material. However, if the ash leachate should ever enter the highly permeable alluvium, minimal attenuation, characteristic of this type of material, would allow the rapid lateral spread of ash leachate.

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Figure 17

TRACER TEST OF CONDUCTANCE AND CHLORIDE

vs

PUMPING TIME
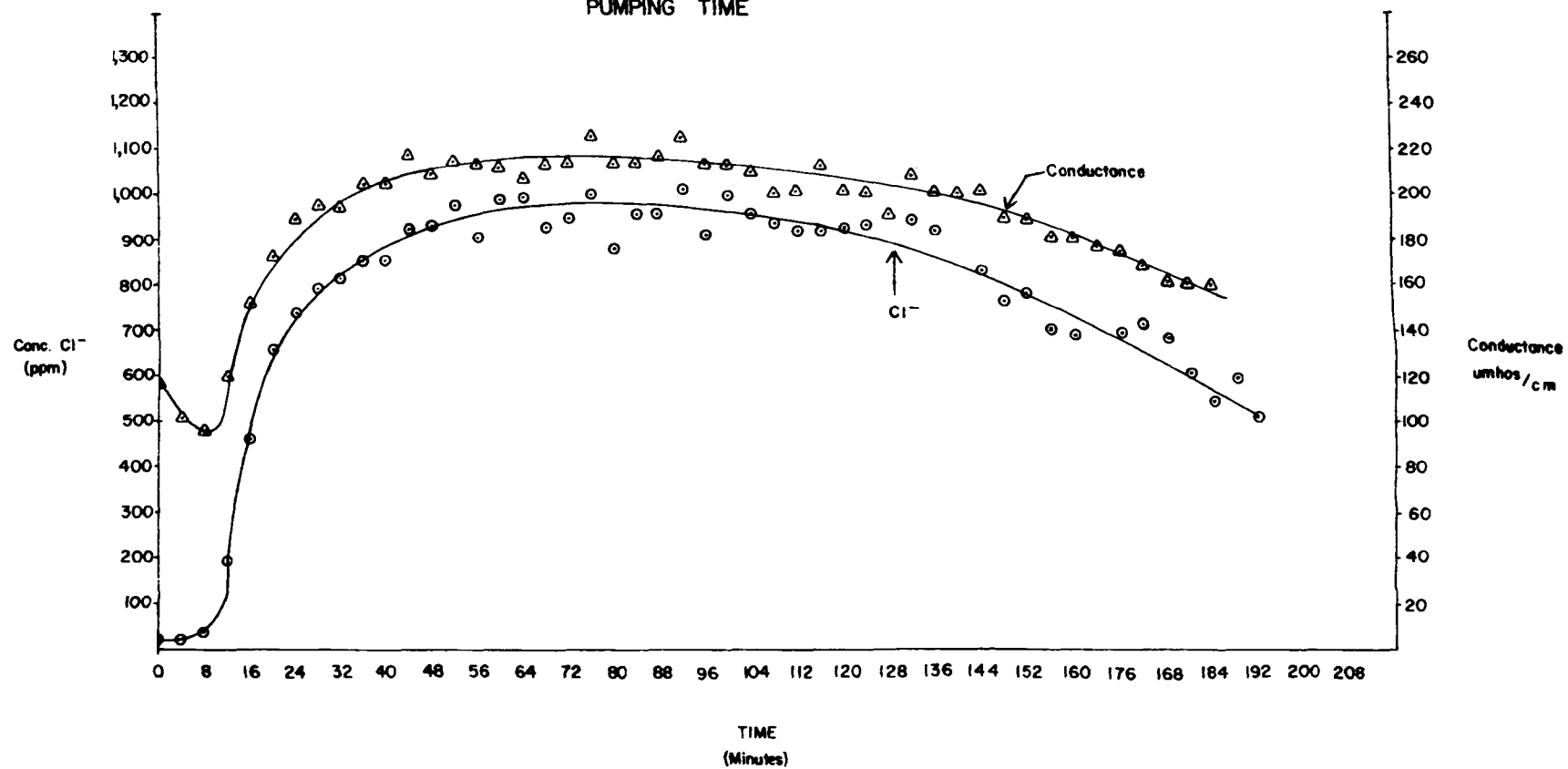
SECTION 8

LABORATORY ATTENUATION STUDIES

This section presents the results of the soil column attenuation studies performed on natural soils from plants L and J, and kaolinite clay.

ATTENUATION BY NATURAL SOIL COLLECTED AT PLANT L

A sample of the ash pond dike material from plant L was one of three soil materials used in the attenuation experiment described in section 4. As previously mentioned in that section, samples of the soil column effluent were analyzed for the constituents listed in Table 8. These data are tabulated, along with an analysis of the ash leachate used in the attenuation studies, and are presented in appendix C. The soil sample from plant L, was also analyzed by X-ray diffraction for quartz, and the clay minerals montmorillonite, kaolinite, and illite. The analysis found the clay minerals present in the following relative abundance: kaolinite > montmorillonite > illite, and quarts was the most predominate crystalline phase present.

Figures 18, 19, 20, and 21 are plots of plant L's soil column effluent concentrations for magnesium, sodium, potassium, aluminum, barium, copper, zinc, nickel, and sulfate, versus the cumulative effluent volumes. Also shown on these plots are the "initial" influent leachate concentrations for the above parameters. In Figure 18, magnesium, sodium, and potassium are plotted. The illustration shows that the concentrations of magnesium and sodium in the column effluent exceeded the initial influent concentration for the duration of the experiment, indicating solubilization of these elements occurred in the soil. Potassium, however, initially had a concentration less than the influent, which increased above the influent in subsequent samples and eventually equilibrated (concentration in equaled concentration out) with the initial leachate concentration. This seems to indicate that potassium was initially attenuated by the soil, and later merely passed through the column unaffected. This is further illustrated in Table 22, where mass balances have been calculated on twelve parameters including those illustrated in Figures 18 through 21.
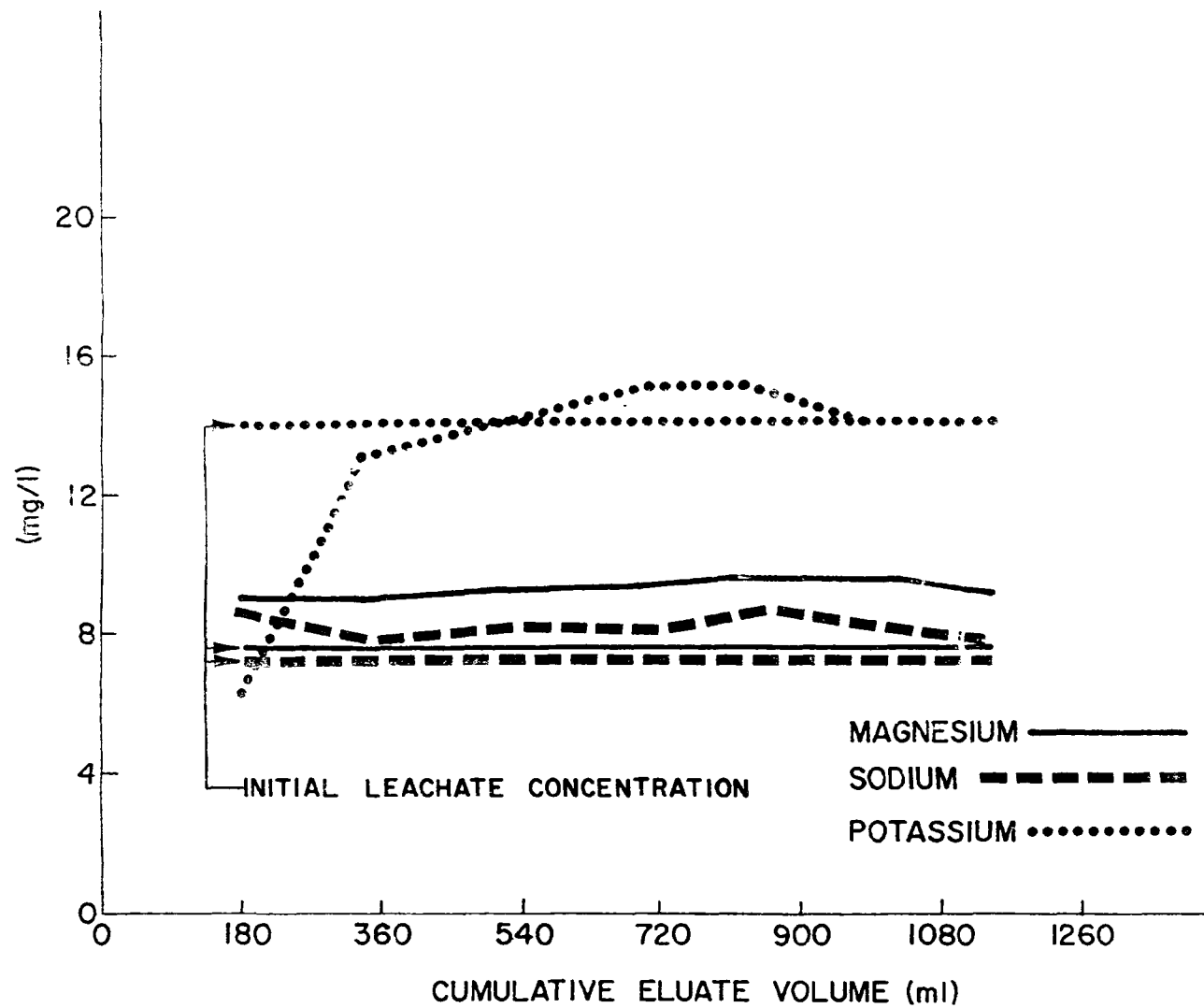
Figure 18. Concentrations of magnesium, sodium, and potassium in the effluent from plant L's soil column.
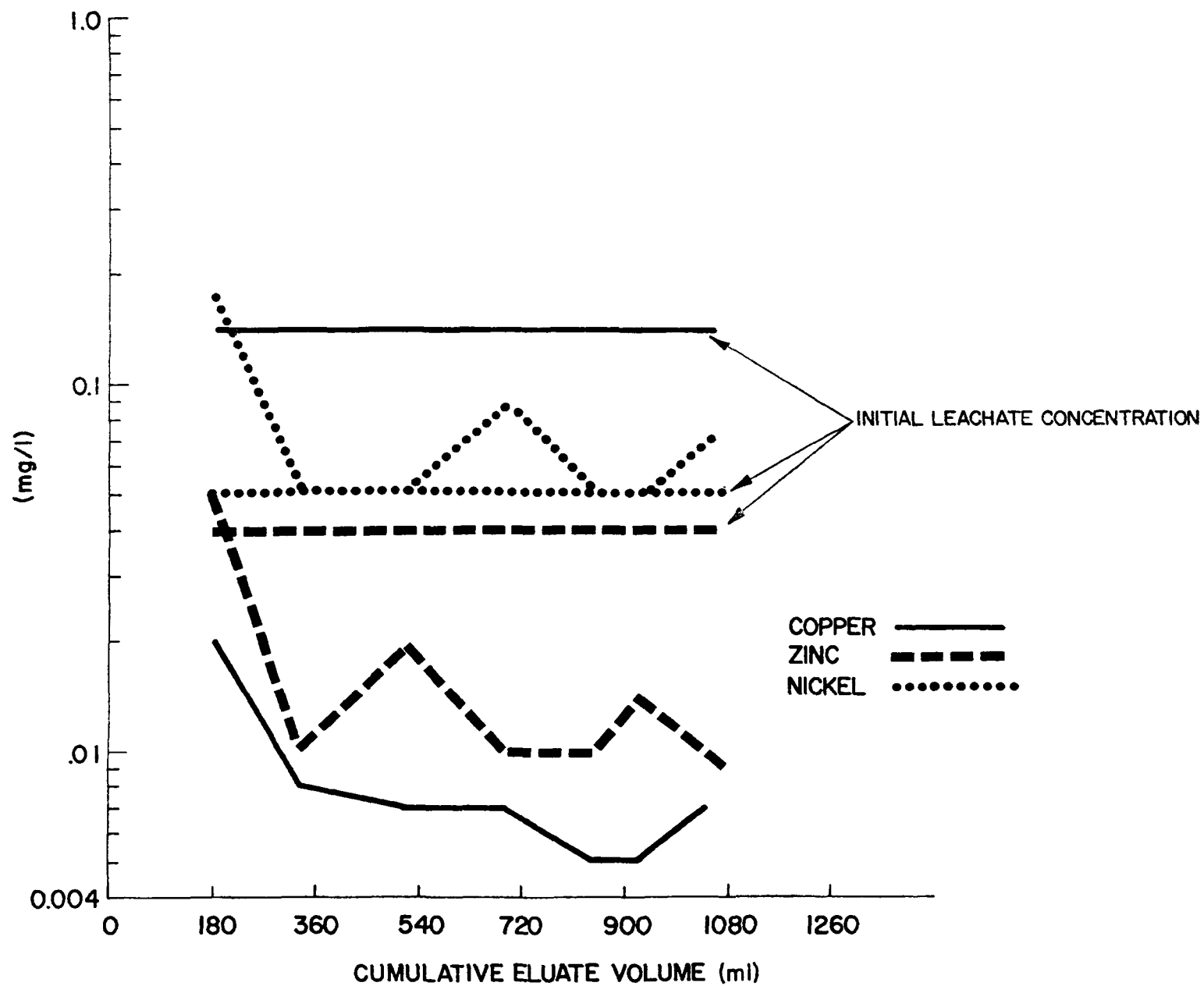
Figure 19. Concentrations of copper, zinc, and nickel in the effluent from plant L's soil column.
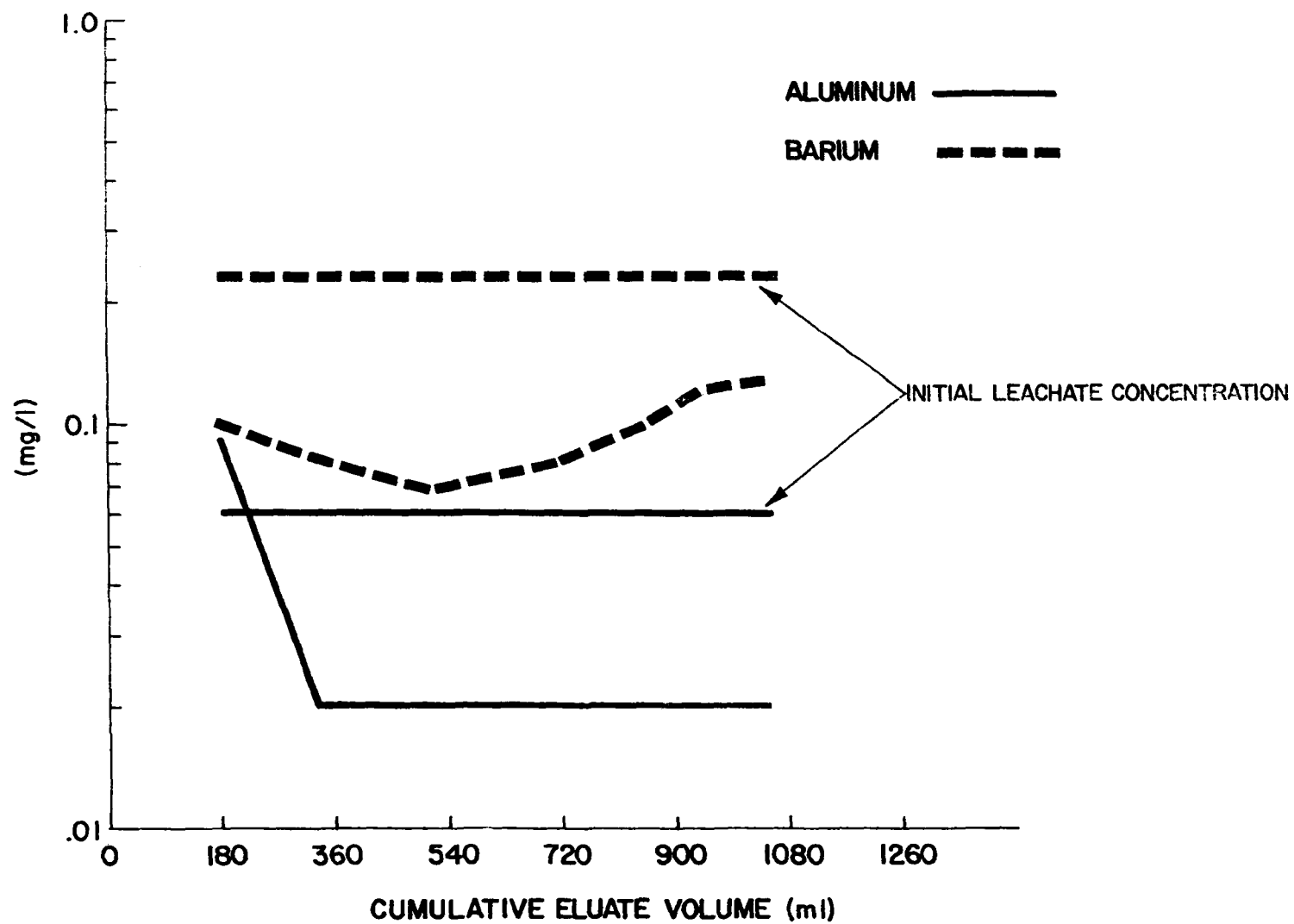
Figure 20. Concentrations of aluminum and barium
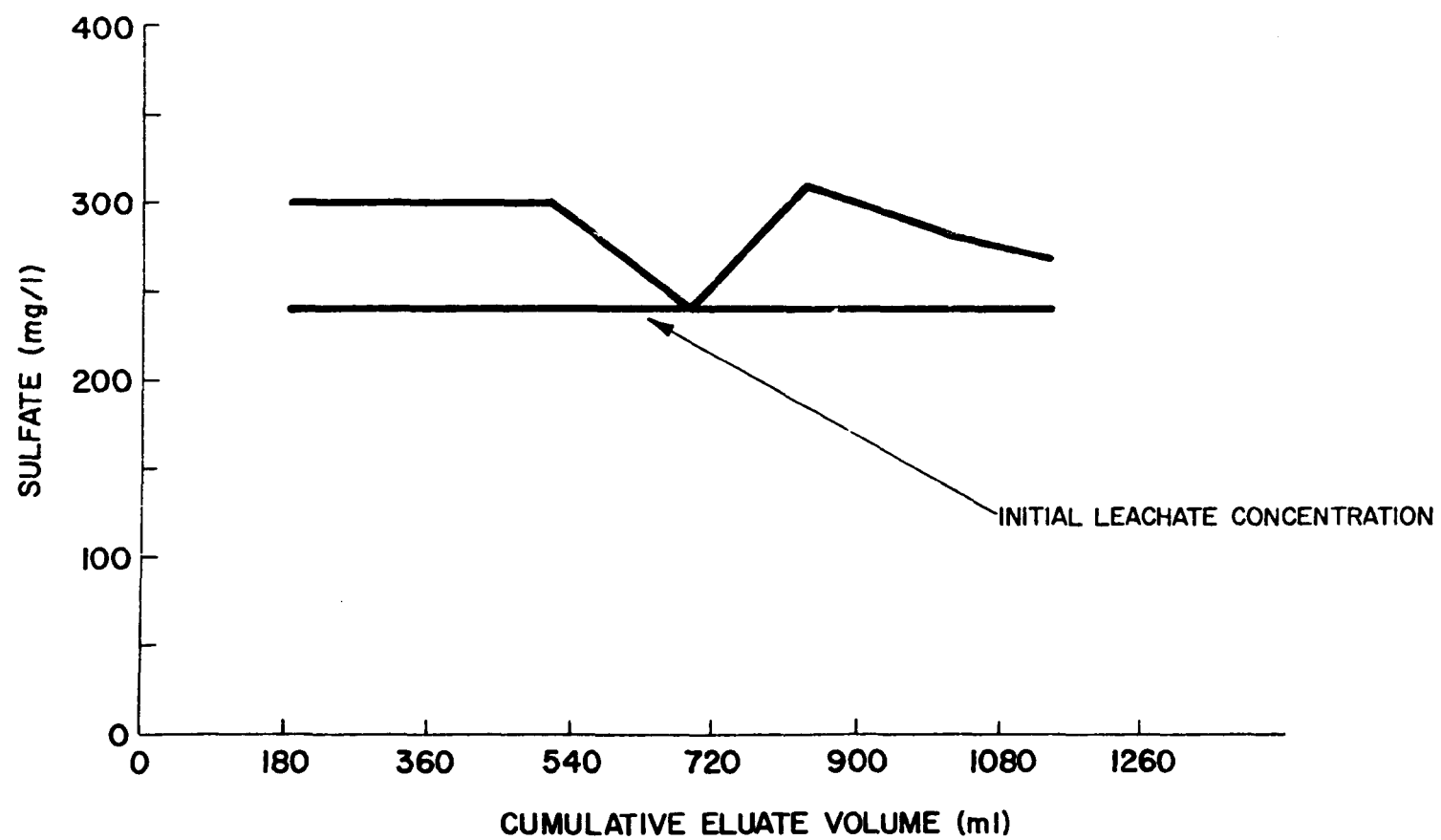in the effluent from plant L's soil
column.

Figure 21. Concentration of sulfate in the effluent
from plant L's soil column.

The mass input to the soil columns was determined by multiplying the total volume of eluate that passed through the column by the initial influent concentration. To determine the mass output of the column, the volume of each composite eluate sample was multiplied by the concentration of the constituent measured therein, and these values summed. The difference between the mass in and mass out is the amount retained, or contributed by the soil column. These data indicate that magnesium and sodium had, respectively, 16 and 5 percent increases in mass between the influent and effluent of the soil column, while 3 percent of the potassium was removed in the soil.

In Figure 19, copper, nickel, and zinc concentrations in the soil column effluent are plotted. Copper concentrations were less than the initial leachate concentration in all samples analyzed, and gradually decreased with the volume eluted. Zinc showed a trend similar to copper. With the exception of the first sample, all samples analyzed for zinc were less than the initial leachate concentration. The concentration of nickel in the leachate was less than the analytical detection limit, which accounts for effluent values not plotted below the initial leachate concentration line. However, samples of the soil column effluent did contain some measureable quantities of nickel indicating solubilization from the soil. In Table 22, the data indicate that copper and zinc had, respectively, 97 and 40 percent removal during the test. Calculations were not performed for nickel because it was not detected in the leachate.

The concentrations of aluminum and barium in the effluent from plant L's soil column are illustrated in Figure 20, and the plot for sulfate in Figure 21. All of the effluent samples analyzed for barium and aluminum had concentrations less than the initial leachate value, with the first eluate volume for aluminum being the only exception. The data in Table 22, show that 70 percent of the aluminum and 57 percent of the barium were removed in the soil column. The sulfate plot in Figure 21, indicates solubilization occurring within the soil column. This indication is supported by the mass balance data in Table 22, which shows a 15 percent increase in sulfate mass between the soil column influent and effluent.

Calcium, manganese, iron, and mercury data are not plotted, but from the mass balance data in Table 22, it is indicated that 11 percent of the calcium, 85 percent of the manganese, essentially 100 percent of the iron, and 87 percent of the mercury were removed in the soil column. Cadmium, chromium, lead, beryllium, and selenium were all less than the minimum detectable limits in all samples, including the initial leachate sample, and for that reason were not addressed in this analysis.

Also presented in Table 22, are the number of chemical equivalents removed from the leachate and the number added to the soil column effluent, with the sum of both listed at the bottom of the table. These

TABLE 22.   MASS BALANCE OF PLANT L'S SOIL COLUMN INFLUENT AND EFFLUENT

Constituent	Mass IN (mg)	Mass OUT (mg)	Percent removal	Equivalents removed	Percent increased	Equivalents added
Magnesium	8.66	10.03	0	0	16	0.11
Sodium	8.44	8.87	0	0	5	0.02
Potassium	15.96	15.45	3	0.01	0	0
Copper	0.16	0.01	97	0.004	0	0
Zinc	0.05	0.03	40	0.0004	0	0
Aluminum	0.07	0.02	70	0.01	0	0
Barium	0.26	0.11	57	0.002	0	0
Sulfate	273.6	315.3	0	0	15	0.87
Calcium	210.9	187.8	11	1.16	0	0
Manganese	4.79	0.74	85	0.15	0	0
Iron	17.1	<0.01	>99	0.92	0	0
Mercury	0.0023	0.0003	87	1.9×10^{-5}	0	0
Total				2.26		1.00

data indicate that 2.26 meq (milliequivalents) were removed in the soil column, while 1.0 meq was added to the column effluent. This amounted to a net decrease of 1.26 meq through the soil column.

ATTENUATION BY NATURAL SOIL COLLECTED AT PLANT J

A sample of plant J's ash pond dike material was used to determine the attenuation of coal-ash leachate by a natural soil type being used in ash pond construction, as was the soil from plant L, previously discussed. The soil sample from plant J, contained a percentage of clay minerals that when subjected to X-ray diffraction analysis were found to be present in the following relative abundance: illite > kaolinite > montmorillonite. In addition to these clay minerals, the soil contained quartz as the most predominant crystalline phase. The cation exchange capacity of the soil was 17.0 meq per 100 grams, as determined by the method of Bascomb, which is described in appendix A.

In Figures 22 through 24, plant J's soil column effluent concentrations are plotted against the cumulative effluent volumes for magnesium, sodium, potassium, aluminum, barium, copper, zinc, and nickel. The initial leachate concentrations for the above parameters are also shown in these plots. In Figure 22, magnesium, sodium, and potassium are plotted. The illustration shows that the concentrations of magnesium and sodium in the soil column effluent exceeded the influent concentration in all samples analyzed during the experiment. Again, as with plant L's soil column study, the plots indicated solubilization of these elements within the soil column. The plot for potassium is similar to the one generated from plant L's attenuation data. Initially the effluent concentration was less than the influent, and then gradually increased to nearly the same concentration, but never quite equilibrated with the influent. This indicates that potassium was being attenuated, to some degree, for the duration of the experiment. This is also illustrated in Table 23, where mass balances have been calculated on eleven constituents, in a manner previously described. As the data in this table indicate, magnesium and sodium had, respectively, 20 and 10 percent increases in mass between the influent and effluent of the soil column, while 12 percent of the potassium was removed in the soil.

Figure 23 illustrates the plotted concentrations of copper, zinc, and nickel in plant J's soil column effluent. Copper concentrations fluctuated, but were always less than the influent value. The zinc concentration in the column effluent initially was higher than the influent, but immediately declined to remain less than the influent value for the experiment's duration. The mass balance data from Table 23 show that copper and zinc, respectively, had 94 and 57 percent removal of mass in the soil column. A balance on nickel could not be calculated because it was not detected in most samples.

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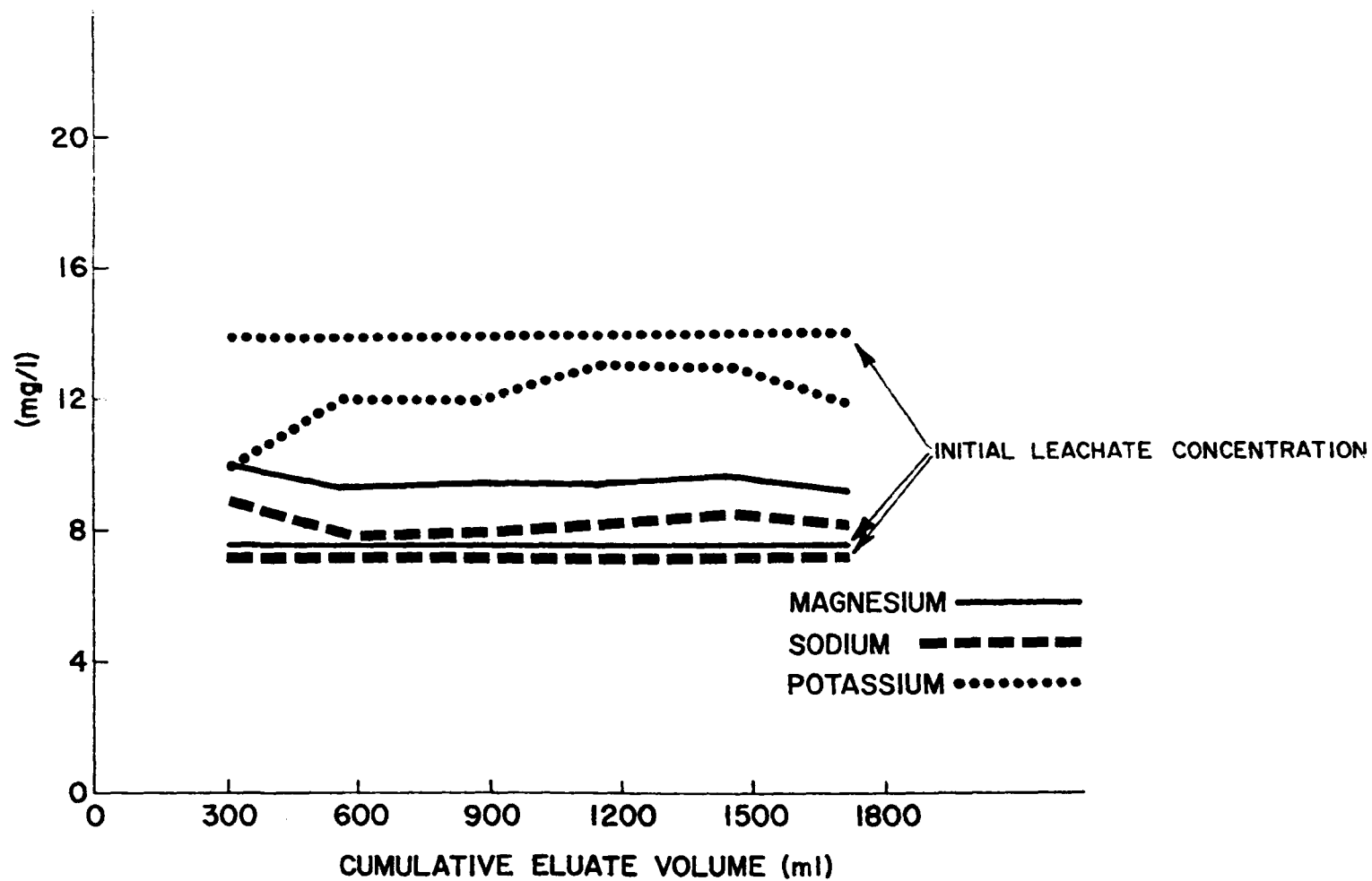
Figure 22. Concentrations of magnesium, sodium, and potassium
in the effluent from plant J's soil column.
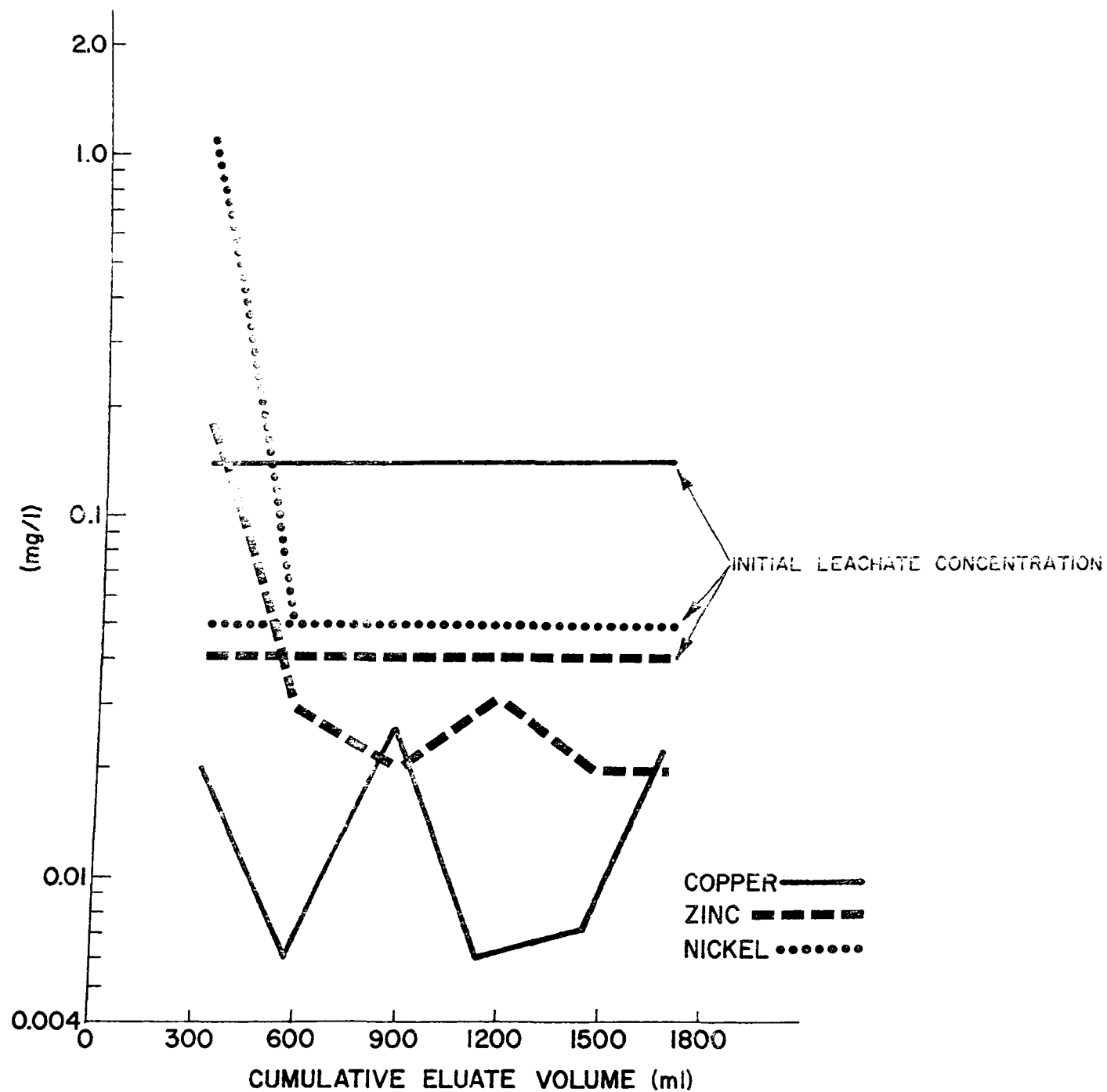
Figure 23. Concentrations of copper, zinc, and nickel in the effluent from plant J's soil column.
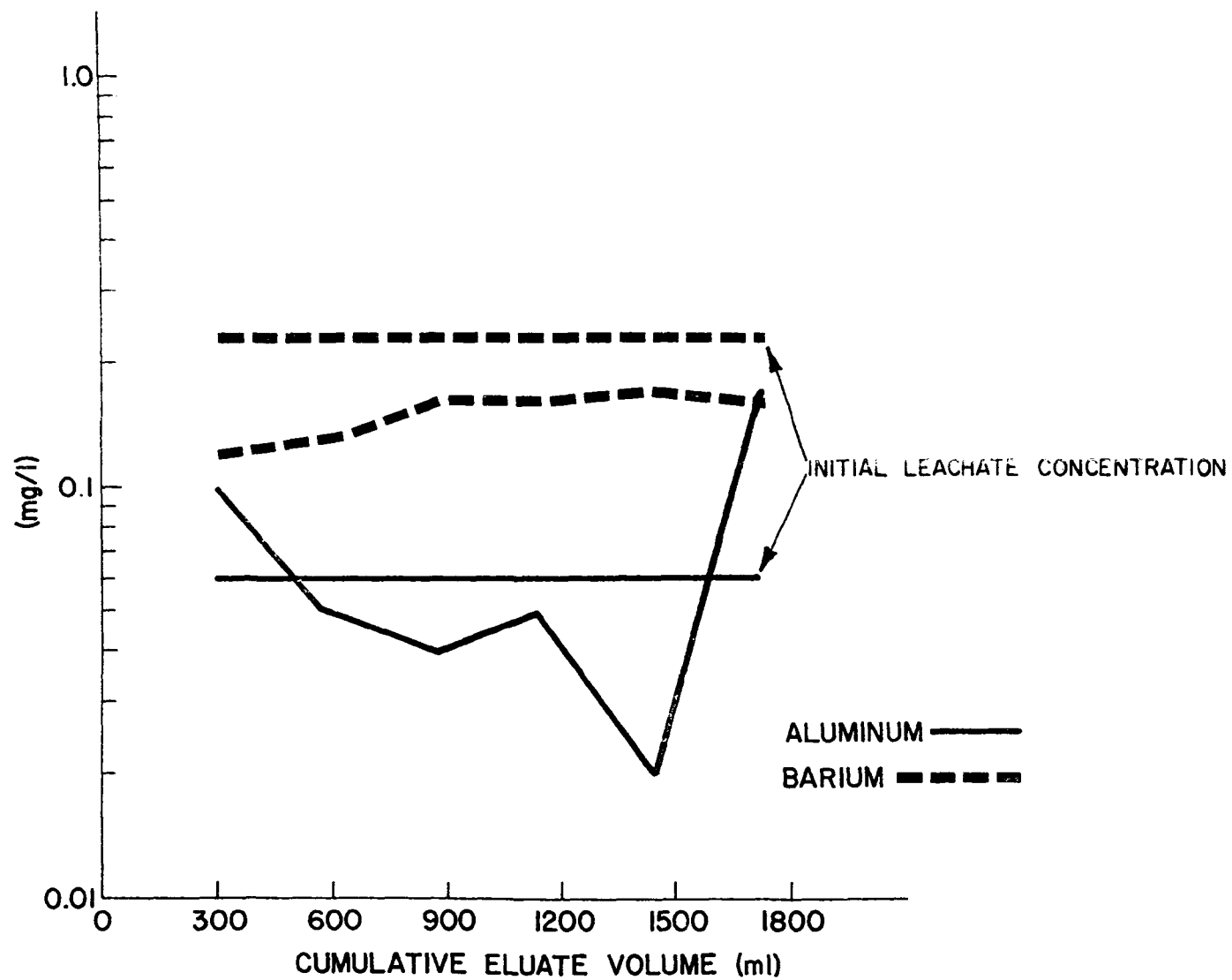
Figure 24. Concentrations of aluminum and barium
in the effluent from plant J's soil
column.

TABLE 23. MASS BALANCE OF PLANT J'S SOIL COLUMN INFLUENT AND EFFLUENT

Constituent	Mass IN (mg)	Mass OUT (mg)	Percent removal	Equivalents removed	Percent increased	Equivalents added
Magnesium	13.00	15.65	0	0	20	0.22
Sodium	12.31	13.59	0	0	10	0.06
Potassium	23.94	21.18	12	0.07	0	0
Copper	0.24	0.01	94	0.01	0	0
Zinc	0.07	0.04	57	0.0009	0	0
Aluminum	0.10	0.11	0	0	10	0.001
Barium	0.39	0.27	32	0.001	0	0
Calcium	316.35	261.30	17	2.75	0	0
Manganese	7.18	1.77	75	0.20	0	0
Iron	26.65	0.08	≈100	1.38	0	0
Total				4.41		0.28

Aluminum, however, began with an effluent concentration greater than the influent, then decreased to below the influent value where it remained for all samples except the last which was again greater in concentration than the influent. Barium had 32 percent removal within the soil column, according to the data in Table 23. Aluminum had a 10 percent increase in mass between influent and effluent.

Although the data for calcium, manganese, and iron were not plotted, the mass balance data in Table 23 indicate that 17 percent of the calcium, 75 percent of the manganese, and nearly 100 percent of the iron was removed in the soil column. Cadmium, chromium, lead, beryllium, mercury, and selenium were all less than their analytical detection limits in nearly all samples. Not enough sample was collected for sulfate analysis. Table 23 also shows that 4.41 chemical meq were removed in the soil column from plant J, while 0.28 meq were added to the column effluent. Thus, a net decrease of 4.13 meq occurred between the soil column influent and effluent.

ATTENUATION BY KAOLINITE

In natural soils there are three major clay minerals that may be present in relative amounts depending on the geographical location; they are kaolinite, montmorillonite, and illite. Each of these clays has the ability to attenuate pollutants from aqueous solutions, although selectively and at different rates.[25] Attempts were made to study the attenuation of coal-ash leachate by each of these clays. However, expansion of the clay minerals montmorillonite and illite upon wetting inhibited percolation to such a degree that experiments with these two clays were abandoned. It was possible, however, to study the attenuation capabilities of kaolinite. The results of that investigation are presented in Figures 25 through 27, and Table 24.

In Figure 25, column effluent concentrations of magnesium, sodium, and potassium are plotted against the cumulative eluate volume from the kaolinite clay column. The illustration shows that concentrations of magnesium and sodium in all effluent samples analyzed, except the last, exceeded the influent leachate concentration. This indicates that solubilization of these two elements occurred within the clay column. Potassium, on the other hand, had concentrations in the column effluent which exceeded the influent concentrations in all samples except the first and last. This would seem to indicate that potassium is also being solubilized to some degree within the clay column. In Table 24, mass balance data agree with the plotted data, in that magnesium, sodium, and potassium respectively, had 23, 17, and 9 percent increases in mass between the clay column influent and effluent.

Figure 26 illustrates the plotted concentrations of copper, zinc, and nickel in the kaolinite column effluent. Copper concentrations were less in the column effluent than in the influent for all samples except
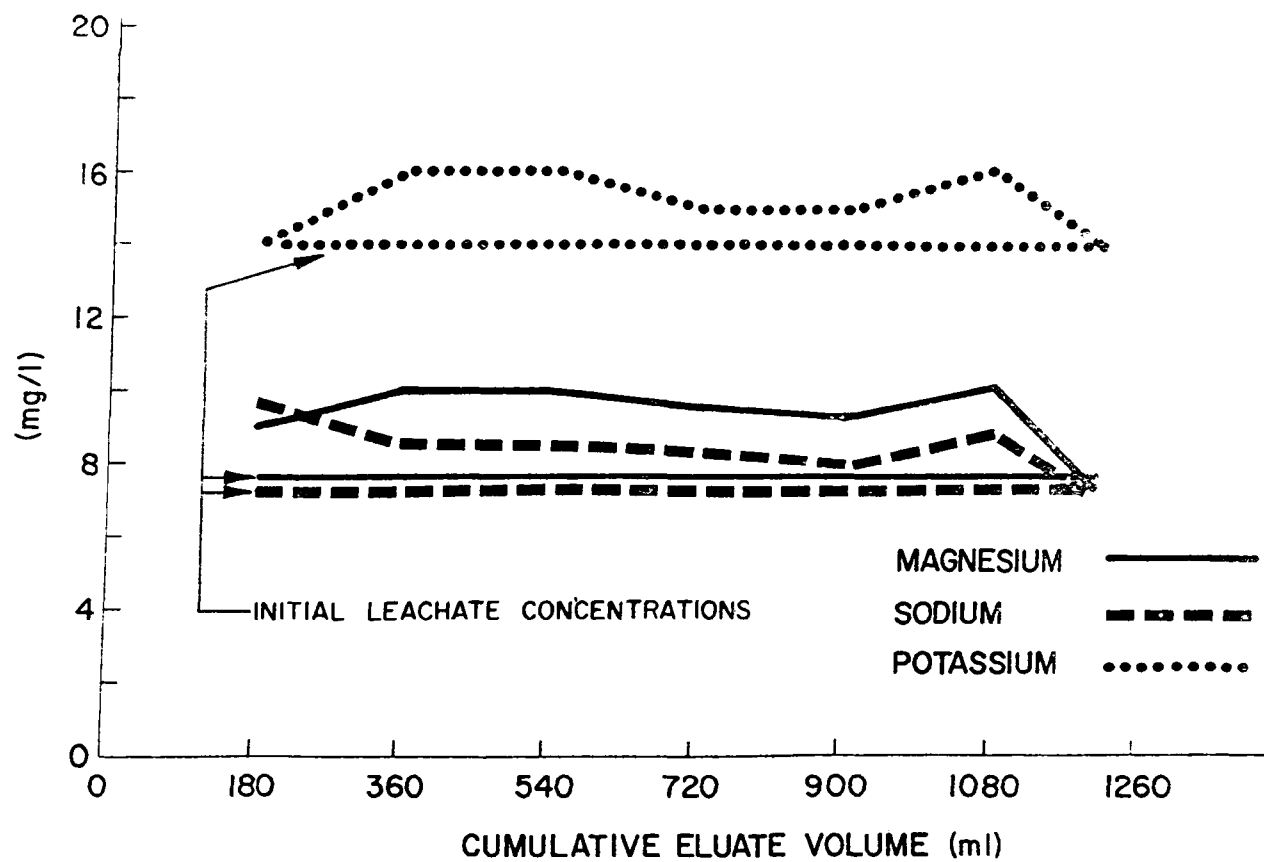
Figure 25. Concentrations of magnesium, sodium, and pottasium in the kaolinite clay column effluent.
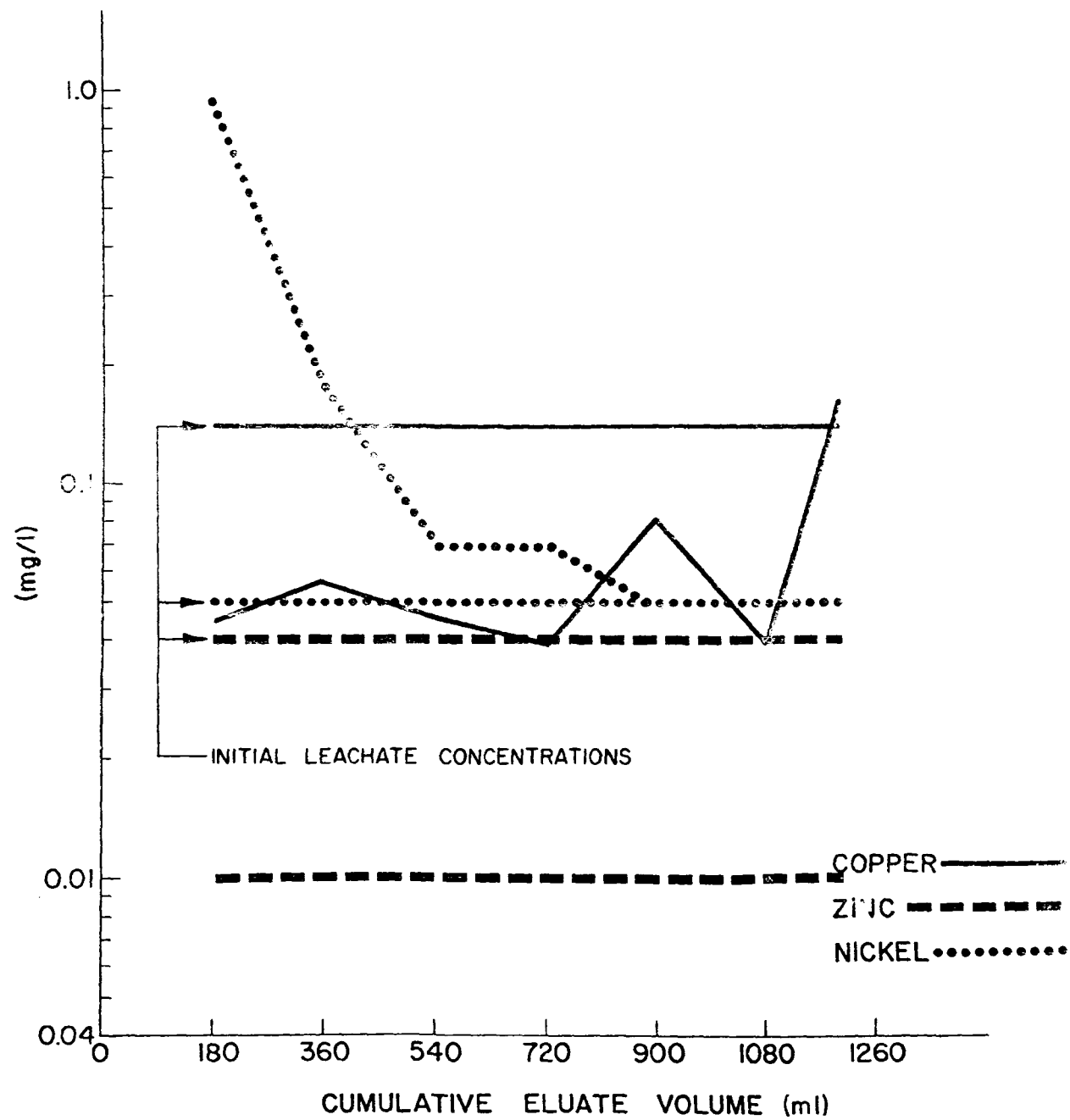
Figure 26. Concentrations of copper, zinc, and nickel in the kaolinite clay column effluent.

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INITIAL LEACHATE CONCENTRATIONS

ALUMINUM ————
BARIUM ━━ ━━ ━━ ━━

CUMULATIVE ELUATE VOLUME (ml)
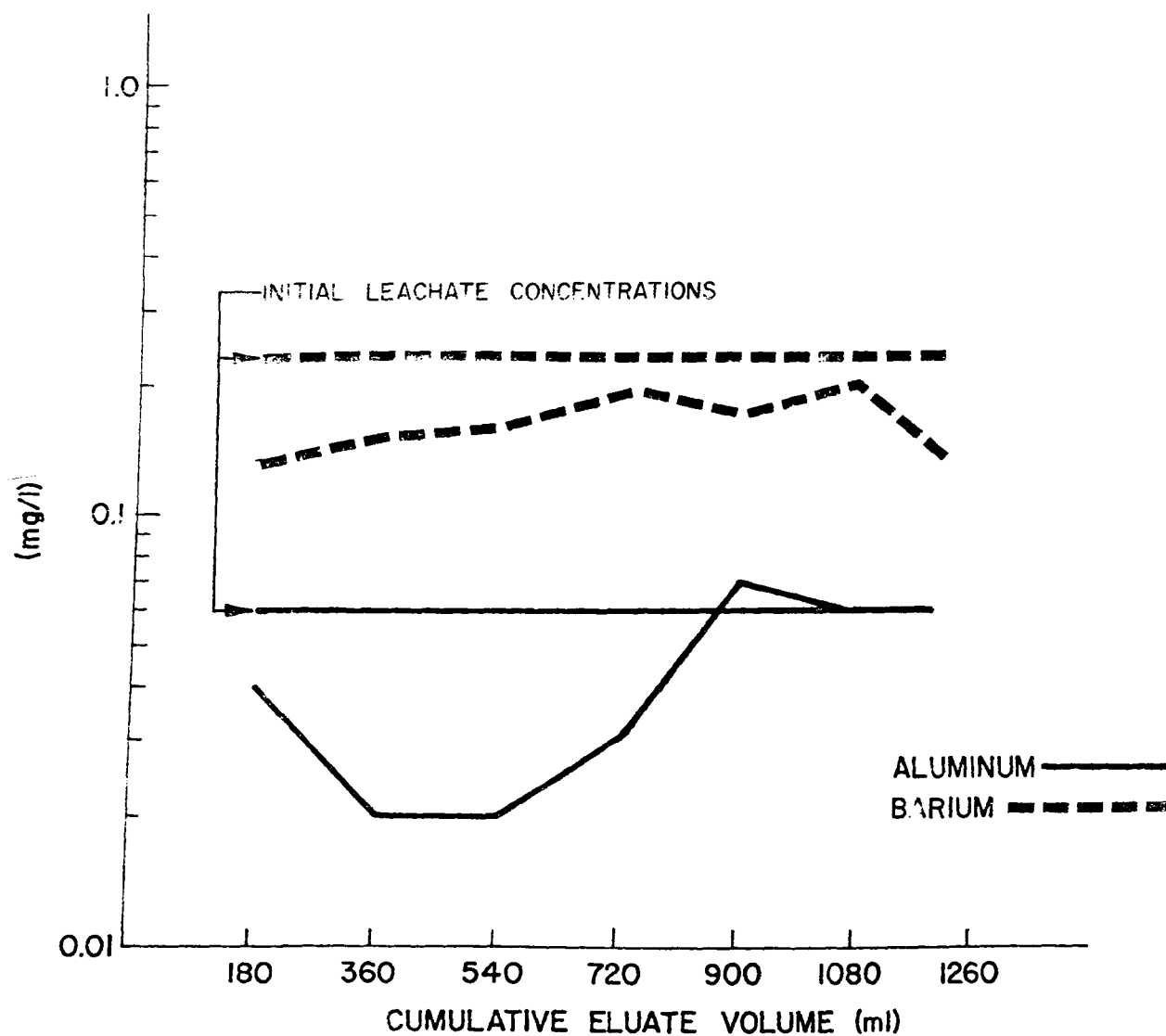
Figure 27. Concentrations of aluminum and barium
in the kaolinite clay column effluent.

TABLE 24. MASS BALANCE OF KAOLINITE PACKED COLUMN INFLUENT AND EFFLUENT

Constituents	Mass IN (mg)	Mass OUT (mg)	Percent removal	Equivalents removed	Percent increased	Equivalents added
Magnesium	9.12	11.21	0	0	23	0.17
Sodium	8.64	10.11	0	0	17	0.06
Potassium	16.80	18.24	0	0	9	0.03
Copper	0.17	0.07	56	0.003	0	0
Zinc	0.05	0.03	50	0.001	0	0
Aluminum	0.07	0.05	30	0.002	0	0
Barium	0.28	0.20	29	0.001	0	0
Calcium	222.00	180.00	19	2.09	0	0
Manganese	5.04	0.99	80	0.14	0	0
Iron	18.00	0.05	99.7	0.96	0	0
Mercury	0.0024	0.0003	88	0.000021	0	0
Sulfate	345.60	429.60	0	0	24	1.75
Total				3.20		2.01

the last. All effluent zinc samples analyzed were less than the influ-
ent leachate concentration. The concentration of nickel in the column
effluent initially exceeded the column influent, then decreased to
equilibrate with the influent concentration, which was nickel's minimum
detectable limit. Copper and zinc removal within the clay column, based
on mass balance data in Table 24, was respectively 56 and 50 percent. A
balance on nickel was not calculated because it was not detected in the
influent and most of the effluent samples.

Plots of aluminum and barium concentration in the kaolinite column
effluent are illustrated in Figure 27. All samples analyzed for barium
had concentrations less than the initial leachate concentration. Alu-
minum concentrations were less than the influent in the early eluate
volumes, then gradually increased to become equal with the influent
concentration. The data from Table 24 indicates that aluminum and
barium were reduced in the kaolinite column by 30 and 29 percent,
respectively.

For calcium, manganese, iron, and mercury, the mass balance data in
Table 24 indicate that 19 percent of the calcium, 80 percent of the
manganese, nearly 100 percent of the iron, and 80 percent of the mercury
was removed in the kaolinite clay column. Sulfate had a 24 percent
increase in mass from influent to effluent.

In Table 24, the data also indicates that 3.20 chemical meq were
removed in the kaolinite column, while 2.01 meq were added to the column
effluent. As a result, a net of 1.19 chemical meq were attenuated from
the initial leachate influent by the kaolinite clay column.

DISCUSSION OF LEACHATE ATTENUATION STUDY

The attenuation processes that occur in soils are a potential
major influence precluding groundwater contamination by leachate from
coal ash disposal areas. In an attempt to determine the extent of ash
leachate attenuation by various soils, the laboratory investigations
reported on above were conducted. Other investigators have used this
technique to determine the attenuation of various aqueous wastes by
different soils, clays, and other porous material.[26,27,28]

For example, in a study by Griffin, et al,[26] the attenuation of
pollutants in municipal landfill leachate by the clay minerals mont-
morillonite, kaolinite, and illite was investigated. Their study
determined the following clay mineral hierarchy for attenuation capa-
bilities, montmorillonite > illite > kaolinite, and that the principal
attenuation mechanism for lead, cadmium, mercury, and zinc was precipi-
tation in the soil columns surface layers. In a study by Fetter[27] to
determine attenuation of secondarily treated wastewater by a calcareous
glacial outwash soil, the findings suggested that heavy metals removal
was through ion exchange. Still other investigators, such as Leeper,[28]
consider the adsorption reactions with hydrous oxides of iron, aluminum,

and manganese to be the major mechanism for the attenuation of metals in soil. As one can see from this small, but representative sampling, the literature is inconclusive as to which mechanism is the major factor influencing attenuation by soils. Indeed, no one mechanism is universally responsible for the attenuation processes that occur in soils receiving liquid wastes or leachate. In fact, the mechanisms may vary for each situation depending on the characteristics of the soil, the characteristics of the liquid waste or leachate, and the hydrological conditions.

The major mechanism influencing attenuation of ash leachate in this study is not readily discernable from the previously discussed data. It may be concluded with a certain degree of confidence, however, that ion exchange is not solely responsible for the removal of constituents from the ash leachate, for the following reason.

The two natural soils from plants L and J, and the kaolinite clay used in the column tests had cation exchange capacities, respectively, of 20.0, 17.0, and 16.0 meq per 100 g (the cation exchange capacity of the silica was negligible at 1.2 meq/100 g). Five grams each of the soils and clay material was used in each column, which yielded a total exchange capacity for the columns of 1.0 meq plant L material, 0.85 meq plant J material, and 0.80 meq for the kaolinite column. The number of meq's attenuated by these same respective columns was 1.26, 4.13, and 1.19 meq. Each column has thus exceeded its available exchange capacity; consequently, ion exchange is not the only mechanism accounting for attenuation in this study. Actually, ion exchange may be only a minor influence, and precipitation and/or adsorption may be the major influencing mechanisms affecting attenuation, but the data do not differentiate between these two mechanisms.

A comparison of the mass balance data in Tables 22, 23, and 24 shows that magnesium and sodium had a net increase in mass between the column influent and effluent for all three attenuation experiments. This indicates that these two elements were solubilized by the ash leachate as it passed through the soil columns. Of the three materials used in this study, the kaolinite clay released the most magnesium and sodium. Potassium was also released from the kaolinite, while the two natural soils, from plants J and L, attenuated potassium.

Copper and zinc were both retained in all three attenuation columns, copper more so than zinc. Copper was attenuated the least by the kaolinite clay. The two natural soils attenuated greater than 90 percent of the copper. Barium was attenuated in all three columns, but the kaolinite clay retained the least. Aluminum was attenuated in the kaolinite column and by the soil from plant L. Aluminum solubilized in plant J's column. Calcium, manganese, and iron were all attenuated in the three columns. Iron had the largest attenuation of the constituents analyzed with essentially 100 percent removal in each column; manganese followed with 75 percent plus attenuation.

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In general, soils that are composed of high percentages of clay minerals will, by one mechanism or another, tend to attenuate solutes more so than will materials containing high percentages of sand.

SECTION 9

THEORETICAL CONSIDERATIONS

Using information reported in the literature to date regarding coal-ash leachate and the attenuation of leachates in general, the results of this study, and some fundamental knowledge of the hydrogeochemical environment, a schematic diagram of ash pond leachate generation and attenuation was prepared (see Figure 28). This diagram illustrates some of the basic concepts associated with solid waste disposal in general and presents some details relating to coal-ash disposal in particular. The diagram does not necessarily indicate conditions and mechanisms associated with all ash disposal sites, but is designed to relate some of the physical and chemical processes that may be pertinent to an environmental evaluation of groundwater degradation at coal-ash disposal areas.

In this illustration, fly ash, bottom ash, and pyrites are sluiced to the ash pond, along with other miscellaneous plant discharges, such as acidic coal-pile drainage. Once in the disposal pond, the ash and other heavy particles settle out, with the supernatant overflowing into an adjacent surface water. Supernatant that does not leave the pond via the surface overflow infiltrates into the settled ash, carrying with it any solutes picked up during sluicing of the ash from the plant and mixing with other wastes discharged to the pond. After infiltration, the pond supernatant percolates down through the saturated aerobic zone where readily soluble ions of calcium, magnesium, sodium, and sulfate are added to solution. Some slight dissolution of certain metals may occur, and sulfite, if not already in solution, solubilizes.

Further downward percolation leads to the anoxic zone, which is created by the depletion of dissolved oxygen by sulfite or other oxidation processes. It is in the anoxic zone, which occurs either below the water table or deep enough to prevent surface aeration, where the percolating water acquires the culmination characteristics of coal-ash leachate. The anoxic zone is a reducing acidic environment in which dissolution of metals can occur, sulfides are formed, and high concentrations of ferrous iron may occur. It is also in this zone where the hydraulic gradient of the groundwater begins to exert its energy potential on the leachate, altering the direction component of its velocity from strictly downward to lateral.

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ASH SLUICING PIPE FROM PLANT
(CONTAINS FLY ASH,BOTTOM ASH,PYRITE,RAW SURFACE WATER,LOW DISS. METALS,HIGH TOTAL METALS)

DIRECT PRECIPITATION
($pH \approx 5.0$)

ORIGINAL
GROUND WATER TABLE

ASH DISPOSAL POND →

GROUND WATER FLOW

CLAY-SILT STRATUM
(LOW HYDRAULIC CONDUCTIVITY
HIGH CATION EXCHANGE CAPACITY
HIGH ADSORPTION SITES)

SAND STRATUM
(HIGH. HYDRAULIC CONDUCTIVITY
LOW CATION EXCHANGE CAPACITY
LOW ADSORPTION SITES)

SUPERNATE   INFILTRATION
SATURATED
AEROBIC ZONE

DEPOSITED ASH

DISSOLUTION OF Cd^{++},Mg^{++},$SO_4^=$,
TDS INCREASES SOME METALS
AND $SO_3^=$ PRODUCED

D.O.DEPLETES $SO_3 + \frac{1}{2} O_2^- = SO_4$

ANOXIC ZONE   DISSOLUTION OF METALS,LOW
pH,REDUCING ENVIRONMENT
FORMATION OF SULFIDE
HIGH CONCENTRATION Fe^{++}
INFILTRATION OF LEACHATE INTO SUBSURFACE

LEACHATE ATTENUATION OCCURING HERE
(ABSORPTION,PRECIPITATION,ION EXCHANGE
SOME DILUTION)

SPECIFIC GRAVITY OF
LEACHATE HIGHER
THAN AMBIENT

GROUND WATER FLOW

ASH POND DIKE
(LOW PERMEABILITY)

ASH POND SUPERNATE DISCHARGE
(MUST MEET NPDES REQUIREMENTS)

SURFACE WATER-LAKE OR RIVER
(NORMALLY SUPPLIES RAW WATER
FOR ASH SLUICING)

OXIDATION AND PRECIPITATION OF
LEACHATE ENTERING SURFACE WATER

LEACHATE MIGRATION FRONT
(HIGH CONC. OF EXCHANGEABLE DIVALENT CATIONS
PRECEEDS LEACHATE FRONT,TERMED "HARDNESS HALO")

MINIMAL ATTENUATION HERE

IMPERMEABLE
BEDROCK

FIGURE 28. SCHEMATIC OF COAL-ASH
LEACHATE GENERATION AND ATTENUATION

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Eventually the leachate begins to infiltrate the bottom of the ash disposal area. This may occur at a relatively high rate if the pond bottom is of a highly permeable material, such as sand, or at a lower rate if the material is a low permeable clay-silt, such as illustrated in Figure 28. In the clay-silt stratum attenuation of solutes in the leachate will begin to occur. Attenuation mechanisms, such as ion exchange, chemical precipitation, and physical adsorption, will act to lessen the leachate migration. As the leachate plume advances through the subsurface environment, metal ions with an affinity for adsorption and exchange sites will replace weakly-bound divalent cations, such as calcium and magnesium. These displaced divalent cations will move along preceding the leachate plume and create what has been termed a "hardness halo." Conservative constituents, such as chloride and sulfate, may even precede this hardness halo. The rate of leachate migration will depend on the hydraulic conductivity and the rate of attenuation.

If the leachate plume should reach a highly permeable stratum such as the sand stratum depicted in Figure 28, then the potential for increased groundwater degradation can occur. A sand stratum or aquifer offers little in the way of attenuation capacity and much as an avenue for leachate migration. Hydraulic conductivities of sandy material are normally greater than those of clay, and the number of adsorption and exchange sites are fewer, which will tend to decrease the rate of attenuation.

Leachate studies thus far suggest that coal-ash disposal sites, and perhaps solid waste disposal sites in general, might be constructed in such a manner as to allow leachate to percolate downward through an attenuation blanket. The attenuation blanket would need to be of design and material that would allow optimum flow and attenuation, thus acting as a treatment system.

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APPENDIX A

ANALYTICAL METHODS

APPENDIX A

ANALYTICAL METHODS

The elements, aluminum, barium, beryllium, calcium, cadmium, chromium, copper, iron, magnesium, manganese, nickel, and lead in the interstitial water, soil column effluents and groundwater well samples, were measured by direct atomic absorption spectrophotometry using techniques described in Standard Methods[29] and by EPA.[30] Mercury was determined by the cold vapor technique recommended by EPA. Arsenic and selenium were measured using the gaseous hydride method, also recommended by EPA.

Sulfate was measured using the turbidimetric method, and pH values were obtained in the field using the glass electrode. Conductivity and alkalinity were also measured in the field using established procedures.[29,30]

Soil core samples were analyzed for aluminum, barium, beryllium, calcium, cadmium, chromium, copper, iron, magnesium, manganese, nickel, and lead by first undergoing a wet acid digestion followed by direct measurement with atomic adsorption spectrophotometry.[30] Mercury was measured in the soil core samples using EPA's recommended methods.[30] Arsenic and selenium were set digested and then determined by the gaseous hydride method previously mentioned.

Standard powder X-ray diffraction techniques were used to determine the relative amounts of quartz, kaolinite, montmorillinite, and illite present in the soils.

Oxidation-reduction potentials were determined with a silver-silver chloride electrode.

Vertical and horizontal permeabilities of soil samples were determined by encasing soil specimens 3.5 cm in diameter and approximately 7.6 cm long in a rubber membrane and placing in a triaxial chamber. Back pressure to 70,310 kgs/m^2 (100 psi) was applied to assure specimen saturation. The average coefficient of permeability was then determined under a constant head test method by measuring the quantity of water flowing through the specimen in a given time.

The moisture content of soil samples was determined as per ASTM procedure D-2216. Bulk densities were determined by weighing a soil specimen 15 cm long and 8.9 cm in diameter, then coating the specimen with paraffin and submerging in water to determine its volume. The moisture content of the specimen was then determined and the bulk density calculated using the specimen's weight, volume, and moisture content.

Grain size classifications were determined by sieve separation of large particles followed by the hydrometer technique for the silt and clay fraction.

Dissolved oxygen concentrations in groundwater samples were determined _in situ_ by lowering a membrane electrode into the monitoring wells. The membrane electrode was also used on groundwater samples brought to the surface for dissolved oxygen determinations.

DETERMINATION OF CATION EXCHANGE CAPACITY[*]

Reagents

Triethanolamine solution: Triethanolamine (commercial) 90 ml diluted to 1:1, and pH adjusted to 8.1 by adding 2N-hydrochloric acid (140-150 ml). This solution is diluted to 2:1. Protect from carbon dioxide during storage.

Barium chloride solution: $BaCl_2$, $2H_2O$ 244 g per 1 (approximately 2N).

Buffered barium chloride reagent: Mix equal volumes of the above solutions.

Magnesium sulphate solution: $MgSO_4$, $7H_2O$ 6.2 g per 1 (approximately 0.05N).

EDTA solution: Sequestric acid disodium salt 3.723 g per 1 (0.02N).

Catechol Violet indicator: 0.1 g dissolved in 100 ml of water.

Method

Transfer 5 g of soil (<2 mm air-dry, of known moisture content) to a tightly stoppered polythene centrifuge bottle. Note the weight of bottle plus soil (W_1). Treat calcareous soils with approximately 100 ml of the buffered barium chloride reagent, preferably with gentle shaking for 1 hr. Centrifuge at 1500 rpm (RCF) 415) for 15 min. and discard the supernatant liquid. For noncalcarous soils this first washing can be omitted. Treat with a further 200 ml of reagent overnight, centrifuge and again discard the supernatant liquid. Add approximately 200 ml of distilled water and shake for a few minutes to break up the soil cake. Centrifuge and discard the supernatant liquid. Weigh the bottle with contents (W_1). Pipette into the bottle 100 ml of magnesium sulphate solution and shake the stoppered bottle occasionally over a period of 2 hr. Centrifuge and transfer the clear liquid immediately to a stoppered flask.

To a 5-ml aliquot of the solution add 6 drops of 2N-aq. ammonia and titrate with standard EDTA solution using 2 drops of Catechol Violet indicator. The end-point is indicated by a color change from clear blue to reddish purple (Titre A_1 ml). This titre must be corrected for the effect of the volume (not chloride content) of liquid retained by the centrifuged soil after the wash water:

Corrected titre (A_2) = $A_1(100 + W_2 W_1)/100$ ml

Aliquots of 5 ml of the original magnesium sulphate solution are also titrated under similar conditions (Titre B).

CEC of the soil = $8(B - A_2)$ meq/100 g

[*]Bascomb, C. L. "Rapid Method for the Determination of Cation-Exchange Capacity of Calcareous and Noncalcareous soils." J. of Sci. Food and Agric., 15, 1964. pp. 821-823.

APPENDIX B

VERTICAL PROFILES OF SUBSTRATUM AT PLANTS J AND L

VERTICAL PROFILE OF THE SUBSTRATUM AT PLANT J's MONITORING WELL LOCATIONS
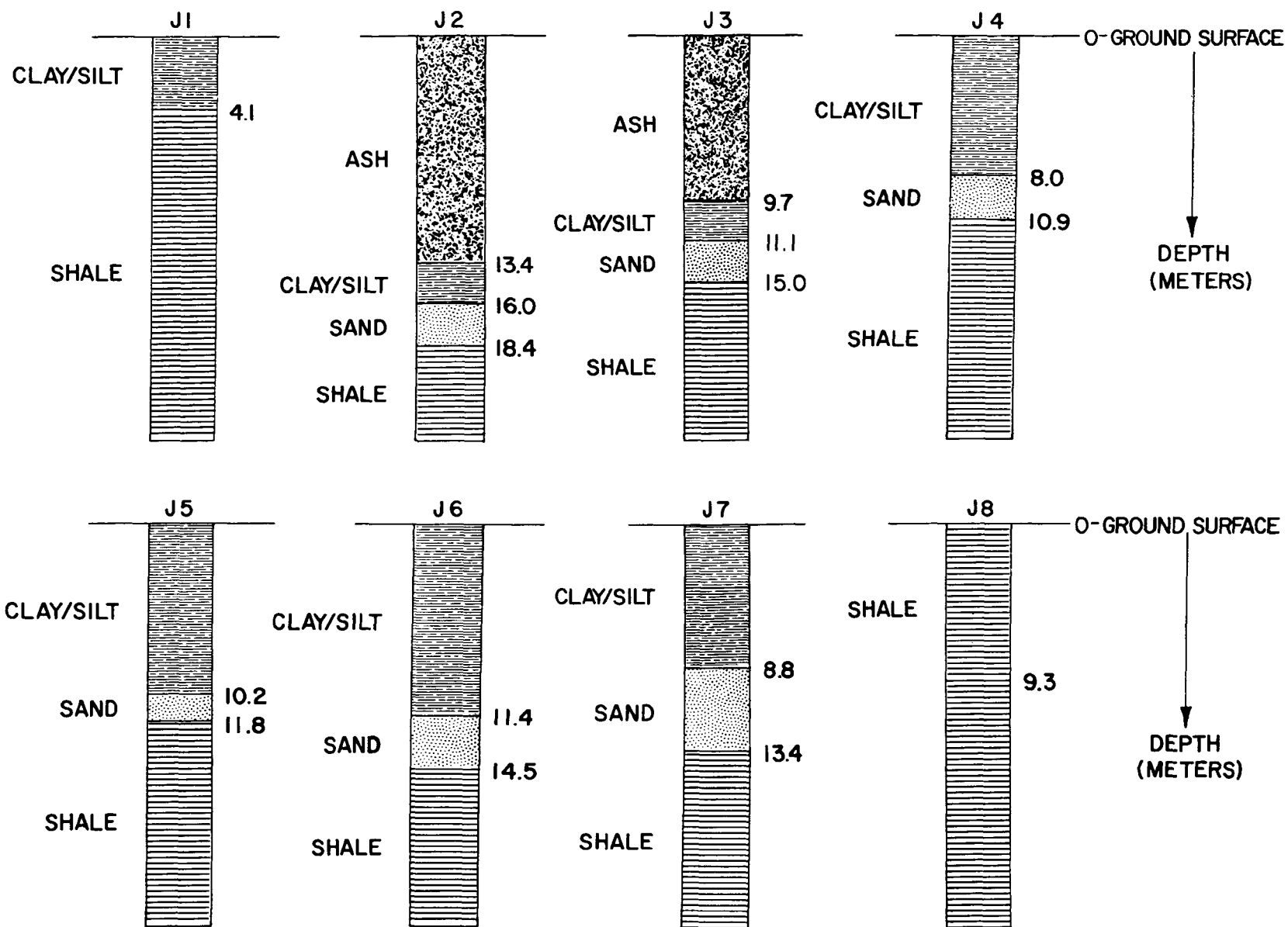
VERTICAL PROFILE OF THE SUBSTRATUM AT PLANT L's MONITORING WELL LOCATIONS

VERTICAL PROFILE OF THE SUBSTRATUM AT PLANT L's MONITORING WELL LOCATIONS

L7

CLAY/SILT

8.9

LIMESTONE

L8

ASH
2.6

CLAY/SILT

11.7

LIMESTONE

L9

ASH

5.4

CLAY/SILT

10.8

LIMESTONE

0-GROUND SURFACE

DEPTH
(METERS)

LIO

CLAY/SILT

6.7

GRAVELLY CLAY

12.3

CLAY/SILT

21.1

LIMESTONE

L II

0-GROUND SURFACE

CLAY/SILT

DEPTH
(METERS)

18.1

LIMESTONE
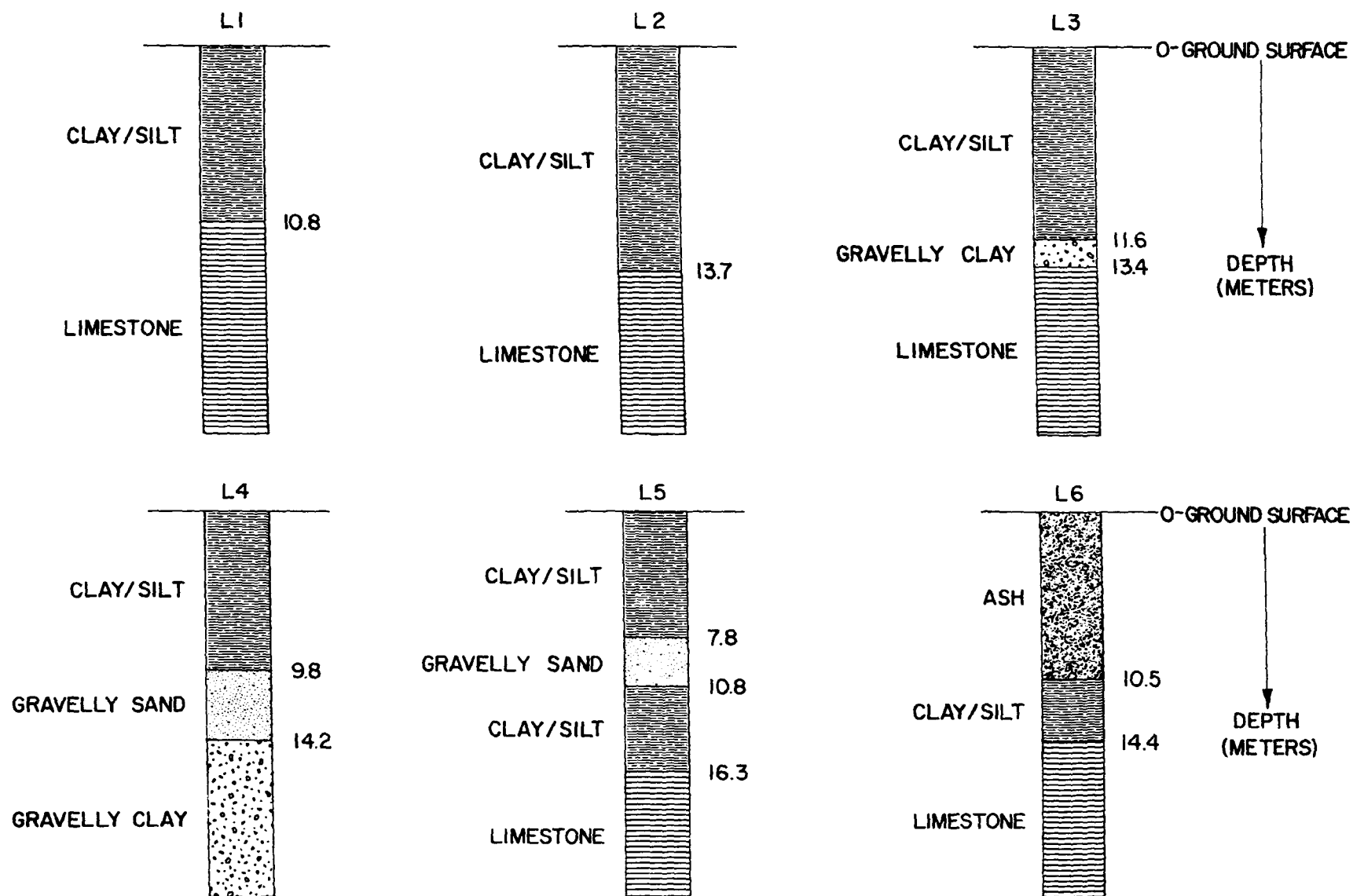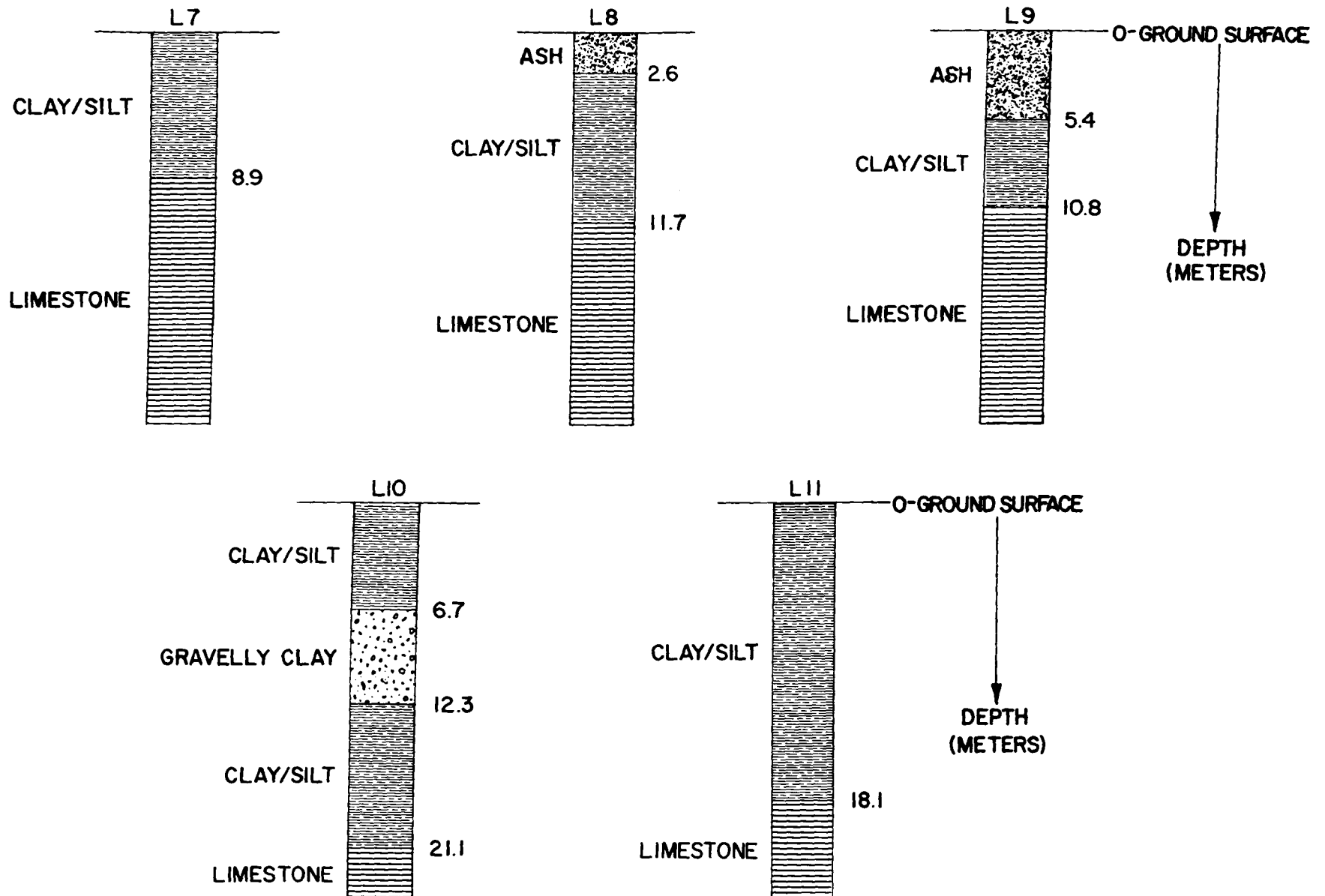
APPENDIX C

ANALYTICAL RESULTS OF COLUMN ATTENUTATION STUDIES

TABLE C-1.   ANALYSIS OF COAL ASH LEACHATE USED IN SOIL
             ATTENUATION STUDY

Constituent	Value	Constituent	Value
pH, std. units	7.8	Lead, mg/l	<0.010
Eh, millivolts	-280	Beryllium, mg/l	<0.002
Temperature, °C	17	Barium, mg/l	0.23
Conductivity, μmhos/cm^2	1800	Manganese, mg/l	4.2
Alkalinity, mg/l		Sodium, mg/l	7.2
as CaCO$_3$	340	Potassium, mg/l	14
Calcium, mg/l	185	Selenium, mg/l	<0.002
Magnesium, mg/l	7.6	Total dissolved	
Iron, mg/l	15	solids, mg/l	720
Copper, mg/l	0.14	Sulfide, mg/l	0.07
Zinc, mg/l	0.04	Sulfate, mg/l	240
Nickel, mg/l	<0.05	Nitrite and Nitrate	
Aluminum, mg/l	0.06	as nitrogen, mg/l	0.02
Cadmium, mg/l	<0.01	Phosphate as	
Mercury, mg/l	0.002	Phosphorus, mg/l	0.08
Chromium, mg/l	<0.05		

TABLE C-2. ANALYTICAL RESULTS OF COLUMN ATTENUATION STUDIES

	Analysis of effluent from soil column—Plant L						
Composite volume, ml	180	150	180	150	150	180	120
Total volume eluted, ml	180	330	510	660	810	990	1110
Calcium, mg/L	120	180	190	180	180	180	170
Magnesium, mg/L	9.0	9.0	9.3	9.4	9.6	9.6	9.2
Sodium, mg/L	8.6	7.9	8.2	8.1	8.6	8.2	7.9
Potassium, mg/L	6.3	13	14	15	15	14	14
Iron, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Manganese, mg/L	0.48	0.06	0.26	0.44	0.650	0.83	0.86
Copper, mg/L	0.02	0.008	0.007	0.007	<0.005	<0.005	0.007
Zinc, mg/L	<0.05	<0.01	0.19	<0.01	<0.01	0.14	<0.01
Nickel, mg/L	0.18	0.05	0.05	0.09	<0.05	<0.05	0.07
Sulfate, mg/L	300	300	300	240	310	280	270
Cadmium, mg/L	0.009	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Lead, mg/L	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Chromium, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Aluminum, mg/L	0.09	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
Barium, mg/L	0.1	0.08	0.07	0.08	0.1	0.12	0.13
Beryllium, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Mercury, mg/L	0.03	0.0005	0.0003	0.0002	0.0003	0.0003	0.0002
Selenium, mg/L	<0.004	<0.002	<0.002	<0.004	<0.002	<0.002	<0.01

TABLE C-2 (continued)

	Analysis of effluent from soil column--Plant J					
Composite volume, ml	300	270	300	270	300	270
Total volume eluted, ml	300	570	870	1140	1440	1710
Calcium, mg/L	160	160	160	160	160	150
Magnesium, mg/L	10	9.4	9.5	9.5	9.7	9.2
Sodium, mg/L	8.9	8.0	8.0	8.2	8.4	8.2
Potassium, mg/L	10	12	12	13	13	12
Iron, mg/L	0.12	0.068	0.063	0.053	0.039	0.03
Manganese, mg/L	0.5	0.49	0.77	1.1	1.1	0.94
Copper, mg/L	0.02	0.006	0.02	0.006	0.007	0.022
Zinc, mg/L	0.16	0.03	0.02	0.03	0.02	0.02
Nickel, mg/L	1.1	<0.05	<0.05	<0.05	<0.05	<0.05
Sulfate, mg/L	*	*	*	*	*	*
Cadmium, mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Lead, mg/L	0.032	<0.01	<0.01	<0.01	<0.01	<0.01
Chromium	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Aluminum, mg/L	0.1	0.05	0.04	0.05	<0.02	0.17
Barium, mg/L	0.12	0.13	0.1	0.16	0.17	0.16
Beryllium, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Mercury	<0.0008	<0.0008	<0.0008	<0.0008	<0.0008	<0.0008
Selenium, mg/L	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004

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TABLE C-2 (continued)

	Analysis of effluent from kaolinite clay column						
Composite volume, ml	180	180	180	180	180	180	120
Total volume eluted, ml	180	360	540	720	900	1080	1200
Calcium, mg/L	130	150	150	160	160	170	120
Magnesium, mg/L	9.0	9.9	9.9	9.5	9.2	10	7.2
Sodium, mg/L	9.6	8.5	8.5	8.3	7.8	8.8	7.0
Potassium, mg/L	14	16	16	15	15	16	14
Iron, mg/L	0.046	0.013	0.013	0.022	0.099	0.063	0.014
Manganese, mg/L	0.75	0.22	0.25	0.96	1.1	1.3	0.92
Copper, mg/L	0.045	0.056	0.046	0.039	0.081	0.04	0.16
Zinc, mg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.12
Nickel, mg/L	0.96	0.18	0.07	0.07	0.05	0.05	<0.05
Sulfate, mg/L	*	*	*	*	*	*	*
Cadmium, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Lead, mg/L	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
Chromium, mg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Aluminum, mg/L	0.04	<0.02	<0.02	0.03	0.07	0.06	0.06
Barium, mg/L	0.13	0.15	0.16	0.19	0.17	0.2	0.14
Beryllium, mg/L	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Mercury, mg/L	<0.0002	<0.0002	<0.0002	<0.0002	0.0003	0.0003	0.0003
Selenium, mg/L	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	<0.01

*Not enough sample for analysis.

APPENDIX D

USED IN COLUMN ATTENUATION STUDIES

TABLE D-1. CHEMICAL CHARACTERISTICS OF CLAY MINERALS
USED IN COLUMN ATTENUATION STUDIES

Illite

Constituent	Value	Constituent	Value
Chlorine, mg/l	460.0	Zinc, µg/g	17
Sulfate, mg/l	480.0	Nickel, µg/g	26
Bromine	1.2	Cadmium, µg/g	<1.0
Calcium, µg/g	4000	Lead, µg/g	12.0
Magnesium, µg/g	2500	Chromium, µg/g	<5.0
Sodium, µg/g	300	Aluminum, µg/g	7700
Potassium, µg/g	5300	Barium, µg/g	830
Iron, µg/g	6100	Beryllium, µg/g	5.0
Total Manganese, µg/g	65	Mercury, µg/g	0.27
Copper, µg/g	52	Arsenic, µg/g	1.2
		Selenium, µg/g	<1.0

Kaolinite

Constituent	Value	Constituent	Value
Chlorine, mg/l	30.0	Zinc, µg/g	52
Sulfate, mg/l	120.0	Nickel, µg/g	21
Bromine	<0.2	Cadmium, µg/g	<1.0
Calcium, µg/g	120	Lead, µg/g	10.0
Magnesium, µg/g	110	Chromium, µg/g	<5.0
Sodium, µg/g	60	Aluminum, µg/g	840
Potassium, µg/g	70	Barium, µg/g	61
Iron, µg/g	1400	Beryllium, µg/g	3.0
Total Manganese, µg/g	830	Mercury, µg/g	0.60
Copper, µg/g	15	Arsenic, µg/g	10.0
		Selenium, µg/g	<1.0

Montmorillonite

Constituent	Value	Constituent	Value
Chlorine, mg/l	26.0	Zinc, µg/g	70
Sulfate, mg/l	1800.0	Nickel, µg/g	<5.0
Bromine	<0.2	Cadmium, µg/g	<1.0
Calcium, µg/g	770	Lead, µg/g	48.0
Magnesium, µg/g	3400	Chromium, µg/g	<5.0
Sodium, µg/g	1100	Aluminum, µg/g	2900
Potassium, µg/g	200	Barium, µg/g	13
Iron, µg/g	1700	Beryllium, µg/g	2.0
Total Manganese, µg/g	22	Mercury, µg/g	0.17
Copper, µg/g	3	Arsenic, µg/g	2.3
		Selenium, µg/g	<1.0

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO. EPA-600/7-80-066	2.	3. RECIPIENT'S ACCESSION NO.
4. TITLE AND SUBTITLE Effects of Coal-ash Leachate on Ground Water Quality		5. REPORT DATE March 1980
		6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) Jack D. Milligan and Richard J. Ruane		8. PERFORMING ORGANIZATION REPORT NO.
9. PERFORMING ORGANIZATION NAME AND ADDRESS Tennessee Valley Authority 1120 Chestnut Street, Tower II Chattanooga, Tennessee 37401		10. PROGRAM ELEMENT NO. INE624A
		11. CONTRACT/GRANT NO. EPA Interagency Agreement D5-E721
12. SPONSORING AGENCY NAME AND ADDRESS EPA, Office of Research and Development Industrial Environmental Research Laboratory Research Triangle Park, NC 27711		13. TYPE OF REPORT AND PERIOD COVERED Final; 5/75-11/79
		14. SPONSORING AGENCY CODE EPA/600/13

15. SUPPLEMENTARY NOTES IERL-RTP project officer is Michael C. Osborne, Mail Drop 62, 919/541-2547. TVA project director is Hollis B. Flora II.

16. ABSTRACT The report gives results of research to: (1) develop methodology for the field-collection of coal-ash leachate; (2) chemically characterize ash leachates from power plants using different coal sources; (3) determine the characteristics of the hydrogeochemical environment in which the leachate occurs; and (4) determine the attenuation of coal-ash leachate by various soil types. Groundwater monitoring wells were installed around ash ponds at two TVA plants. Continuous soil-core samples were collected and analyzed periodically. Ash leachate was percolated through different clays and soils to study attenuation rates. Results include indications that: (1) coal-ash leachate is highly variable, but characteristically high in dissolved solids, B, Fe, Ca, Al, and SO4; ash leachate is acidic, with measured pH as low as 2; (2) the coal sources associated with the study produced ash leachate with similar characteristics; (3) an inert-gas lift pump was effective in collecting anoxic groundwater samples while minimizing oxidation; (4) differences were found in the characteristics of leachate samples obtained by extracting the interstitial soil water and samples collected from the monitoring wells; interstitial water samples contained higher concentrations of metals and were more acidic than well samples; and (5) the flux of metals in the ash pond leachate was negligible compared to ash pond surface overflow.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Pollution Leaching Water Quality Ground Water Coal Ashes	Pollution Control Stationary Sources Coal Ash	13B 07D,07A 08H 21D 21B

18. DISTRIBUTION STATEMENT Release to Public	19. SECURITY CLASS *(This Report)* Unclassified	21. NO. OF PAGES 125
	20. SECURITY CLASS *(This page)* Unclassified	22. PRICE

EPA Form 2220-1 (9-73)