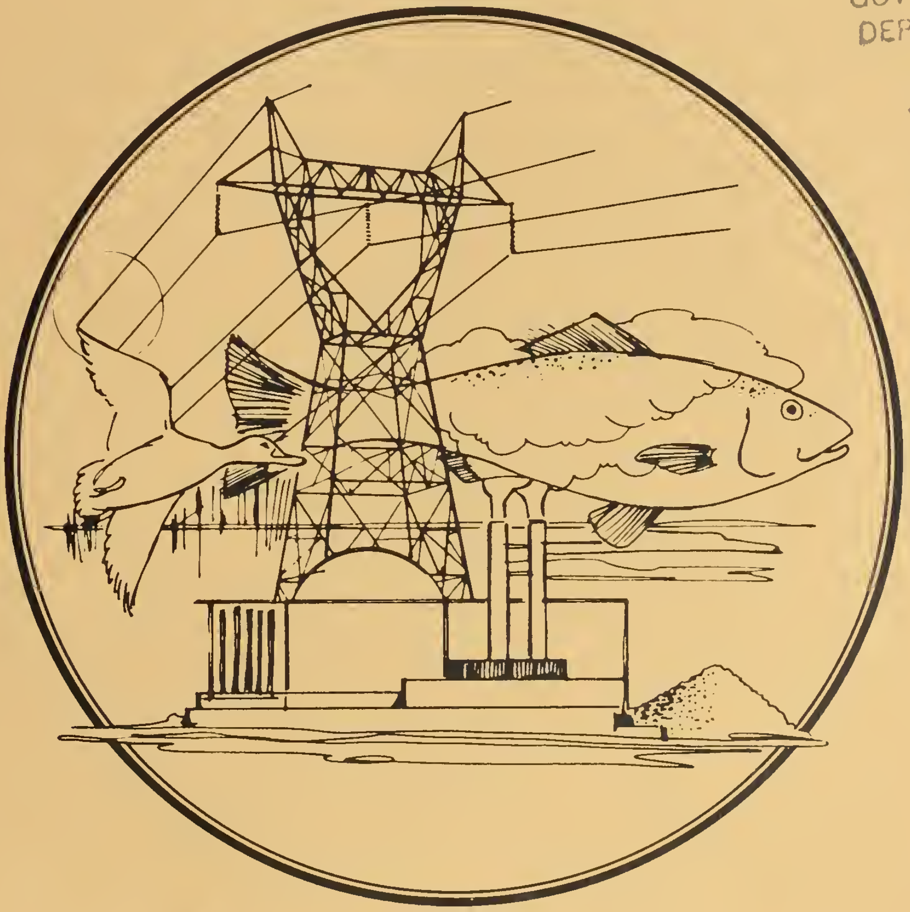
Biological Services Program

FWS/OBS-80/33
September 1980

Handling of Combustion and Emission-Abatement Wastes from Coal-Fired Power Plants: Implications for Fish and Wildlife Resources



Office of Research and Development
U.S. Environmental Protection Agency
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2. To gather, analyze, and present information that will aid decision makers in the identification and resolution of problems associated with major land and water use changes.

3. To provide better ecological information and evaluation for Department of the Interior development programs, such as those relating to energy development.

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HANDLING OF COMBUSTION AND EMISSION-ABATEMENT WASTES FROM COAL-FIRED POWER PLANTS: IMPLICATIONS FOR FISH AND WILDLIFE RESOURCES

Lars F. Soholt, Project Leader
Robert W. Vocke, Assistant Project Leader

N. J. Beskid
W. K. Derickson
M. J. Knight
D. D. Ness
B. Siskind
W. S. White

D. J. Wyman, Editor

Division of Environmental Impact Studies
Argonne National Laboratory

Michael Avery/James P. Bennett
Project Officers
National Power Plant Team
U.S. Fish and Wildlife Service
2929 Plymouth Road
Ann Arbor, Michigan 48105

Prepared for
National Power Plant Team
Office of Biological Services
Fish and Wildlife Service
U.S. Department of the Interior

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The correct citation for this report is:

Soholt, L.F., et al. 1980. Handling of Combustion and Emission-Abatement Wastes from Coal-Fired Power Plants: Implications for Fish and Wildlife Resources. U.S. Fish and Wildlife Service, Biological Services Program, National Power Plant Team, FWS/OBS-80/33. 184 pp.

ABSTRACT

Increased emphasis upon using coal as an energy source and the concomitant requirements for flue-gas-emission control have led to increased production of solid wastes, which is expected to accelerate over the next several decades. The goals of this report are to (1) provide a basic introduction to handling of wastes from coal combustion and emission abatement and (2) present a procedure for evaluating the potential for these wastes to impact fish and wildlife resources.

Coal combustion ashes and flue-gas-desulfurization (FGD) sludges, the solid waste products from coal-fired facilities, contain a number of trace elements that can be toxic to biota if they are available in sufficient quantities. Both ashes and FGD sludges are usually deposited in pond or landfill storage areas. Dispersal of constituents from waste-storage sites occurs primarily by runoff, seepage, and wind erosion. This report contains qualitative and quantitative methods for evaluating the potential impacts from these routes of dispersal in site-specific situations. Generally, pond storage methods, even when properly managed, have a greater impact upon fish and wildlife resources than do landfill methods. Proper management of storage sites reduces the amount of waste constituents that are dispersed into the environment.

It is difficult to make site-specific predictions regarding the toxicity of materials mobilized from the wastes. The potential for uptake of trace elements to toxic levels is dependent upon a number of factors including (1) pH of the dispersal and growth media, (2) capacity of the dispersal and growth media to bind elements in a form unavailable for uptake, (3) magnitude of biological concentration of elements in primary producers and succeeding trophic levels, and (4) tolerances of individual species. In this report, we have provided some generalized information that can be used to estimate the relative likelihood of toxicity problems resulting from dispersal of trace elements from coal ashes and FGD sludges.

After the active lifetime of a waste-storage site, revegetation is desirable as a means of controlling erosion and regaining potential fish and wildlife habitat. A number of plant species have been shown to successfully establish on fly ash; however, toxic effects of the ash constituents have been demonstrated in several cases, and wildlife forage plants have been shown to accumulate some of these constituents to potentially toxic levels. Revegetation of FGD sludge has not been well-documented, although ongoing research may provide methods to successfully revegetate this material.

Four model waste-storage sites are used to illustrate the methods of assessment presented in this report. Future research needs are identified.

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INTRODUCTION

Increased use of coal in the generation of electricity has become national policy. With the anticipated acceleration of coal combustion, a concomitant increase can be expected in the potential for impacts to fish and wildlife resources. In the past, attention has focused upon the potential for deleterious effects from the aerial emissions of coal combustion residues, including ash particulates and oxides of sulfur. This has led to regulatory emphasis on restricting the amount of emissions to the atmosphere. New source performance standards promulgated by the USEPA will require restriction of atmospheric emissions at virtually all coal-fired electric generating stations. However, disposition of both the pollutants extracted from flue gases and the reagents used in the extraction process poses a problem that has only recently received much attention.

Personnel of the U.S. Fish and Wildlife Service are responsibile for assessing the impact of these flue-gas control wastes upon the nation's fish and wildlife resources. These responsibilities are met through consultation with other agencies and through review of environmental assessment documents. The purpose of this report is to provide a data base that will allow the fish and wildlife biologist to effectively critique plans for the disposition of flue-gas control wastes and boiler ash residues in the context of potential impacts upon biotic resources.

This report is an outgrowth of an earlier report prepared by Dvorak et al. (1978) for the U.S. Fish and Wildlife Service, entitled "Impacts of Coal-Fired Power Plants on Fish, Wildlife, and Their Habitats" (FWS/OBS-78/29). The intent of the authors of this document is to expand the discussion of combustion and emission-abatement wastes. Because of the desire to ensure that the current document can be used without reference to the earlier document, there is, necessarily, overlap between the two documents.

The scope of this document is restricted to the combustion and emission-abatement waste-handling system. For general background information about the operation of a coal-fired, electric generating facility, the reader is referred to Dvorak et al. (1978). We have not described in detail the mechanisms for extracting flue-gas pollutants and have restricted our discussion to the period following collection of the wastes. However, the reader may find descriptions of flue-gas control systems in Dvorak et al. (1978) as well as other documents, including "Air Pollution. Vol. IV, Engineering Control of Air Pollution," edited by Stern (1976).

The approach taken in this report is to divide the waste-handling system into three facets: storage, management, and ultimate reclamation. We have chosen to term the initial part of waste handling as storage rather than disposal because the latter term implies that the wastes have been discarded and require no further attention. This is often not the case if one wishes to adequately protect biotic resources; nor is such "open dumping" allowable under current federal regulations. Discussion of waste storage includes description of the nature of the wastes, alternative methods of processing and storage, and potential impacts to biotic resources both from dispersal of waste constituents from storage sites and from competitive uses of biotic habitat. Discussion of management emphasizes techniques with which the waste handler can mitigate the potential for impacts and decrease the likelihood of wastes contacting biota. Lastly, discussion focuses on stabilization and revegetation of sites after their useful lifetime as active storage areas. A goal of this last phase of the waste-handling system can be to accelerate reclamation of fish and wildlife habitat.

Throughout the report, we have given the reader tools that can be used to evaluate specific situations that may be encountered in reviewing proposed plans. These presentations will serve as the foundation for a concise handbook for evaluating the potential effects upon biotic resources from combustion and emission-abatement waste-storage projects. The handbook will be published in the near future.

The reader must bear in mind that the approach taken for evaluating impacts from storage facilities is neither the only approach nor a unique approach. The ideas of others have been borrowed from substantially and molded into a format pertinent to the needs of the fish and wildlife biologist. The data base upon which this report was developed is rapidly changing: new data are being generated for innovative waste-handling techniques, flue-gas control technologies, and the responses of biota to these wastes. This document does not provide a final answer; however, it does provide an introduction to a topic that is of great concern in the area of environmental protection.

The International System of Units (SI) has been used in this report with a few exceptions (e.g., Btu/lb). Definitions and conversion factors (Appendix A) follow the American Society for Testing and Materials (1976) "Standard for Metric Practice." A glossary of technical terms and acronyms that may be unfamiliar to the reader is provided in Appendix B.

COAL COMBUSTION AND EMISSION-ABATEMENT WASTES

The physical and chemical properties of combustion and emission-abatement wastes are major determinants of how the wastes must be handled (Duvel et al. 1979; DiGioia et al. 1979; GAI Consultants 1979). These properties also influence the probability of waste constituents being mobilized into the environment surrounding the waste-handling facility. The various constituents have varying toxicities to biota; thus, one must know the chemical makeup of wastes before assessing the potential impacts. This chapter provides an overview of the properties of ash and sludge from coal combustion and emission abatement. This discussion can be used in a general assessment of the potential environmental hazards from these wastes. However, the great variability evident in this discussion indicates that site-specific assessments must be based upon data collected from the actual wastes produced by a coal-fired plant.

COAL ASH WASTES AND THEIR PROPERTIES

Three kinds of products are formed from the combustion of coal: power plant aggregate (bottom ash and slag), fly ash, and gases. Power plant aggregate is that part of the residual combustion solids that has fused into particles heavy enough to drop out of the boiler gas stream (i.e., the air and combustion gases). Fly ash, by contrast, is the fraction of the ash that becomes entrained in the gas stream leaving the boiler. Gases include those portions of the coal material that have been volatilized during the combustion process. The inorganic residue, remaining either as aggregate or as fly ash after coal has been burned, originates chiefly from the inorganic mineral matter that was present in the unburned coal, although it may vary considerably in composition from the mineral matter originally in the coal. This inorganic mineral matter, known as the ash content of the coal, usually ranges from about 3 to 30% by weight of the unburned coal. A series of measurements of the ash content and major ash components for each rank of coal from lignite to anthracite is presented in Table 1. Discussions of the classification of coal by rank are readily available, e.g., Ergun (1979). For most of the elements comprising the inorganic mineral matter in coal, more than 95% of the quantity present will be found with the fly ash or bottom ash fractions upon combustion. The remaining portion of less than 5% is discharged into the atmosphere as gases (Ray and Parker 1977).

Table 1. Variation in Coal Ash Composition with Rank[a]

	% composition (range) per rank			
Constituent	Anthracite	Bituminous	Subbituminous	Lignite
Silica (SiO_2)	48 -68	7 -68	17 -58	6 -40
Alumina (Al_2O_3)	25 -44	4 -39	4 -35	4 -26
Ferric oxide (Fe_2O_3)	2 -10	2 -44	3 -19	1 -34
Titanium dioxide (TiO_2)	1.0-2	0.5-4	0.6-2	0.0-0.8
Calcium oxide (CaO)	0.2-4	0.7-36	2.2-52	12.4-52
Magnesium oxide (MgO)	0.2-1	0.1-4	0.5-8	2.8-14
Sodium oxide (Na_2O)	-	0.2-3	-	0.2-28
Potassium oxide (K_2O)	-	0.2-4	-	0.1-1.3
Sulfur trioxide (SO_3)	0.1-1	0.1-32	3.0-16	8.3-32
Total ash	4 -19	3 -32	3 -16	4 -19

[a] Data from Ray and Parker (1977).

The quantitative partitioning of the ash residue between the aggregate and fly ash fractions is a function of the firing method, ash fusion temperature, and aggregate collection method. In the United States, direct-suspension firing of pulverized coal is commonly used; the coal is usually pulverized to about 0.075-mm-diameter particles and, while suspended in a moving air stream, is delivered to the burners in a single continuous operation. In cyclone and spreader-stoker firing, the coal is typically crushed to about 5 mm in diameter; the finer particles are burned in suspension whereas the coarser ones are thrown either to a wall coated with molten slag (cyclone) or to a grate (spreader-stoker firing), where they are burned at rest. Reid et al. (1973) and Babcock and Wilcox (1978) present further details on firing methods.

In the various pulverized-coal units, usually 65 to 85% of the ash is produced as fly ash and the remainder as aggregate; in cyclone units, the ash fusion temperature (i.e., melting point of the ash) is exceeded, and about 90% of the ash is collected as aggregate (Table 2).

Table 2. Comparison of Distribution Between Bottom Ash and Fly Ash by Type of Boiler and Method of Firing[a]

		% (typical)	
Type of boiler	Type of firing	Bottom ash	Fly ash
Wet-bottom	Pulverized-coal suspension	35	65
Dry-bottom	Pulverized-coal suspension	15	85
Wet-bottom	Cyclone	90	10
Not applicable	Spreader-stoker	35	65

[a] Data from Ray and Parker (1977).

The small quantities of fly ash from cyclone units consist of very fine particles, 90% of which are less than 10 μm in diameter. In general, ashes with lower fusion temperatures tend to be melted within the boiler and collected as slag (Ray and Parker 1977). Initial deformation of the coal ash may occur at temperatures ranging from about 1100°C (2000°F) for lignitic coals to more than 1600°C (2900°F) for low-volatile bituminous coals. Ash fusion-temperature data for some U.S. coals and lignites are presented in Table 3 (Babcock and Wilcox 1978).

Newer installations commonly use the so-called "dry bottom" boilers, in which the aggregate falls as bottom ash through a grate into an ash hopper usually filled with water. "Wet-bottom" or slag-tap boilers are more common in older installations; the aggregate is tapped in the molten state and falls into a water-filled ash hopper. Wet-bottom boilers are designed to produce and process a much larger fraction of the aggregate (as slag) than are dry-bottom boilers (Table 2).

The chemical composition of coal ash depends largely on (1) the geologic and geographic factors related to the coal deposit and (2) the combustion conditions. The inorganic constituents of ash are those typical of rocks and soils--primarily Si, Al, Fe, and Ca (Table 1); the oxides of these four elements comprise 95 to 99% of the composition of ash. Ash also contains smaller amounts (0.5 to 3.5%) of Mg, Ti, S, Na, and K, as well as very small quantities (on the order of parts per million) of up to 50 other elements (Table 4).

One must use caution in attempting to characterize the effluents from a power plant based on the average ash content from coal of any given rank, because the maxima and minima of some elements vary greatly among ashes from coals of the same rank (Table 4). The inorganic ash content varies even within a single seam (Babcock and Wilcox 1978). Thus, precise prediction of the composition of the fly ash, slag, and bottom ash cannot be made at any given time without an analysis of the particular coal being used in the facility at that time.

Analysis of various ashes shows that the distribution of major elements is approximately the same in the bottom ash and fly ash fractions. However, for certain of the trace components, there is a very definite partitioning between the bottom ash and fly ash. As seen in Table 5, there can be differences of an order of magnitude in the concentrations of some trace elements between these two fractions; for example, Se is much more concentrated in the fly ash fraction.

Table 3. Ash Content and Ash Fusion Temperatures of Some U.S. Coals and Lignite[a]

	Low-volatile bituminous	High-voltatile bituminous				Sub-bituminous	Lignite
Seam	Pocahontas No. 3	No. 9	Pittsburgh	No. 6			
Location	West Virginia	Ohio	West Virginia	Illinois	Utah	Wyoming	Texas
Ash, dry basis (%)	12.3	14.10	10.87	17.36	6.6	6.6	12.8
Sulfur, dry basis (%)	0.7	3.30	3.53	4.17	0.5	1.0	1.1
Analysis of ash (% by weight)							
SiO_2	60.0	47.27	37.64	47.52	48.0	24.0	41.8
Al_2O_3	30.0	22.96	20.11	17.87	11.5	20.0	13.6
TiO_2	1.6	1.00	0.81	0.78	0.6	0.7	1.5
Fe_2O_3	4.0	22.81	29.28	20.13	7.0	11.0	6.6
CaO	0.6	1.30	4.25	5.75	25.0	26.0	17.6
MgO	0.6	0.85	1.25	1.02	4.0	4.0	2.5
Na_2O	0.5	0.28	0.80	0.36	1.2	0.2	0.6
K_2O	1.5	1.97	1.60	1.77	0.2	0.5	0.1
Total	98.8	98.44	95.74	95.20	97.5	86.4	84.3
Ash fusibility							
Initial deformation temperature (°C)							
Reducing conditions	>1590	1110	1110	1090	1130	1090	1080
Oxidizing conditions	>1590	1330	1240	1260	1160	1200	1130
Fluid temperature (°C)							
Reducing conditions		1440	1300	1270	1230	1250	1230
Oxidizing conditions		1470	1390	1430	1350	1266	1250

[a] Data from Babcock and Wilcox (1978).

Table 4. Concentration of Some Trace Elements in Coal Ashes[a]

	Concentration (ppm)														
	Anthracites			High-volatile bituminous			Low-volatile bituminous			Medium-volatile bituminous			Lignites and subbituminous		
Element	Min.	Max.	Avg.[b]	Min.	Max.	Avg.[c]	Min.	Max.	Avg.[d]	Min.	Max.	Avg.[e]	Min.	Max.	Avg.[f]
Barium	540	1340	866	210	4660	1253	96	2700	740	230	1800	896	550	13900	5027
Beryllium	6	11	9	4	60	17	6	40	16	4	31	13	1	28	6
Boron	63	130	90	90	2800	770	76	180	123	74	780	218	320	1900	1020
Chromium	210	395	304	74	315	193	120	490	221	36	230	169	11	140	54
Cobalt	10	165	81	12	305	64	26	440	172	10	290	105	11	310	45
Copper	96	540	405	30	770	293	76	850	379	130	560	313	58	3020	655
Gallium	30	71	42	17	98	40	10	135	41	10	52	*	10	30	23
Germanium	20	20	*	20	285	*	20	20	*	20	20	*	20	100	*
Lanthanum	115	220	142	29	270	111	56	180	110	19	140	83	34	90	62
Lead	41	120	81	32	1500	183	23	170	89	52	210	96	20	165	60
Manganese	58	365	270	31	700	170	40	780	280	125	4400	1432	310	1030	688
Nickel	125	320	220	45	610	154	61	350	141	20	440	263	20	420	129
Scandium	50	82	61	7	78	32	15	155	50	7	110	56	2	58	18
Silver	1	1	*	1	3	*	1	1.4	*	1	1	*	1	50	*
Strontium	80	340	177	170	9600	1987	66	2500	818	40	1600	668	230	8000	4660
Tin	19	4250	962	10	825	171	10	230	92	29	160	75	10	660	156
Vanadium	210	310	248	60	840	249	115	480	278	170	860	390	20	250	125
Ytterbium	5	12	8	3	15	10	4	23	10	4	13	9	2	10	4
Yttrium	70	120	106	29	285	102	37	460	152	37	340	151	21	120	51
Zinc	155	350	*	50	1200	310	62	550	231	50	460	195	50	320	*
Zirconium	370	1200	688	115	1450	411	220	620	458	180	540	326	100	490	245

[a] Data from Ray and Parker (1977). An asterisk indicates insufficient data to compute an average value.
[b] Number of samples used to compute average values equals 5, except cobalt (4).
[c] Number of samples used to compute average values equals 24, except tin (22) and zinc (14).
[d] Number of samples used to compute average values equals 8, except gallium (7) and tin (7).
[e] Number of samples used to compute average values equals 7--except cobalt (6), nickel (6), and zinc (6).
[f] Number of samples used to compute average values equals 13--except gallium (12), nickel (8), and scandium (10).

Table 5. Comparison of Fly Ash and Bottom Ash from Various Utility Plants[a]

Compound or element	Plant 1		Plant 2		Plant 3		Plant 4		Plant 5		Plant 6	
	Fly ash	Bottom ash	Fly ash	Bottom ash	Fly ash	Bottom ash	Fly ash	Bottom ash	Fly ash	Bottom ash	Fly ash	Bottom ash
							%					
SiO_2	59	58	57	59	43	50	54	59	NR	NR	42	49
Al_2O_3	27	25	20	18.5	21	17	28	24	NR	NR	17	19
Fe_2O_3	3.8	4.0	5.8	9.0	5.6	5.5	3.4	3.3	20.4	30.4	17.3	16.0
CaO	3.8	4.3	5.7	4.8	17.0	13.0	3.7	3.5	3.2	4.9	3.5	6.4
SO_3	0.4	0.3	0.8	0.3	1.7	0.5	0.4	0.1	NR	0.4	NR	NR
MgO	0.96	0.88	1.15	0.92	2.23	1.61	1.29	1.17	NR	NR	1.76	2.06
Na_2O	1.88	1.77	1.61	1.01	1.44	0.64	0.38	0.43	NR	NR	1.36	0.67
K_2O	0.9	0.8	1.1	1.0	0.4	0.5	1.5	1.5	NR	NR	2.4	1.9
P_2O_5	0.13	0.06	0.04	0.05	0.70	0.30	1.00	0.75	NR	NR	NR	NR
TiO_2	0.43	0.62	1.17	0.67	1.17	0.50	0.83	0.50	NR	NR	1.00	0.68
							ppm					
As	12	1	8	1	15	3	6	2	8.4	5.8	110	18
Be	4.3	3	7	7	3	2	7	5	8.0	7.3	NR	NR
Cd	0.5	0.5	0.5	0.5	0.5	0.5	1.0	1.0	6.44	1.08	8.0	1.1
Cr	20	15	50	30	150	70	30	30	206	124	300	152
Cu	54	37	128	48	69	33	75	40	68	48	140	20
Hg	0.07	0.01	0.01	0.01	0.03	0.01	0.08	0.01	20.0	0.51	0.05	0.028
Mn	267	366	150	700	150	150	100	100	249	229	298	295
Ni	10	10	50	22	70	15	20	10	134	62	207	85
Pb	70	27	30	30	30	20	70	30	32	8.1	80	6.2
Se	6.9	0.2	7.9	0.7	18.0	1.0	12.0	1.0	26.5	5.6	25	0.08
V	90	70	150	85	150	70	100	70	341	353	440	260
Zn	63	24	50	30	71	27	103	45	352	150	740	100
B	266	143	200	125	300	70	700	300	NR	NR	NR	NR
Co	7	7	20	12	15	7	15	7	6.0	3.6	39	20.8
F	140	50	100	50	610	100	250	85	624	10.6	NR	NR

[a] Data from Ray and Parker (1977).

NR indicates that data were not reported.

The chemical elements that constitute the minerals in coal and ash may be grouped into three general classes as follows (Mann et al. 1978):

Class I. Elements that have been reported to be approximately equally distributed (i.e., the same proportion by weight) in all ash fractions are Al, Ba, Ca, Fe, Mg, Rb, Si, Sn, and Sr.

Class II. Elements that have been reported to be enriched relative to the aggregate in the smaller fly ash particles are As, Br, Cd, Cl, Cr, Cu, F, Ga, Hg, I, Mo, Pb, S, Sb, Se, Zn. Of these, Br, Cl, F, Hg, S, Se, and Zn are generally thought to be emitted from the plant stack either partially or totally as vapors that may condense on the fly ash particles.

Class III. Elements for which enrichment in or on fly ash has been reported in some studies, but not in others, are Be, Co, Mn, Ti, and V.

The organic composition of coal ashes has not been well characterized, although a few studies have been carried out (Tables 6 and 7). The presence of organic compounds in coal ash is the result of incomplete combustion; the amount and nature of the organic residue then depends on the chemical nature of the coal and completeness of combustion. The organic species, in particular the polycyclic aromatic compounds, have been found to be present in much higher specific concentrations in fly ash emitted from the stack than in fly ash collected in bulk by particulate control devices within the coal-fired plant (e.g., 19 μg/g as compared to 0.02 μg/g). Fisher and Natusch (1979) proposed that the organic compounds are present as a component of the flue gases within the plant and are not collected in precipitators but that they do condense onto the surface of fly ash particles as the flue stream is emitted from the stack and cools. It is also possible that the organic forms may be modified within particulate control devices, leading to differences in carcinogenicity between collected and released ashes. These organic species are thus more likely to pose serious problems in aerial releases than in solid-waste releases.

Fly Ash Fraction

The fly ash fraction of the combustion products generally consists of fine spherical particulates ranging from 0.5 to more than 100 μm in diameter. Up to 5% by weight (20% by volume) of these particulates consists of cenospheres, i.e., silicate glass spheres filled with nitrogen and carbon dioxide and ranging from 20 to 200 μm in diameter. The measured particle-size distribution of the fly ash depends on the type of collector employed, some collectors being more efficient than others in collecting the very small particles (Ray and Parker 1977). The densities of 12 fly ashes at 24°C were found to range from about 2 to 3 g/cm^3 by Coltharp et al. (1979b). Hart and DeLaney (1978), quoting Cooper (1975), give the normal dry bulk density range of fly ash as 75 to 95 lb/ft^3, equivalent to 1.2 to 1.5 g/cm^3. The difference between the two reported densities may reflect differences in measuring techniques as well as differences in compaction of the ash material in the two studies.

One of the major environmental concerns about ash wastes is mobilization of ash constituents from waste-handling sites by wind and water. The small size of fly ash particles makes them prone to being entrained in high winds if the wastes are stored in open areas. The pozzolanic (cement-like) property of the fly ash reduces this problem if the ash remains undisturbed because a hardened crust forms on the waste surface after exposure to water.

Water can permeate ash wastes and transport waste constituents into the surrounding soil. Leachate may be defined as the liquid that has percolated through or drained from waste (e.g., FGD sludge) or other materials and contains soluble, partially soluble, or miscible components removed from such materials. Permeability is a measure of how easily water will pass or flow through a material and is thus an important factor in determining the volume of leachate that can drain through the material. A reduction in the coefficient of hydraulic conductivity, which is a component of permeability, results in a corresponding decrease in the flow of leachate (Coltharp et al. 1979b; Duvel et al. 1979). For fly ash, this parameter has been found to range from 5×10^{-7} to 8×10^{-5} cm/s (4×10^{-4} to 7×10^{-2} m/day), corresponding to drainage characteristics ranging from practically impervious to slow (Frascino and Vail 1976).

The composition of fly ash leachate may be inferred from the chemical characteristics of ash pond liquors, although such data are not as readily available as data about coal ash itself. The largest contributors to the dissolved solids in pond liquors are the Ca, Mg, K, and Na sulfates and anhydrous oxides in the ash. Other soluble components in the ash include compounds of Fe, Ni, and Zn (often as sulfates), as well as trace quantities of B, Cr, Cu, Pb, As, and Cd compounds (Table 8). Fly ash typically contains higher concentrations of soluble material than does bottom ash; therefore, fly ash liquors contain higher concentrations of trace elements than bottom ash liquors (Hart and DeLaney 1978).

The smaller fly ash particles have relatively large surface-to-volume ratios, affording a comparatively large adsorption surface; thus, they may provide the principal source for the trace elements in leachates from fly ash and from scrubber sludge containing fly ash (Duvel et

Table 6. Estimated Concentrations of Saturated n-Hydrocarbons in Ash[a]

Component	Concentration (ppb)	Component	Concentration (ppb)
C_{15}	trace	C_{25}	319
C_{16}	192	C_{26}	366
C_{17}	608	C_{27}	516
C_{18}	740	C_{28}	664
C_{19}	383	C_{29}	816
C_{20}	308	C_{30}	660
C_{21}	528	C_{31}	596
C_{22}	548	C_{32}	344
C_{23}	480	C_{33}	199
C_{24}	308	C_{34}	66
Total concentration of all components = 8.6 ppm			

[a] From Hart and DeLaney (1978). Original data from Van Hook (1976).

Table 7. Estimated Concentrations of Polycyclic Aromatic Hydrocarbons in Ash[a]

Polycyclic Aromatic Hydrocarbon	Concentration (ng/g)
Naphthalene	8.3
2-Methylnaphthalene	5.0
1-Methylnaphthalene	5.2
Biphenyl	10.3
1,6- and/or 1,3-Dimethylnaphthalene	trace
2,6-Dimethylnaphthalene	trace
1,5- and/or 2,3-Dimethylnaphthalene	trace
9,10-Dihydroanthracene	12.6
Phenanthrene	17.6
2-Methylanthracene	9.1
1-Methylphenanthrene	<24.8[b]
Fluoranthene	<13.4[b]
Pyrene	<19.0[b]
1,2-Benzofluorene	36.8
2,3-Benzofluorene	11.8
1-Methylpyrene	trace
Picene	trace
Total	<200[b]

[a] Data from Hart and DeLaney (1978).
[b] Interference allows estimate only of maximum possible concentration.

Table 8. Dissolved Constituents of Ash Pond Liquors and Sluicing Waters[a]

Constituent	Concentration (mg/L) in ponds			Concentration (mg/L) in sluicing waters	
	Fly ash	Bottom ash	Ash	Fly ash	Bottom ash
Aluminum	3.6 -8.8	0.5 -8.0	0.4	0.57 -9.2	1.69 -2.31
Antimony	-	-	0.007 -0.012	0.004 -0.021	0.034 -0.41
Arsenic	<0.005 -0.023	0.002 -0.015	0.006 -<0.12	<0.0001-0.001	0.004 -0.008
Barium	0.2 -0.4	<0.10 -0.30	<0.1 -<3.0	<0.5 -<0.6	<0.5 -0.5
Beryllium	<0.01 -0.02	<0.01	<0.002	<0.002 -0.003	0.001 -0.002
Boron	-	-	1.0 -24.67	0.5 -2.41	0.25 -2.46
Cadmium	0.023 -0.052	<0.001 -0.002	0.005 -<0.025	0.001 -<0.002	0.001 -0.004
Calcium	94 -180	23 -67	<200 -563	46 -114	45 -785
Chlorine	5 -14	5 -15	85 -189	14.9 -17.2	15.7 -27.7
Chromium	0.012 -0.17	<0.005 -0.023	<0.02	<0.05	<0.05
Cobalt	-	-	<0.001 -<0.7	<0.003	0.004 -0.005
Copper	0.16 -0.45	<0.01 -0.14	<0.003 -0.05	0.008 -0.022	0.011 -0.024
Fluorine	-	-	1 -14.85	0.023 -0.70	0.25 -16.2
Iron	0.33 -6.6	1.7 -11	<0.05 -<0.2	0.01 -1.38	0.31 -2.25
Lead	<0.01 -0.2	<0.01 -0.031	0.0028-0.08	0.006 -0.025	0.007 -0.024
Lithium	-	-	0.4 -0.08	-	-
Magnesium	9.4 -20	0.3 -9.3	0.1 -102	15.7 -24.1	25.8 -67.7
Manganese	0.29 -0.63	0.07 -0.26	<0.005 -0.49	0.016 -0.096	0.055 -0.77
Mercury	<0.0002-0.0006	<0.0002-0.026	<0.001	<0.0004-0.0005	<0.0004-0.0005
Molybdenum	-	-	0.030 -0.49	<0.012 -0.015	0.016 -0.055
Nickel	0.06 -0.13	0.5 -0.12	0.003 -<0.2	0.007 -<0.02	0.001 -0.015
Phosphorus	<0.01 -0.06	<0.01 -0.23	-	-	-
Potassium	-	-	6.6 -7	-	-
Selenium	<0.001 -0.004	<0.001 -0.004	0.021 -<0.05	0.001 - 0.004	0.001 -0.031

(continued)

Table 8. (Concluded)

Constituent	Concentration (mg/L) in ponds			Concentration (mg/L) in sluicing waters	
	Fly ash	Bottom ash	Ash	Fly ash	Bottom ash
Silicon	10 -15	6.1 -8.6	1 -51	-	-
Silver	<0.01 -0.01	<0.01	<0.02	<0.0002	<0.0002-0.0004
Sodium	-	-	170 -294	-	-
Strontium	-	-	0.8	-	-
Tantalum	-	-	<0.02	-	-
Titanium	-	-	<0.02	<0.1 -0.1	-
Uranium	-	-	-	0.004 -0.005	0.003 -0.005
Vanadium	-	-	<0.02	<0.005 -0.072	<0.005 -0.192
Zinc	1.1 -2.7	0.02 -0.16	<0.022 -0.09	0.008 -0.028	0.013 -0.075
Zirconium	-	-	<0.07	-	
Total hardness ($CaCO_3$)	185 -520	-	-	-	-
Total alkalinity (as $CaCO_3$)	-	30 -160	-	-	-
pH	3.6 -6.3	4.1 -7.9	8.47	-	-
Dissolved solids	141 -820	69 -404	1100 -3335	-	-
Suspended solids	2 -256	5 -657	-	-	-
Bicarbonate (HCO_3^-)	-	-	31.0	-	-
Sulfate (SO_4^{2-})	-	-	580 -2300	-	-

[a] Data from Hart and DeLaney (1978).

al. 1979). The surface-to-volume ratio becomes important because some trace elements are more highly concentrated near the surfaces rather than in the interiors of the fly ash particles, probably as a result of elements that are vaporized during combustion condensing on the particles. Because the surfaces of the particles are in direct contact with the external environment, the surface predominance of the trace elements promotes their leachability even though the bulk structure of the particle is effectively insoluble. The total amount of a given trace element in fly ash may be relatively small, but the localized surface concentration of such an element may render its contribution to the dissolved solids in the leachate disproportionately high (Natusch 1978).

Aggregate Fraction

Slag (also termed "boiler slag" or "black beauty") is that portion of the total ash that melts to a viscous fluid at burner operating temperatures. It is usually recovered from the bottom of the boiler by tapping the molten slag from the boiler into a tank of water. This produces a glassy, angular material that ranges in particle size from about 0.6 to 5 mm in diameter. Coarse "clinker", produced by melting within the furnace, is usually crushed to a 5-cm-diameter maximum size prior to disposal. This coarse fraction comprises less than 10 to 20% of the boiler slag. Gases may be trapped in the slag as it is withdrawn from the furnace and cools, so the resulting slag will be somewhat porous or vesicular. Slags from the western lignite coals are reported to be more vesicular than slags from the eastern bituminous coals; the bulk density of slags in one series of measurements ranged from 2.65 to 2.76 g/cm^3 (Usmen and Anderson 1979). The slag may contain water-soluble particles, such as iron sulfates, which were dropped as part of the slag after having concentrated in parts of the furnace; but, in general, slag is insoluble.

Bottom ash (also called "cinders") is dry ash that does not melt but is too heavy to be entrained in the flue gas. It falls from the bottom of the boiler through a grate into an ash hopper filled with water. Some of the ash may melt and resolidify on the surfaces inside the furnace and thus have the appearance of boiler slag. Bottom ash has a coarse texture (similar to sand), with the size of its particles ranging from 0.08 to 20 mm in diameter. Porous, easily crushed particles are present in some bottom ashes. Dry bottom ashes tend to absorb water more readily than boiler slags, and they also tend to have somewhat lower bulk densities, e.g., 2.31 to 2.68 g/cm^3. Small quantities of sulfates and other soluble salts occur as deposits on the surfaces of the ash particles in many bottom ashes (Usmen and Anderson 1976). For bottom ash, which is considered a free drainage material, values on the order of 10^{-2} cm/s (10 m/day) have been reported (Frascino and Vail 1976).

Prior to the passage of the Resource Conservation and Recovery Act, air pollution problems received more attention than solid-waste-disposal problems; consequently, fly ash has been much more extensively characterized, both chemically and physically, than bottom ash or slag. The recent increase in concern about solid-waste disposal should encourage further investigation of power plant aggregate.

FLUE-GAS-DESULFURIZATION WASTES AND THEIR PROPERTIES

General Considerations

A number of flue-gas-desulfurization (FGD) systems for coal-fired power plants have been or are being developed in order to control SO_2 emissions. The two processes in widest use (as of 1977) represent over 80% of the systems in operation or under construction (Figure 1). The lime/limestone and double-alkali scrubbing processes are classified as "throwaway" systems because they produce a waste sludge by-product, FGD sludge. Similarly, the sodium carbonate process may be classified as "throwaway" because of the large quantities of sodium sulfite effluent. The magnesia scrubbing and Wellman-Lord processes are considered "regenerable" systems because most of the sorbent is regenerated and recycled; liquefied SO_2, sulfuric acid, or sulfur is produced as a by-product (Ottmers et al. 1975; Federal Power Commission 1977). A brief description of these processes is given in this subsection as an introduction to the discussion of FGD sludge properties in the next subsection.

The chemical reactions presented below are given as overall reactions for the purpose of brevity. Further details about the intermediate steps may be found elsewhere, e.g., Federal Power Commission (1977). Also, there is some variation in the possible reagents and reactions--for example, the description of the magnesia process given by Ottmers et al. (1975) differs from that given by the Federal Power Commission (1977). The description given by Ottmers is used in the following discussion.

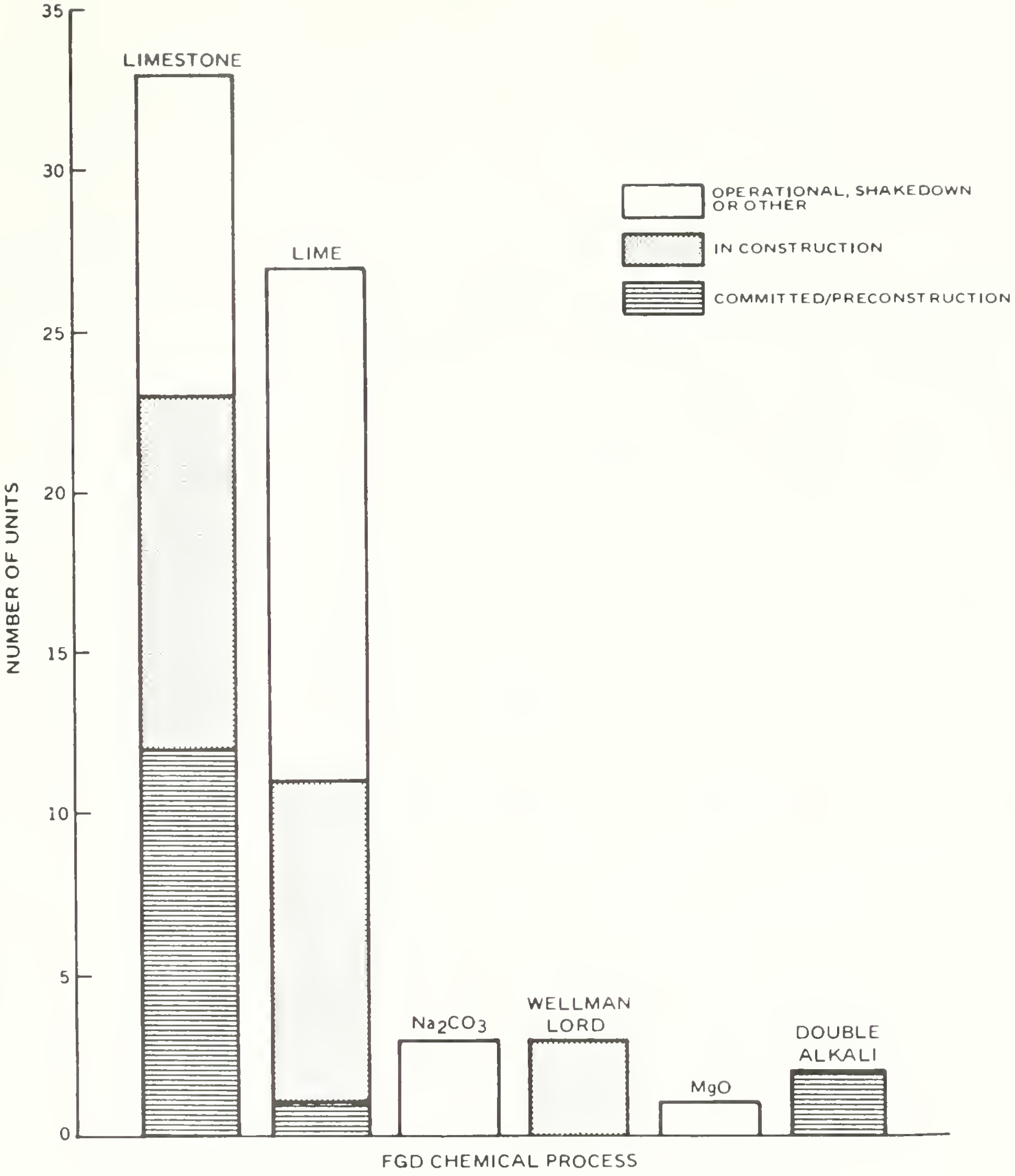

Figure 1. Distribution of Units by Process Type.
From Federal Power Commission (1977).

In the lime/limestone scrubbing processes, SO_2 is extracted from the flue gas by a slurry containing lime or limestone. Lime consists primarily of CaO, which in water solution becomes $Ca(OH)_2$; limestone consists primarily of $CaCO_3$. Lime and limestone react with SO_2 as follows:

$$SO_2 + Ca(OH)_2 \rightarrow H_2O + CaSO_3 \qquad (1)$$

$$SO_2 + CaCO_3 \rightarrow CO_2 + CaSO_3 \qquad (2)$$

Some of the resulting sulfite will be oxidized to sulfate:

$$2CaSO_3 + O_2 \rightarrow 2CaSO_4 \qquad (3)$$

The waste sludge consists principally of hydrated $CaSO_3$, $CaSO_4$, and any fly ash entrained in the scrubber stream (Ottmers et al. 1975; Dvorak et al. 1978).

A variant of the lime/limestone scrubbing processes is the alkaline ash scrubbing process. The subbituminous and lignitic coals plentiful in the western and south-central parts of the United States are frequently characterized by low sulfur levels (averaging 0.7%) and high alkaline ash contents. Because the mole ratio of lime in the ash to sulfur in the coal is often greater than unity, the ash itself may be used as a scrubbing reagent. Lime or limestone may be used as a supplemental source of alkali (Ness and Talty 1980).

Dry scrubbing techniques are now becoming available. In this type of scrubbing, an aqueous suspension of an alkaline reagent such as lime (CaO) or soda ash (Na_2CO_3) is sprayed into the flue-gas stream. The sprayed droplets dry while removing the sulfur dioxide so that the resulting product is a dry powder rather than a wet sludge. Although the dry waste product presents fewer handling problems than the wet scrubber sludge product, it has not yet been characterized to the extent that wet scrubber sludge has been. Dry scrubbing would be of particular interest in regions where the water supplies are limited (Crowe et al. 1979; Janssen and Eriksen 1979; Meyler 1979; Ness and Talty 1980).

In the double-alkali scrubbing process, two solutions are used: (1) a sodium sulfite (Na_2SO_3) liquor which absorbs the SO_2 and (2) a lime or limestone solution which regenerates the sulfite. The absorption reaction is:

$$SO_2 + Na_2SO_3 + H_2O \rightarrow 2NaHSO_3 \qquad (4)$$

The regeneration reaction is either

$$2NaHSO_3 + Ca(OH)_2 \rightarrow Na_2SO_3 + CaSO_3 + 2H_2O \qquad (5)$$

or

$$2NaHSO_3 + CaCO_3 \rightarrow Na_2SO_3 + CaSO_3 + H_2O + CO_2 \qquad (6)$$

for a lime or limestone regenerant, respectively (see also Figure 2 for a schematic of the reaction cycle). Some of the sulfite is oxidized to sulfate (Equation 3). The product is in the form of a waste sludge (Ottmers et al. 1975; Dvorak et al. 1978).

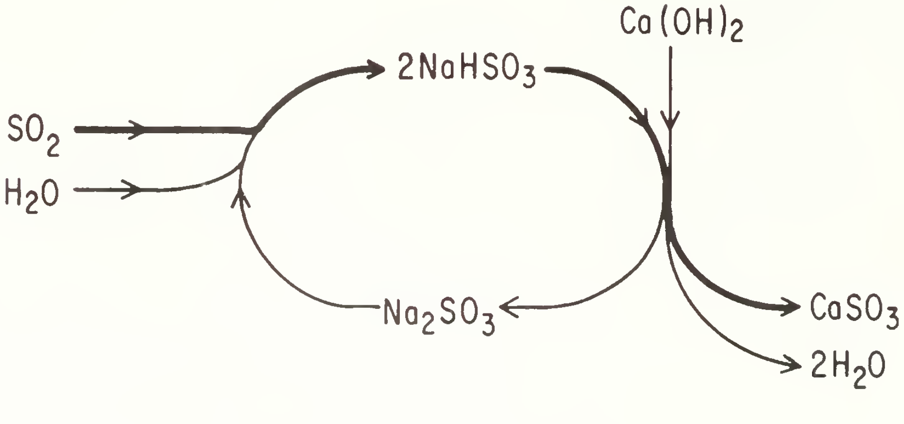

Figure 2. Schematic of the Double-Alkali Scrubbing Process. The boldface line denotes the path of SO_2 from coal combustion. Based on Ottmers et al. (1975).

The sodium carbonate process is based on the absorption of SO_2 by an aqueous solution of sodium carbonate (Na_2CO_3) to produce sodium sulfite:

$$SO_2 + Na_2CO_3 \rightarrow Na_2SO_3 + CO_2 \qquad (7)$$

A small amount of sodium bisulfite is also formed (Equation 4). As with the lime/limestone or double-alkali systems, the effluent is produced in large quantities, and large disposal ponds are needed (Federal Power Commission 1977).

In the Wellman-Lord and magnesia processes, the absorbed SO_2 is separated from the absorbent with which it had chemically reacted so that the absorbent may be reused--the separated SO_2 being converted to sulfuric acid, elemental sulfur, or liquefied SO_2, all of commercial value. For example, in the Wellman-Lord process, the SO_2 in the flue gas is absorbed by reaction with Na_2SO_3 (Equation 4). The spent absorbent is then regenerated by heating to drive off the SO_2 as follows:

$$2NaHSO_3 \rightarrow Na_2SO_3 + H_2O + SO_2 \qquad (8)$$

The sulfite and water are returned to the scrubber for reuse; the SO_2 is converted to one of the above-mentioned commercial products (Ottmers et al. 1975; Dvorak et al. 1978).

Because of the oxidation of some of the sulfite to sulfate in the Wellman-Lord process, some amount of makeup sulfite is needed during operation, and a corresponding amount of waste sulfate is produced. Duvel et al. (1979) stated that regenerable scrubbing processes such as the Wellman-Lord process do not generate significant quantities of a waste product, which, in any case, would consist mostly of a sodium sulfate solution. The Federal Power Commission (1977), however, considers the quantity of solid sodium sulfate and other sodium salts requiring disposal to be "significant" and to pose a potential water pollution problem.

In the magnesia scrubbing process, a magnesium sulfite ($MgSO_3$) slurry is used to absorb the SO_2 as follows:

$$SO_2 + H_2O + MgSO_3 \rightarrow Mg(HSO_3)_2 \qquad (9)$$

The regeneration is accomplished in two steps. First, magnesium oxide (MgO)--also called magnesia--is used to regenerate $MgSO_3$:

$$MgO + Mg(HSO_3)_2 \rightarrow 2MgSO_3 + H_2O \qquad (10)$$

About half of the $MgSO_3$ is recycled to the scrubber for further SO_2 absorption; the remainder is heated, thus releasing the SO_2 for further processing and regenerating the MgO for further use (Figure 3) (Ottmers et al. 1975; Dvorak et al. 1978).

$$MgSO_3 \rightarrow MgO + SO_2 \qquad (11)$$

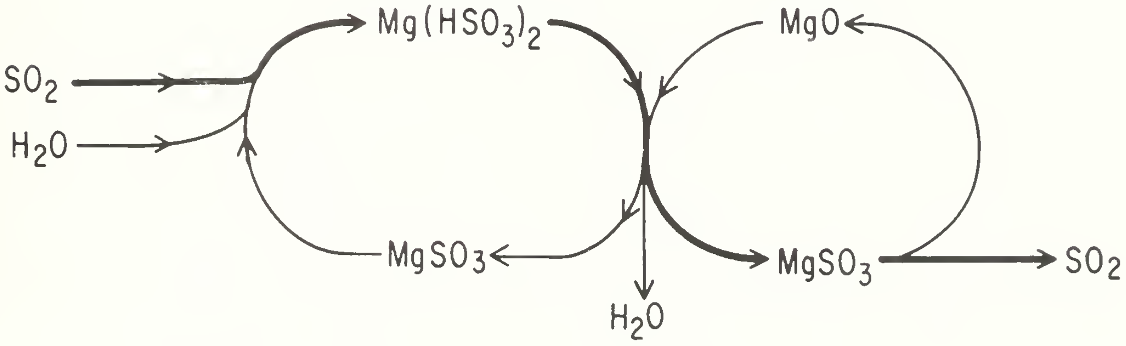

Figure 3. Schematic of the Magnesia Scrubbing Process. The boldface line denotes the path of SO_2 from coal combustion. Based on Ottmers et al. (1975).

Because of the predominance of "throwaway" wet scrubbers using lime or limestone as the reagent (Figure 1), only details relevant to FGD sludge waste product from these types of scrubber systems will be considered in any detail in the following discussion.

Chemical and Physical Properties of FGD Scrubber Sludge

The waste-product bleed from a "throwaway" scrubber system is a thin water slurry containing from 5 to 15% solids and a variety of soluble materials. After the slurry is partially dewatered, the final solids content may range from 30 to 80%. Because the dewatering is not complete, the resulting product will have the consistency of toothpaste or heavy mud and, in many cases, the thixotropic behavior of quicksand. Thixotropic substances become less viscous when disturbed and return to the original state upon standing undisturbed. In the sludge solids, gypsum ($CaSO_4 \cdot 2H_2O$) and calcium sulfite hemihydrate ($CaSO_3 \cdot \frac{1}{2}H_2O$) are the principal sulfur products (Table 9). The other major components of the sludge solids include fly ash, resulting from either separate or simultaneous fly ash collection, and calcium carbonate ($CaCO_3$), resulting from unreacted limestone absorbent and/or carbonate formation by absorption of carbon dioxide (CO_2) from the atmosphere (Elliot 1974; Leo and Rossoff 1978; Duvel et al. 1979). The fly ash is the major contributor of trace elements to FGD sludge.

Table 9. Flue-Gas-Desulfurization Waste Solids from Samples of Eastern and Western Scrubber Liquors[a]

Constituent	Samples[b]	Composition (weight percent)
$CaSO_4 \cdot 2H_2O$	13	6.3-84.6
$CaSO_3 \cdot \frac{1}{2}H_2O$	13	0.2-69.2
$CaSO_4 \cdot \frac{1}{2}H_2O$	1	19.2
$CaCO_3$	13	0.2-38.7
$MgSO_4 \cdot 6H_2O$	4	1.9-4.6
$Na_2SO_4 \cdot 7H_2O$	2	6.9,7.8
NaCl	1	1.5
$CaSO_4$[c]	1	17.7
Fly ash	13	<1.0-59.7
Carbon	1	3.5
Other[d]	2	8.2,10.7

[a] Data from Leo and Rossoff (1978). Analyses were conducted on 13 samples from eight power plants burning eastern or western coal and using lime, limestone, or double-alkali scrubbing processes.

[b] Number of samples in which constituent was detected.

[c] Phase not explicitly measured; presence deduced from X-ray study.

[d] Soluble salt. Phase not determined; quantity by difference.

There are likely to be impurities in the scrubber reagent which are assumed not to react with the SO_2 and may thus be present in the final sludge product. These components usually include various carbonates (other than $CaCO_3$), silicates, aluminates, sulfates, and ferrites and are collectively termed "grit". The grit contributes a small amount to the final volume of sludge that must be dealt with (Duvel et al. 1979).

The ratio of sulfite to sulfate in the sludge is influenced by several factors and may vary considerably; perhaps there will be exceptions to any generalizations made. The sulfate tends to predominate when limestone is used as the scrubbing reagent. Excess oxygen in the system will also favor the formation of sulfate. The sulfur content of the coal will also influence the sulfite-sulfate ratio; use of lower-sulfur coals will favor the formation of the sulfate product. Sludges in which calcium sulfite predominates usually retain water and, unless stabilized in some manner, they are unsuitable for storage in a landfill because the material will behave more like a fluid than a solid (Duvel et al. 1979).

Fly ash particles are considered to be the principal source of trace elements in scrubber sludge for all but the most volatile elemental species (e.g., mercury and selenium) that may be scrubbed directly from the flue gases. The properties of fly ash have been discussed briefly in an earlier section. A list of the trace elements reported in several analyses of scrubber sludge is given in Table 10.

Table 10. Concentrations of Chemical Constituents in Flue-Gas-Desulfurization Sludges from Samples of Eastern and Western Scrubber Liquors[a]

Constituent	Sludge concentration range Liquor[b] (mg/L)	 Solid[c] (mg/kg)
Aluminum	0.03-2.0	-
Arsenic	<0.004-1.8	0.6-52
Beryllium	<0.002-0.18	0.05-6
Cadmium	0.004-0.11	0.08-4
Calcium	180-2,600	105,000-268,000
Chromium	0.015-0.5	10-250
Copper	<0.002-0.56	8-76
Lead	0.01-0.52	0.23-21
Magnesium	4.0-2,750	-
Mercury	0.0004-0.07	0.001-5
Potassium	5.9-100	-
Selenium	<0.0006-2.7	2-17
Sodium	10.0-29,000	n.d.-48,000[d]
Zinc	0.01-0.59	45-430
Chloride	420-33,000	n.d.-9,000[d]
Fluoride	0.6-58	-
Sulfate	600-35,000	35,000-473,000
Sulfite	0.9-3,500	1,600-302,000
Chemical oxygen demand	<1-390	-
Total dissolved solids	2,800-92,500	-

[a] Data from Leo and Rossoff (1978).

[b] Liquor analyses were conducted on 13 samples from seven power plants burning eastern or western coal and using lime, limestone, or double-alkali scrubbing processes; pH of liquors ranged from 4.3 to 12.7.

[c] Solids analyses were conducted on six samples from six power plants burning eastern or western coal and using lime, limestone, or double-alkali scrubbing processes.

[d] No lower value given by Leo and Rossoff (1978); we have assumed that the lower limit was below the sensitivity of the sampling technique and was not detectable.

The water retention characteristics of scrubber sludges will affect the storage volume needed, the waste-handling methods, and the condition of the wastes when they are disposed of. The retention of water by a sludge depends on the distribution of particle sizes in the sludge and on the composition and crystalline structure of these particles. The sludges most readily dewatered are those with coarse particle sizes, generally those produced by limestone scrubber systems. Sludges from double-alkali scrubbers have the finest particle-size distributions and the poorest dewatering characteristics (Leo and Rossoff 1978). As noted above, calcium sulfite crystals have a high affinity for water, and a predominance of the sulfite over the sulfate will render the sludge more difficult to dewater. The presence of entrained water will also influence the bulk density of the sludge (Table 11).

Table 11. Water Retention and Bulk Density Characteristics of Flue-Gas-Desulfurization Wastes from Lime, Limestone, and Double-Alkali Scrubber Systems[a]

	Lime		Limestone		Double-alkali	
Dewatering method	Solids (%)	Density (g/cm^3)	Solids (%)	Density (g/cm^3)	Solids (%)	Density (g/cm^3)
Settled	40-48	1.34-1.40	47-67	1.39-1.65	37-40	1.30-1.35
Settled and drained	43-53	1.36-1.50	56-67	1.44-1.67	41-44	1.33-1.44
Centrifuged	50-57	1.39-1.52	60-77	1.56-1.86	50-62	1.38-1.62
Vacuum-filtered	56-57	1.48-1.54	53-80	1.48-1.78	55-58	1.50-1.61

[a] Data from Leo and Rossoff (1978).

The density of sludge solids is reported to range from 2.34 to 2.68 g/cm^3 by Coltharp et al. (1979a), but for purposes of estimating the volume of sludge solids, Duvel et al. (1979) suggest that the bulk density of these solids be assumed to have a value of 2.4 g/cm^3 unless better information is available.

Characteristics of Scrubber Sludge Leachate

As with ashes, sludge constituents can be leached from the waste into surrounding soils. The results of laboratory leaching tests with FGD sludge indicate that the initial concentrations of dissolved solids in the leachate are similar to those in the scrubber liquor retained in the sludge at the time of disposal. However, as the interstitial scrubber liquor of ponded sludge is displaced by infiltrating water from precipitation or pond supernatant, it becomes more dilute with time and more of the dissolved solids in the leachate will originate from dissolution of the soluble sludge components (Duvel et al. 1979). Thus, concentrations of the major species in the leachate decrease rapidly as the first few pore volumes of displacement (PVD)--i.e., a volume of leachate equal to three times the volume of the interstitial liquor--are percolated through the sludge, gradually displacing the scrubber liquor. A PVD is the volume of liquid equal to the volume of the interstitial liquor, so that each PVD of leachate percolated through the sludge represents the equivalent of one complete flushing of the interstitial liquor in the sludge. Concentrations of the major dissolved species level off after about the fifth PVD, although some trace elements may exhibit a more protracted decline (Figure 4).

The permeability of the sludge, i.e., the capacity of the sludge for transmitting water, is an important factor in determining the volume of leachate that can drain from the sludge. Both the presence of fly ash in the sludge and the compaction or consolidation of the sludge have been found to reduce the permeability. Coefficients of hydraulic conductivity for untreated sludge typically range from about 0.05 to 0.2 m/day (Leo and Rossoff 1978; Duvel et al. 1979). This is within the range of the hydraulic conductivities found in loamy to fine sandy soils, and represents a moderately low permeability (Bouwer 1979).

QUANTITIES OF WASTES PRODUCED AND WATER REQUIREMENTS

It is possible to estimate the quantities of solid waste produced by a coal-fired power plant and the water requirements for handling these wastes. The method for such estimation presented here is basically that presented by Duvel et al. (1979) in the "FGD Sludge Disposal Manual" prepared for the Electric Power Research Institute (EPRI). Often the utility will provide data on the amount of waste that will be produced by emission abatement and ash collection. We present here a method for calculating the approximate amount of waste produced for those cases in which the data base may be imcomplete.

Information Needed for Calculating Quantities of Wastes

The mass and volume of coal ash and FGD scrubber sludge are functions of: (1) rate of coal consumption, (2) ash content of the coal, (3) sulfur content of the coal, (4) overall particulate-removal efficiency, (5) upstream particulate-removal efficiency (i.e., upstream of the FGD scrubber), (6) SO_2-removal efficiency, (7) efficiency of dewatering, (8) sulfate/sulfite ratio, (9) specific gravity of sludge solids, (10) excess quantity of scrubbing reagent, and (11) grit content of reagent.

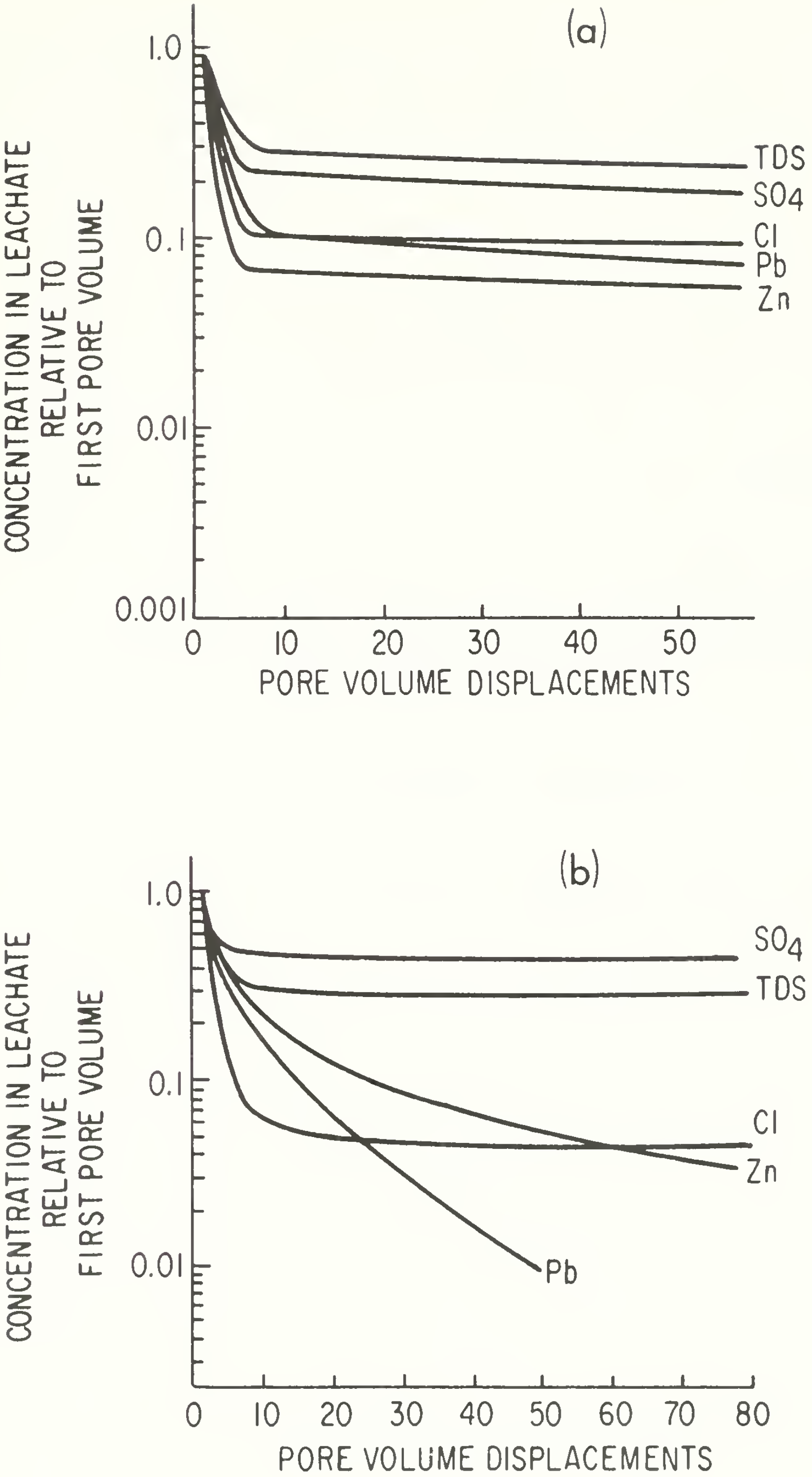

Figure 4. Analysis of Leachate Under Aerobic Conditions from (a) TVA Shawnee Limestone Sludge and (b) Duquesne Phillips Sludge. Modified from Leo and Rossoff (1978).

The quantity of coal ash produced is calculated separately from the quantity of scrubber sludge. For simplicity, it is assumed that most of the coal ash is removed from the flue gas upstream of the scrubber so that it does not contribute significantly to the weight of the FGD-sludge solids. Because the reagent grit and excess scrubbing reagent also comprise only a small percentage of the FGD sludge solids by weight, reasonably good estimates may usually be made without knowledge of the values of (4), (5), (10), and (11) above. Because the data bases are generally presented in English units, we have presented the calculation also using English units. Factors for converting to SI units are given in Appendix A.

The rate of coal consumption, C, may be calculated from:

$$C\ (\text{t/yr}) = \frac{\left\{\begin{matrix}\text{Plant}\\ \text{capacity (kW)}\end{matrix}\right\} \times \left\{\begin{matrix}\text{Heat}\\ \text{rate}\end{matrix}\ \left(\frac{\text{Btu}}{\text{kWh}}\right)\right\} \times \left\{\begin{matrix}\text{Plant}\\ \text{factor}\end{matrix}\right\} \times 8760\ \text{h/yr}}{(2000\ \text{lb/t}) \times \left\{\begin{matrix}\text{Heating value}\\ \text{of coal (Btu/lb)}\end{matrix}\right\}} \tag{12}$$

The plant capacity or rated capacity is the nominal capacity of the power plant unit for the production of electricity and may be expressed in kilowatts (kW), as above. The heat rate is a measure of the efficiency of conversion of boiler heat to electricity, i.e., so many Btu of boiler heat are needed to produce so many kilowatt-hours (kWh) of electrical energy. Because 3413 Btu equals 1 kWh, the heat rate for a plant that converts boiler heat to electrical energy with 100% efficiency would be 3413 Btu/kWh. Typical heat rates for coal-fired plants have ranged from about 8860 to 9000 Btu/kWh (Babcock and Wilcox 1978). Several older units in the Northeast, which are candidates for conversion from oil to coal firing, have heat rates on the order of 11,000 to 13,000 Btu/kWh (Energy Resour. Co. 1977). The plant factor (on an annual basis) is the ratio of the electrical energy actually generated during the course of a year to the electrical energy that could have been generated if the plant had been operated at its nominal rated capacity for the entire year (8760 h):

$$\text{Plant factor} = \frac{\text{Actual electrical energy generated (kWh)}}{\text{Plant capacity (kW) v 8760 h}} \tag{13}$$

Heating value is the quantity of heat released when the coal is burned. The heating value of the coal must be obtained from a fuel analysis; based on the characteristics of the coal intended for use in the plant, typical values should be available from the plant operators.

Similarly, the ash content and the sulfur content of the coal must be obtained from a fuel analysis; the values of these parameters should also be available from the operators of the power plant.

The upstream particulate-removal efficiency depends upon the type of particulate-removal system placed in the flue-gas stream upstream of the scrubber. The overall particulate-removal efficiency depends, in addition, upon the ash content of the coal and the details of the scrubbing system. The SO_2-removal efficiency, or scrubbing efficiency, depends on the kind of scrubber used, but it may be assumed that the scrubber will reduce SO_2 emissions to an extent sufficient to meet regulations. The reagent type and purity depends on the kind of scrubber used and on the source of supply of the reagent. The efficiency of sludge dewatering also depends on the equipment used. All of these data should be available from the plant operators based on a knowledge of similar systems in operation or on information from the manufacturers of the equipment and the vendors of the reagent.

The sulfate/sulfite ratio in the scrubber sludge is difficult to predict. In general, a ratio of 50/50 may be assumed, or 80/20 for limestone scrubbers and 20/80 for lime scrubbers. With the use of a forced oxidation process, it is possible to obtain almost 100% sulfate.

The specific gravity of the sludge solids is not considered an important variable by Duvel et al. (1979), who suggest an assumed value of 2.4 unless more reliable values are available. Coltharp et al. (1979a) report values of the specific gravity of FGD sludge solids ranging from 2.34 to 2.68.

Methodology for Calculating Quantities of Wastes

Using the input described above, the quantities of solid waste may be estimated by means of the following methodology. Sample calculations are presented in Tables 12 and 13.

Ash production is calculated as:

$$\text{Total ash produced (t/yr)} = C \times \left\{\frac{\%\ \text{ash}}{100}\right\} \tag{14}$$

The percent ash is the ash content of the coal, and C is the annual rate of coal consumption in tons of coal per year.

Total fly ash production is calculated as:

$$\text{Total fly ash produced (t/yr)} = \text{Total ash} \times \left\{\frac{\%\ \text{fly ash}}{100}\right\} \tag{15}$$

The percent fly ash is the so-called "split factor", that percentage of the ash which becomes entrained in the flue gases as fly ash. This is a system-dependent parameter, being dependent to a great degree on the type of boiler. Information on the expected split factor should be available from the plant operator.

The fly ash collected in the precipitator is calculated as:

$$\text{Precipitator fly ash (t/yr)} = \text{Total fly ash} \times \left\{\frac{\%\ \text{precipitator efficiency}}{100}\right\} \tag{16}$$

The precipitator efficiency is identical to the upstream particulate-removal efficiency. Other particulate-removal devices may be used instead of a precipitator upstream of the scrubber, and removal efficiency will vary according to manufacturer's specifications.

The total ash collected upstream of the FGD scrubber (including the aggregate, i.e., the bottom ash and slag) may now be calculated:

$$\begin{aligned}\begin{matrix}\text{Weight of}\\ \text{total ash collected (t/yr)}\end{matrix} &= \text{Precipitator fly ash + aggregate}\\ &= \text{Precipitator fly ash + total ash - total fly ash} \qquad (17)\end{aligned}$$

$$\begin{matrix}\text{Volume of}\\ \text{total ash collected (ft}^3\text{/yr)}\end{matrix} = \left\{\left(\frac{\text{Precipitator fly ash}}{\text{Bulk density of precipitator fly ash}}\right) + \left(\frac{\text{Total ash - Total fly ash}}{\text{Bulk density of aggregate}}\right)\right\} \times \left\{2000\ \text{lb/t}\right\} \tag{18}$$

The volume is calculated by dividing the values of the weight of the precipitator fly ash and the weight of the aggregate by their respective bulk densities (in lb/ft^3) and then adding these values. Based on the reported range of ash densities given above (see section on Coal Ash Wastes and Their Properties), the bulk density of the precipitator fly ash may be taken as about 80 lb/ft^3 if no better data are available. Similarly, the bulk densities of bottom ash and slag may be taken as 160 and 170 lb/ft^3, respectively.

The volume required for ash disposal (i.e., conversion of the volume units from ft^3/yr to acre-ft/yr) is calculated as:

$$\begin{matrix}\text{Volume for}\\ \text{ash disposal (acre-ft/yr)}\end{matrix} = \begin{matrix}\text{Volume of}\\ \text{collected ash (ft}^3\text{/yr)}\end{matrix} \times 2.3 \times 10^{-5} \tag{19}$$

Table 12. Sample Calculations for Determining Volume of Ash Produced by Operation of a Coal-Fired Power Plant

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ASSUMPTIONS:

Plant capacity	2100 MWe
Plant factor	0.70 days operation/day
Heat rate	8980 Btu/kWh
Precipitator efficiency	99.5%
Coal heat capacity	8200 Btu/lb
Coal ash content	6%
Split factor	85 fly ash:15 aggregate

CALCULATIONS:

Coal consumption, C (from Eq. 12)

$$= \frac{2100\text{ MWe} \times 8.98 \times 10^{6}\text{ Btu/MWh} \times 0.7 \times 8760\text{ h/yr}}{2000\text{ lb/t} \times 8200\text{ Btu/lb}}$$

$$= \frac{1.156 \times 10^{14}}{1.640 \times 10^{7}}$$

$$= 7.049 \times 10^{6}\text{ t/yr}$$

Total ash produced (from Eq. 14)

$$= 7.049 \times 10^{6}\text{ t coal/yr} \times \frac{6\text{ t ash}}{100\text{ t coal}}$$

$$= 4.229 \times 10^{5}\text{ t ash/yr}$$

Fly ash product (from Eq. 15)

$$= 4.229 \times 10^{5}\text{ t ash/yr} \times \frac{85\text{ t fly ash}}{100\text{ t ash}}$$

$$= 3.595 \times 10^{5}\text{ t fly ash/yr}$$

Fly ash collected (from Eq. 16)

$$= 3.595 \times 10^{5}\text{ t fly ash/yr} \times \frac{99.5}{100}$$

$$= 3.577 \times 10^{5}\text{ t fly ash/yr}$$

Total ash collected (from Eq. 17)

$$= 3.577 \times 10^{5}\text{ t fly ash/yr}$$

$$+ (4.229 \times 10^{5}\text{ t} - 3.595 \times 10^{5})\text{ t aggregate/yr}$$

$$= 4.211 \times 10^{5}\text{ t ash/yr}$$

Volume ash collected (from Eq. 18)

$$= \frac{(3.577 \times 10^{5}\text{ t/yr}) \times 2000\text{ lb/t}}{80\text{ lb/ft}^{3}}$$

$$+ \frac{(6.344 \times 10^{4}\text{ t/yr}) \times 2000\text{ lb/t}}{165\text{ lb/ft}^{3}}$$

$$= 9.711 \times 10^{6}\text{ ft}^{3}\text{/yr}$$

$$= 223\text{ acre-ft/yr}$$

Table 13. Sample Calculations for Determining Volume of FGD Sludge Produced by Operation of a Coal-Fired Power Plant

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ASSUMPTIONS:

Plant capacity	2100 MWe
Plant factor	0.70 days operation/day
Heat rate	8980 Btu/kWh
Precipitator efficiency	99.5%
Coal heat capacity	8200 Btu/lb
Coal sulfur content	0.48%
Sulfur to unit heat	0.58 lb/10^6 Btu
Scrubbing reagent	Lime
Sulfate to sulfite ratio	1:4

CALCULATIONS:

Coal consumption, C (from Table 12) $= 7.049 \times 10^6$ t/yr

% SO_2 removal required for coal containing 0.58 lb S/10^6 Btu (from Figure 5) = 70%

Weight of sulfate solids (from Eq. 20)

$$= 7.049 \times 10^6 \text{ t/yr} \times 0.48 \text{ t S/100 t coal} \times 0.70\ SO_2 \text{ removal} \times 1/5\ SO_4 \text{ ratio} \times 172 \text{ g } SO_4/32\text{g S}$$

$$= 2.546 \times 10^4 \text{ t/yr}$$

Weight of sulfite solids (from Eq. 21)

$$= 7.049 \times 10^6 \text{ t/yr} \times 0.48 \text{ t S/100 t coal} \times 0.7\ SO_2 \text{ removal} \times 4/5\ SO_3 \text{ ratio} \times 129 \text{ g } SO_3/32 \text{ g S}$$

$$= 7.638 \times 10^4 \text{ t/yr}$$

Weight of sludge solids (from Eq. 22)

$$= (2.546 + 7.638) \times 10^4 \text{ t/yr}$$

$$= 1.018 \times 10^5 \text{ t/yr}$$

Weight of sludge water (from Eq. 23)

$$= 1.018 \times 10^5 \text{ t/yr} \times 85 \text{ t water/15 t solids}$$

$$= 5.771 \times 10^5 \text{ t/yr}$$

Sludge volume (from Eqs. 25, 26)

$$= \left(\frac{1.018 \times 10^5 \text{ t/yr}}{2.4}\right) \times \frac{2000 \text{ lb/t}}{62.4 \text{ lb/ft}^3} + \frac{5.771 \times 10^5 \text{ t/yr} \times 2000 \text{ lb/t}}{62.4 \text{ lb/ft}^3}$$

$$= 1.986 \times 10^7 \text{ ft}^3\text{/yr}$$

$$= 456 \text{ acre-ft/yr}$$

The weight of the sulfate solids in FGD sludge is calculated as:

$$CaSO_4 \cdot 2H_2O \text{ (t/yr)} = C \times \left\{\frac{\%\text{ S in coal}}{100}\right\} \times \left\{\frac{\%\text{ SO}_2\text{ removal}}{100}\right\} \times \left\{\text{Fraction of S solids as } CaSO_4 \cdot 2H_2O\right\} \times \left\{\frac{172}{32}\right\} \tag{20}$$

The scrubber efficiency or percentage of SO_2 removed from the flue gas stream required to meet USEPA SO_2 emission-abatement standards can be obtained from Figure 5. The fraction of S solids as $CaSO_4 \cdot 2H_2O$ is taken from the sulfate/sulfite ratio. The factor (172/32) is simply the ratio of the molecular weight of $CaSO_4 \cdot 2H_2O$ to the atomic weight of S; C is the annual rate of coal consumption.

The weight of the sulfite solids is calculated as:

$$CaSO_3 \cdot \tfrac{1}{2}H_2O \text{ (t/yr)} = C \times \left\{\frac{\%\text{ S in coal}}{100}\right\} \times \left\{\frac{\%\text{ SO}_2\text{ removal}}{100}\right\} \times \left\{\text{Fraction of S solids as } CaSO_3 \cdot \tfrac{1}{2}H_2O\right\} \times \left(\frac{129}{32}\right) \tag{21}$$

This is similar to the preceding equation except that the sulfite fraction and the molecular weight of the $CaSO_3 \cdot \tfrac{1}{2}H_2O$ (i.e., 129) are used.

The weights of sludge solids, sludge water, and total wet sludge are calculated as in the following equations:

$$\text{Weight of sludge solids (t/yr)} = \text{Sum of Equations 20 and 21} \tag{22}$$

$$\text{Weight of sludge water (t/yr)} = \left\{\frac{\%\text{ of total as water}}{\%\text{ of total as solids}}\right\} \times \left\{\text{Weight of sludge solids}\right\} \tag{23}$$

$$\text{Total weight of wet sludge (t/yr)} = \text{Weight of sludge solids} + \text{Weight of water} \tag{24}$$

The volume of sludge solids, sludge water, and total wet sludge are calculated as in the following equations:

$$\text{Volume of sludge solids (ft}^3\text{/yr)} = \left\{\frac{\text{Total weight of dry solids}}{\text{Specific gravity of sludge solids}}\right\} \times \left\{\frac{2000\text{ lb/t}}{\text{Density of water (62.4 lb/ft}^3\text{)}}\right\} \tag{25}$$

$$\text{Volume of sludge water (ft}^3\text{/yr)} = \text{Total weight of water} \times \left\{\frac{2000}{62.4}\right\} \tag{26}$$

$$\text{Total volume of wet sludge (ft}^3\text{/yr)} = \text{Volume of sludge solids} + \text{Volume of sludge water} \tag{27}$$

The volume required for sludge storage (i.e., conversion of units from ft^3/yr to acre-ft/yr) is calculated as:

$$\text{Volume for sludge storage (acre-ft/yr)} = \text{Total sludge volume (ft}^3\text{/yr)} \times 2.3 \times 10^{-5} \tag{28}$$

The volume of water may also be expressed in units of acre-ft/yr by multiplying the value in ft^3/yr by the conversion factor 2.3×10^{-5}.

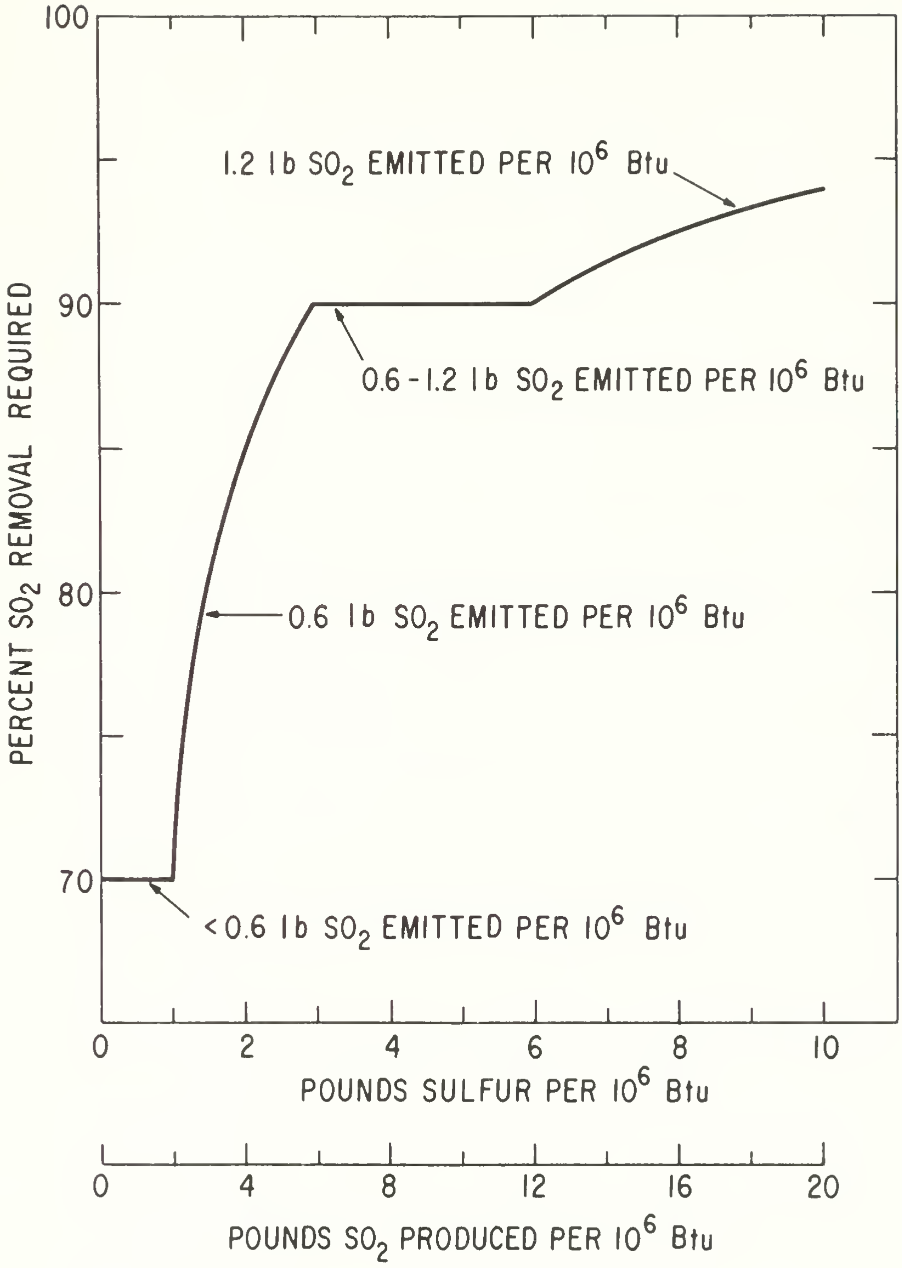

Figure 5. Percentage SO_2 Removal Required to Meet USEPA Emission Standards.

In summary, the following results may be obtained:

1. Annual dry weight of collected ash (in tons).
2. Annual volume of collected ash (in acre-ft).
3. Annual dry weight of sludge solids (in tons).
4. Annual volume of water used for FGD scrubbing (in acre-ft/yr).
5. Annual disposal volume needed for wet sludge (in acre-ft).

Water use will vary with the actual design of waste handling. Slurrying the ash to storage sites will require more water than dry handling. Therefore, waste-handling plans must be known before a complete picture of water use can be obtained.

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CURRENT PRACTICES FOR HANDLING AND STORING COAL COMBUSTION AND EMISSION-ABATEMENT WASTES

FLUE-GAS-DESULFURIZATION SLUDGE PROCESSING

Flue-gas-desulfurization (FGD) sludge is highly unstable and fluidizes when disturbed. This makes handling the sludge quite difficult, particularly if dry storage is desired. Thus, several ways to stabilize FGD sludges have been developed (Duvel et al. 1979): (1) dewatering, (2) underdraining impoundments, (3) chemical fixation, (4) blending with ash, and (5) forced oxidation.

Dewatering

Dewatering and thickening reduce the moisture content and volume of waste requiring storage (Duvel et al. 1979). The lower volume of the waste reduces the land requirements for storage, and the higher solids content reduces the potential for seepage of soluble elements into the surrounding environment. Duvel et al. (1979) indicate that over 75% of the nation's operating and proposed FGD sludge-handling systems use some dewatering prior to ultimate disposition. Dewatering can be accomplished by one or a combination of several mechanical means, including settling ponds, thickeners, vacuum filters, and centrifuges. The method of choice depends on the characteristics of the waste sludge, the method of storage, the availability of land, and the economics involved. Because of their inherent simplicity and effectiveness, thickeners and settling ponds are used nearly universally to initially concentrate scrubber solids. The primary dewatering can increase the solids content of sludge bleed from 5 to 15% to about 20 to 45% (sulfate sludges up to 60 to 65% solids). Where vacuum filtration or centrifuging (secondary dewatering) is used in conjunction with thickeners or settling ponds, an additional increase in solids content is effected. Either method can increase the solids content to 50 to 60% for sulfite sludges and up to 80 to 85% for sulfate sludges.

Settling ponds. Settling ponds are usually flat-bottomed impoundments created by constructing an earthfill dike or excavation in a rectangular configuration or by building a dam across a natural valley (Duvel et al. 1979). Scrubber bleed is discharged at one end of the pond, the solids settle out as it flows at a low velocity to the opposite end, and the supernatant is drawn off for recycle or discharge. Interim ponding has proven effective in dewatering sludges from lime/limestone scrubbing systems, up to a maximum of about 50% solids. Settling ponds are advantageous because they are less sensitive than thickeners to variations in system bleed flow and solids content, and because they ordinarily require little maintenance. Settling ponds are disadvantageous because substantial commitments of land are required and the removal of sludge is often difficult, time-consuming, and expensive. Moreover, because of the presence of a constant hydraulic head, settling ponds have higher potential for seepage than other dewatering alternatives. The ponds can be lined to reduce seepage of soluble substances into the surrounding environment. As of 1978, fewer than 15% of the operating facilities used settling ponds for dewatering.

Thickeners. A conventional gravity thickener operates very much like a settling tank: sludge enters in the center of the tank and is distributed radially. The solids are collected as underflow in a sludge sump, and a clear supernatant exits the top over weirs. The bottom of the tank is usually sloped to the center, and the settled sludge is gently pushed by two flow blades, aided by the sloped bottom, toward the central outlet and discharged. In normal operations, the input slurry (6 to 15% solids) is typically concentrated to 20 to 45% solids; if sulfate is predominant, thickeners may produce sludges with up to 65% solids. Thickeners require a commitment of less acreage than is needed for settling ponds. Containment of the sludges is more secure in a thickener; therefore, the potential for seepage of soluble materials into surrounding habitat is much less than for impoundments. Thickeners, then, do have a lower potential than settling ponds for affecting fish and wildlife resources. Most operators (> 60%) use a thickener system for primary dewatering of FGD sludges.

Vacuum filters. Two types of vacuum filters are most applicable to dewatering sludges from lime/limestone scrubber systems: drum and belt (Duvel et al. 1979). In rotary-drum vacuum filters, the drum is divided into sections, each of which is connected through ports to a discharge head and vacuum source. The slurry is fed to a tank and maintained in suspension by an agitator. As the drum rotates, the faces of its sections pass successively through the slurry, and the vacuum draws filtrate through the filter medium, impinging suspended solids (filter cake) on the drum filter. While the caked filter rotates with the drum, it undergoes dewatering by the simultaneous flow of air and filtrate. The cake is removed from the drum by a scraper, which can be assisted by air reversal through the valved filter ports. The belt filter is an improved version of the rotary-drum filter. After the vacuum filtering portion of the cycle is completed, the filter medium (a continuous belt) is lifted from the drum and passed over a small-diameter roller to remove the cake. Vacuum filters are typically used in secondary dewatering, taking sludges of 20 to 45% solids from thickeners and concentrating them to 50 to 85% solids. The increased solids content of filtered sludge decreases the amount of area required for ultimate storage and reduces the potential for dispersal of waste materials via seepage. As of late 1978, only about 25% of the operating and proposed plants used vacuum filtration.

Centrifuges. The typical solid-bowl centrifuge consists of a combination cylindrical/conical rotating element or bowl, partially closed at the ends. Slurry is piped into the bowl and forms a peripheral ring due to the centrifugal force generated by rotation. Solid particles (which are denser) are forced to the outside of the ring of slurry, leaving clear liquor toward the inside. As more slurry is added, clear liquor fills the bowl to a preset level and overflow is discharged through ports in the bowl. The object of this operation is to fill the machine with solids as more and more are settled out against the walls of the bowl. The solids are removed from the bowl with a concentrically mounted screw conveyor rotating at a speed different from that of the bowl, pushing the solids toward the conical end of the bowl and out discharge ports. Currently, few operators have chosen to use centrifuges for dewatering FGD sludges.

Dewatering aids. Scrubber sludges usually contain some very small particles that are dispersed as a stable colloidal suspension. Suspended particles tend to remain dispersed and do not agglomerate and settle out. Polyelectrolytes (high-molecular-weight polymers) added to the sludge can destroy this stability and promote formation of free-draining solid aggregations; these additives have proved to be successful in improving all types of dewatering operations.

Underdraining Impoundments

Underdraining impoundments provides a simple, economic, and widely applicable method of reducing a sludge impoundment (simple pond) to an acceptable landfill available for other ultimate uses (Duvel et al. 1979). Prior to introduction of sludge, a drainage bed is placed in the floor of an impoundment to collect seepage and rainfall percolating through the sludge. The drainage can be collected and reused as makeup water in the scrubbers, reducing consumptive water requirements of the system. Only a minor accumulation of liquor provides a small hydraulic head, and leachate infiltrating into the soil and ultimately to groundwater is greatly reduced. Therefore, pond liners and/or low permeability soils are not necessary for siting sludge impoundments except in cases where soils are highly permeable or the water table is high. After the impoundment is filled and retired, the land can be reclaimed for other uses. Apparently this method is applicable to all scrubber sludges currently being produced (Rossoff et al. 1979). However, because impoundment is used rather than mechanical dewatering, the area committed for this process can be large.

Chemical Fixation

Fixation/stabilization involves treatment of sludge with one or more chemical additives to improve its chemical and/or physical properties (Duvel et al. 1979). The treated (stabilized) sludge is suitable for ponding or landfill storage, depending on the treatment process used. There are a number of companies that offer fixation methods but, at the time of this report, only two have full-scale operating experience: Dravo Corporation and IU Conversion Systems (IUCS). These two systems use chemical additives (having cementitious properties) that cause chemical reactions similar to those of Portland cement (calcined clay and limestone). The additives are: Dravo "Calcilox," ground blast furnace slag; and IUCS "Poz-o-Tec," lime and fly ash. Chemical fixation of FGD sludges results in a solidified waste which can be handled much more readily than the original sludge. The fixed waste can be landfilled, requiring less land for storage. The permeability of the waste is reduced by fixation, thereby reducing the amount of material that can leach from the waste into the environment. The fixing processes can be expensive and, to date, few (10-15%) operators have chosen to use them for secondary dewatering.

Forced Oxidation

Forced oxidation involves forcing air through the sludge, thereby accelerating the oxidation of calcium sulfite to calcium sulfate (gypsum) (Duvel et al. 1979). Gypsum is more desirable because of its higher settling rate, ease in subsequent dewatering and consequent reduction in waste volume, and reduced thixotropic properties. The gypsum then can be landfilled or pumped to a settling pond without mixing or use of additives. The degree of oxidation attained depends on pH, air stoichiometry, and design of the air tank. Likewise, the properties of the oxidized sludge vary with the degree of oxidation and amount of fly ash added. The calcium sulfate sludges are more readily handled and dewatered than calcium sulfite sludges, and less land is required for disposal of the sulfate sludges. The gypsum produced may be marketable, obviating the need for long-term storage. However, forced oxidation is more costly than simply transporting the sludge to a pond, and very few operators use it.

Blending with Ash

Blending scrubber sludge with coal ash (primarily fly ash) is essentially the same as the fixation processes discussed above. However, no lime is added, and the natural cementing (pozzolanic) property of the fly ash is used to bind the sludge. This is beneficial for many utilities because sludge and ash can be stored in one simple process, making them suitable for landfilling which takes up less land area. Again, the blend has a lower permeability than the original sludge, reducing the potential for seepage. However, the addition of ash to FGD sludge results in a higher concentration of potentially mobile elements than is found in the sludge alone. About 25% of FGD waste facilities have chosen to blend ash with scrubber sludges.

COAL ASH AND FLUE-GAS-DESULFURIZATION SLUDGE STORAGE

The increasing complexity and rising costs of modern ash and sludge storage techniques have focused attention on the benefits accruing from maximum possible utilization of these wastes. However, it is anticipated that even under optimum market conditions, utilization of ash will only be about 25% of total production in the United States (Frascino and Vail 1976). Thus, the remaining 75% must necessarily be suitably handled while satisfying all applicable requirements. Similarly, recycling of FGD sludge for other uses is expected to account for only a fraction of the amount produced (Duvel et al. 1979). Regulations and guidelines for managing waste-storage sites require that the wastes be stored in a well engineered, well monitored, and environmentally acceptable manner (e.g., lined ponds). The most stringent requirements relate to maintaining surface water and groundwater quality and are designed to ensure that potentially toxic wastes do not contact humans or other biota. In order to adequately protect fish and wildlife resources, operators must strive to contain the ash and sludge wastes, and our discussion of storage techniques should be read in this context.

There are two basic types of storage systems for coal ash and FGD sludge wastes: wet (ponding) and dry (landfilling). In wet-storage systems, the wastes are transported to and deposited in the storage site (pond) in a fluid state. Dry-storage systems often employ one or more processing steps that solidify the waste in order to facilitate its handling and storage as a more stable solid material. Modifications of these two basic storage methods include a number of combinations that result in a broad range of storage practices, some of which overlap. The two basic methods are equally applicable for storing coal ash (slag, bottom ash, and fly ash) as well as scrubber sludge. Scrubber sludge, however, presents some special handling considerations as a result of its physicochemical character. Sludge has thixotropic properties and as such is not amenable for use in structural landfills. As a result, most sludge is stored by ponding, and where ponding is impractical, the need to dewater or chemically stabilize the sludge for landfilling is unavoidable. Table 14 is a summary of the following discussion of methods for ultimate storage, including applicability to ash and various sludges. This discussion represents a compilation of the information contained in Frascino and Vail (1976), Ansari et al. (1979), Duvel et al. (1979), DiGioia et al. (1979), and GAI Consultants (1979).

Wet Storage (Ponding)

In all wet storage systems, the by-product waste (ash or FGD sludge) is transported and deposited in the storage pond as a fluid (slurry). The waste stream is usually piped directly to the storage pond; alternative means of transport include railroad tank car, tank truck, or barges--depending on location, access, and costs. In some cases, after settling, the ponded slurry supernatant is returned and recycled in the scrubber system. In others, the sludges in the waste stream may be processed or treated to varying degrees prior to storage.

Wet storage is versatile and has wide applicability. It can be utilized by power plants having wet or dry ash-handling systems, alkaline ash, or lime/limestone scrubbing systems. These systems produce waste streams as a solid/liquid slurry, a form amenable to wet storage. Double alkali sludges are not amenable to wet storage because they are dewatered to recover and reuse the soluble scrubbing agent. The inherent simplicity and low cost of operation and maintenance make wet storage attractive to waste managers. Currently, wet storage systems are handling both coal ashes (GAI Consultants 1979) and FGD sludges (Duvel et al. 1979).

Wet storage systems use pipelines for the transportation of the wastes, either as FGD sludge or ash slurries. Because of the difficulties in handling sludges and slurries, other means of transport (e.g., truck or railroad) are not as technologically efficient as pipelines. Under normal operations, pipelines contain the wastes, reducing the potential for a general dispersal of the materials into the environment. Construction of a pipeline preempts some land from other uses and may obstruct the free movement of large game animals. However, a properly maintained pipeline system will be of minor environmental concern compared to the storage area itself.

Table 14. Comparative Summary of Waste Storage/Disposal Options[a]

	Wet storage	Dry storage	Mine disposal	Ocean disposal
Method	Ponding of waste	Landfilling of waste	Backfilling mines with waste	Depositing waste in the ocean
Applicability	Most wastes and handling systems except double alkali	Dry or fixed waste	Mainly dry or fixed wastes	Dry or fixed wastes from coastal sources
Advantages	Simple	Low land requirement	Sites available	No new land preempted
	Versatile	No impoundment required	No new land preempted	
	Low traffic potential	Low seepage potential	Aids in mine stability	
	Low dust potential	Reclamation practicable		
		No attraction for biota		
Disadvantages	High land requirement	Sludge fixation required	High leaching potential	Coastal areas limited
	Impoundment construction	High dust potential	Potential for acid mine drainage synergisms	May be environmentally unacceptable
	High potential for seepage	High traffic potential	May require dewatering	
	Sludge instability	Requires diversion of runoff	Plant must be near mine site	
	Reclamation uncertain	May require further processing		
	Liners may be required			
	Ponds may attract biota			

[a] Data from Frascino and Vail (1976), Ansari et al. (1979), and Duvel et al. (1979).

Although seemingly most advantageous in terms of ability to handle a wide variety of waste streams, wet storage is not universally applicable. Climate, regulatory requirements, and/or site-related constraints can render wet storage undesirable or uneconomical. For example, in areas of high rainfall where overflow or discharge is not permitted or is strictly regulated, storage ponds may be infeasible. On the other hand, excessive evaporation from ponds in arid regions would be unacceptable where water conservation and recycling is an important consideration. Wet storage systems would generally have a higher consumptive water requirement than dry systems.

A major disadvantage of wet waste systems is that the volumes of waste are larger than in dry systems, and because of the low bulk densities of wet wastes, larger land areas are required for storage. Thus, wet storage has a greater potential for preempting land from use by wildlife. Once an impoundment is constructed to meet an anticipated need, habitat is disturbed and discontinuation of the storage process cannot mitigate habitat loss impacts beyond initiating early recovery. Successful reclamation of these impoundments is uncertain, and recovery of wildlife habitat may not be possible because the sludges remain structurally unstable even after prolonged settling and dewatering. The potential for seepage of substances from containment is high because of the hydraulic head maintained in the ponds, and thus liners are often required. Pond supernatant liquid may be attractive to biota as watering sites, and potentially toxic substances could be ingested with the pond waters. Extensive hydrological studies are required to ensure safe operation of the pond, and a relatively high capital investment in pipelines and impoundment structures is required.

A number of pond configurations (Figure 6) exist for the impoundment of wastes (Duvel et al. 1979). Diked ponds, probably the most common pond configurations, require a nearly level site and are contained within a perimeter embankment or dike. The dike may be constructed either from materials excavated from below the existing grade at the pond site or from borrow sources above the grade.

An incised pond is contained in an excavation entirely below the existing grade. This type of pond is most appropriate where the water table and bedrock are deep; it is preferable where space is limited for dikes or where excavated materials are unsuitable for dike construction or are valuable for other uses. For example, if a storage pond was excavated into an aggregate-grade sand deposit, the sand could be stockpiled for future sale.

A variation of the incised pond is the use of existing basins, or abandoned surface mines and quarries. If available and approved by the appropriate regulatory agencies, abandoned surface mines or quarries may provide suitable storage sites.

The side-hill pond configuration takes advantage of local hilly terrain to provide one or two sides of an impoundment. However, it may be difficult to safely construct a large side-hill pond on steeply sloping sites.

A cross-valley pond is formed by constructing a dam across a portion of a natural valley between the valley walls, similar to typical water-impounding dams. In this configuration, both the natural drainage, as well as waste materials, are impounded behind the dam. The design is more critical here than with other pond configurations because, in addition to waste storage, it must provide for the controlled storage and discharge of the natural flow in the valley as well as handle design flood flows. Because it is placed in a natural drainageway, potential for seepage is high.

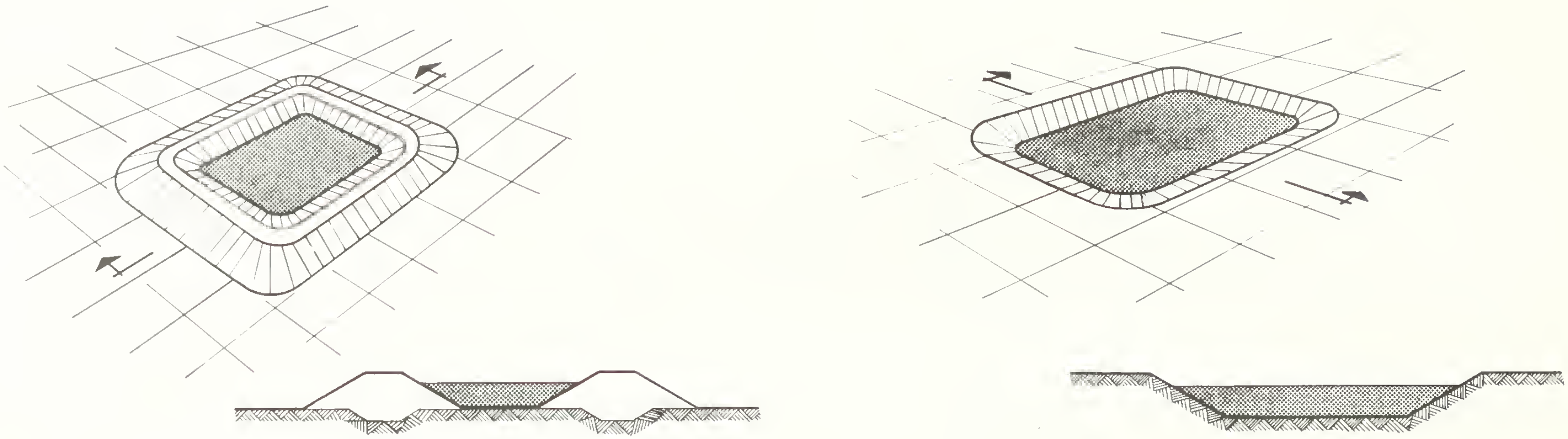
Dry Storage (Landfilling)

In dry-storage systems, the waste is collected dry or processed so that it can be deposited in the landfill site as a solid or stabilized material, or as one which will solidify (set) quickly. In these cases, dikes or embankments are not required. In most dry-storage systems, the FGD sludges are processed mechanically or treated chemically so that they can be handled as a solid material.

The trend in waste storage appears to be moving toward dry storage. One report indicates that as of 1979, 65% of online power plants have capabilities for dry ash collection and handling. About half of all ash is trucked to the storage area and the balance is sluiced; 40% of the fly ash is separated from bottom ash before storage and 60% is stored together (Natl. Ash Assoc. 1979a). The same trend toward dry storage can be seen in FGD sludge management (Duvel et al. 1979). However, dry storage, overall, is less versatile than wet storage, and its relative applicability is much smaller. Dry storage of sludge requires blending of dry material (e.g., ash) with sludge to aid in reducing its moisture concentration. As a result, dry storage can be difficult or uneconomical where waste streams include little or no dry fly ash. Thus, utilities employing cyclone-fired boilers, wet ash-handling systems, and oil-fired boilers generate waste streams of a fluid nature that are therefore not amenable to dry storage (Duvel et al. 1979). Dry storage systems are most applicable in arid regions (where water availability is low), in power plant systems with high ash/sludge ratios, and in areas where land availability is low.

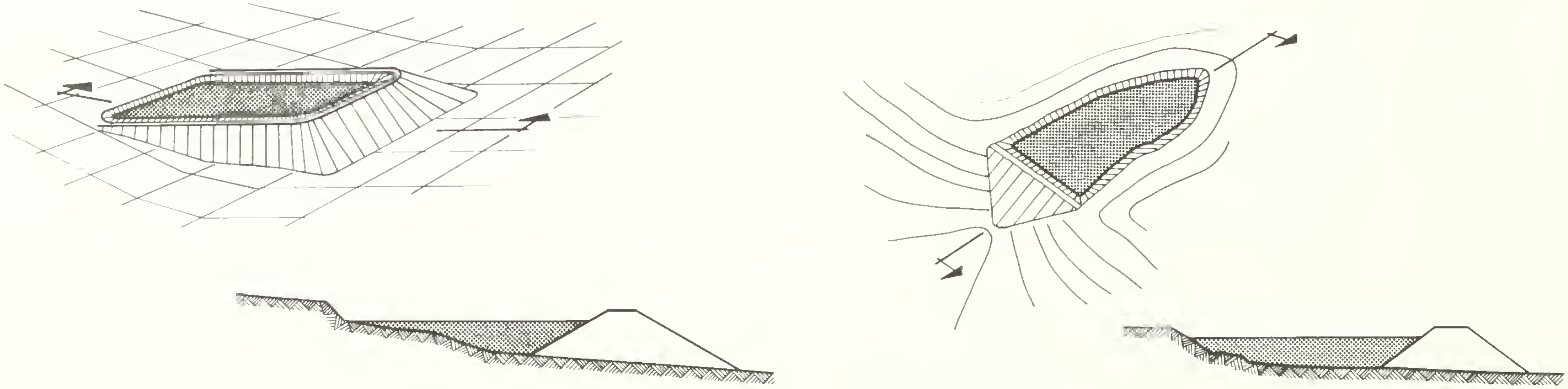
The higher bulk density of dry wastes results in less land required for storage of the wastes and more efficient use of the storage site. This means that per unit mass of waste, less potential habitat will be lost. The need for liners is less than in impounded storage sites. The methods for reclaiming landfills are much more highly developed than for wet-storage sites; thus, successful reclamation is more likely on dry-storage sites. The absence of ponded water above the dry waste reduces the potential for seepage of leachate from the site. Use of landfills can be readily terminated short of their maximum lifetime without disturbance of unused land. In general, landfill systems pose less of a potential for adverse impact to biota than pond systems because of the lower land requirements and lower potential for leaching out soluble substances.

For dry storage, the need to fix FGD sludges makes the process more expensive than for wet systems without fixation. Large amounts of materials (e.g., fly ash and soil) are required to aid in stabilizing sludges. The waste may require further processing to remove compounds of a high pollution potential (e.g., pyrites). Dry storage usually requires construction of facilities to divert runoff from the landfill areas.



Diked Pond Constructed Above-Grade

An Incised Storage Pond



A Side-Hill Storage Pond

A Cross-Valley Pond Configuration

Figure 6. General Types of Storage Pond Configurations. From Duvel et al. (1979).

Dry storage systems use trucks as the predominant means of transporting the waste; rail is a less frequently used option. Containment of wastes in these systems is not as complete as in pipelines, and wind dispersal of the wastes is much more likely than in wet transport systems. Proper handling of the dry waste can mitigate dusting problems. When new waste-storage sites are developed, new roads or railbeds are required, removing land from other users such as wildlife. Again, impacts to fish and wildlife resources from the transportation systems are minor in comparison with impacts from development of the waste-storage site itself.

Options for landfill configurations (Figure 7) also exist (Duvel et al. 1979). The simplest configuration for dry storage of waste ash and sludge is a heaped fill. This type of landfill is typically used in areas where the terrain is level. When the site is properly managed, problems of groundwater pollution, slope stability, and site preparation are usually minimal. Heaped landfills, however, do not blend with the surrounding level terrain and are highly visible. Because of the high visibility, this method of landfill storage may be the least preferred.

In hilly or gently sloping terrain, side-hill construction is often preferred. The fill can be constructed along the side of a hill or valley. Properly constructed, side-hill fills may blend well with the existing terrain and may provide valuable property when reclaimed. Site preparation and slope stability work are generally more complex than with heaped fills.

The most common type of landfill in areas of hilly terrain is valley fill. Since valleys are natural avenues of surface runoff and, in some cases, of springs along the side slopes, control of surface water and groundwater is necessary. It is important that runoff be directed under or around the fill to avoid impoundment and to help control erosion and runoff pollution. Similarly, to be covered by fill, springs must be collected and channelled to reduce leachate generation and slope instability. In general, site preparation for valley fill is more complex than for other configurations.

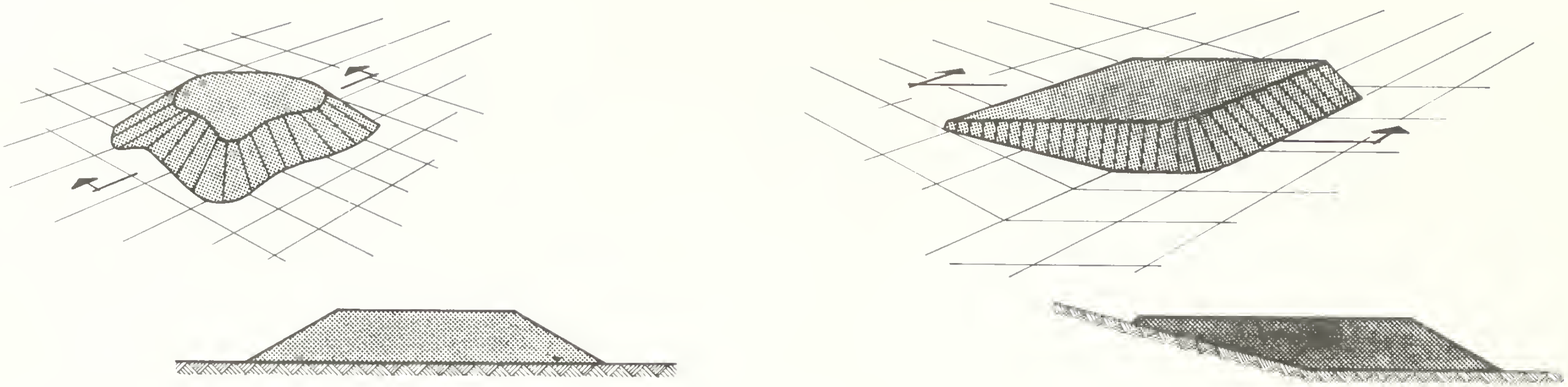
Mine Disposal

Use of surface or deep mines and quarries (active or abandoned) for disposal of coal ash and FGD sludge wastes represents a special case in waste handling. Overall, the concept of mine/quarry disposal of ash and sludge wastes has merit and should receive consideration in the site-selection process, especially where suitable locations, active or abandoned, are within reasonable hauling distances. For FGD sludges, mine disposal has received consideration in a number of instances, but no full-scale operation of this kind has yet been developed (Duvel et al. 1979). For coal ash, it is practiced at a number of mine-mouth plants currently in operation.

The return of waste materials (ash and scrubber products) to their point of origin is conceptually attractive and offers certain advantages. However, there are also problems and/or constraints that need to be evaluated. Disposal of wastes must not interfere with or disrupt current or future mining activities. Mine/quarry disposal has potential for impacts on groundwaters and surface waters where the mining has encountered these waters. The disruption of these features can enhance the potential for dispersal of soluble materials from wastes deposited in mines. The degree of impact is dependent on site-specific geology, groundwater hydrology (including pathways to surface water), and waste characteristics. Protective or mitigative measures can be costly and could outweigh other cost advantages, making such an option unattractive to waste managers. In addition to general siting and design criteria, mine/quarry disposal would necessitate development of an expanded monitoring program at the mine disposal site. Analysis of the monitoring results can aid in understanding what effects, if any, the disposal operation has on groundwater quality.

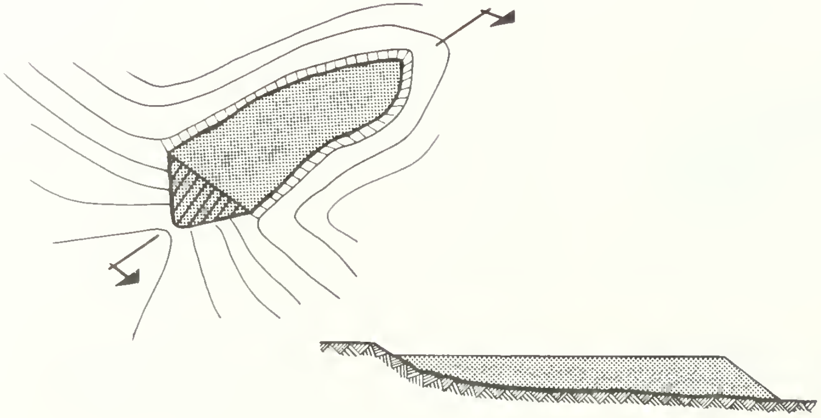
Before burial, the waste material should be mixed with overburden; this requires dewatering and/or fixation for wastes in wet-handling systems. Transportation problems may include some means of holding the sludges in suspension during transit and difficulties in unloading. If certain ash and sludge wastes are exposed to the elements when shipped long distances, the existing coal cars may not be capable of handling the wet sludge. Specially equipped tank cars may be required to alleviate unloading difficulties. Overall, in conjunction with transportation costs, the problems of in-transit suspension, unloading, and liquor disposal combine to render railroad transport infeasible. For deep mine disposal of wastes (particularly wet sludges), additional problems of economical placement, containment, and monitoring must be considered. Viscosity and settling characteristics of the wastes must be carefully controlled. In all cases, such disposal is only feasible where the mine or quarry and the generating plant are close enough for transport costs to be acceptable or there are no other available options.

Provisions for backfilling and grading of both surface and deep mines are included in the rules and regulations pertaining to Surface Coal Mining and Reclamation Operation (U.S. Dep. Inter. 1979) and must be complied with. Backfilling with coal ash or FGD sludge could require meeting guidelines for covering toxic-forming materials (30 CFR 816.103, 817.103). These



A Heaped Landfill Configuration

A Side-Hill Landfill



A Valley-Fill Storage Configuration

Figure 7. General Types of Landfill Configurations. From Duvel et al. (1979).

guidelines require a minimum cover of nontoxic material to a depth of 1.2 m (4 ft). Additional precautions may be required to reduce the potential for soluble substances leaching from the waste, erosion of waste materials, upward migration of soluble salts, or uptake of toxic substances by plants.

Ocean Disposal

Ocean disposal of waste sludge may be an available alternative for some coastal utilities, but it does not have wide applicability. No stations currently utilize this option; and, although the USEPA has examined the concept, neither the technical practicality, economic feasibility, nor regulatory acceptability has been fully demonstrated (Lunt et al. 1977). In fact, ocean dumping in general is being discouraged by governmental agencies. However, the utility industry is pursuing research to develop means for ocean dumping of ash and sludge that would be environmentally acceptable (Woodhead et al. 1979).

The primary advantages of ocean disposal would be minimal incremental land use, no disposal-site development costs, and availability of relatively inexpensive transport facilities (barge). It would be limited to coastal utilities not having land area within a reasonable hauling distance. The option is particularly attractive to the urbanized Atlantic Coast states. The system is not applicable to untreated, sulfite-rich FGD sludges because of potential sulfite toxicity effects. Disposal of stabilized sludges by pipeline or quick-dumping craft is not suitable on the continental shelf because of sedimentation and suspension impacts. For all practical purposes, transportation is limited to barging.

Handling options include (1) dispersal of sulfate-rich wastes on the continental shelf or in the deep ocean, (2) concentrated disposal of stabilized (solidified) wastes on the continental shelf, and (3) concentrated disposal of both sulfate-rich and stabilized wastes in the deep ocean.

If ocean dumping is to be seriously considered as a disposal option, the regulations and criteria with respect to ocean dumping (USEPA 1977) must be complied with. The USEPA (1980) is currently proposing modifications to existing regulations. In order to obtain a permit for ocean dumping, the operator of a waste facility must demonstrate that there are no other options economically and environmentally available. Certain types of wastes may not be disposed of in the ocean. Although current regulations (40 CFR 227) have not specifically identified utility wastes, these wastes do contain compounds that are listed as toxic pollutants in 40 CFR 401, and control of these pollutants would be required if an ocean dumping permit were to be granted.

UTILIZATION OF COAL ASH AND FLUE-GAS-DESULFURIZATION SLUDGE

Ash Utilization

Currently, less than 25% of the coal ash and slag produced nationally is recovered for use. In 1978, 17.4% of fly ash, 34.0% of bottom ash, and 58.8% of boiler slag collected in the United States were utilized (Table 15). Other areas in which ash utilization is under study include the following:

- Agricultural applications and land reclamation projects (soil modification, neutralizing soils).
- Water treatment applications (phosphate removal, and treatment of eutrophic ponds).
- Grouting mixes (to shore up deep mines to alleviate subsidence problems).
- Combining with sludge and other additives to form inert solids for use in the shallow ocean to create artificial, shallow ocean reefs (Natl. Ash Assoc. 1977).
- Fire abatement in landfills or coal mine refuse piles.
- Mineral recovery.
- Use of cenospheres (tested and approved) in nose cones of space shuttles (Natl. Ash Assoc. 1979b).

Recovery of ash and slag for such uses will reduce the land area required for waste storage. However, recovery and use of these wastes is not expected to increase at the same rate as waste production.

Table 15. Ash Collection and Utilization, 1978[a]

Description	Fly ash	Bottom ash	Boiler slag[b]
	Millions of tons		
Total ash collected	48.3	14.7	5.1
Ash utilized	8.4	5.0	3.0
	Percentage commercial utilization		
Mixed with raw material before forming cement clinker	4	-	3
Mixed with cement clinker or with Type 1-P cement	4	3	-
Partial replacement of cement in concrete and blocks	24	-	-
Lightweight aggregate	3	3	-
Fill material for roads, construction sites, land reclamation, dikes, etc.	11	21	12
Stabilizer for road bases, parking areas, etc.	3	5	2
Filler in asphalt mix	10	-	-
Ice control	-	22	15
Blast grit and roofing granules	-	-	45
Miscellaneous	4	10	19

[a] Data from National Ash Association (1979a).
[b] If separated from bottom ash.

Sludge Utilization

In general, the prospects for large-scale utilization of scrubber sludge appear small (Duvel et al. 1978). About 40 different uses or products (Table 16) have been considered for sludge, but none have been developed in the United States to the point of being practical commercial alternatives to storage. The sludge situation is somewhat similar to the fly ash utilization experience. There are some markets that could accommodate small portions of the waste, as well as others where all could be utilized. For some time to come, however, it is likely that the majority of scrubber sludge will be stored (Duvel et al. 1978). Applications of sludge shown to be feasible at least in small-scale demonstration include:

- Soil amendment to improve porosity or to enhance nutrient-deficient soil.
- Mixed with fly ash for landfill and surface reclamation.
- Fixed sludge as fill in road bases (parking and driveways).
- Fixed sludge used in liners for wastewater ponds.
- Use in wallboard manufacture (calcium sulfate-gypsum) and in cement as set-retardant.
- Structural fill if properly treated with a chemical fixative to meet structural specifications.

Table 16. Summary of Proposed Schemes for Flue-Gas-Desulfurization Sludge Recovery/Waste Conversion[a]

	Level of development				
Proposed scheme	Concept developed	Laboratory studies	Pilot demonstration tests	Full scale	Comments
Recovery of chemicals:					
Aluminum	+	+	+	-	Use of fly ash more feasible
Ammonium sulfate	+	+	+	+	b, c, d
Calcium oxide	+	-	-	-	
Magnesium oxide	+	+	+	-	b
Sulfur	+	+	+	+	c, d
Sulfuric acid	+	+	+	+	b
Sodium sulfate	+	+	+	+	b, c, d
Sodium sulfite	+	+	+	+	b, c, d
Sulfur dioxide	+	+	+	+	b, c, d
Ammonia	+	+	+	+	c, d
Calcium carbonate	+	+	-	-	
Manufacture of building materials:					
Portland cement	+	+	-	-	
Concrete admixture	+	+	+	+	c
Sinter brick	+	-	-	-	
Calcium silicate brick	+	+	+	-	
Aerated concrete	+	+	+	-	
Poured concrete	+	+	+	-	
Concrete block	+	+	+	-	
Gypsum wallboard	+	+	+	+	c, demonstration in U.S.
Lightweight aggregate	+	+	-	-	e
Mineral aggregate	+	+	-	-	
Cenospheres	+	-	-	-	Feasible with fly ash only
Mineral wool	+	+	+	-	e
Gypsum-plastic	+	+	+	+	c
Structural fill:					
Land recovery	+	+	-	-	
Surface-mine reclamation	+	+	-	-	
Deep-mine reclamation	+	+	+	-	
Highway or similar embankments	+	+	+	+	Work done with fly ash
Paving materials:					
Road base	+	+	+	-	
Agriculture:					
Soil stabilizer	+	+	-	-	
Pollution control/ environmental:					
Artificial reefs	+	+	+	-	
Filter aid for sewage sludge dewatering	+	-	-	-	
Pond liners	+	+	+	-	
Neutralize acid mine drainage	-	-	-	-	Excessive quantities required

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[a] Data from Duvel et al. (1978). The + sign designates that use is developed at the indicated level; the - sign designates that use is not developed to the indicated level.

[b] These do not involve calcium-base sludges.

[c] Full-scale application in Japan.

[d] These sludges are not from power plant scrubbers.

[e] SO_2 is expelled in the process and must be controlled.

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POTENTIAL IMPACTS FROM HANDLING AND STORING COAL COMBUSTION AND EMISSION-ABATEMENT WASTES

IMPACTS TO TERRESTRIAL ECOSYSTEMS

The major sources of impacts to terrestrial ecosystems from ash and sludge storage are: (1) loss of habitat, (2) runoff and seepage of toxic elements from storage sites, (3) fugitive dust, and (4) wildlife use of active ponds. These sources are listed by their overall relative potential for impacts; however, the magnitude of the resultant impacts will vary with particular storage practices and with regional abiotic and biotic characteristics. The abiotic and biotic pathways by which potential impacts are expressed are elucidated in the following discussion.

Loss of Habitat

Potential impacts. The continuing escalation of contemporary land values reflects in part the ever-increasing competition for use of the land resource. Thus, the land requirement for storage of coal ash and FGD sludge wastes is frequently a major or limiting determinant in decisions involving waste management practices or siting of new storage facilities. Socioeconomic pressures often favor siting these facilities in economically unexploited areas that may be important habitat for fish and wildlife. Land storage of utility wastes will require the preemption of such habitat from use by fish and wildlife for the active lifetime of the storage facility. Estimates of land required for ash and FGD-sludge storage are as high as 1500 ha for waste from a model 2100-MWe utility during a 40-year operating life (Table 17). Land requirements for a proposed waste-storage system are primarily dependent on the volume of wastes to be stored, the design specifications relative to the ultimate thickness (depth) of the surface water layer, and the condition of the wastes. The assumptions upon which Table 17 is based must be borne in mind when extrapolating to a specific situation.

The estimates are specific for the required storage area and do not include other land requirements such as those for access and service rights-of-way, dams or peripheral dikes, staging areas, and erosion-control structures such as settling ponds. These additional land requirements will vary, depending on site-specific conditions. Assessing the effects of land preemption will require a working knowledge of all design characteristics of the proposed storage system (Figures 6 and 7), as well as a thorough understanding of how the site will be altered prior to, during, and following waste storage. The long-term impacts, beyond the active life of the site, will be determined by the success of reclamation of the site after closure. As will be seen in the section on reclamation, success is by no means certain.

Withdrawal of land from use by wildlife may have marked impacts upon local faunal populations. Some of the less mobile species may be killed by clearing and construction activities. Although mobile species can move into adjacent habitats, the resulting increased competitive pressures may prove to be detrimental to the population as a whole. It is difficult to assess the potential magnitude and impact of the increased competition pressures due to displacement of individual wildlife. Available information is largely anecdotal, and predictions of negative impact are based upon the assumptions that habitats are normally at carrying capacity and increased competition is detrimental. There has not been rigorous testing of these assumptions.

Of particular concern is the displacement of wildlife populations from habitat that is important to their life history, e.g., winter foraging, nesting, or breeding areas. If such areas are rare in a given locale, their removal from use by wildlife may markedly reduce the wildlife abundance by reducing reproductive or foraging success. This may be of special concern if rare, endangered, or other sensitive wildlife populations are involved. Therefore, in assessing the impact of land preemption due to storage of coal ash or FGD sludge wastes, one must first evaluate the kinds, extent, and value of habitat available to local wildlife resources.

Land-use impacts are expected to be more severe in the Northeast and Southeast because of existing intensive land uses and high population densities. The implications for wildlife may be serious because of the general paucity of undisturbed habitat. In the Midwest, land-use impacts will be moderate because of the lower human population density and predominance of agricultural land uses. However, the socioeconomic pressures could force the preemption of

Table 17. Estimated Forty-Year Requirements for Coal Ash and FGD Sludge Storage at Four Model Coal-Fired Power Plants[a]

Plant size (MWe)	Land area required for ash storage[b]					
	Western coal		Northern Appalachian coal		Eastern Interior coal	
	(ha)	(acres)	(ha)	(acres)	(ha)	(acres)
100	11	28	8	20	16	40
350	40	98	28	70	57	140
700	79	195	57	140	113	280
2100	238	588	170	420	340	840

Plant size (MWe)	Land area required for FGD sludge storage[b]							
	Northern Appalachian coal				Eastern Interior coal			
	Limestone		Lime		Limestone		Lime	
	(ha)	(acres)	(ha)	(acres)	(ha)	(acres)	(ha)	(acres)
100	32	80	29	72	58	144	45	112
350	113	280	102	252	204	504	159	392
700	226	558	204	503	406	1003	316	781
2100	680	1680	612	1512	1225	3024	953	2352

[a] Data from Dvorak et al. (1978).

[b] Calculated using data from Cooper (1975) on land area requirements for sludge and ash disposal. "Ash" refers to precipitator fly ash and bottom ash combined (density = 80 lb/ft^3, dry); depth of waste ponds assumed to be 3 m (10 ft). Sludge assumed to be 50% solids; depth of ponds assumed to be 3 m (10 ft).

relatively undisturbed land from use by wildlife. In the West, land-use impacts are expected to be minimal because of the sparse population densities and large areas of land existing in their natural conditions. Impacts could be substantial, however, if waste storage preempts wildlife access to important or critical resource areas, e.g., winter foraging areas.

Evaluating land requirements. Comparatively inefficient land use is illustrated by the above-grade diked pond because much of the site is occupied by peripheral embankments constructed of borrowed materials (Figure 6). A modification of this configuration entails excavating central portions of the pond and using the excavated materials in dike construction, thereby increasing the space-effectiveness of the impoundment and reducing the need for offsite borrow materials. The degree of modification will be limited by the depth to groundwater or bedrock. Circular or square ponds are more space-effective than elongated ponds, since the latter require greater length of embankment to provide comparable storage volume.

The incised storage pond shown in Figure 6 illustrates below-grade storage. This configuration is space-effective; however, evaluating the total land-use impact entails considerations of the disposition of excavated materials during construction of the pond. The advantages of side-hill and cross-valley ponds are that the existing terrain serves as an impoundment structure for one or more sides of the storage pond (Figure 6). The space-effectiveness of these configurations is strongly conditioned by topography; effectiveness decreases with increasing steepness of the terrain.

Comparisons between Figures 6 and 7 indicate the relative significance of dams and peripheral embankments with regard to land requirements for storage ponds and landfills. For landfills, the area occupied by the waste materials can be estimated as the volume of the waste divided by the depth of the landfilled waste. For example, 2100 ha-m (2.1×10^7 m^3) landfilled to a depth of 10 m would occupy about 210 ha (2.1×10^6 m^2) of land area. Impounded wastes require an additional preemption of land by the berm or dike surrounding the storage area. If more accurate information is not available, one may use an approach for estimating area under the dike as illustrated in Figure 8. This figure presents a schematic of the distance beyond the waste covered by a dike with a 5:1 external slope and 3:1 internal slope, i.e., 3 horizontal meters for each vertical meter. For this estimate, it was assumed that waste depth was 10 m, freeboard (height from waste surface to top of dike) was 2 m, and a 5-m wide roadway ran along the top of the dike. Under these assumptions, simple geometric calculations yield a dike basal width of 101 m. In adding the dike width to area preempted by waste alone, a correction for displacement by the internal slope can be estimated as half the horizontal distance from the contact points of the dike and the substrate to that of the dike and the waste surface (15 m in our example). The total area preempted by impounding 2100 ha-m of waste can then be approximated as: the area preempted by the sludge (210 ha) plus the additional area at the perimeter of the impoundment occupied by the dike (e.g., 86 m × length of dikes required). In our example, 86 m is the width of the dike (101 m) minus the correction for displacement of waste by the dike (15 m). In some areas (particularly urban areas), a 30-m buffer area may also be required around the site (Duvel et al. 1979). In our example, then, if we assume the area is square, the maximum amount of land preempted would be 210 ha plus about 40 ha.

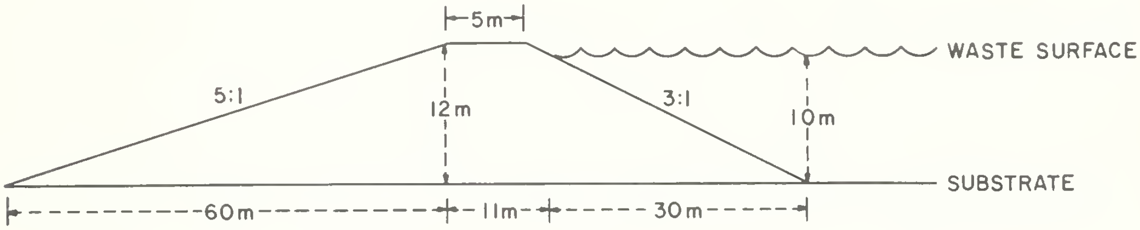


Figure 8. Schematic of a Generalized Impoundment Dike.

Evaluating impacts. Over the past several years, the U.S. Fish and Wildlife Service has been developing a methodology for evaluating the value of land as wildlife habitat, i.e., "The Habitat Evaluation Procedure" (cf. Flood et al. 1977, Schamberger and Farmer 1978). These procedures provide the wildlife biologist with a means of comparing the value of different habitats that may be affected by development of a waste-storage facility. In addition to knowing the value of the habitat to wildlife, it is also necessary to know the availability of that habitat for use by wildlife populations. If a habitat is rare and of high wildlife value, it is a less desirable site for those concerned with wildlife than is a more common habitat of moderate value to wildlife. In addition to quantifying habitat quality and availability, wildlife biologists must rely greatly upon their own experience and knowledge to evaluate the potential for adverse impacts from developing a waste-storage facility.

Runoff and Seepage

Ash and FGD-sludge waste constituents can be mobilized and dispersed from the storage site into the terrestrial environment by runoff and seepage. Runoff is a surface phenomenon, whereas seepage occurs within the waste and soil. The potential for waste transport into the environment is a function of (1) the method of ash and FGD-sludge storage, (2) local climatic conditions, (3) topography, and (4) soil characteristics. The magnitude of resultant impacts will depend primarily on the degree of erosion (due to runoff) and leaching (due to seepage) within the wastes and underlying strata. Impacts to vegetation and animals are, in turn, dependent upon the degree of erosion and leaching and on the soil's capacity to attenuate waste constituents.

Accidental release from waste impoundments does occur as a result of excessive rainfalls and/or poor dike construction; however, wastes from these releases generally only affect the soils and biota in the immediate area of the break. The short-term terrestrial environmental effects of dike washout would be soil erosion and destruction of herbaceous vegetation and small animals; the long-term effects would be the addition of large quantities of potentially toxic ash and sludge components to the soils in the path of the washout and future runoff of wastes into an expanded area. Ash and sludge waste-storage areas located in natural drainageways are more susceptible to runoff and/or washout, and impoundments must be designed to handle the maximum rainfall expected in the area. Open, uncovered or unimpounded wastes are highly susceptible to runoff of wastes into the surrounding habitat.

The major source of seepage will be from unlined coal ash and FGD sludge impoundments. Less seepage will occur at landfill storage sites because these wastes initially will have lower moisture contents than impounded wastes. Seepage from impoundments can be significantly reduced or eliminated by the use of clay, plastic, or rubber liners or by mixing fly ash, soil, or chemical additives with sludge in order to form complexes that bind water and trace elements within the waste (Dvorak et al. 1978; Hart et al. 1979). The major impacts associated with seepage will be the addition of potentially toxic substances to the soil solution. Toxicity to biota will depend on differences in their ability to accumulate these substances from the soil solutions and on differences in species tolerance.

Runoff and erosion potential. Open piles of waste can be readily eroded, resulting in dispersal of potentially toxic substances beyond the storage site. Under provision of the Resource Conservation and Recovery Act of 1976, such open dumping of waste will be curtailed. However, erosion of landfill areas and impoundment dikes can lead to dispersal of waste constituents from containment areas.

Erosion results from raindrop impact and runoff (overland flow of water). Although raindrop impact contributes to displacement of erodible materials, runoff is the principal transport mechanism of the erosion-sedimentation process induced by water. The principal surface-material characteristics influencing runoff and erosion are the infiltration capacity (maximum rate at which water enters the material) and the structural stability of the waste materials (Brady 1974). Runoff does not occur unless rainfall intensity exceeds the infiltration capacity. The low hydraulic conductivities of coal ashes and FGD sludges result in low rates of infiltration into these materials and enhances the potential for runoff.

Evaluating the erosion potential at a given waste-storage site is a prerequisite to assessing the potential for dispersal of waste constituents from containment. For this evaluation, one must determine the relative significance of the various contributing factors or conditions, the manner by which these influences interact under site-specific conditions, and the manner by which such interactions will be altered during site development and waste-storage activities. Local and state offices of the U.S. Soil Conservation Service (SCS) will be among the more productive sources for data collection, as well as for guidance in acquisition of additional published and unpublished data. If available, county soil survey reports prepared by SCS and cooperating agencies will be particularly useful as a basis for assessing erosion potential at a given site. Data may be available from local state universities as well. The content of these reports varies but typically includes soil maps, detailed soil descriptions, land-use capabilities and limitations, management guidelines, and information concerning local climate, geology, and topography. However, the soils in many counties of the United States have not been intensively surveyed. In the absence of detailed local surveys, regional soil survey data can be used to interpret soil and other characteristics of a given site. The reviewer must remember that regional survey information is usually of a general nature, and therefore of limited utility as a basis for site-specific assessment. In some instances, an investigator may have to rely heavily on observations made during a reconnaissance of the site.

Field evaluation of erosion potential at the proposed storage site entails recognition of the various types of erosion--generally referred to as sheet (or interill), rill, and gully erosion. Sheet erosion results from relatively uniform removal of particles from the exposed surface by runoff. Evidence of sheet erosion is apparent by the presence of perched pebbles and stones; in general, the higher the pedestal, the greater the intensity of active erosion. Sorting of surface materials may also be apparent because the larger particles are less readily dislodged by runoff. Rill erosion results from channelized flow of runoff; the erosive capacity increases with increasing accumulation of water. Rills enlarge due to continued abrasion of water and entrained sediments, as well as sloughing from the sides of the rill. The closer the spacing and the greater the depth of the rills, the greater is the intensity of active erosion. Gullies are essentially large rills, the distinction being that the rills can be obliterated by normal cultivation practices. Gully formation is initiated by the coalescence of rills; however, with continued erosion, the headwall of the gully is typically displaced in an upslope direction.

Runoff from a steeply sloping land surface may include particles attributable to all three types of erosion. In relatively level terrain, sedimentation may result primarily from sheet erosion. During reconnaissance of a proposed waste-storage site, the length of uninterrupted slope should be noted because long slopes erode readily. Further, the investigator will have to envision the manner in which the topography will be altered during development of waste-storage facilities and how this will influence the erosion potential of the site.

After integrating the information obtained by literature review and site reconnaissance, the investigator will usually be able to determine if development of a given site will result in unacceptable levels of water erosion and sedimentation. However, further investigations may be necessary in some instances. At sites where water erosion is a critical issue, the investigator may elect to use the Universal Soil Loss Equation (USLE) for predicting erosion potential at the various sites. The equation is used for estimating the sediment generated and displaced from a given area by sheet and rill erosion during a future period of time. Sediments from small areas such as waste-storage sites result primarily from sheet and rill erosion, and estimates based on the USLE are usually adequate for characterizing water-erosion potential (Wischmeier and Smith 1978). The USLE is a useful tool for comparing the management practices of alternative sites. The equation is expressed as:

Erosion Loss = (Rainfall Factor) · (Erodibility Factor) · (Topographic Factor)
· (Cover Factor) · (Support Practice Factor)

As presented here, the USLE should not be interpreted as a precise prediction of erosion loss. Many of the factors influencing erosion have been generalized in order to obtain a tool that can be readily used without a background in soil science.

Rainfall factors (R) are calculated values representing the integration of raindrop impact effect and the amount of runoff; R values have been calculated for numerous areas throughout the contiguous United States and are the basis for the isorodent (lines of equal R values) delineations shown in Figure 9. The R value for a given site can be established by interpolation between two adjacent isorodents. For example, R values for the southern third of Illinois range between 200 and 250; the value for a site equidistant between the isorodents is 225. The isorodents are based on rainfall characteristics, and R values for areas where significant runoff results from ice and snowmelt must be adjusted as follows: given an R value of 20 and precipitation from 1 December through 31 March equivalent to 12 inches of water, the adjusted R value equals 1.5 (12) + 20, or 38.

The erodibility factors (K) are quantitative values experimentally determined. Tables of established K values for many specific soils are available from state SCS offices. However, such values may or may not be appropriate for subsurface and other materials exposed during site development and management operations. In some instances, subsurface materials are substantially less or more erodible than surface soils. Given the textural composition, organic matter content, structural characteristics, and permeability of the materials to be exposed, the K values can be approximated by use of the nomograph presented as Figure 10. Much of the necessary information should be obtainable from the utility. Other information can usually be extracted or approximated from published soil surveys or other literature, but analysis of materials will probably be necessary in some instances. For dry ash, K is approximately 0.15--assuming particle size distribution of a clay, no organic matter, very fine granular structure, and very slow permeability.

Topographic factors (LS) are a combination of the slope length (L) and slope gradient (S) factors. The L factor is the ratio of soil loss from the field slope length of the area in question to that from a 72.6-ft slope length under identical conditions. The S factor is the ratio of soil loss from a field slope gradient to that from a 9% slope under otherwise identical conditions. Topographic factors are presented in Figure 11. In using the figure, identify a field-measured length of slope on the horizontal axis; move vertically to intercept the appropriate percent slope measured in the field; then read the LS value on scale at the left. For example, the LS value for a 200-ft-long slope with a 14% gradient is about three. The LS values derived in this manner are appropriate only for uniform slopes.

The cover factors (C) of the USLE represent the effects of vegetative cover (including crop residues) and land-management variables such as those associated with agricultural cropping practices. However, site preparation and subsequent development of construction sites usually results in the removal of all vegetation. In the event that all aboveground vegetation and plant roots are removed, as in the case of an unrevegetated waste-storage pile, C for the denuded area will be equal to one. Numerous measures can be initiated to reduce the C value, including applications of various types of mulch. Some examples of the effects of mulching are illustrated in Table 18.

The support practice factor (P) of the USLE is the ratio of soil loss with a support practice (contouring, stripcropping, or terracing) to that with straight-row farming up and down the slope. P factors for open waste dumps will usually be equal to one, and thus will not affect estimates based on other USLE factors. P factors for managed waste dumps can be less. Contour terracing could be used to alter slope characteristics, but the erosion-reducing effects due to terracing would be accounted for in the determinations of LS values.

In conclusion, the USLE can be used for approximating erosion losses from waste-storage sites. Further, the calculations provide a general guide for indicating the capacity of control structures necessary to control sediment depositions in adjacent areas and local water bodies.

Regional variation in the magnitude of impacts from runoff of ash and sludge wastes will mainly be a function of climatic and soil characteristics. Brief intense rainfalls, sparse vegetative cover, low soil infiltration capacity, and hilly topography will promote erosion.

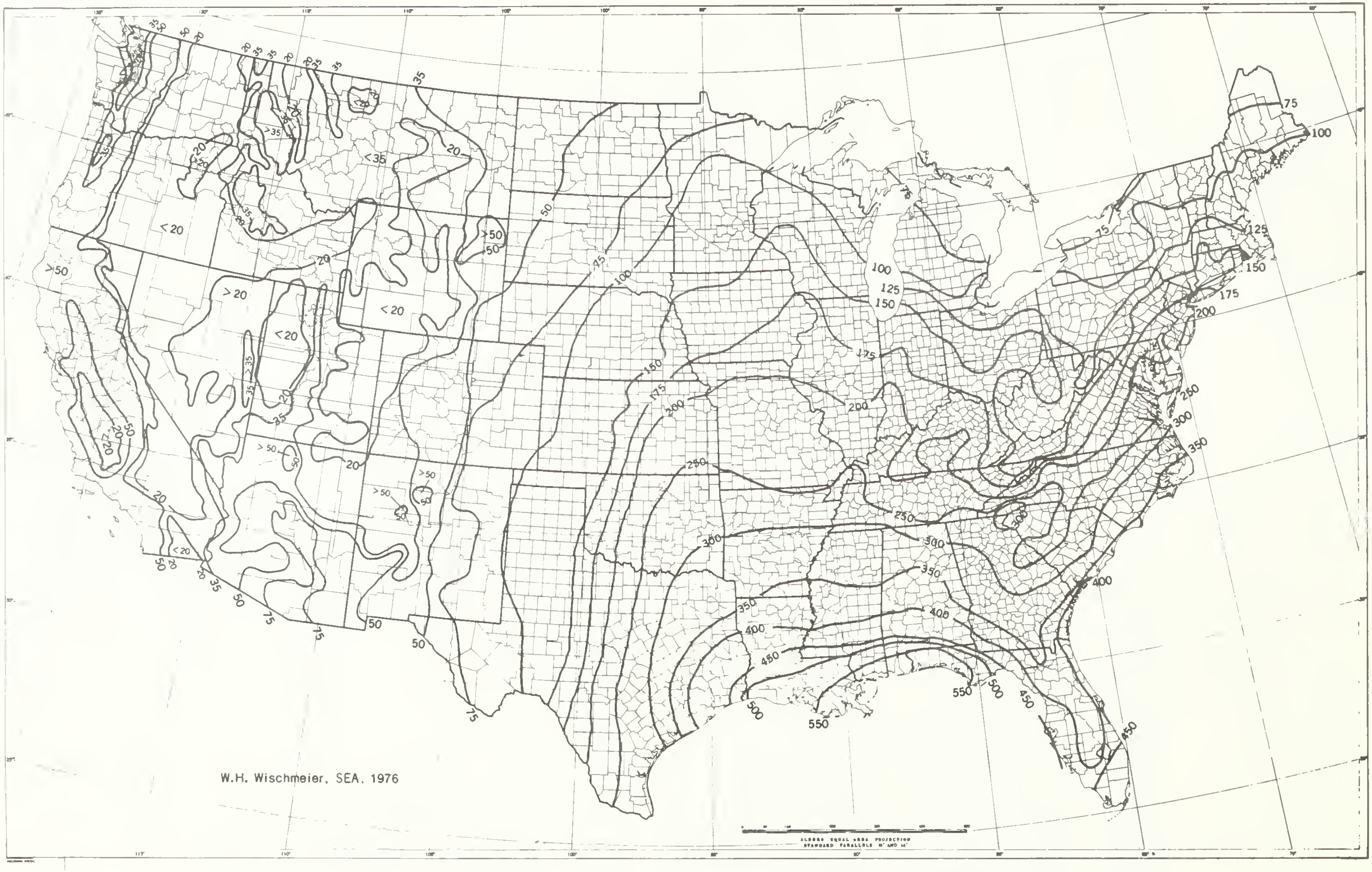
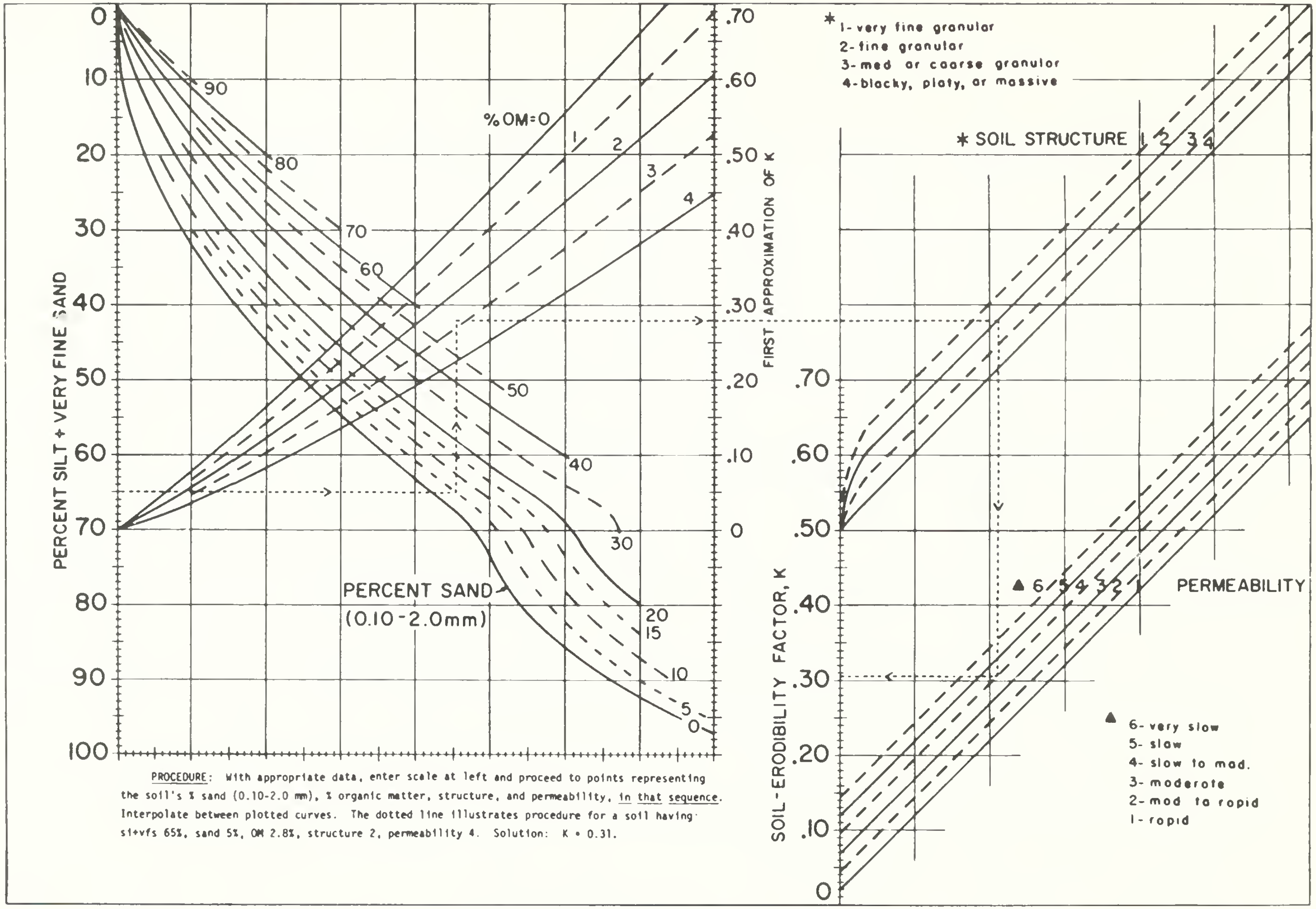


Figure 9. Average Annual Values of the Rainfall Erosion Index. From Wischmeier and Smith (1978).



PROCEDURE: With appropriate data, enter scale at left and proceed to points representing the soil's % sand (0.10-2.0 mm), % organic matter, structure, and permeability, in that sequence. Interpolate between plotted curves. The dotted line illustrates procedure for a soil having si+vfs 65%, sand 5%, OM 2.8%, structure 2, permeability 4. Solution: K = 0.31.

Figure 10. The Soil-Erodibility Nomograph. Where the silt fraction does not exceed 70%, the equation is 100 K = 2.1 $M^{1.14}$ (10^{-4}) (12 - a) + 3.25 (b - 2) + 2.5 (c - 3) where M = (percent si + vfs) (100 - percent c), a = percent organic matter, b = structure code, and c = profile permeability class. From Wischmeier and Smith (1978).

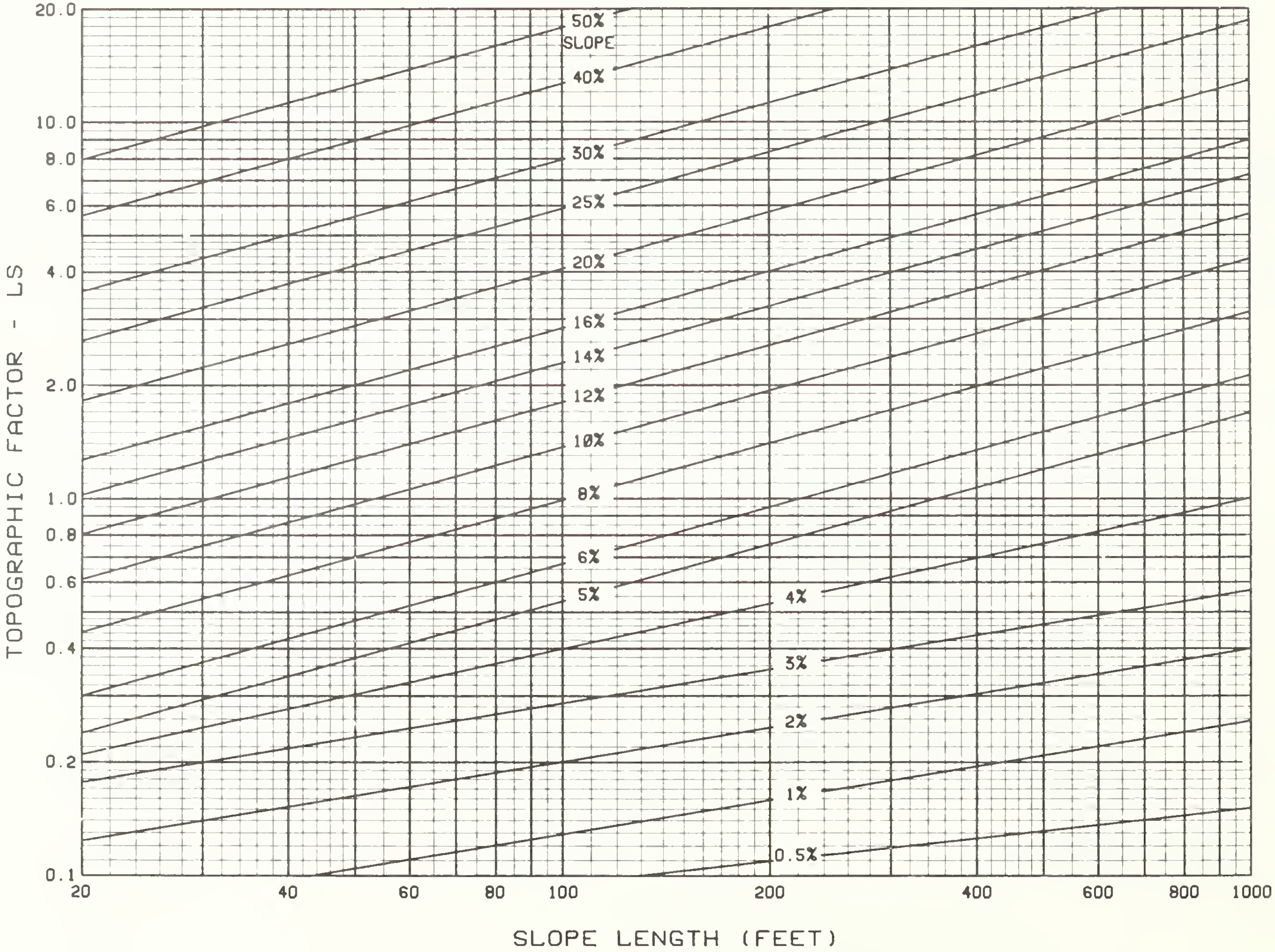


Figure 11. Slope Effect Chart. From Wischmeier and Smith (1978).

Table 18. Mulch Factors and Length Limits for Construction Slopes[a]

Type of mulch	Mulch rate (MT/ha)	Land slope (%)	Factor C	Length limit[b] (m)
None	0	all	1.0	-
Straw or hay (tied down by anchoring and tacking equipment)[c]	2.2	1-5	0.20	60
	2.2	6-10	0.20	30
	3.3	1-5	0.12	90
	3.3	6-10	0.12	45
	4.4	1-5	0.06	120
	4.4	6-10	0.06	60
	4.4	11-15	0.07	45
	4.4	16-20	0.11	30
	4.4	21-25	0.14	22
	4.4	26-33	0.17	15
	4.4	34-50	0.20	10
Crushed stone (¼ to 1½ inches)	297.	<16	0.05	60
	297.	16-20	0.05	45
	297.	21-33	0.05	30
	297.	34-50	0.05	22
	528.	<21	0.02	90
	528.	21-33	0.02	60
	528.	34-50	0.02	45
Wood chips	15.	<16	0.08	22
	15.	16-20	0.08	15
	26.	<16	0.05	45
	26.	16-20	0.05	30
	26.	21-33	0.05	22
	55.	<16	0.02	60
	55.	16-20	0.02	45
	55.	21-33	0.02	30
	55.	34-50	0.02	22

[a] Adapted from Meyer and Parts (1976). Originally developed by an interagency workshop group on the basis of field experience and limited research data.

[b] Maximum slope length for which the specified mulch rate is considered effective. When this limit is exceeded, either a higher application rate or mechanical shortening of the effective slope length is required.

[c] When the straw or hay mulch is not anchored to the soil, C values on moderate or steep slopes of soils having K values greater than 0.30 should be taken at double the values given in this table.

Soils that contain a high proportion of clay will have low infiltration capacities which will enhance erosion. Climatic and soil characteristics in the Northeast and Southeast are expected to give rise to moderate erosion and leaching problems. The soils in these regions have a moderate to high clay and organic matter content, the degree of slope is variable from low to moderate, and, although annual precipitation levels are high, the intensity of rainfall tends to be low. In the Midwest, erosion problems are generally low because of the high clay and organic matter content, flat topography, and moderate annual rainfalls of moderate intensity. However, cropland management of these soils may enhance erosion. The greatest likelihood of erosion is in the Great Plains Region, particularly the southern plains, because of the low to moderate clay and organic matter content, sparse vegetation cover, low to high degree of slope, and intense rainfalls.

Seepage and movement of waste constituents. Seepage of water through the wastes can leach potentially toxic substances from ash and FGD sludge into the adjacent soil. Both vertical and lateral seepage of leachate can occur from ash and sludge waste-storage sites, particularly where the waste material is deposited as a slurry. The major impact of seepage is addition of potentially toxic waste constituents to groundwater and soil. Release of these contaminants can result in eventual contamination of fish and wildlife water resources. Quantitative estimates of leachate quantity, composition, and movement (seepage) at ash and sludge waste-storage sites are difficult to obtain. Quantity, composition, and seepage of leachate are influenced by the physicochemical properties of the wastes and surrounding substrates, climatic conditions, and storage-site design and management practices that are site-specific. Leachate formation and movement are influenced by (1) nature of the wastes, (2) leachate pH, (3) physicochemical properties of soil, (4) permeability of the waste materials and storage-site substrates, and (5) rainfall zone (Dvorak et al. 1978; Duvel et al. 1979).

Nature of the wastes. When ash and sludge wastes are deposited dry, or with no standing water, there will be very little or no seepage of waste liquors (Dvorak et al. 1978). Leaching by rainfall may occur. If the waste is deposited as a slurry, chances of contamination of both groundwater and the food chain are higher, unless the impoundment area is lined or impervious layers are present. The presence of the slurry liquor accelerates the percolation of leachate through the waste material because maintenance of a hydraulic head in an impoundment exacerbates the seepage problem.

Leachate pH. The pH of the waste liquors and leachate also determines the mobility of certain elements (Figure 12). Many trace elements are insoluble under alkaline conditions--e.g., Al, Be, Cd, Cr, Co, Cu, and Zn (Brady 1974). Therefore, these elements may be less soluble in alkaline ash-pond liquors (pH 6-11) than in the more acidic scrubber sludge liquors (Cooper 1975), and leachates percolating to the soils below alkaline ash storage areas would not contain large concentrations of these elements. However, these elements are more likely to reach the soils under the more acidic FGD sludge wastes. Arsenic, boron, molybdenum, and selenium are soluble in alkaline solutions and would likely be more mobile in the soil solution below ash disposal wastes; molybdenum and selenium are insoluble in acid leachates and would be less likely to reach the soil solution below scrubber-sludge wastes. Factors besides pH affect the mobility and availability to plants of a given element, but, in general, trace-element-toxicity effects should be of more concern when the absorbing medium (soil) and transporting medium (pond leachates) are acidic than if they are neutral to alkaline.

Physicochemical properties of soil. At most waste-storage sites, it can be assumed that the waste will be in physical contact with the surrounding soils unless the impoundment is lined (Dvorak et al. 1978). The physical properties of a soil that influence movement of leachate laterally and vertically from a waste-storage site include textural grade (proportion of sand, silt, and clay) and bulk density (the weight of a given volume of soil)--properties that affect hydraulic conductivity. The textural names used in soil science to designate the proportions of sand, silt, and clay size classes are given in Table 19. "Sand" refers to particle sizes between 2 and 0.05 mm diameter, "silt" between 0.05 and 0.002 mm diameter, and "clay" less than 0.002 mm diameter [USDA Classification (Brady 1974)]. The size range of "sand" is further classified as follows: very coarse sand, 2.0-1.0 mm; coarse sand, 1.0-0.5 mm; medium sand, 0.5-0.25 mm; fine sand, 0.25-0.10 mm; very fine sand, 0.10-0.05 mm. In general, bulk density decreases and hydraulic conductivity increases as one proceeds from clays through gravels, i.e., as particle size increases (Figure 13).

The chemical properties of a soil that influence the transport of solutes include ion-adsorption capacity, which in turn is influenced by the type of clay mineral present and the quantity and quality of organic matter present. A given soil has a finite capacity to adsorb cations and very little or no capacity to adsorb anions. The elements Al, Ba, Be, Cd, Cr, Co, Cu, Pb, Mn, Mi, Sn, V, and Zn occur most often as cations in the soil (Romney and Childress 1965; Pratt 1966; Berry and Wallace 1974; USEPA 1975b). Mobility and availability of these cations is controlled in part by the cation exchange capacity (CEC) of the soil (total of the exchangeable cations that a soil can adsorb)--which is a function of soil texture, organic matter content, the amount and kind of clay present, and, ultimately, by the amount of surface area of the soil colloids (Table 20). Generally, the higher the CEC, the greater the soil's ability to adsorb cations from coal ash or FGD-sludge leachates. In general, the higher the clay content and organic matter of a soil, the greater is its cation exchange capacity. The particular clay mineral in a given soil also influences its cation exchange capacity, i.e., soils with a higher proportion of montmorillonitic clays have higher cation exchange capacities than those with the same amount of kaolinitic clays. The clay fraction present in a given soil usually consists of a mixture of clay minerals; the types of minerals present are primarily dependent on climatic conditions and the nature of the parent material from which the soil was formed. Very generally, the greater the weathering process, the higher will be the proportion of hydrous oxides and kaolinitic clay in the soil. Parent materials high in bases, e.g., magnesium, tend to produce montmorillonitic clays. Again very generally, montmorillonites and illites tend to predominate in the soils of the semiarid and arid United States, whereas kaolinitic clays and hydrous oxides tend to predominate in the humid East or South (cf. Brady 1974).

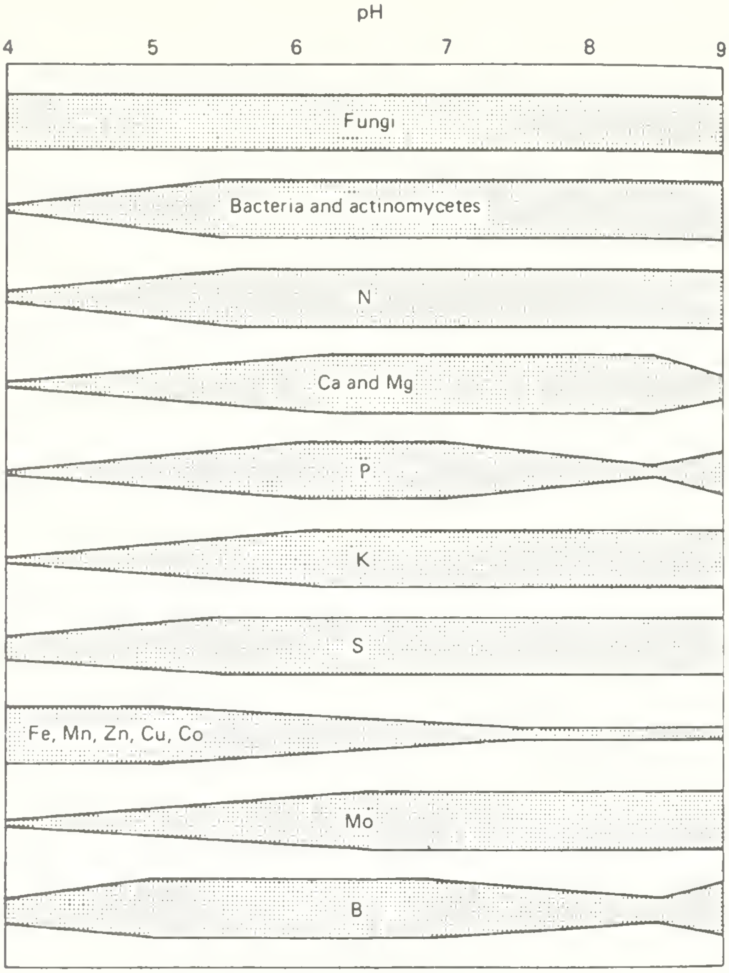


Figure 12. Relationships Existing in Mineral Soils Between pH on the One Hand and the Activity of Microorganisms and the Availability of Plant Nutrients on the Other. The wide portions of the bands indicate the zones of greatest microbial activity and the most ready availability of nutrients. From Brady (1974) (with permission, see credits).

The ion-adsorption ability of a soil can be misleading, since trace elements that are bound by a soil today can accumulate until the soil's adsorption capacity is exceeded. These ions may then be released and percolate to the lower soil horizons, perhaps reaching groundwater. In the short term, a high CEC may protect lower soil horizons and groundwater in the area; but in the long term, retention of these potentially toxic substances in the rooting zone could result in toxic effects to biota (Dvorak et al. 1978).

One of the most important factors influencing the effects of trace elements added to the soils is the endogenous concentrations of elements (Dvorak et al. 1978). Regardless of the source, the addition of trace elements to soils from coal ash and FGD sludge effluents will have a greater impact in areas where endogenous trace-element concentrations are already close to the tolerance limit for any of the biota living there than in areas of lower endogenous concentrations. Conversely, areas suffering from deficiencies of essential trace elements (e.g., Cu, Mo, B, Zn, and Mn) may benefit from the addition of these elements to the soil as a result of coal combustion. Dvorak et al. (1978) present a discussion of the regional distribution of trace elements in U.S. soils, and Table 21 presents a generalized summary of endogenous concentrations for selected elements in worldwide soils.

Table 19. Percentages of Sand, Silt, and Clay in the Textural Classes[a]

Textural name (Soil class)	Range in percent		
	Sand	Silt	Clay
Sand[b]	85-100	0-15	0-10
Loamy sand[b]	70-90	0-30	0-15
Sandy loam[b]	43-80	0-50	0-20
Loam	23-52	28-50	7-27
Silt loam	0-50	50-88	0-27
Silt	0-20	8-10	0-12
Sandy clay loam	45-80	0-28	20-35
Clay loam	20-45	15-53	27-40
Silty clay loam	0-20	40-73	27-40
Sandy clay	45-65	0-20	35-55
Silty clay	0-20	40-60	40-60
Clay	0-45	0-40	40-100

[a] From U.S. Department of Agriculture (1961).

[b] These textural names can be modified by the following designations: coarse = greater than 25% coarse sand; fine = 50% or more fine sand, less than 25% coarse sand; very fine = 50% or more very fine sand.

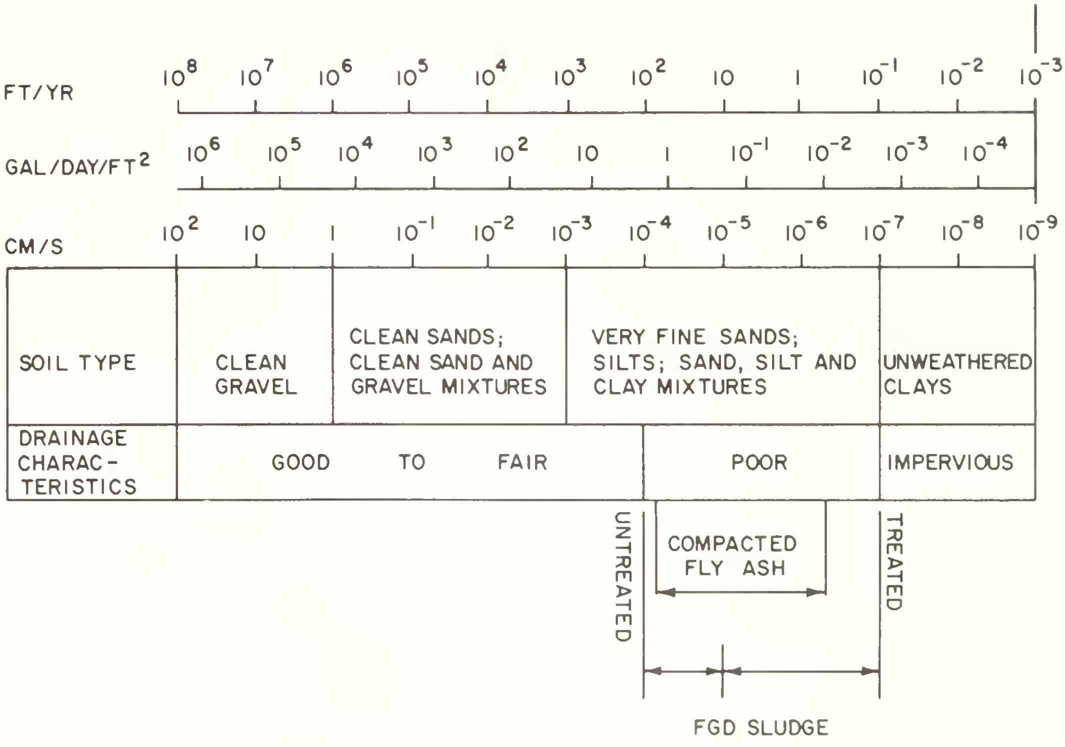


Figure 13. Saturated Hydraulic Conductivities for Different Soil Types at Unit Gradients. Modified from Duvel et al. (1979); compacted fly ash data based on Frascino and Vail (1976).

Table 20. Factors Affecting Soil Cation Exchange Capacity (CEC)[a]

Soil factor	Relative CEC
Texture	
Sand	Low
Loam	Moderate
Clay	High
Organic content	
Low	Low
High	High
Clay type	
Hydrous oxides	Low (4 meq/100 g)
Kaolinite	Low (8 meq/100 g)
Chlorite	Low (30 meq/100 g)
Hydrous micas	Low (30 meq/100 g)
Montmorillonite	High (100 meq/100 g)
Vermiculite	High (150 meq/100 g)

[a] Based on data in Brady (1974).

Table 21. Worldwide Total Endogenous Soil Concentrations of Selected Elements[a]

Element	Soil range (μg/g)	Average soil concentration (μg/g)	Reference
Arsenic	0.1 - 40	6.0	Allaway (1968)
Barium	100 - 3000	500	Swaine (1955); Bowen (1966)
Beryllium	1 - 40	6.0	Allaway (1968)
Boron	2 - 100	10.0	Allaway (1968)
Cadmium	0.01 - 7.0	0.06	Allaway (1968)
Chromium	5 - 3000	100	Allaway (1968)
Cobalt	1 - 40	8	Allaway (1968)
Copper	2 - 100	20	Allaway (1968)
Fluorine	30 - 300	200	Allaway (1968)
Lead	2 - 100	10	Allaway (1968)
Manganese	100 - 4000	850	Allaway (1968)
Mercury	0.01 - 4(?)	-	Lisk (1972)
Molybdenum	0.2 - 5	2	Allaway (1968)
Nickel	10 - 1000	40	Allaway (1968)
Selenium	0.01 - 80	0.5	Trelease (1945)
Vanadium	20 - 500	100	Allaway (1968)
Zinc	10 - 300	50	Allaway (1968)

[a] Compiled by Dvorak et al. (1978).

Permeability of waste materials and storage-site substrates. Where the storage area is not excavated to bedrock, the underlying unconsolidated material is the natural soil (Dvorak et al. 1978). In cases where the impoundment is excavated to bedrock, the material forming the sides of the impoundment is usually surface soil and subsoil. In either case, unless the impoundment basin is completely lined (e.g., with clay or a synthetic liner), lateral seepage through the soil can occur, particularly if the waste is deposited as a slurry.

Permeability of ash and sludge wastes and storage-site substrates is one of the most important parameters in determining leachate quantities seeping from storage sites (Duvel et al. 1979). Hydraulic conductivities of compacted fly ash vary between 8×10^{-5} and 5×10^{-7} cm/s. Conductivities of FGD sludges vary between 1×10^{-4} and 1×10^{-7} cm/s. Soil hydraulic conductivities are generally greater, ranging from 30 cm/s for clean gravels to 1×10^{-9} cm/s for some clays. Permeabilities for natural soils and ash and sludge wastes are presented in Figure 13. Stratification of soils and wastes can also markedly affect permeability by creating layers of differing compaction. Other factors affecting permeability are density, trapped air pockets, and dissolved salt content of the leachate--all of which are inversely correlated with bulk water movement (Duvel et al. 1979).

Contamination of groundwater is related to the permeability of the impoundment material; in general, the permeability of such material increases in the order: granite < shale < sandstone < soil < sand. In regions where the rainfall is low (i.e., less than 25 cm/yr), seepage to groundwater will be unlikely in areas underlain by all but the most permeable material unless the water table is high (e.g., less than 5 m below the surface). In areas of high water table, installation of a liner--particularly if the impoundment is underlain by sandstone, soil, or sand--would be essential to preclude contamination of the groundwater, even if the waste were deposited in a dry state.

Hughes and Cartwright (1972) have indicated that a well-compacted earth impoundment bottom has a hydraulic conductivity of about 1×10^{-6} cm/s. Addition of soil cements during compaction can reduce hydraulic conductivity several orders of magnitude below 1×10^{-6} cm/s (Williams 1975). Impoundments lined with clay could have hydraulic conductivities on the order of 10^{-8} cm/s (Figure 13). Synthetic liners such as plastic or rubber are reputed to have essentially zero permeabilities, but the durability of these liners has not been proven (Dvorak et al. 1978).

Rainfall zone. The amount of rainfall entering a waste-storage site and its environs markedly affects the potential for adverse effects from the waste at sites where the waste-storage impoundments are not lined (Dvorak et al. 1978). If the average annual rainfall is low, seepage from the waste-storage site will tend to remain in the upper layers of the soil, thus increasing the chances for uptake by vegetation; however, seepage to groundwater will be low, depending on the depth at which the water table occurs. In zones of high rainfall, ionic constituents of waste will tend to be leached rapidly to groundwater, particularly where the substrate is sandy or otherwise relatively permeable. High rainfall will also tend to move dissolved materials laterally into the soil. In sandy soils, these dissolved materials would tend to be leached from plant root zones, whereas in soils with higher proportions of clay, a larger fraction of the dissolved constituents would be retained in the soils and might eventually be available for plant uptake.

Average annual and net precipitation (difference in inches between precipitation and evaporation) tend to increase west to east across the United States, excluding coastal areas (Figures 14 and 15). Average net precipitation values in the eastern United States are positive, whereas in most of the western United States values are negative. In general, leachate quantities are likely to be greater in the eastern part of the country.

Predicting quantities of leachate seepage. Accuracy of estimating leachate seepage depends on accuracy of permeability estimates. Extensive field and laboratory testing are required to accurately determine permeability of ash and sludge wastes and storage-site substrates. However, one can obtain reasonable estimates of leachate production.

Landfill leachate production. Order-of-magnitude estimates of leachate quantities from landfill storage sites are obtained by assuming an overall infiltration rate, with overall hydraulic conductivity (k) of the waste or substrate being a limiting condition (Figure 16) (Duvel et al. 1979); either the overall infiltration rate or the overall permeability of the site can limit the leachate quantity. A net infiltration rate of 20% of the precipitation is a reasonable estimate for many situations. However, when site-specific measurements are available, 30%, 40%, and 50% net infiltration may be more appropriate. If the average rainfall is 30 in./yr, with 20% infiltration and $k = 5 \times 10^{-7}$ cm/s, the leachate seepage rate from the storage site would be 450 gal/acre/day. However, if $k = 2.5 \times 10^{-7}$ cm/s, the leachate seepage from the storage site would be 225 gal/acre/day. By selecting the appropriate lining materials and proper compaction of the fill, permeability and infiltration can be adjusted at a landfill site to achieve a desired leachate seepage rate.

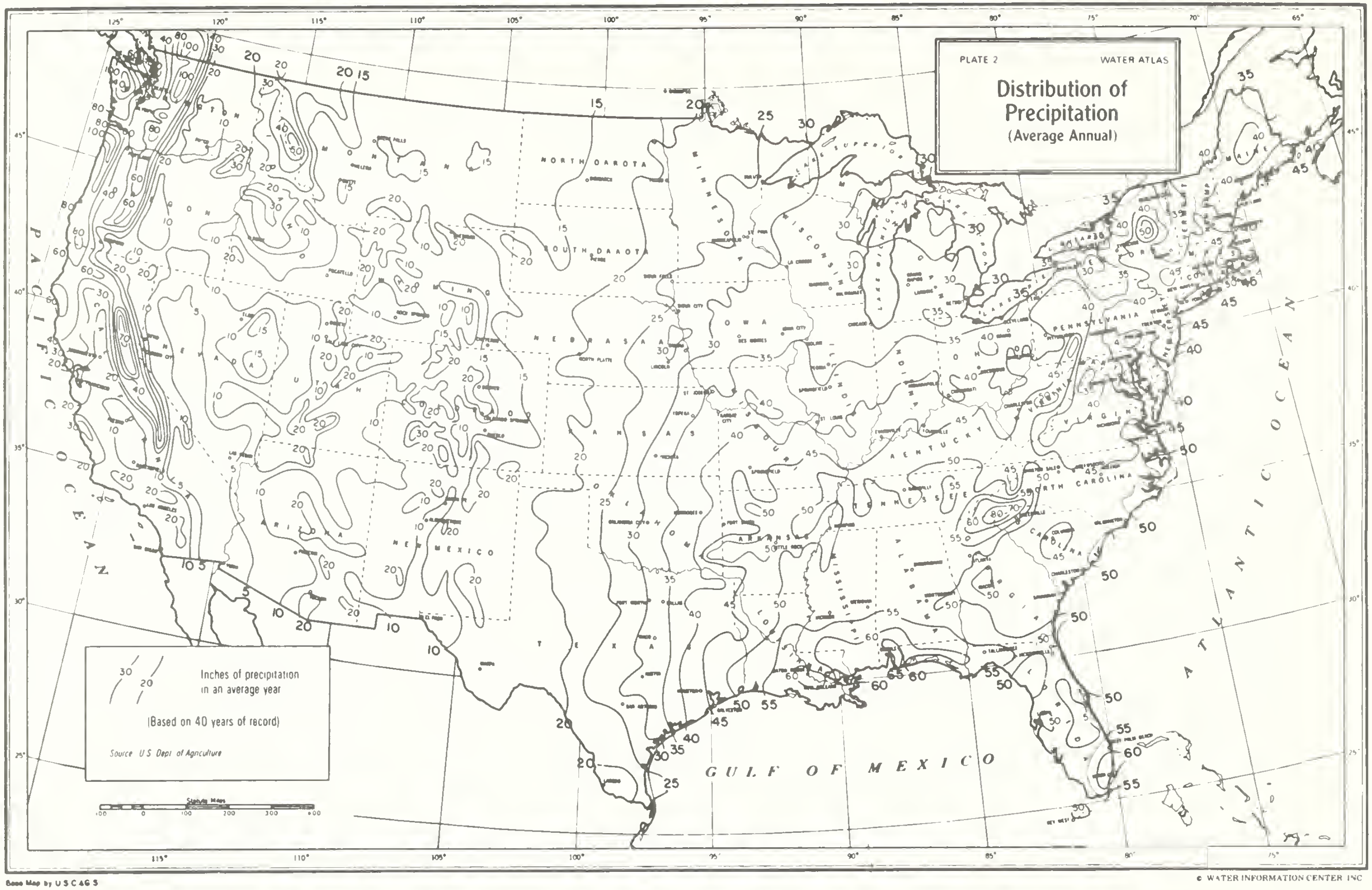


Figure 14. Average Annual Precipitation in the United States. From Geraghty et al. (1973) (with permission, see credits).

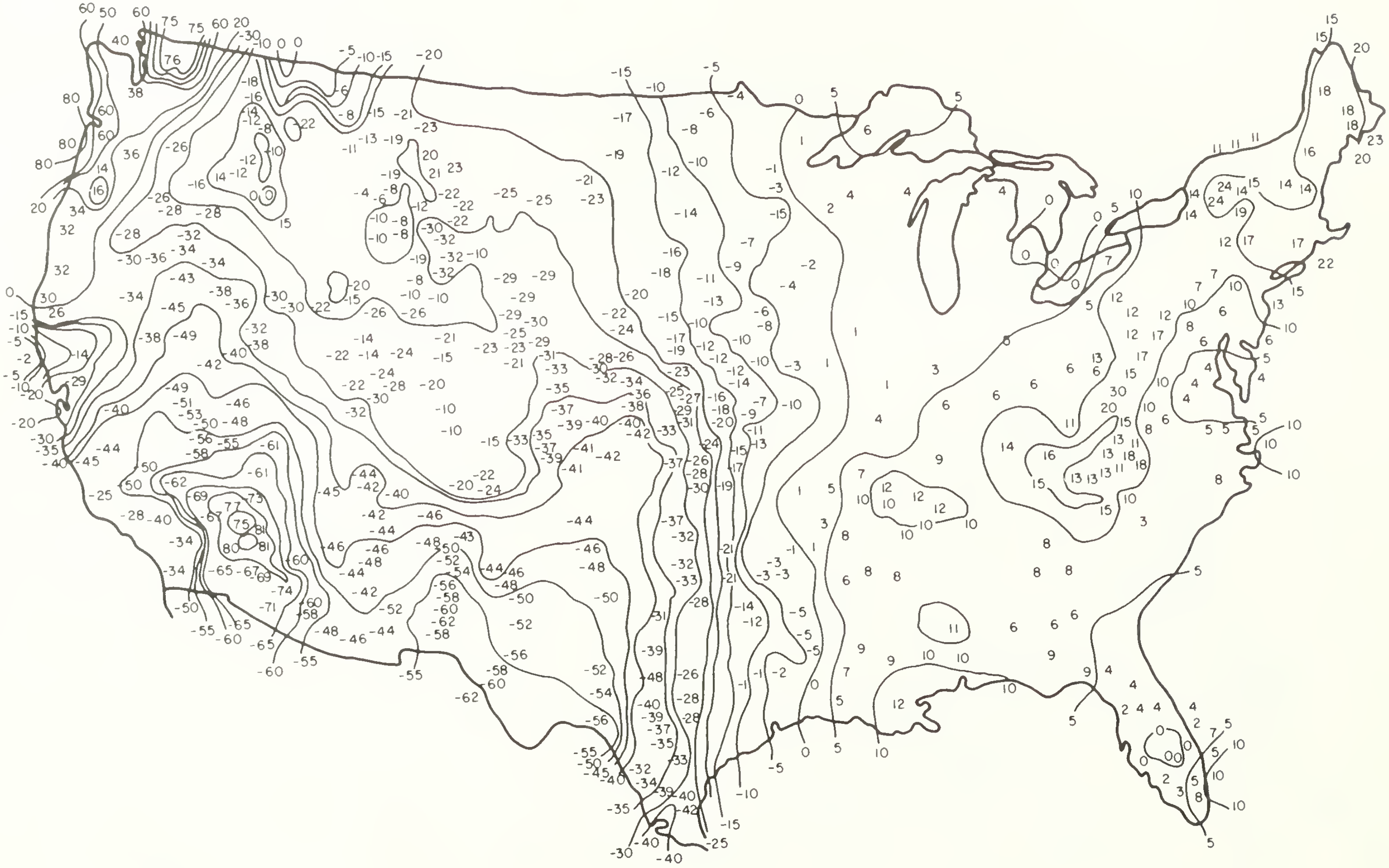


Figure 15. Average Net Precipitation in the United States. Figures represent difference in inches between precipitation and evaporation. From Duvel et al. (1979).

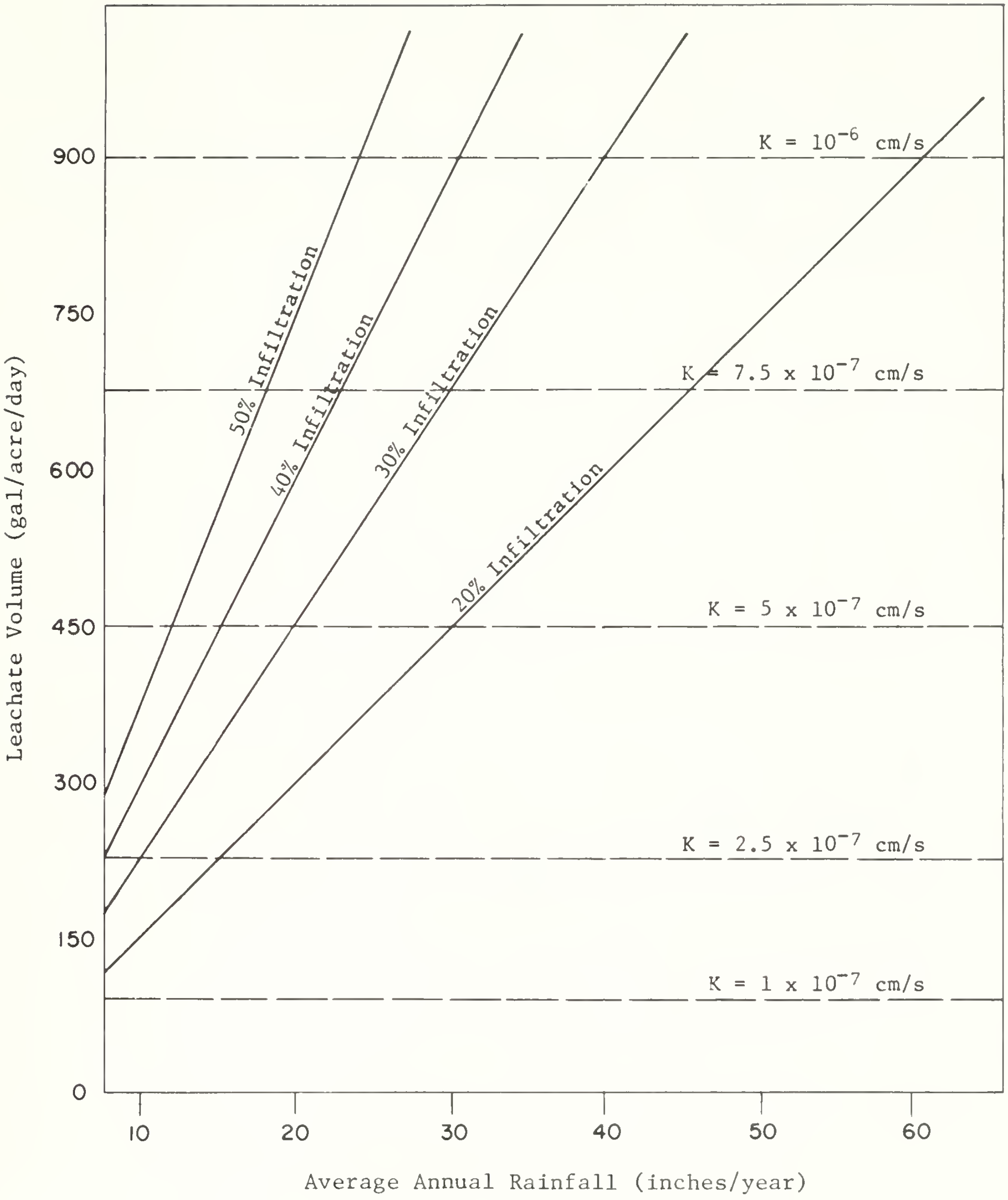


Figure 16. Quantity of Leachate from a Landfill. From Duvel et al. (1979).

Pond leachate production. The quantity of seepage from a pond storage system is influenced by permeability of the wastes and substrate, dimensions and configuration of the pond, and boundary conditions of the entire system. Unlike landfill sites, supernatant is present as a recharge source for leachate generation. Figure 17 can be used to obtain an approximate estimate of seepage quantities if (1) the substrate beneath the pond is more permeable than the wastes, (2) the depth to any impervious stratum is much greater than pond depth, (3) the depth of supernatant is small compared to sludge depth, and (4) there are no complex subsurface conditions. For example, if the hydraulic conductivity of the sludge is 10^{-5} cm/s, the volume of leachate generated is 9000 gal/acre/day. If substrate permeability is less than waste permeability or if depth to an impervious layer beneath the pond is not much greater than pond depth, the seepage quantities will be less than predicted by using Figure 17 (Duvel et al. 1979). If the depth of pond supernatant is large, seepage quantities will be increased; depth of pond supernatant is dependent on net precipitation (Figure 15) and storage practices. A method of estimating seepage quantities for cases with more complex boundary conditions than those assumed in Figure 17 has been developed by Witherspoon and Narasimhan (1973).

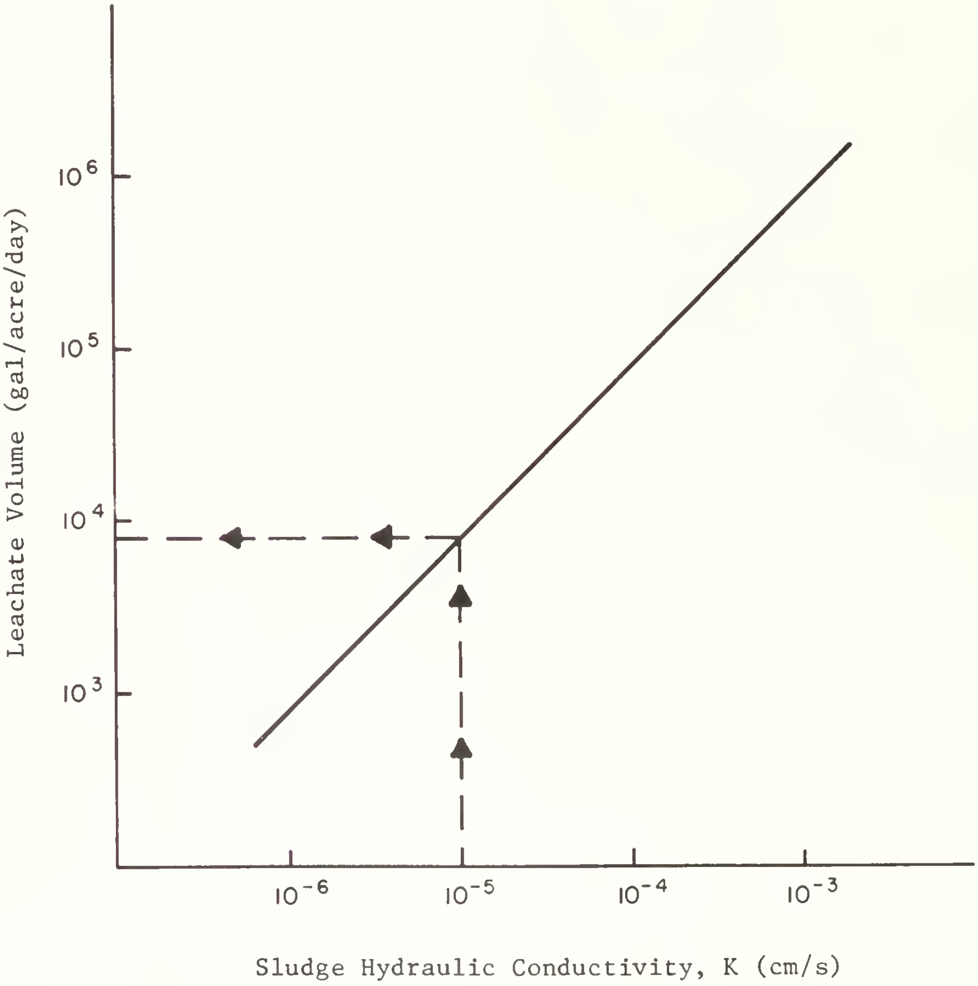


Figure 17. Effect of Sludge Hydraulic Conductivity on the Volume of Leachate from a Pond. From Duvel et al. (1979).

The problem of calculating the flow path and flow rate of leachate from a pond is complex and extremely site-specific (Dvorak et al. 1978). Models can be formulated which, with the use of a digital computer, can be used to simulate the flow of leachate from a given pond under given conditions. A rough estimate of pond seepage and thus the liner thickness required for "total attenuation" of leachate during active pond use can be predicted using Darcy's equation (Cartwright et al. 1977):

$$Q = kA \frac{dH}{dL} \qquad (29)$$

The variables in Darcy's equation are: Q = flow rate in cm^3/s, k = hydraulic conductivity in cm/s, A = area of storage site in cm^2, dL = thickness of liner in cm, dH = head of water across the liner in cm. The hydraulic conductivity, k, can be estimated by laboratory determinations on undisturbed substrate core samples.

Holland et al. (1975) conducted laboratory tests with ash and scrubber sludge leachates through soil samples in cylindrical tubes. The characteristics of the soils used are given in Table 22. Leachates and soils from five electric generating stations, as well as soils of varying clay content from other sites, were tested. Soil columns of 15 cm length and 3.3 cm inside diameter were used; the soils were disturbed and homogenized. The results indicated that a clay soil, with an average permeability of 7.4×10^{-6} cm/s, removed selenium and chromium from the leachate at a rate equivalent to the removal of over 95% of these elements after 10 years of continuous flow through a 12-m (40-ft) soil depth. A sandy soil did not remove selenium and chromium to the same level as the clay soil, but it was concluded that 15 m (50 ft) of the sandy soil would remove over 95% of copper, arsenic, or zinc after 10 years of flow. The vertical depth required of a given soil to reduce the concentration of certain trace elements to less than 5% of the pond concentration is given in Figure 18 and Table 23. Table 23 was prepared with values derived from results of laboratory tests. Extrapolation from these data to the assessment of field behavior should be done with caution, and may not be applicable in many cases. However, the study does provide data indicating that the following trace elements should be of concern regarding contamination of groundwater: selenium, boron, chromium, barium, and mercury. With regard to contamination of soils and the food chain, these and other elements adsorbed by the soil should be of concern (Dvorak et al. 1978).

The pond leachates that move through a particular soil will displace ions that are already present in the soil, and move them downward with the seepage flow. Most soils are relatively low in trace elements (except for iron and aluminum), but there are areas, e.g., the arid West and Southwest, that have abnormally high concentrations of selenium, molybdenum, or boron (see Dvorak et al. 1978). In such areas, which can be identified most often by consultation with the district Soil Conservation Service, attention should be given to the displacement of these elements by leachate constituents.

Potential for adverse effects to groundwater from seepage from unlined ash and sludge waste-storage sites. In summary, the seepage and transport of potentially toxic ions from ash and sludge waste-storage sites is a complex process, the magnitude of which is site-specific (Dvorak et al. 1978). It will be difficult to obtain quantitative estimates of seepage concentrations and patterns of flow without input from hydrologists. A qualitative assessment will be useful in siting and pond-construction decisions. The following discussion includes a table from which the potential severity of a seepage problem at a given storage site can be determined.

Currently, all soils in the United States are being evaluated for their limitations for waste storage. These ratings for the limitations can be found in published Soil Survey Reports for various counties. Unfortunately, completed Soil Survey Reports have not been published for most counties in a large number of states. Other reports are of older vintage when such ratings were not routinely made. Inquiries to the district Soil Conservation Service regarding published and unpublished data on the soils of a specific site should be an initial step in any assessment of land-storage impacts. Soil Survey Reports also contain a wealth of information on the vegetation, wildlife, and land use of a given county, as well as data on the geology, soils, climate, topography, and depths to the seasonally high water table.

In the absence of soil survey information, Table 24 can provide some measure of the potential for impact from seepage, given some general information about a particular site and assuming that the impoundment area is unlined. (Lining of an impoundment area implies, here, the correct placement of clay or a synthetic material on the bottom and sides of the impoundment.) It must be emphasized that Table 24 does not include all factors involved in the evaluation of seepage effects; rather, it lists the major factors related to specific site characteristics that could be readily evaluated. It is essential to realize that the factors must be considered concurrently, rather than as separate entities.

Table 22. Characteristics of Soils Used in the Leachate Experiments of Holland et al. (1975)

Soil designation	Textural classification[a] (grain-size analysis)	Mineralogy of < 2-mμ (clay) fraction[b]	Specific gravity[c]	Average[d] TOC (ppm)	Average[e] CEC (meq/100 g)	Average permeability[f] (cm/s)
Station 1	Sand	70% illite, 20% montmorillonite, 10% kaolinite	2.58	2,540	3.8	5.1×10^{-4}
Station 2	Clay	90% illite, 5% montmorillonite, 5% kaolinite	2.43	14,390	N.D.[g]	5×10^{-8}
Station 3	Clay	90% kaolinite, 10% montmorillonite	2.76	1,270	31	7.4×10^{-6}
Station 4	Silt loam-silty clay loam	60% illite, 20% montmorillonite, 20% kaolinite	2.30	72,700	30	1.2×10^{-5}
Station 5	Loam-clay loam	60% illite, 30% montmorillonite, 10% kaolinite	2.45	7,710	21	2.1×10^{-5}

[a] Dry-sieving, pipetting, and centrifugation.

[b] X-ray diffraction analysis, 1 μm = 10^{-6} m.

[c] Pycnometer determination with tetrachloromethane.

[d] Oxidation to CO_2 by persulfate and wet combustion analyzer (TOC = Total organic carbon).

[e] 1 N NH_4OAc displacing NA^+ at pH = 8.2. (CEC = Cation exchange capacity in units of milliequivalents of cation per 100 g of soil.)

[f] Disturbed, homogenized soil samples, and using both ash and sludge leachate as the fluid.

[g] N.D. = Not determined.

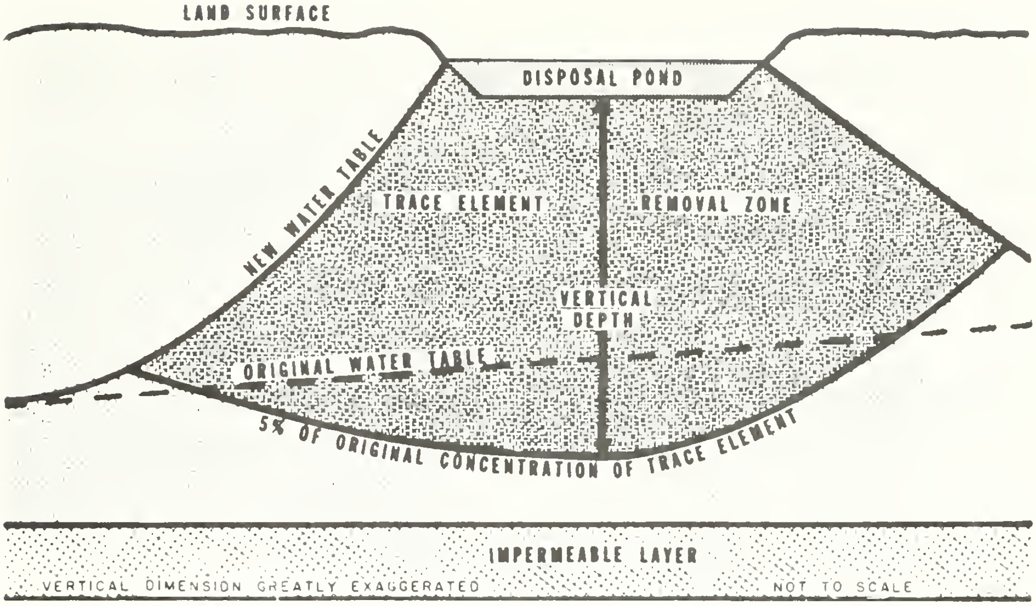


Figure 18. Removal of Trace Elements from Pond Leachate as a Function of Time. Characteristics of the soils at these stations are given in Table 22. The tables for determining approximate depth required for reduction of trace-element levels to less than 5% of the pond concentration are given in Table 23. From Holland et al. (1975).

Table 23. Tables for Determining Approximate Vertical Depth Required for Reduction of Trace-Element Levels to Less than 5% of the Pond Concentration[a]

In soils similar to station no.[b]	Depth (m) required - ash leachate					Depth (m) required - sludge leachate				
	Arsenic	Chromium	Copper	Fluorine	Selenium	Chromium	Copper	Fluorine	Mercury	Zinc
				After two years						
1	<4.9	60	<4.9	610	37	60	<0.9	60	27	0.9
3	<2.7	<2.7	<2.7	>15	3	18	<1.2	10	<1.2	<1.2
4	<1.2	2.1	<1.2	21	5.2	17	<0.9	24	<1.5	<0.9
5	<2.1	>30	N.D.	<29	4.6	29	<0.9	29	<1.2	<1.2
				After ten years						
1	<15	215	<15	3,050	150	305	3	305	130	4.9
3	<10	10	<10	>85	11	88	<5.8	49	<5.8	<5.8
4	<3.4	7	<3.4	>110	19	67	<3	94	<5.5	<3
5	<7	>130	N.D.	120	16	170	<3	170	3.7	3.7
				After 100 years						
1	<150	2,285	<150	30,480	1,555	305	27	3,050	1,220	37
3	<79	79	<79	>855	85	855	<49	425	<49	<49
4	<23	55	<23	> 1	170	670	<19	945	<43	<19
5	<58	1,280	N.D.	1,130	70	1,280	24	1,280	26	2.6

[a] Data derived from Holland et al. (1975). This table is to be used with Figure 18. Table values assume depth to lower boundary of removal zone << depth to impermeable layer. N.D. = Not determinable.

[b] Effective porosities of 10% were assumed for all soils except soil from Station No. 1 which was assumed to be 50% due to its high sand content. Soil from Station No. 2 was very clayey, permitting very little flow and offering virtually complete protection.

Table 24. Potential for Adverse Effects to Groundwater from Seepage from Unlined Ash and Sludge Waste-Storage Sites[a]

Factor	Relative probability of groundwater contamination
Nature of waste	
Dry	Low to moderate
Slurry	High
Acid	High
Alkaline	Low to moderate
Nature of substrata[b]	
Granite	Extremely low
Shale	Low
Sandstone	Moderate
Soil	
Clays	Low
Loams	High
Sands, sandy loams	Very high
Rainfall zone[c]	
< 25 cm (< 10 inches)	Low
25-76 cm (10-30 inches)	Low to high
> 76 cm (> 30 inches)	High

[a] Derived from Dvorak et al. (1978).

[b] Defined as the layer or layers of natural material beneath the waste, or between the waste impoundment and the groundwater aquifer, and may include the soil.

[c] Annual average precipitation.

Wind Erosion and Fugitive Dusting

Fugitive dust impacts associated with wind erosion of ash and sludge waste piles are generally expected to be minimal. Wind erodibility of these wastes will primarily be a function of their surface texture and moisture content. Both ponded ash and sludge wastes typically have a high moisture content and surface crusting is common, thereby minimizing the likelihood of wind erosion. Exposed ash wastes in semiarid and arid regions represent more of a fugitive dust hazard. These ash wastes will be subject to a high degree of wind erosion because of their low moisture content and the strong hot-dry winds typical of these regions. Given these latter conditions, winds in excess of 19 km/h (12 mph) will be sufficient to suspend ash particles in the atmosphere, with quantity proportional to the wind velocity cubed (Brady 1974). Studies (U.S. Nucl. Reg. Comm. 1979) of wind dispersal of uranium mill tailings have shown that the tailings may be dispersed in measurable quantities as far as 400 m downwind. The inverse relationship between fly ash particle size and trace-element concentrations indicates that the particles most likely to be airborne also have the greatest potential for carrying toxic trace elements.

The deposition of fugitive ash may alter the surface of the affected soils in extreme situations. Turner et al. (1979) found that if fly ash, which is pozzolanic, is deposited on the soil in extreme quantities ($\sim$ 100 MT/ha), it will form a surface crust. However, these authors found no difference in the moisture content of untreated and fly ash-treated plots (some of the heavily treated plots had surface crusting). Apparently, fly ash must be mixed in the soil before it can impact moisture-retention capability (Plank et al. 1975; Page et al. 1979; Terman 1978).

For a given ash and sludge storage method, the potential for wind erosion and fugitive dust will vary as a function of climatic factors (precipitation, evaporation, and wind speed). Regions with high annual precipitation rates, low annual evaporation rates, and low annual wind speeds will have the least wind erosion, because ash and sludge wastes will (1) retain their moisture for a longer period of time and (2) be subject to lower wind energy. Roughly 80% of the United States falls into this category (Brady 1974). The greatest potential for wind erosion will occur in those regions [2-3% of the United States (Brady 1974)] with low annual precipitation rates, high annual evaporation rates, and high wind speeds. The ash and sludge wastes in these latter regions will lose their moisture quickly and will be subjected to high wind energy.

Given these considerations and information on regional mean annual precipitation, evaporation, and wind speed (Table 25), it is possible to indicate the potential for wind erosion in various regions: the Northeast and Southeast will have the least potential, the western Midwest and lower near West regions the highest potential, and the eastern Midwest and upper near West regions a moderate potential. Brady (1974) indicates that there are two areas within the western Midwest and lower near West regions that have the highest potential for wind erosion in the United States: (1) the central portions of North Dakota, South Dakota, and Nebraska; and (2) the western portions of Kansas, Oklahoma, and Texas; and southeastern Colorado and northeastern New Mexico.

Table 25. Regional Variation in Annual Mean Precipitation, Evaporation, and Wind Speed[a]

Region	Precipitation (cm)	Evaporation (cm)	Wind speed (m/s)
Northeast	80-120	80-120	6-8
Southeast	100-160	120-160	4-8
Midwest, Upper			
East	40-100	80-120	8-10
West	20-60	80-200	8-10
Midwest, Lower			
East	80-142	120-160	6-8
West	20-100	160-285	6-10
Near West			
Upper	20-60	80-120	6-10
Lower	20-60	120-285	6-8

[a] Data from U.S. Department of the Interior (1970).

Local topographic features also modify wind erosion and associated atmospheric dispersal of particles by reducing their exposure to wind energies. Ash and sludge wastes stored in flat, exposed areas will be more subject to wind energies than those stored in hilly, forested areas. In this regard, ash and sludge wastes in the Northeast and Southeast will be least subject to wind energies because of the often hilly terrain and large forested areas. The Prairie and Great Plains regions have large flat areas with no forests, and ash and sludge waste piles in these areas will be subject to high wind energies. Wind energy effects on ash and sludge wastes in the eastern Midwest will be moderate because of an interspersion of hilly and flat areas with prairie and forested areas.

Analyses and integration of information obtained by literature review and field reconnaissance will usually provide an adequate basis for evaluating wind-erosion potential at a waste-storage site. However, further investigations may be necessary; in some instances, determinations as to the feasibility of using the Wind Erosion Equation for estimating erosion potential may be warranted (Skidmore and Woodruff 1968). The equation is used by the U.S. Soil Conservation Service (SCS) in designing control practices and advising farmers on conservation programs (Woodruff et al. 1977a, 1977b). Thus, the investigator is advised to consult with SCS personnel regarding appropriate applications of the equation under considerations specific to a given storage site. The Wind Erosion Equation is expressed as:

Soil Loss = Function of Erodibility, Surface Roughness, Climate, Open Field Length, and Vegetative Cover

Functional aspects for variables of the equation are not illustrated herein because resolution of the equation entails use of numerous tables and figures. The following discussion provides a qualitative evaluation of how the components of the Wind Erosion Equation interact.

Characteristics of the climate that affect wind erosion include wind, precipitation, temperature, and humidity. Wind is the energy source for the erosion process, and the effects of the process vary with the velocity, turbulence, direction, and duration of windflow during an erosion event. Erosion is initiated when wind action is sufficient to dislodge and transport surface particles. Beasley (1972) reported that the minimum wind velocity required to initiate movement of the most erodible soil particles [about 0.1 mm (0.004 inch) in diameter] is about 16 km/h (10 mph) at 0.3 m (1 ft) above the soil surface. However, Woodruff et al. (1977a) indicated that under field conditions where the exposed surface consists of a mixture of single-grained particles, the "practical" wind speed to initiate erosion is about 21 km/h (13 mph) at 0.3 m height. Given initial particle transport, the rate of erosion increases with incremental increases in wind speed; i.e., under otherwise comparable conditions, the rate of erosion for a 48-km/h (30-mph) wind is more than three times that for a 32-km/h (20-mph) wind (Woodruff et al. 1977a).

The erosive effects of wind are strongly influenced by other climatic variables. The occurrence of precipitation contributes to soil moisture which, in turn, is depleted by evaporation and transpiration. The rate of depletion is dependent on the prevailing temperature, humidity, and wind conditions; the residual moisture is critical in that the cohesion of water films surrounding exposed particles is a strong deterrent to erosion, even at low levels of soil moisture. Erosion of air-dry soil is about 1.3 times more than is the case when soil moisture approximates the wilting coefficient of plants (Woodruff et al. 1977a).

The climatic elements also variously affect the structural stability of surface materials which, in turn, influences erosion potential. Alternate freeze-thaw and wet-dry cycles as well as raindrop impact tend to cause disintegration of surface aggregates resulting in increased erosion potential. On the other hand, heavy rainfall may consolidate certain fine-grained materials; subsequent drying results in formation of a crust that is relatively resistent to wind erosion. Aside from structural relationships, the density and texture of surface materials also influence erosion. For a given size fraction, the lighter particles are more readily displaced. Particles of 0.1 mm (0.004 inch) in diameter are more erodible than other particle size fractions of similar density (Beasley 1972). Materials comprised of a high proportion of fine particles [less than 0.02 mm (0.0008 inch) in diameter] are strongly cohesive and relatively resistant to displacement by direct wind force, particularly if the exposed surface is smooth. However, if the surface is disturbed, fine particles projecting into the turbulent airflow are readily displaced. Soil particles of up to 2.0 mm (0.08 inch) in diameter may be displaced at high wind speeds, but a diameter of 0.84 mm (0.03 inch) can be used as an approximate index for differentiating erodible and nonerodible particles (Woodruff et al. 1977a).

The presence of living vegetation and/or vegetative residues reduces the erosion potential of a given area. The vegetation not only absorbs some of the wind energy, thereby reducing wind velocity, but also binds the soil with its root systems. The degree to which the surface is protected depends on the orientation of the vegetation. For example, the protection afforded by horizontally oriented residues is primarily limited to the surface underlying the residues. In contrast, standing vegetation promotes stability of the immediate surface as well as additional area in the downwind direction. The greater the height of the vegetation, the greater is the downwind area protected from direct wind force. Vegetation also intercepts or entraps windborne particles, thereby reducing the erosive force of the wind as well as the amount of sediment removed from the eroding surface.

Small ridges and depressions, clods, and surface aggregates collectively contribute to the roughness of an erodible surface. These surface irregularities alter wind speed by absorbing and deflecting some of the wind energy. Entrained particles moving via saltation may be entrapped, thus reducing abrasion and buildup of the sediment load downwind. However, surface roughness is also conducive to turbulent windflow. Microrelief of 5 to 12 cm (2 to 5 inches) is considered the most effective in limiting wind erosion losses (Woodruff et al. 1977a). Greater microrelief causes increased wind turbulence and therefore increased erosion.

For a single erosion event, the erosion loss from an unprotected area is strongly dependent on the length of eroding surface in parallel with the direction of the erosive wind. With an effective wind barrier at the windward edge of the eroding area, the rate of erosion is zero. However, the erosion rate increases with increased distance downwind, as dislodged particles progressively contribute to increasing the erosive force of the wind. If the eroding area is of sufficient length, the sediment load increases to the maximum that the wind can sustain, and the rate of erosion remains constant regardless of additional length of eroding surface.

Interactions of Waste Constituents with Biota

Once dispersed into the environment from containment, constituents of ash and FGD wastes can interact with biota. Indeed, organisms can also serve as agents for dispersal of potentially toxic constituents from ash and sludge waste-storage sites. To date, no single encompassing theory adequately describes either the cycling of potentially toxic ash and sludge waste constituents through the biotic and abiotic components of an ecosystem or the potential for deleterious impacts to biotic resources.

Soil microorganisms. Soil microbiota can influence the availability of trace elements in ash and sludge leachates. It is known that microbes can in some cases decrease or enhance the toxicity of chemical compounds by either demethylating (i.e., methylmercury) or methylating (i.e., arsenic, lead, selenium, tin) trace elements, or they can toxify by increasing trace-element solubilities through altering the soil pH, chelating, or forming ligands (Van Hook 1978; Cataldo and Wildung 1978). Insufficient data are available, however, to make any more specific observations. Whether the net effect of microbial activity is to reduce or to enhance trace element toxicity is not known at present. In addition to modifying the availability of elements to terrestrial macrophytes, soil microbiota may themselves be affected adversely by elements found in coal ash and FGD waste-storage sites. A number of heavy metals have been implicated in the dysfunction of soil microbiota altering decomposition processes (Jordan and Lechevalier 1975; Ehrlich 1978; Brierley and Thornton 1979; Coughtrey et al. 1979). The implications of this for wildlife resources remain obscure. Disruption of decomposition processes and nutrient cycling could lead to ecosystem degradation. The magnitude of the potential for such an impact from combustion waste-storage sites is not currently known.

Vegetation. It is difficult to predict the potential for phytotoxicity from uptake of elements dispersed from coal ash and FGD sludge storage sites (Dvorak et al. 1978). Generalized maximum, nonphytotoxic plant tissue concentrations for several trace elements are presented in Table 26; these can be used as indicators of relative phytotoxicities of the elements considered.

Table 26. Normal Range and Suggested Maximum Nonphytotoxic Concentrations of Selected Elements in Plant Leaves[a]

Element	Concentration (μg/g, dry wt) Range	Maximum
Arsenic	0.1 -1.0	2
Barium	10 -100	200
Boron	7 -75	150
Cadmium	0.05 -0.20	3
Cobalt	0.01 -0.30	5
Copper	3 -40	150
Chromium	0.1 -0.5	2
Fluorine	1 -5	10
Iodine	0.1 -0.5	1
Iron	20 -300	750
Manganese	15 -150	300
Molybdenum	0.2 -1	3
Nickel	0.1 -1.0	3
Lead	0.1 -5.0	10
Mercury	0.001-0.01	0.04
Selenium	0.05 -2.0	3
Vanadium	0.1 -1.0	2
Zinc	15 -150	300

[a] Compiled by Lewis et al. (1978).

Most of what is known about phytotoxicity of the constituents from ash and sludge wastes is based upon studies in which plants were exposed to high concentrations of trace elements for short periods of time, i.e., acute exposure. These results are not directly applicable to the situation of long-term exposure to low concentrations (i.e., chronic exposure) of trace elements dispersed from ash or sludge storage sites. However, these studies indicate that soil characteristics, ability of plants to concentrate trace elements, and evolved tolerances are the primary factors in determining the phytotoxicity of elements that may be dispersed from these wastes [see Dvorak et al. (1978) and Gough et al. (1979) for more detailed discussion].

Because of their influence on ion mobility and solubility, the pH and CEC of soils play a large role in determining the potential for impacts to plants from coal ash and FGD sludge waste leachates. Most trace elements tend to be less soluble and less available for uptake in soils with a high pH and high CEC, and there is less potential for toxic effects to vegetation in these soils. However, even at high pH, such elements as As, B, Mo, and Se are readily soluble, can be taken up by plants, and are potentially toxic (Walsh et al. 1976; Dvorak et al. 1978; Menzie 1979).

Dispersal of soluble salts into soils can also add to osmotic stress of plants in saline soils, particularly if these plants are on the threshold of reduced productivity (Adriano et al. 1978). This could be particularly true in catastrophic releases of impoundment liquors into the adjacent environment.

Plant uptake of potentially toxic substances will also be governed by its solute demand, net permeability to solutes, and the supply of solutes in the soil (Van Hook 1978). Transpiration rates are a determinant of the rate of passive solute uptake; the higher the transpiration rates, the greater the uptake of soil solution. Active uptake of elements is also an important pathway into the plants (Tiffin 1977; Cataldo and Wildung 1978). Active uptake mechanisms of the roots exclude or reduce uptake rates of some elements (Cataldo and Wildung 1978). Like the soil, roots have their own CEC and may adsorb elements such as aluminum and manganese to the exchange sites on the root, thus avoiding toxic effects from these elements (Vose and Randall 1962). The presence of elements such as zinc and copper in the roots may also competitively inhibit the uptake of other potentially toxic trace elements such as cadmium and nickel (Cataldo and Wildung 1978). In contrast, some elements may enhance the uptake of others. Finally, the uptake of solutes may be more efficient at lower concentrations than at higher concentrations; for example, a fivefold increase in soil concentration of magnesium resulted in less than a doubling of plant uptake rate (Cataldo and Wildung 1978).

Evolved adaptations are also factors that determine the potential for trace-element toxicities. For example, plants growing in areas that have high endogenous levels of potentially toxic trace elements are more tolerant of these trace elements than plants growing in areas of low endogenous concentrations (Cataldo and Wildung 1978). This is true for plant species growing in the Southwest on soils with high concentrations of selenium (Dvorak et al. 1978). Smith and Bradshaw (1979) have identified a number of plant populations that have evolved tolerances to the high metal contents of metalliferous mine wastes in Britain. This indicates that some plants do have the evolutionary plasticity to tolerate high levels of potentially toxic substances introduced by human activity. The evolution of these tolerances to naturally high concentrations of trace elements would tend to counter toxic effects from additional introduction of these elements. It would, however, enhance the potential for transfer of the element to higher trophic levels, particularly if the mechanisms of tolerance involve concentration of the elements in the plant tissues rather than preclusion from plant uptake.

The deposition of airborne fly ash on vegetation in the immediate storage area can result in reduced productivity through (1) reduction of gaseous exchange rates, (2) alteration of leaf light absorption and reflectance abilities, and (3) uptake and translocation of toxic trace elements. Ricks and Williams (1974) found that ash particles deposited on leaves of *Quercus petraea* blocked open the stomates, thus preventing their closure during the dark period. This not only interferes with normal gaseous diffusion of CO_2, H_2O vapor, and O_2, but it also enhances the uptake of SO_2 and allows entrance of pathogenic fungal hyphae. Ricks and Williams postulated that fly ash deposition on leaves can also affect light absorption and reflectance of incident solar radiation, thereby affecting heat exchange and photosynthetic rates. These effects will be greater for vegetation having coarse, hispid or pubescent leaves than for vegetation having smooth shiny leaves because the coarse, hispid (or pubescent) leaves will retain more fly ash particles.

There is some dispute as to whether trace elements associated with fly ash deposited on leaves can be absorbed and translocated within the plant (Zimdahl and Arvik 1973). Ash particles may be washed or blown from the leaves to the soil below where, as a result of leaching and seepage into the soil, the trace elements can be taken up by the plant roots and translocated (Little and Martin 1972).

The variability in rates of uptake and biological concentration makes it extremely difficult to quantify the removal of trace elements from the soil by plants. Van Hook (1978) has defined biological concentration as the accumulation by adsorption and absorption of a constituent by an organism, resulting in an increased concentration in the organism. Biological concentration can be expressed in terms of a ratio (the biological concentration ratio) of the concentration of a constituent in the organism to the concentration in the surrounding medium (air, water, or soil) and/or the food consumed. The medium in aquatic systems is typically water, and in terrestrial systems is typically soil. When the biological concentration ratio is greater than unity, the phenomenon is referred to as biomagnification. In an attempt to provide at least some generalized information on plant uptake, Dvorak et al. (1978) used plant:soil concentration ratios (CR) as determined by the method of Hodgson (1970) (Table 27). The CR was calculated by taking the ratio of the average concentration of each element in plants to the average concentration of each element in soils; in this case, worldwide averages from Bowen (1966) and Chapman (1966) were employed. In reality, of course, trace-element uptake and accumulation by plants cannot be defined by single values; CRs will vary with type of soil, plant species and variety, environmental conditions, and other variables. Lewis et al. (1978) used Table 26 in combination with Table 27 to evaluate the potential phytotoxicity of elements dispersed in the soil. For example, cadmium has the highest estimated CR (Table 27) and one of the highest toxicities (Table 26) and would be considered to have a high potential for toxic effects.

Most plant species do not readily translocate As, Be, Cr, Pb, Ni, and V from the root; whereas other elements are readily translocated to the shoot (Romney and Childress 1965; Allaway 1968; Lisk 1972; Wiltshire 1972; Berry and Wallace 1974; Dvorak et al. 1978). Differential accumulation of toxic elements in plant tissues may determine which elements are ingested by

Table 27. Plant:Soil Concentration Ratios[a]

Element	Concentration ratio[b]
Arsenic	0.14
Barium	0.03
Beryllium	0.02
Boron	5.3
Cadmium	10.7
Chromium	0.02
Cobalt	0.11
Copper	0.47
Fluorine	0.03
Lead	0.45
Manganese	0.066
Mercury	0.02-0.5
Molybdenum	0.57
Nickel	0.045
Selenium	1.0
Vanadium	0.01
Zinc	0.64

[a] From Dvorak et al. (1978).

[b] This is a generalized approximation of the ability of plants to accumulate trace elements similar to the method employed by Hodgson (1970). The concentration ratio is the ratio of the average concentration of each trace element in plants to the average concentration of each trace element in soils. The concentrations in plants were taken from Chapman (1966) except for barium, beryllium, cadmium, mercury, and selenium which were taken from Bowen (1966). The concentrations in soil were taken from Bowen (1966).

wildlife foraging on different plant tissues. Based on the literature and their own experiments, Wallace and Romney (1977) have tentatively placed a number of trace elements into three groupings regarding element distribution between roots and shoots:

1. Reasonably uniformly distributed: Zn, Mn, Ni, Li, B.
2. Usually more in roots than in shoots, but often moderate with sometimes large quantities in shoots: Fe, Cu, Al, Cd, Co, Mo.
3. Mostly in roots with very little in shoots: Pb, Sn, Ti, Ag, Cr, V, Zr, Ga.

Animals. Little is known about the potential for toxic effects to animals from dispersed materials from coal ash and FGD sludge. Research has emphasized the toxicity of potential constituents under acute exposure regimes in the laboratory. Relative toxicities of some of the elements found in ash and sludge wastes can be obtained from Table 28; this table contains recommended toxicity threshold limits for livestock drinking water. Among factors that will influence the toxicity of waste constituents to wildlife include the chemical form of the element, species-specific tolerances, mode of entry into the body, and physiological condition of the animal. Dvorak et al. (1978) and Gough et al. (1979) discuss the toxicity of potential waste constituents in greater depth.

Toxic effects to primary consumers will be dependent, in part, on the ability of their forage to concentrate trace elements (Dvorak et al. 1978). Obviously, if the food plants do not take up toxic trace elements from the ash and sludge waste leachate, no toxic effects will occur to the consumers. Moreover, the potential for toxicity will depend on the location of the concentrated trace elements within the food plant. As noted above, trace elements more readily translocated to the shoot generally represent a greater hazard, because it is these portions of the plants that are eaten by most terrestrial wildlife. However, studies done on trace-element uptake from forage grown on fly ash have only demonstrated the animals' ability to concentrate

Table 28. Recommended Limits for Concentrations of Elements and Ions in Livestock Drinking-Water Sources Above Which Toxic Effects May Occur[a]

Element or ion	Recommended limit[b] (mg/L)
Aluminum	5
Arsenic	0.2
Boron	5.0
Cadmium	0.05
Chromium	1.0
Copper	0.5
Fluorine	2.0
Lead	0.1
Mercury	0.01
Molybdenum	uncertain[c]
Nitrate	100
Nitrite	10
Selenium	0.05
Vanadium	0.1
Zinc	25
Total soluble salts	5000

[a] Compiled by Dvorak et al. (1978).

[b] Compiled from National Academy of Sciences-National Academy of Engineering (1973).

[c] Toxicity influenced by many factors. Natural surface waters rarely contain over 1 mg/L.

these elements in various organs (DeJong et al. 1977; Furr et al. 1978a; Stoewsand et al. 1978; Fleming et al. 1979). Pathological responses have not been demonstrated in these experiments. Turner et al. (1979) did demonstrate that beetles and mites are affected by 25% fly ash supplements to their diet; larval development was prolonged, water uptake rates were increased, and fecundity was reduced. Dvorak et al. (1978) suggested that certain life stages may be more vulnerable to the toxic effects of trace elements because of inherent differences in tolerances as well as differences in feeding habits and physiology.

Inhalation of airborne fly ash particulates may cause physical damage to lung tissue or toxic effects due to trace elements adsorbed to these particulates. Most of what is known of the effects of inhaled particles is based on human studies; however, these results should generally be applicable to other vertebrate species. Particles < 1 μm represent the greatest concern since they are able to bypass respiratory filtering mechanisms and can be deposited in alveolar sacs (Natusch et al. 1974; Davidson et al. 1974; Fennelly 1975). Of the inhaled particles, 25% settle in lung tissues, 25% are exhaled, and 50% are swallowed (Schroeder 1971). Particles ≤ 1 μm entering the alveolar sacs are generally absorbed with absorption efficiencies of 50-80% (Natusch et al. 1974). Because of the large surface area-to-volume ratios of these submicron particles, soluble trace elements are preferentially concentrated on the smaller particles (Natusch et al. 1974; Davidson et al. 1974; Klein et al. 1975; Lee et al. 1975; Linton et al. 1976). Physical damage to the lung tissues results from impaired oxygen transfer due to inert materials (e.g., silicates) in the particulates leading to such diseases as silicosis (Fennelly 1975). In addition, irritation from these particles combined with other harmful substances (e.g., SO_2) can lead to lung diseases such as chronic bronchitis, bronchial asthma, and pulmonary edema. Exposure to fly ash particles associated with vanadium at levels ≥ 50 $\mu g/m^3$ in air can become a lung irritant due to the accumulation of vanadium in an insoluble form (Piperno 1975).

Of the trace elements found in coal ash, known or suspected carcinogens are As, Be, Cd, Co, Cr, Cu, Fe, Hg, Ni, Pb, Se, and V (Natusch 1978). Potentially carcinogenic polycyclic organic matter (POM) has been shown to adsorb to fly ash particles; however because of its volatile nature, POM tends to adsorb to fly ash particles after they are emitted from the stack and not to fly ash particles entrained within the emission-abatement system (Natusch 1978). This potential for carcinogenicity in the bacterium *Salmonella syphimurium* was demonstrated by Crisp et al. (1978), utilizing fly ash collected downstream of electrostatic precipitators. However, a study by Fisher et al. (1979), utilizing the same bacteria species and precipitator fly ash, failed to demonstrate any carcinogenicity. Thus, the potential carcinogenicity of ash and sludge wastes remains unresolved.

Ecosystem effects. Dispersal of constituents from coal ash and FGD sludge into the environment involves transport among and chemical transformation within the compartments of terrestrial ecosystems. Pathological effects induced by these constituents may disrupt the interaction of ecosystem compartments resulting in indirect impacts to biota. The trophic or food web is the major pathway of transport of potentially toxic elements into wildlife populations. Except for fluorine, mercury, and selenium, the constituents of ash and sludge wastes are not sequentially concentrated as they are transported along the trophic chain of terrestrial systems (Table 29).

Dvorak et al. (1978) reviewed the potential for emitted coal fly ash to impact terrestrial ecosystems. Analysis of several simulations indicated that arsenic, cadmium, fluorine, and selenium could achieve toxic thresholds in some of the model ecosystems. The major factor influencing the simulation was the background level of these elements in the soil. Van Hook et al. (1977) indicated that cadmium, lead, and zinc were accumulating in a deciduous forest ecosystem subject to fallout from a coal-fired plant. However, within the time frame of the study, no ecosystem responses were associated with this accumulation of metals.

Studies of fly ash have indicated the potential for impacts from coal combustion wastes due to both dispersal of constituents and concentration of trace elements to toxic levels by plants and animals (Page et al. 1979; Van Hook 1978; Furr et al. 1978a, 1978b, 1979). Several studies have examined the effects of fly ash amendments to soils (Bradford et al. 1978; Page et al. 1979; Turner et al. 1979). For example, Turner et al. (1979) found that application of 30 and 100 MT/ha of fly ash to desert soils reduced annual plant abundance and number of species. Although concentrations of several elements were elevated when treated plants were analyzed, Turner et al. attributed this to external accumulation of ash rather than uptake. Thus, it is unclear whether the observed effects are due to phytotoxicity of the ash or the physical impact of larger quantities of ash. Application of 5 MT/ha of ash also reduced the number of soil microarthropods for a period of 6 months after application, but toxicological effects were not demonstrated in the experiment. Other studies have indicated that constituents of ash and sludge wastes may actually increase local productivity and diversity in ecosystems that have low pH and soil nutrient levels (Plank et al. 1975; Wochok et al. 1976; Martens and Beahm 1978). Finally, a study of cadmium dynamics in terrestrial food webs demonstrated

Table 29. Generalized Biological Concentration Factors for Elements in Aquatic and Terrestrial Ecosystems[a]

	Concentration factor ([biota]/[growth medium])[b]							
	Terrestrial		Freshwater			Marine		
Element	Plant	Animal	Macrophyte	Invertebrate	Fish	Plant	Invertebrate	Fish
Aluminum	0.007	0.001	-[c]	-	-	6,000	-	-
Antimony	0.03	0.003	-	10	1	-	5	40
Arsenic	∿0.03	0.03	1,000	300	300	10,000	300	300
Barium	0.03	0.002	-	-	-	1,000	-	-
Beryllium	0.02	0.0003	-	10	2	2,000	200	200
Boron	5	0.02	-	-	-	30	-	-
Cadmium	10	8	4,000	2,000	200	4,000	200,000	3,000
Chromium	0.002	0.0008	2,000	-	-	20,000	-	-
Cobalt	∿0.06	0.004	4,000	-	-	2,000	-	-
Copper	∿0.7	0.1	200	1,000	200	4,000	2,000	1,000
Fluorine	∿0.2	3	-	-	-	3	-	-
Lead	∿0.3	0.2	500	100	300	300,000	1,000	200
Manganese	∿0.7	0.0002	200	-	-	30,000	-	-
Mercury	0.5	2	200,000	100,000	1,000	1,000	30,000	2,000
Molybdenum	∿0.4	0.1	1,000	-	-	40	-	-
Nickel	∿0.08	0.02	3,000	100	100	600	200	100
Selenium	1	10	-	200	200	10,000	1,000	4,000
Vanadium	0.016	0.002	-	-	-	1,000	-	-
Zinc	∿2	3	5,000	10,000	1,000	20,000	100,000	2,000

[a] Data from Bowen (1966), Braunstein (1978), and Hutchinson (1975).

[b] Growth medium: soil for terrestrial biota, water for aquatic biota.

[c] A hyphen indicates data not available.

higher cadmium levels in a coal ash basin than in an old field ecosystem (Skinner et al. 1978). Cadmium was actually concentrated sequentially up the food web; however, no pathological symptoms were detected in the time frame of the study. Thus, although the potential for toxic effects from stored ash and sludge is recognized, to date there has been no clear demonstration of impacts to terrestrial biota from operating storage sites.

Evaluating consequences to biota. The actual magnitude of impacts to fish and wildlife from ash and sludge wastes is site- and species-specific. Only after extensive studies of a given situation can one make precise predictions of impacts to biota contacting constituents of these wastes. In most cases, such studies will not have been carried out on projects which the fish and wildlife biologist reviews, and a more generalized approach must be taken. Lewis et al. (1978) proposed to use biological concentration ratios as predictors of potential toxicity to terrestrial biota (cf. p. 67). If one knows of the relative abundance of waste constituents and the potential for their dispersal from containment, this approach can be used to obtain an indication of which constituents may present problems to terrestrial biota. If data are available that predict the environmental concentrations of constituents dispersed from storage sites, a slightly more sophisticated approach may be used.

Generalized criteria for determining the potential for harm to human health and the environment have been developed under the sponsorship of the USEPA by Cleland and Kingsbury (1977); these criteria have been termed "estimated permissible ambient concentrations" (EPC). These values represent indicator thresholds above which deleterious effects may occur to biota (including wildlife resources) during chronic exposure. Permissible concentrations for the protection of health (EPC_H) were derived from laboratory animal toxicological studies using acute exposures. These values can be used as indicators of the potential for adverse direct impacts to wildlife. The EPC_Hs for soils represent threshold limits for wildlife via their food, whereas the EPC_Hs for water represent threshold limits for ingestion of water. Permissible concentrations in soils for the protection of the environment (EPC_E) were derived from studies of plant toxicology. These values may be used as indicators of the potential for adverse indirect impacts to wildlife, i.e., impacts to wildlife habitat. EPC values fall below acute toxicity threshold values. Dilution factors were applied to toxicity threshold values in order to reflect the lower concentrations required to elicit responses during chronic exposure, the type of exposure most likely for wildlife in waste-handling areas. The elemental concentrations presented in Table 30 are for constituents in solution; thus, in general, the values represent amounts potentially available for biological uptake. For soils, this amount can be considerably less than the total amount of the element in a unit of soil. If the estimated amount of a given constituent of coal combustion waste exceeds an EPC, it does not necessarily mean an adverse impact will occur but indicates there is a potential for deleterious effects that requires further scrutiny.

Where data on ambient concentrations of constituents dispersed from wastes are unavailable, a worst-case scenario may be developed for analysis. As illustrated in Table 31, maximum soil concentrations of waste constituents can be estimated from estimated concentrations in the leachate from a storage site. Leachate concentrations were assumed to be as in Holland et al. (1975); soil bulk density was assumed to be 1.5 g/cm^3 and soil water content 33%. For the site-specific case, the operator of the proposed facility will be the primary source of this information. If leachate replaces all soil water, concentrations of the elements in the soils are given by:

$$C_s = \frac{C_l \times 0.33}{1.5\ g/cm^3 \times 1000\ cm^3/L} \qquad (30)$$

where C_s is the soil concentration ($\mu g/g$) and C_l is the leachate concentration ($\mu g/L$). Maximum water concentrations of the elements can be taken as the concentrations in the leachate. In our example (Table 31), the elements most likely to cause problems for wildlife are boron, nickel, and vanadium--all of which markedly exceed EPCs.

The many complex interactions that may occur among constituents of ash and sludge wastes have not been taken into account for the values listed in Table 30. These interactions include: (1) synergistic effects, in which the response to two constituents together may be greater than the sum of the responses to each alone; (2) antagonistic effects, in which the response to two constituents together may be less than the sum of the responses to each alone; and (3) additive effects, in which the response to two constituents together equals the sum of the responses to each alone. For general assessment purposes, it can be assumed that the interactions are additive and that the potential for adverse effects exists if any waste constituent present in the environment occurs at a concentration higher than the EPC value for that constituent as given in Table 30.

Sophisticated levels of assessment cannot be accomplished without more detailed site-specific information, including more complex models of (1) the interactions of the abiotic and

Table 30. Estimated Permissible Ambient Concentrations (EPC) of Ash and Sludge Waste Constituents[a]

	EPC_H		EPC_E	
Constituent	Water (μg/L)	Soil or sediment (μg/g)	Water (μg/L)	Soil or sediment (μg/g)
Aluminum	73	0.15	200	0.4
Antimony	7	0.014	40	0.08
Arsenic	50	0.1	10	0.02
Barium	1,000[b]	2[b]	-	-
Beryllium	-	-	11[b]	0.022[b]
Boron	43	0.09	5,000	10
Cadmium	10	0.02	0.4	0.0004
Chromium	50	0.1	50	0.1
Cobalt	0.7	0.001	50	0.1
Copper	1,000	2	10	0.02
Lead	50	0.1	10	0.02
Manganese	50	0.1	20	0.04
Mercury	2	0.004	0.05[b]	0.0001[b]
Molybdenum	70	0.14	1,400	0.02
Nickel	1.4	0.003	2	0.004
Selenium	10	0.02	5	0.01
Strontium	27	0.05	-	-
Vanadium	7	0.014	75	0.15
Zinc	5,000	10	20	0.04

[a] Data from Cleland and Kingsbury (1977), except as indicated. EPC_H are permissible concentrations for health effects; EPC_E are permissible concentrations for environmental effects. EPCs in soil or sediment represent amounts available for biological uptake, i.e., that dissolved in soil solution.

[b] Data from USEPA (1976).

biotic components of the affected ecosystem and (2) the dispersal and interactions of waste constituents. In most instances, however, these detailed data and analyses will not be available for assessing the impacts of waste-handling systems to wildlife resources.

Use of Waste Impoundments by Wildlife

Little is known of the extent to which wildlife use active ash and sludge storage ponds. Anderson et al. (1975) observed waterfowl utilizing a slag pit near a power plant in Illinois. This pit supported lush aquatic vegetation (mostly brittle naiad) which apparently attracted the waterfowl. Alimentary tract analyses of ducks utilizing the slag pit indicated that they were ingesting slag. Anderson et al. (1975) detected no adverse effects, but did suggest that the ingested slag could supply needed nutrients as well as toxic metals. Waterfowl mortality has been attributed to ingestion of food and water from waterways contaminated by metalliferous mine wastes (e.g., Chupp and Dalke 1964). However, in these cases, the ambient concentrations of metals were two to three orders of magnitude higher than the concentrations associated with coal ash or FGD wastes. The U.S. Fish and Wildlife Service is currently studying waterbird use of an ash holding pond in Texas, but there are no results to report as yet.

Waterbirds and other wildlife may use the ash and sludge ponds for resting and watering, respectively, particularly in areas where the ponds are adjacent to wildlife habitat or croplands (Dvorak et al. 1978). If the ponds are near transmission lines, the potential for bird collisions with the conductors and towers is a problem. Use of these ponds could also result in some increase in hunting pressure in areas adjacent to the ponds. It is unlikely that large numbers of animals will be affected, however, because of limited pond sizes and human activity associated with maintenance and operation of these ponds. If wildlife ingest the pond water,

Table 31. Factors by Which Maximum Ambient Concentrations Exceed Estimated Permissible Ambient Concentrations (EPC) for a Model Waste-Storage Site

Element	Concentration (μg/L) in leachate[a]	Based on health effects		Based on environmental effects
		Water	Soil or sediment	Soil or sediment
Antimony	16	2	0	0
Arsenic	19	0	0	0
Barium	640	1	0	-
Beryllium	2	-	-	0
Boron	1840	43	4	0
Cadmium	1	0	0	1
Chromium	171	3	0	0
Copper	19	0	0	0
Lead	5.4	0	0	0
Manganese	2	0	0	0
Mercury	0.6	3	0	1
Molybdenum	158	2	0	2
Nickel	50	36	4	0
Selenium	92	9	1	2
Vanadium	100	14	2	0
Zinc	20	0	0	0

[a] Derived from Holland et al. (1975).

the potential toxic effects would be as described above for animals and ecosystems. The values in Table 31 suggest that concentrations of some elements in impoundment liquors can exceed estimated threshold limits for drinking water, but the magnitude of potential impacts needs further investigation.

The potential for impacts to waterfowl and other wildlife as a result of ash and sludge pond usage will be a function of the availability of water and, for waterbirds, the proximity to flyways. In the Northeast, Southeast, and Midwest, there are numerous reservoirs and ponds that will be more suitable for resting or watering places, and use of ash and sludge ponds should be low. In the near West, resting and watering areas are scarce, and use of these ponds should be much higher. Finally, the closer the ponds are located to major waterbird flyways, the greater the likelihood that these ponds will be utilized.

IMPACTS TO AQUATIC ECOSYSTEMS

The major sources of impacts to aquatic systems are (1) modification of the habitat by consumption of water for waste handling and (2) dispersal of waste constituents into aquatic environments. If waste materials are discharged directly into aquatic systems or enter via erosion or seepage from waste-storage sites, there is the potential for toxic impacts to fishery resources. Both toxic effects and effects of modification of aquatic habitat are highly dependent upon site-specific parameters of abiotic and biotic components of the ecosystem.

Consumptive Water Use

Consumptive use of water takes place in the FGD scrubbing process because water is the medium in which scrubber reactions occur. In addition, water may be used in transporting coal ash and FGD sludge to processing and storage sites. Water loss during ash and sludge waste handling occurs by three main methods: evaporation, mechanical carryover and condensation from the stack, and inclusion with the ash and sludge waste. Because water content in final combustion waste products has the greatest influence on physical properties of the waste, it is the most important and troublesome constituent during the waste-handling phase.

Numerous water bodies (lakes, ponds, reservoirs, and rivers) have competing water users; thus, the addition of a coal-fired power plant or a change in processing combustion waste products may require an additional demand on water consumption in an aquatic system also managed for fish and wildlife. If cumulative demands of industrial, utility, municipal, and agricultural consumptive water are substantial, regional analysis of consumptive use is necessary. Piecemeal consideration may be misleading, and one may dismiss impacts on a case-by-case basis as negligible although the cumulative effect to aquatic resources may be marked.

Ottmers et al. (1975) estimated that water consumption for lime/limestone scrubbing is 16 to 30% of the cooling-water requirement of a coal-fired power plant, depending on the nature of the ash slurry and whether fly ash is removed in the FGD scrubber. Estimates for water consumption of model operations are presented in Table 32.

Table 32. Water Requirements for Waste Disposal at a Coal-Fired Power Plant[a]

Waste		Water (L/MWh)	
Type	Weight (kg/MWh as dry waste)	No recycling	Recycling
Bottom ash	5	20[b]	2[d]
Fly ash	20	79[b]	8[d]
Lime sludge	27	64[c]	12[d]
Limestone sludge	34	79[c]	15[d]

[a] From Dvorak et al. (1978). Assumes 70% plant capacity and use of a coal with 10% ash and 3.5% sulfur content.

[b] Assumes slurry with 20% solids by weight.

[c] Assumes sludge with 30% solids by weight.

[d] Assumes 70% solids by weight.

The volume of water required for solid-waste disposal will depend on the specific waste-handling procedure employed. Thickening and dewatering are used on the scrubber bleed stream to reduce the water content by utilizing settling ponds, thickeners, vacuum filters, and centrifuges. Increasing the sulfate content of the sludge (e.g., by forced oxidation) improves the dewatering potential of the wastes. If ash and FGD sludge are handled separately and slurried or sluiced to onsite settling basins or holding ponds and no water recycling is practiced, large volumes of water will be transported to these basins. Transport of combined wastes to onsite basins will also require large volumes of water. After settling of the solids, the supernatant water may be discharged to surface waters, evaporated, or recycled. Consumptive use is greatest if dewatering is by evaporation, least if the supernatant liquid is discharged to surface waters, and intermediate if the water is recycled. Recycling can reduce the amount of water consumed by an order of magnitude over dewatering by evaporation (Table 32). These options for reducing consumptive water use are particularly important where water resources are scarce, e.g., the arid West.

The changes that occur in aquatic ecosystems from which water is drawn are directly related to water "loss". These changes will be greater where the percentage change from baseline characteristics is greatest; this is more likely in small watersheds or in more arid areas. The effects of water consumption upon lakes or ponds will require a calculation of lake or pond volume and recharge rate relative to the amount of water required for combustion waste disposal. Magnitude of the impact will be directly proportional to the ratio of the volume and recharge rate of the water body. A reduction in the total volume of water in these bodies of water can cause stress to aquatic biota by changes in littoral (inshore) zone production, loss of littoral zone habitat, and changes in species composition.

The location and extent of the littoral zone could change with decreasing water volume. Daily or seasonal fluctuations of the water level in lakes or ponds with limited shallow areas may eliminate or reduce the growth of sensitive rooted aquatic plants (macrophytes) and other littoral species (e.g., macrophyte-associated epiphytes, invertebrates, and vertebrates). This effect can be considered serious if the littoral zone is reduced or eliminated, because it is in this zone that forage grows and becomes available to support the many interrelated organisms within the lake ecosystem. This effect is more likely to be serious in lakes having limited shallow shorelines and steep sloping basins with limited littoral zone production than it is in shallow lakes or ponds having gradual sloping shorelines.

Water withdrawal and subsequent decrease of water volume in the littoral zone result in habitat loss for biota. Organisms occupying this zone become concentrated, thereby increasing both competition for resources and interactions with other species. Small fish prey species may be displaced from plant cover and predation temporarily increased. The severity of these effects is determined by the rate and frequency of drawdown associated with consumptive water use as well as the season of the year.

The construction of reservoirs as a water source may be undertaken in more arid areas where existing water resources are limited and/or unable to meet water requirements for combustion waste disposal. If a reservoir is constructed for multipurpose use (aquatic sports, fishing, and wildlife), consumptive water demands may not be compatible. Caution is advised whenever groundwater resources are utilized to fill a reservoir, since any reduction of groundwater supplies may cause supply problems for other groundwater users.

Water removed from rivers for consumptive use will require investigation of potential problems created downstream from the removal site. In arid (<25 cm rain/yr) or semiarid (25 to 50 cm rain/yr) regions of the western United States, competition for available water is high. If river impoundments are created (the majority of large rivers in the United States are not free-flowing), important ecological changes will occur both upstream and downstream of the impoundment. The dam becomes a physical barrier preventing the migration of fish and invertebrate "drift". Impoundments allowing a significant portion of the suspended solids to settle out will cause increased river scouring downstream and thus affect benthic habitats. Above the dam, the riverine habitat is replaced by a variety of lake habitats; thus, different plant and animal species will colonize the newly created impoundment. The primary plant material, which is available for food supply to aquatic consumers, will shift from predominantly leaf litter (and its associated bacteria, fungi, and invertebrate fauna) to rooted aquatic macrophytes and phytoplankton.

Impounded waters could cover large surface areas for long periods of time and thus be subjected to more surface warming by the sun; however, temperature changes relative to baseline conditions will depend on impoundment morphometry and location and on season of the year. This change in thermal regime of a river may favor increased phytoplankton growth and can effect changes in species composition of the downstream community (Spence and Hynes 1971a, 1971b; Lehmkuhl 1972). Planktonic algal growth within the impoundment could become the major plant food source for downstream organisms; consequently, invertebrates and vertebrates dependent upon a detritus-based (leaf litter) trophic structure may be replaced by organisms capable of utilizing the planktonic food source for some distance downstream.

Generally, the power plant operator will provide estimates of the solids:water ratio of the components in the waste-handling stream. If numbers are not available, the following assumptions can be made (Duvel et al. 1979): ash slurry, 70% solids; scrubber bleed, 10% solids; sludge after primary dewatering, 35% solids; and sludge after secondary dewatering, 65% solids. The amount of water in the scrubber bleed can be calculated from Equation 26. The net amount of water actually consumed can be estimated by subtracting the amount recycled from the dewatering process from the quantity in the scrubber bleed. For example, in Table 13, approximately 6 million tons of water is used per year in the scrubber bleed. If, after dewatering, the sludge solids are increased from 15% to 65% and the removed water is recycled, net water consumed is: $(6 \div 0.85) \times 0.35$ = approximately 2.5 million tons of water per year.

In assessing the significance of water withdrawal from an aquatic habitat, the biologist must rely heavily upon his own knowledge of the habitat requirements of the populations inhabiting the source of the makeup waters. Impacts can be evaluated by determining the habitat alterations that will occur due to the withdrawal of water. The Western Energy and Land Use Team of the U.S. Fish and Wildlife Service is developing instream flow strategies for many states. As part of this effort, weighted criteria are used to assess the impacts of altered stream-flow regimes on a stream habitat (Bovee and Cochnauer 1977). This information base can be used to evaluate the impacts of withdrawing water from stream ecosystems. For lake or pond systems, the habitat alteration due to lowering the water level can be estimated from knowledge of the system's morphometry. The significance of habitat attenuation to the affected fishery resources can be evaluated by determining if the habitat requirements of the fish populations are compatible with the expected habitat changes.

Behavior of Constituents of Ash and Sludge Wastes in Aquatic Systems

Trace elements enter the aquatic system via weathering of rock and erosion of soils. Storage and handling systems for coal ash and FGD sludges are anthropogenic sources of elements. Mobilization of waste constituents into aquatic systems occurs by means of seepage, erosion, and direct discharge. Leachate movement through soil and subsoil can introduce waste constituents into surface waters and groundwaters. Generally, as the leachate moves, the concentrations of waste constituents in the leachate are reduced as a result of attenuation by the soil and dilution by soil water. Runoff from waste-storage sites can introduce ash or FGD sludge constituents either in solution or as suspended solids. The impact of waste constituents in aquatic systems is a function of this chemical and physical behavior in the systems as well as species-specific rates of uptake and tolerance.

Once the trace elements are introduced into lakes or rivers, the rules of complex formation (Stumm and Morgan 1970) will apply and determine trace-metal behavior or activity (mobility, availability to biota, and ultimate fate in the aquatic ecosystem). This activity is influenced by redox potential and pH (Table 33); however, primary activity of trace elements is determined by the associations they form with water molecules (hydration) and/or organic molecules (chelation) (DeGroot 1973; Brooks 1977). It is very difficult to model or predict the behavior, chemical species, and activity of trace elements in freshwater (or seawater) since dynamic biological, chemical, and physical processes continuously modify concentrations and chemical equilibria. In spite of these difficulties, it is possible to generalize and note that trace elements are present (as variable and continually changing percentages) in surface water in any of these forms: free ionic elements, inorganic ion pairs, inorganic complexes, organic complexes, inorganic colloids, and in living organisms and their remains. Temperature, solubility, water hardness, chemical speciation, bioconcentration, and other environmental and physicochemical factors will affect the concentration and forms of trace elements in aquatic ecosystems.

Trace elements liberated from geological strata by weathering and erosion, or discharged as a result of human activities, are transported in particulates and in solution. Studies related to trace-element transport have focused attention on four major forms of the particulate input or load: carbonate particles, suspended particles, clay sediments, and organic compounds. Perhac (1974) found that heavy metals, dissolved from carbonates in the Tennessee River, were transported in solution and associated with clay sediments at a downstream dam; Hartung (1974) has noted that Cr, Cu, Fe, Mn, Na, Ni, Pb, and Zn concentrations in sediments of a river in Arkansas decreased downstream as the amount of carbonate rock increased and shale decreased; Ramamoorthy and Kushner (1975) reported that carbonate and bicarbonate ions are major binding agents in waters with high inorganic mineral content. These data indicate that carbonate and bicarbonate ion concentrations play an important role in solubility of trace elements. Zitko and Carson (1976) suggested that bicarbonate and carbonate ions provide active sites for bonding competition, but some cations are not involved since they are strongly bound to organic compounds (e.g., Cu and Hg). In his study of the lower Mississippi River, Hartung (1974) reported

Table 33. Effect of Redox Potential and pH on Relative Mobility of Eight Trace Elements[a]

	Relative mobility			
	Redox potential		pH	
Trace element	Oxidizing (aerobic)	Reducing (anaerobic)	Acid	Neutral to alkaline
Cadmium	Medium	Very low to immobile	Medium	Medium
Chromium	Very low to immobile	Very low to immobile	Very low to immobile	Very low to immobile
Copper	Medium	Very low to immobile	High	Very low to immobile
Lead	Low	Very low to immobile	Low	Low
Manganese	Very low to immobile	Low	Low	Low
Mercury	Medium	Very low to immobile	High	Very low to immobile
Nickel	Medium	Very low to immobile	High	Very low to immobile
Zinc	High	Very low to immobile	High	Very low to immobile

[a] Data from Brooks (1977).

that As, Cd, Cr, Hg, Mn, and Pb were transported primarily in suspended particles. Steele and Wagner (1975) suggested that fragmented substrate particles transported Ca, Cd, Mg, Pb, Mn, whereas Co, Cr, Cu, K, Mn, and Ni were associated with hydrous iron oxides. These data indicate that heavy metals are probably present as inorganic complexes or associated with surfaces of inorganic particles. A number of studies on trace-element transport suggest that trace elements are commonly associated with clay particles. The large surface areas and high ion-exchange capacity of clay particles would account for the association with trace elements. Other trace-element transport studies indicate that metals are chelated and transported primarily as organometallic particles (Gibbs 1973; Trefry and Presley 1976). The organometallic particles form in relationship with water-staining organic acids, especially humic acids. These organic acids are known to have strong binding (chelating) capabilities with trace elements, and they constitute the major portion of organic matter in soils and water. Benes et al. (1976) reported that cobalt, mercury, and zinc can be strongly bound to humic acids, and Ramamoorthy and Kushner (1975) determined that bivalent metals could be bound by humic substances in water, but not fulvic acid. Ramamoorthy and Rust (1976) concluded that, in general, transport and deposition of trace elements are directly related to the particulate surface area and organic content of sediments.

Potential Impacts from Constituents of Ash and Sludge Wastes

Most aquatic biota have evolved in environments containing minute quantities of trace elements; thus organisms have the ability to concentrate sufficient quantities of essential micronutrient trace elements for growth. Under conditions that allow trace-element concentrations to increase noticeably in waterways, aquatic organisms can continue to accumulate the elements; consequently, elevated or toxic concentrations may be reached. Unfortunately, some of the same metabolic mechanisms allowing essential micronutrients to be taken up do not prevent nonessential trace elements (e.g., arsenic or cadmium) from being taken up and concentrated.

The uptake and biological concentration of trace elements is affected by a number of biological, chemical, and physical factors; furthermore, the behavior and toxicity of trace elements taken up by aquatic biota is extremely difficult to characterize quantitatively (Becker and Thatcher 1973; Dvorak et al. 1978) because complete information is unavailable on (1) trace-element concentration in the total diet and water intake by individual species; (2) biological concentration potential and quantity of food-chain transfer among species; (3) the effects of physiological, biochemical, and synergistic factors on trace-element assimilation rates, retention times, and excretion rates of each aquatic species; and (4) the effects to be expected if aquatic species are subjected to low-level, long-term exposure (chronic exposure).

Cherry et al. (1979a, 1979b) reported that the density of aquatic biota in a swamp system receiving coal ash effluent was periodically altered by three major perturbations: (1) heavy ash siltation from inefficient basin operation, (2) lowered pH from the fly ash addition, and (3) coal ash-associated increases in elemental concentrations. Siltation appeared to be most influential in reducing the invertebrate densities, whereas low pH was more effective in reducing mosquitofish populations and retarding the recovery of invertebrates. Dipterans and odonates were the invertebrates most tolerant to coal ash stress. The discussion that follows will emphasize the potential for toxicity from waste constituents.

Microorganisms. Microorganisms can affect the ionic concentrations and chemical forms of elements in aquatic ecosystems (Dvorak et al. 1978). Microorganisms are primarily responsible for degrading organic detritus in aquatic ecosystems and play an important role in the conversion of some elements to forms more available to biota. Kuznetsov (1970) reported that microorganisms play a role during several stages of the biogeochemical cycles of carbon, oxygen, sulfur, nitrogen, phosphorus, iron, and manganese. Conversions by bacteria are also important in the cycling of various trace elements within aquatic systems (Dvorak et al. 1978). During the metabolic conversion of detritus, microorganisms can concentrate metals from the sediments and water (McLerran and Holmes 1974; Patrick and Loutit 1976) and thus make the biologically concentrated metals available to other organisms in the food chain. In a laboratory study, marine bacteria from Corpus Christi Bay were found to remove 85% of the zinc and 70% of the cadmium from solution and transport them to the substrate (McLerran and Holmes 1974). Bacterial removal of these elements from solution was accomplished through two possible mechanisms: (1) the metal was associated with bacterial cells (either adsorbed to or incorporated in the cell) and the cell settled out of solution, or, (2) the metal was precipitated from the solution by metabolic activity, probably as a sulfide or coprecipitated with iron sulfide. Microorganisms can have an alternate effect and compete with sediments for uptake of certain heavy metals such as mercury (Ramamoorthy et al. 1977).

Some trace elements may inhibit the important nutrient cycling process by inhibiting microbial action. However, the toxic effects of trace elements on aquatic microorganisms are poorly known.

Aquatic plants. The uptake of trace elements by aquatic plants from settling ponds or discharges from storage ponds is a subject of increasing concern since the detrimental effects are not well known. This is further complicated because the aquatic plants that may be involved range from the single cell alga (planktonic and/or benthic) to larger, rooted aquatic plants (macrophytes). The relative importance of nutrient uptake via roots or shoots has not been established for most rooted macrophytes, although roots have been identified as important organs of nutrient uptake (Denny 1972; Carignan and Kalff 1980). Uptake via the roots is important to consider because the sediments could contain a large portion of trace elements precipitated from coal combustion waste leachates.

Biological concentration and toxicity of trace elements in aquatic plants are element- and plant species-specific (Dvorak et al. 1978). Trollope and Evans (1976) found that copper, iron, nickel, lead, and zinc concentrations varied among plant genera and that some genera preferentially concentrated some trace elements. Cherry and Guthrie (1979) reported that duckweed biomagnified mercury and zinc above sediment concentrations in the drainage system of a coal ash basin. Additionally, Al, As, Ba, Cd, Co, Cr, Cu, Fe, Mn, Mo, Se, Sr, and Ti were biomagnified above water concentrations. Guthrie and Cherry (1979b) also found that duckweed and cattail were the most efficient aquatic producers to bioconcentrate the 23 elements studied. However, all elements were concentrated by the majority of the aquatic plants to levels higher than the concentrations of elements in the water in which the plants grew. These high concentration ratios in aquatic plants are of concern because plants are the primary food source for food webs.

A number of environmental and physiological factors can affect uptake, accumulation, and toxicity of trace elements (Dvorak et al. 1978). Chelating agents have been shown to counteract inhibition of growth in algal toxicity studies (Hart 1975; Katagiri 1975; Hart and Scaife 1977). Katagiri (1975) and Hart and Scaife (1977) reported that cadmium accumulation was pH dependent. Temperature-trace metal interactions have been shown to affect algal growth responses (Knowles and Zingmark 1975, 1978; Zingmark 1975). Ionic strength of the growth medium has been inversely correlated with plant responses (Fujita and Hashizume 1972; Hannan and Patouillet 1972; Kinkade and Erdman 1975; Greene et al. 1975; Foster 1976).

Uptake rates of iron, manganese, and titanium for some marine algae were greater in light than dark (Gryzhankova et al. 1975), but light conditions did not affect copper uptake by two freshwater algae (Mierle and Stokes 1976). Uptake rates also depend on population growth rates. Cadmium uptake was higher during the exponential growth phase of a diatom population than during other growth phases (Cossa 1976), and copper and nickel uptake declined in older cultures of *Scenedesmus* (Stokes 1975).

The effects of exposure to a single trace contaminant can be modified by the addition of one or more different trace contaminants (Wissmar 1972; Hart 1975; Hutchinson and Czyrska 1975; Hutchinson and Stokes 1975; Stokes 1975; Dvorak et al. 1978). Discharges, including seepage, from coal combustion waste-storage sites are likely to contain complex mixtures of potentially toxic materials, making it imperative that the complex mixtures, themselves, and not just the individual constituents be studied for potential toxicity. Vocke et al. (1980) conducted one of the few tests on the toxicity of a mixture of waste constituents by exposing algae to various dilutions of scrubber ash slurry extract. They reported that the first significant inhibition of *Ankistrodesmus*, *Scenedesmus*, *Selenastrum*, and *Microcoleus* occurred in 1, 5, 10, and 15% extracts, respectively. Algistatic-algicidal responses for *Ankistrodesmus*, *Scenedesmus*, and *Selenastrum* were indicated in 50, 100, and 75% extracts. *Microcoleus* approached an algistatic response at 100% scrubber ash slurry extract.

Invertebrates. Trace-element uptake by invertebrates depends primarily upon metabolism and feeding behavior of the organism as well as the form of the trace element in the environment. Sediments can become a primary source of trace elements for invertebrates, especially if the sediments are ingested (Dvorak et al. 1978). In a radioactive tracer study of oysters, Harrison et al. (1976) found uptake to occur from both food items and water. Uptake of Cs-137 was mostly from water, Co-60 uptake was mostly from particles, and Zn-65 and Mn-54 uptake were from both sources. Luoma and Jenne (1975a, 1975b) reported that silver, cadmium, cobalt, and zinc uptake by deposit-feeding clams depended on the physicochemical characteristics of the sediment. Uptake increased as the metal binding capacity of the sediment decreased.

Uptake of trace elements is not constant but will vary seasonally and with the life history stage and physiology of the organism. Frazier (1975) found that uptake of manganese and iron, but not zinc and copper, was correlated with oyster shell growth. Body burdens, the amount of the trace element in the body, showed a gradual increase during spring and summer followed by rapid losses during late summer to early fall. During this period, 30% of the zinc and 50% of the copper were lost in less than a month. Cadmium was similar in behavior to zinc and copper. Temporal fluctuations in trace-element concentrations in animals can also reflect changing environmental conditions or changing contamination levels. Mercury uptake in filter-feeding clams increased with elevated mercury concentrations in the water, but temperature had no apparent effect on the uptake or elimination rates (Smith et al. 1975). Conversely, Pringle et al. (1968) found that uptake rate was closely correlated with temperature. The work of Craig (1967) and Smith et al. (1975) suggests that there is a diffusion of the trace element into the organism followed by the formation of stable complexes which are eliminated at a steady rate. Pringle et al. (1968) hypothesized an active-transport uptake mechanism which is dependent on the metabolic activity of the cells and therefore has greater temperature dependence. In both cases, the rate of accumulation would also depend on the binding strength of the complex formed and its stability (Dvorak et al. 1978). Concentrations of trace elements in invertebrates vary among body tissues (Dvorak et al. 1978). Whole body concentrations can be much greater than the concentrations found in the environment (Table 29). The tendency for aquatic invertebrates to concentrate many trace elements to much higher levels than ambient suggests that these organisms could form major pathways of exposure of their predators (e.g., fish) to high levels of waste constituents.

Toxicity of trace elements to invertebrates varies according to the specific element and its form, and among the different species of organisms (Dvorak et al. 1978). Baudouin and Scoppa (1974) found that three species of zooplankton have different sensitivities to Cd, Co, Cr, Cu, Mn, Hg, Pb, and Zn but that the order of sensitivity among the three species was similar. Winner and Farrell (1976) compared acute and chronic toxicities of copper in four different-sized species of *Daphnia*. The two larger species were significantly more tolerant than the two smallest. However, concentrations which reduced life spans were not significantly different.

The biological retention time of trace elements within the organism depends on binding strength and stability of the complex formed, the biological form of the trace element, the metabolic rate of the organism, and toxicity of the trace elements. Elimination of trace elements from organisms can be accomplished by diffusion, active transport, spawning, secretion, molting, defecation, and a variety of other mechanisms (Dvorak et al. 1978).

Fish. The incorporation of trace elements into fish tissues occurs by active or passive absorption through the gills and by ingestion. The chemical form of the trace element--which is influenced by water quality parameters including alkalinity, pH, temperature, dissolved oxygen, organic ligands, and hardness--is an important factor in uptake (Dvorak et al. 1978). Buhler et al. (1977) reported that hexavalent chromium is easily diffused through the gills of rainbow trout, and that both uptake and elimination is rapid until an equilibrium with the medium is reached. Trivalent chromium, however, is much slower to be taken up and eliminated due to binding with proteins. Olson et al. (1973) demonstrated that both inorganic mercury and methylmercury can be absorbed through the gills, but uptake of the methylmercury form is more rapid.

Uthe et al. (1973) suggested that uptake of mercury by rainbow trout is related to seasonal changes in temperature. They found that the accumulation rate was greatest during the first warm period of summer and that uptake was greatly reduced during the remainder of the year. These data also suggest that uptake is a function of feeding. During the early summer, trout maximize feeding and growth. Under such conditions, uptake from contaminated food items would also be greatest.

The pH of the environment has a pronounced effect on the form and therefore the uptake of certain trace elements (Dvorak et al. 1978). When the solution is acidic, most heavy metal ions are liberated into solution and available for uptake through the gills. Merlini and Pozzi (1977) found that the uptake of lead was three times greater at pH 6 than at 7.5. Tsai et al. (1975) determined that inorganic mercury was less available for uptake by fish in alkaline water than in acidic water. Mercury complexes formed under alkaline conditions were postulated as the reason for reduced uptake.

Trace elements can affect fish in two ways: direct lethal effects and indirect sublethal effects (Dvorak et al. 1978). The direct lethal cause of death is often related to physiological changes in tissues or organs of fish. Frequently, death is caused by some enzyme system being poisoned by a trace element. The gills are often involved initially, followed by impaired oxygen uptake (Lloyd 1960; Skidmore 1970). The mechanism that causes death appears to involve the production of lactic acid, a product of anaerobic glycolysis (Hodson 1976) in oxygen-impaired uptake by the gills. The other major tissue involved in trace-element accumulation is the liver (Jackim et al. 1970). Indirect sublethal effects are often very obvious; however, the fish may live for a long period of time. The manifestations of these effects at the organism level include: inhibition or interference with neurophysiological activity, enzyme activity, and hormonal balance; increased susceptibility to disease or parasites; and teratogenic, carcinogenic, and mutagenic effects. These effects also include reduced growth, behavioral modifications, reduced survival, reduced reproductive capacity, and reduced fitness (Pakkala et al. 1972; Drummond et al. 1973; Holcombe et al. 1976; Lett et al. 1976). Within a species, resistance to a toxicant can vary with the age, sex, life history stage, physiological condition, exposure history, or even from individual to individual. Some species are more tolerant to some toxicants than others, and within a species some individuals may be more tolerant than other individuals.

Ecosystems. Trace elements in aquatic systems undergo differential uptake by biota. Since the study of trace-element cycling is in its infancy, few studies are available for discussion. Currently, no single encompassing theory has adequately described trace-element cycling through biota. This is no doubt due to the differential behavior of trace-element forms in both biotic and abiotic components of different aquatic systems. Uptake and transfer through trophic interactions are modified by the physicochemical form (ionic or particulate) of the trace element, and by the nature of the organisms and their habitat, substrate-sediment associations, and food habits. The studies that have been done to date indicate that biological concentration of trace elements deserves greater consideration.

In a study of an unpolluted stream, Enk and Mathis (1977) found biota to accumulate cadmium and lead above water concentrations (biological magnification). Concentrations were generally highest in invertebrates (mayflies, damselflies, caddis flies, and snails), intermediate in sediments, and lowest in fish. Food chain magnifications were not observed. Concentrations of lead and cadmium appeared to be more a function of food and habitat. Organisms associated with the sediments and/or with detrital feeding habits accumulated more of the metals than predatory species.

Individual variability in metal concentration, partly owing to size difference of individuals, can mask trends in elemental trophic transfer. Concentrations of metals in an organism can show positive, negative, or no relationship with size, depending on a number of physical, chemical, and biological factors related both to the organism and metal(s) in question. In fish, most heavy metals, except for mercury, that are associated with coal in more than trace quantities either show no increase in concentration with size or a decrease with size (Brooks and Rumsey 1974; Giesy and Wiener 1977; Mathis and Kevern 1975; Vinikour 1977).

Guthrie and Cherry (1976) examined the biota of a stream receiving fly ash settling-basin effluent. In almost every case, fish had the lowest concentrations of the trace elements; notable exceptions were calcium and selenium where fish contained the highest concentrations. The major role of each biotic form in the cycling of specific trace elements varied, and no single species was found to concentrate any element to a greater degree than all other species. Plants were more efficient concentrators of manganese and potassium than animals; however, no distinction was made among rooted, floating, or algal plants. As a group, the primary producers did not rank higher than third as concentrators of any group of elements. Midges were the most efficient concentrators of cobalt, mercury, copper, chromium, arsenic, and antimony. Additionally, Cherry and Guthrie (1977) reported that plants had high accumulations of titanium, manganese, arsenic, and mercury; invertebrates had high concentrations of cobalt, mercury, copper, chromium, cadmium, and arsenic; and vertebrates greatly biologically magnified selenium and zinc from the same settling basin. In another paper on the same basin, Guthrie and Cherry (1979a) reported that among the biota, *Hydrodictyon* sp. and *Lemna perpusilla* had the highest concentrations of aluminum and iron, whereas other macrophytes were the major accumulators of manganese and barium. Invertebrates generally concentrated high amounts of copper and zinc, although cadmium and mercury were accumulated by most crayfish. Selenium was selectively concentrated by bacteria, crayfish (*Procambus* sp.), and mosquitofish (*Gambusia affinis*).

Trace elements can have a variety of effects on aquatic biota other than acute toxicity, including changes in physiology, productivity, community composition, and species abundance (Dvorak et al. 1978). Elevated concentrations of trace elements, e.g., copper at 25 μg/L, inhibit photosynthesis in phytoplankton populations (Nielsen and Laursen 1976). This inhibitory action is also affected by pH, humic acid content of the water, copper tolerance of individual species, and population density. Bartlett et al. (1974) found that the initial effect of low concentrations of either copper, zinc, or cadmium was a reduction in growth rate; increased concentrations stopped growth, and a further increase was lethal. Patrick (1975) suggested that a change in community structure from primarily diatoms to blue-green algae was related to an increase in trace-element concentration. Vocke et al. (1980) reported interspecies variation in sensitivity of algae (*Ankistrodesmus, Scenedesmus, Selenastrum,* and *Microcoleus*) to arsenic, cadmium, mercury, selenium, and scrubber ash slurry extract.

Sublethal or chronic effects of trace elements are perhaps more significant to the ecosystem than the acute toxic or lethal effects. Sublethal effects can be classified as either (1) affecting the fitness of the organism, or (2) affecting the structure or function of the community; the second is a function of the first. Trace-element effects which reduce the fitness of the organism include: changes in physiology, such as modification of osmoregulation (Schmidt-Nielsen 1974); shell deposition (Frazier 1975); and growth, development, and reproduction (Reinhart and Myers 1975). Sublethal effects on the population or community level include reductions in abundance, diversity, and production. All of these effects reduce the available biomass, nutrients, and/or energy for transfer to higher trophic levels (Dvorak et al. 1978).

Phillips and Russo (1978) evaluated the tendency of fish and shellfish (crustaceans) to biologically concentrate trace elements. Although the distribution is known of some metals in tissues of a number of aquatic species, more information is needed about the mechanisms of toxicity, particularly during chronic exposure. Only then can we more accurately predict what constitutes harmful ambient concentrations.

Although specific responses are important for understanding how a toxicant affects an organism, it is important to keep in mind how these responses are related to species fitness, energy transfer through trophic structures in the aquatic ecosystems, and other species interactions at the population and community level. One important effect is increased susceptibility to predation. Contaminated fish may be preyed upon at a higher rate than unaffected fish. Kania and O'Hara (1974) suggested that bioaccumulation of trace elements in predators may be increased by selective ingestion of more contaminated than uncontaminated prey.

In an aquatic system, trace elements are predominantly associated with the sediments, which act as both sink and reservoir; relatively small amounts are found dissolved in the water. From the sediments, trace elements are accumulated by both rooted vegetation and benthic invertebrates. Phytoplankton both adsorb trace elements to their cell walls and absorb them. Grazers and lower-order consumers seem to concentrate trace elements to the highest degree. The greatest biological concentration factors are found in sediment or detrital feeders. Higher-order consumers or predators not associated with the sediment accumulate trace elements both from water and food, but food appears to be the major source. Some discrimination appears to occur at this trophic transfer since trace-element concentrations are usually lower in predators than in their prey. The ultimate fate of trace elements is return to the sediments or translocation downstream, unless components of the aquatic system are removed from the system by human harvest or predation by birds or mammals.

Evaluating consequences to biota. Generalized criteria for determining the potential for harm to aquatic biota have been developed by Cleland and Kingsbury (1977). These criteria are the EPC_Es for water listed in Table 30 and are equivalent to the USEPA's "quality criteria for water." As discussed on p. 71, expected concentrations of trace elements in the waste liquors can generally be obtained from the operator of the proposed facility. With this information, one can calculate a dilution factor (D_f) or the factor by which leachate concentration exceeds the EPC:

$$D_f = \frac{C_e}{EPC} \qquad (31)$$

C_e is the concentration of a constituent in the waste leachate or discharge effluent and EPC is the estimated permissible concentration of that constituent (from Table 30). The dilution factors can be used as indicators of which waste constituents discharged or leached into surface waters could pose potential hazards to aquatic biota. For example, for the model in Table 34, the elements mercury, selenium, and nickel will require the greatest amount of dilution before they can be brought to levels that will ensure protection of aquatic life. When the concentrations of elements in waste discharge are known, the same approach can be used to indicate potential problem areas for other situations.

If effluents, including flowing leachate seepage, from ash and sludge waste-storage sites are discharged into surface waters, the following relationship can be used to conservatively predict receiving-stream flow rates that are required to achieve acceptable EPC_E values for potentially toxic discharge constituents, with no losses after complete mixing:

$$D_r = \frac{D_e(C_e - EPC)}{EPC - C_r} \qquad (32)$$

D_r is the receiving-stream flow rate; D_e is the effluent flow rate; C_e is the effluent concentration of a given constituent; C_r is the ambient receiving-stream concentration of a given constituent before effluent addition (generally considered to be zero for nonpolluted streams); and EPC is the permissible concentration of a given constituent in the receiving water after complete mixing (Table 30). Complex interactions--which can be antagonistic, additive, or synergistic--occur between discharge constituents and receiving-stream biota. For general assessment, an additive relationship may be assumed for environmental protection; this additive relationship can be derived by totaling the estimated receiving-stream flow rates required for each constituent. Information on receiving-stream and discharge flow rates and chemistry may be available from the operators of the proposed facility. If receiving-stream flow rates are above the calculated D_r (Equation 32) for a given constituent or combination of constituents, one can generally conclude that aquatic biota in the receiving stream will be unaffected by the operation. Where the measured flow of the receiving stream is less than D_r, the potential for impact is indicated. The actual degree of environmental impact caused in aquatic ecosystems by ash and sludge waste storage will be dependent on the quantity and quality of storage-site discharges, receiving-stream flow rates, and many other site-specific variables.

In using the EPCs as criteria for assessing a given site, one must remember that they serve only as indicators for potential problems. Precise prediction of the actual magnitude of impact to fish and wildlife resources is not possible because of the highly variable nature of the

Table 34. Dilution Factors Required to Achieve Estimated Permissible Ambient Concentrations (EPC) for Water of Coal Combustion Waste Constituents from a Model Waste-Handling Facility

Element	Concentration (μg/L) in discharge or seepage[a]	Dilution factors
Antimony	16	0
Arsenic	19	2
Barium	640	-
Beryllium	2	0
Boron	1840	0
Cadmium	1	2
Chromium	171	3
Copper	19	2
Lead	5.4	1
Manganese	2	0
Mercury	0.6	12
Molybdenum	158	0
Nickel	50	25
Selenium	92	18
Vanadium	100	1
Zinc	20	1

[a] Derived from Holland et al. (1975).

physical and biological systems involved with coal waste dispersal into the environment. This means that the assessor is operating under severe constraints regarding the ability to determine the effect of a project upon fish and wildlife resources. Further research into the dispersal of coal waste constituents is required before the precision of predicting impacts to fish and wildlife resources can be increased.

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MANAGEMENT OF STORAGE SITES FOR COAL COMBUSTION AND EMISSION-ABATEMENT WASTES

RESOURCE CONSERVATION AND RECOVERY ACT

In the past decade, there has been increasing public concern over detrimental health and environmental effects from poorly contained solid wastes. Responding to this concern, the U.S. Congress amended the Solid Waste Disposal Act of 1965 (Pub. L. 89-272) by replacing it with the Resource Conservation and Recovery Act (RCRA) of 1976 (Pub. L. 94-580). The goals of RCRA, as outlined in Section 1003 of the act, include:

1. Prohibiting open, uncontrolled dumping;
2. Providing assistance in developing improved techniques of solid-waste management;
3. Providing for the promulgation of regulations and guidelines for solid-waste management that will reduce adverse effects to health and environment; and
4. Promoting a national research and development program to improve resource conservation and recovery.

The act provides (Sec. 2001) that it be administered by an Office of Solid Waste within the U.S. Environmental Protection Agency (USEPA). The USEPA (1978, 1980) has issued regulations and guidelines for the management of solid waste, including coal ash and flue-gas-desulfurization sludge.

Nonhazardous waste must be disposed of in conformance with regulations of 40 CFR 257 (USEPA 1979a). These regulations provide that a waste facility in a floodplain should not restrict the base flood, reduce the water storage capacity of the flood, or result in a washout of solid waste so as to pose a hazard to human life, wildlife, or land and water resources. Moreover, a waste facility may not contribute to the further degradation of an endangered or threatened species of biota. Nonhazardous waste storage facilities may not discharge effluent into surface waters in violation of requirements of the National Pollutant Discharge Elimination System (NPDES). The NPDES effluent standards for steam-electric generating stations, including effluent from coal ash and FGD sludge handling, are presented in 40 CFR 42.3. Standards regulating these effluents are based upon effluent pH and concentrations of total suspended solids and oil and grease in the effluent. A nonhazardous waste facility may not cause an underground drinking water source to exceed the National Interim Primary Drinking Water Regulations (40 CFR 141). The USEPA proposes to include the requirement that National Secondary Drinking Water Regulations (40 CFR 143) also may not be exceeded. The USEPA interim regulations restricting the use of nonhazardous waste near food-chain croplands are based upon pH limitations and application rates of cadmium (USEPA 1979a). Requirements for restricting pathogen dispersal and open burning of wastes would not apply to wastes from coal combustion. However, safety requirements to restrict access would be applicable. Although birds may be attracted to waste impoundments, regulations for reducing bird hazards to aircraft have been promulgated only for putrescable (decomposable organic) wastes, and these regulations would not currently apply to impoundments of ash and sludge wastes.

In 1978, the USEPA proposed to list utility wastes such as combustion ashes and FGD sludges as "special wastes", requiring application of only a select few of the hazardous-waste standards for their handling. In the Interim Final Standards for Owners and Operators of Hazardous Waste Treatment, Storage and Disposal Facilities, this designation has been dropped by the USEPA (1980, pp. 33173-33175); utility wastes have been excluded from the listing as hazardous wastes and currently may be handled as nonhazardous wastes (USEPA 1980, p. 33120). However, it is the intent of the USEPA to further investigate the potential hazards of these wastes, and the possibility remains of their being declared hazardous.

The USEPA (1980) has recently promulgated permanent and interim regulations controlling hazardous waste management systems. For ignitable, reactive, volatile, or incompatible mixtures of hazardous wastes, the USEPA requires storage in tanks or containers capable of retaining the wastes. The USEPA has also required the treatment and storage of other hazardous wastes in landfills or impoundments to reduce mobilization of hazardous substances into surrounding areas. The proposed standards for closure and management of inactive sites for hazardous waste storage provide security against contact of human or animal life with the hazardous materials. There are, however, no specific reclamation criteria in the proposed regulations. To meet USEPA requirements, choice among the design and reclamation alternatives discussed in later sections will be to a large degree dependent upon the requirements imposed by site-specific constraints.

The USEPA (1979b) has also proposed guidelines for landfill storage of solid wastes. These guidelines are designed to provide sound waste management direction but do not represent a guarantee of compliance with the Criteria for Classification of Solid Waste Disposal Facilities and Practices (USEPA 1979a). The guidelines provide recommendations for site selection and facility design that will protect public health and the environment. Relevant to utility waste storage, the guidelines provide recommendations for controlling seepage and leachate mobilization into the environment as well as surface flow diversion. Operational and monitoring practices are also recommended with a goal of ensuring maintenance of landfill integrity.

Possible Classification of Ash and Sludge Wastes as Hazardous

Utility wastes are not currently on the list of hazardous waste, nor are utility waste handling processes listed as producing hazardous waste (USEPA 1980, pp. 33122-33127). However, a waste may be considered hazardous if it can be categorized as ignitable, corrosive, reactive, or toxic (USEPA 1980, pp. 33121-33122), and utility wastes may meet some of these criteria.

Wastes are considered ignitable if they meet any of the following criteria: (a) waste is a liquid with a flash point of 60°C; (b) waste is not a liquid and is liable to cause fire through friction, absorption of water, spontaneous chemical change, retained heat, or by burning so vigorously when ignited as to create a hazard during management; (c) waste is an ignitable compressed gas; or (d) waste is an oxidizer. Neither scrubber sludges nor combustion ashes would be considered hazardous by these criteria.

Corrosive wastes meet one or both of the following criteria: (a) waste is a liquid and has pH $\leq$ 2 or pH $\geq$ 12.5; or (b) waste corrodes steel at a rate greater than $\sim$ 0.6 cm/yr (0.250 in./yr) at $\sim$ 54°C (130°F). A literature review by Hart and DeLaney (1978) found that the pH of fly ash slurries ranged from 3 to 12, but extremes were rare; the pH of several samples of FGD scrubber sludge and sludge leachate approached the pH corrosive criterion, but no documentation was found where the criterion was equalled or exceeded. No information on the effects of ash and sludge on steel corrosion was found, but Hart and DeLaney concluded that it was unlikely the wastes would meet this criterion. It appears that FGD scrubber sludge, fly ash, or bottom ash may meet the corrosive criteria on specific occasions, but on the whole these wastes would generally not be classified as corrosive and, therefore, hazardous.

Reactive wastes are those which are unstable and undergo violent reaction, are capable of detonation or explosion, or are forbidden explosives. There is no evidence that FGD scrubber sludges, fly ash, or bottom ash would meet any criteria for being reactive wastes.

The proposed criteria for toxic wastes are that extracts from the waste must not contain the substances listed in Table 35 in excess of 100 times the National Interim Primary Drinking Water Regulations. The USEPA is also considering basing criteria levels on their "Quality Criteria for Water" (USEPA 1976). A factor of 100 is used because the USEPA believes that leachate will be diluted by that factor by the time a leachate plume enters a well. Hart and DeLaney (1978) and Weeter and Bahor (1979) have reviewed the potential for coal combustion ashes and scrubber sludge to meet or exceed the criteria for toxicity; Tables 36 and 37 summarize their reviews. The leachates that have been analyzed do not conform to the extraction procedure specified by the proposed USEPA guidelines. However, comparison of levels measured in the past to the criteria under consideration can give one an impression of the likelihood of considering ashes and sludges as hazardous. The criteria listed in Tables 36 and 37 are 100 times the USEPA (1976) quality criteria for water, analogous to the criteria given in Table 35.

It appears likely that few ashes and FGD sludges would be considered toxic. Ash is the most likely to be considered hazardous; maximum levels in ash leachate greatly exceeded criteria for arsenic, cadmium, chromium, lead, and manganese. Leachate from FGD sludges also occasionally exceeded criteria for iron and manganese. It is apparent, however, that leachate concentrations of contaminants are quite variable from case to case because makeup of the leachate is highly dependent upon the nature of the boiler operation, the coal burned, the emission-abatement techniques, and the storage methods. Thus, a generalization of the nature of these wastes cannot be made. Classification of these wastes as hazardous will probably require site-specific evaluations, and even this classification may change as operation parameters are modified during the course of a combustion facility's lifetime.

Table 35. Proposed USEPA Toxicity Criteria for Classifying Waste as Hazardous[a]

Contaminant	Criterion Concentration in Extract (mg/L)
Arsenic	5.0
Barium	100.0
Cadmium	1.0
Chromium	5.0
Lead	5.0
Mercury	0.2
Selenium	1.0
Silver	5.0
Endrin	0.02
Lindane	0.40
Methoxychlor	10.0
2,4-Dichlorophenoxyacetic acid (2,4-D)	10.0
2,4,5-Trichlorophenoxypropionic acid (2,4,5-TP)	1.0

[a] Data from USEPA (1980). Criteria are 100 times the USEPA National Interim Primary Drinking Water Regulations.

Table 36. Maximum Concentrations of Chemicals from Coal Combustion in Various Waste Systems and Potential Criteria for Hazardous Waste Classification[a]

	Concentration (mg/L)					
Element or ion	Fly ash pond liquor	Bottom ash/slag pond liquor	Fly ash overflow	Ash pond leachate	Sludge leachate	Criterion[b]
Antimony	0.012	0.012	0.03	0.03	-	5
Arsenic	0.023	0.015	0.02	0.084	0.30	5
Barium	0.40	0.3-3.0	0.30	40.00	2.00	100
Beryllium	0.02	0.01	0.003	0.003	1.00	1
Boron	24.60	24.60	1.03	16.90	40.00	75
Cadmium	0.052	0.025	0.04	0.01	0.047	1
Calcium	180.00	563.00	-	1.00	-	-
Chromium	0.17	0.023	0.139	0.092	0.25	5
Copper	0.45	0.14	0.09	17.30	0.56	100
Fluoride	1.00	14.85	10.40	<0.10	-	200
Lead	0.20	0.08	-	0.024	0.039	5
Manganese	0.63	0.49	0.02	<0.002	-	5
Mercury	0.0006	0.006	0.0002	0.015	0.07	0.2
Molybdenum	-	0.49	0.10	0.69	-	-
Nickel	0.13	0.20	0.015	0.046	0.05	-
Selenium	0.004	0.05	0.015	0.47	0.54	1
Vanadium	-	0.02	0.20	<0.20	0.20	-
Zinc	2.70	0.16	2.50	0.19	4.20	500

[a] Data from Hart and DeLaney (1978).

[b] Criteria are 100 times the quality criteria proposed by USEPA (1976).

Table 37. Concentrations of Constituents of Ash and Sludge Liquors and Leachates Relative to USEPA Quality Criteria for Water[a]

	Concentration (mg/L)						
Element or ion	1× Standard	100× Standard	Bottom ash liquor	Fly ash liquor	Combined ash liquor	Fly ash Leachate	FGD Leachate
Arsenic	0.05	5	0.006-0.018 (0)[b]	0.01-1.1 (0)	0.005-0.10 (0)	0.006-72.9 (2)	0.0045-0.34 (0)
Barium	1	100	0.1 -0.2 (0)	0.2,0.3 (0)	0.1 -0.2 (0)	-	0.14 -1 (0)
Cadmium	0.01	1	0.001-0.003 (0)	0.001,0.037 (0)	0.001-0.005 (0)	0.15 -1.1 (1)	0.002 -0.06 (0)
Chloride	250	25,000	0.8 -7 (0)	6,7 (0)	3 -14 (0)	40,120 (0)	0.2 -2820 (0)
Chromium	0.05	5	0.009-0.01 (0)	0.02,0.067 (0)	0.004-0.043 (0)	1.9 -9.4 (1)	0.002 -0.11 (0)
Copper	1	100	0.041-0.065 (0)	0.02,0.31,2.4 (0)	0.01 -0.08 (0)	0.36 -15.6 (0)	0.005 -0.98 (0)
Cyanide	1	500	-	-	0.01 -0.05 (0)	-	-
Fluoride	∿2	∿200	-	-	-	-	0.05 -3.1 (0)
Iron	0.3	30	5.29,5.98 (0)	1.44,1.93,630 (1)	0.23 -2.3 (0)	0.1 -20 (0)	0.035 -93 (6)
Lead	0.05	5	0.02 (0)	0.01,0.06,0.91 (0)	0.01 -0.025 (0)	0.73 -9.1 (1)	0.005 -0.25 (0)
Manganese	0.05	5	0.16 -0.58 (0)	0.13,0.48 (0)	0.01 -0.39 (0)	∿17 (1)	0.001 -6.5 (5)
Selenium	0.01	1	0.002-0.01 (0)	0.002,0.15,0.33 (0)	0.003-0.065 (0)	-	0.002 -0.022(0)
Silver	0.05	5	0.01 (0)	-	-	-	0.001 -0.038(0)
Sulfate	250	25,000	49 -139 (0)	209,358 (0)	59 -156 (0)	17.5 -ca.200 (0)	21 -5760 (0)
Zinc	5	500	0.09 -0.14 (0)	0.06,1.51,2.2 (0)	0.03 -0.12 (0)	0.26 -25.7 (0)	0.01 -3.1 (0)

[a] Data from Weeter and Bahor (1979). Ranges are given where there were four or more samples.

[b] Number of cases in which concentrations exceeded 100 times the standard of USEPA (1976).

Implications for Fish and Wildlife Resources

The U.S. Congress passed RCRA in order to protect human health and the environment from the deleterious effects of hazardous solid wastes. Implicit in the protection of the environment is protection of the nation's fish and wildlife resources. The primary thrust of the regulations promulgated and proposed under RCRA is to contain toxic wastes in the storage area. In general, this should lead to a reduction in the amounts of hazardous material reaching areas where they might affect fish and wildlife resources.

The major effect from the storage of ash and sludge wastes is loss of habitat, at least for the duration of the storage site's use. Regulations promulgated under RCRA will not substantially alter this effect for most areas. However, the use of some areas as storage sites will be restricted. The USEPA Criteria for Classification of Solid Waste Disposal Facilities and Practices (USEPA 1979a) indicate that a storage facility shall not adversely impact an endangered or threatened species. This will tend to protect resources associated with these species, but will probably not add substantially to the protection provided under the Endangered Species Act (as amended). Requirements restricting the siting of storage sites in floodplains will reduce impacts to the biotic resources of these habitats. This may be of particular import in the western states where floodplains support riparian habitats that provide nesting and foraging areas for a variety of fish and wildlife species (Johnson and Jones 1977). Under the proposed regulations, hazardous waste-storage sites may not be located in a wetland habitat unless appropriate permits can be obtained under authority of the Federal Water Pollution Control Act. The USEPA has not imposed such a restriction on the siting of nonhazardous waste-storage facilities. Use of wetlands as storage sites could result in loss of these areas as fish and wildlife habitat. The future impact from utility wastes is dependent upon the ultimate classification of those wastes.

The proposed and promulgated regulations call for ensuring the integrity of waste impoundments by siting and design methods. Additionally, discharges into surface waters must comply with regulations promulgated under the Clean Water Act. This will reduce the potential for adverse effects upon water quality and upon fish and wildlife using the receiving waters. One potential problem that has not been addressed in the framework of RCRA is use of impoundments by biota. Fencing to prevent access by unauthorized humans and livestock will prevent large mammals from using impoundments. However, smaller terrestrial wildlife and flying animals may still make use of open impoundments. The extent to which ash and sludge liquors would be deleterious to biota ingesting them remains uncertain. The potential hazards should be considered in the promulgation of further guidelines.

The interim regulations for closure of hazardous waste sites call for ensuring that the wastes will not come into contact with humans or the environment; they do not explicitly call for revegetation. The extent to which development of wildlife habitat occurs will depend upon the goals and success of any revegetation program. If revegetation is not initiated after closure, storage sites for coal ash and FGD sludge may be lost from use by biota for an extended period after storage has ceased.

Other Laws

There are several other federal laws that affect the production and storage of coal combustion and emission-abatement wastes. Several of these overlap with RCRA. The USEPA guidelines under RCRA must integrate with regulations promulgated under these laws. Four of these laws with import to ash and sludge wastes are the Powerplant and Industrial Fuel Use Act (Pub. L. 95-620), the Clean Air Act (Pub. L. 90-148, as amended), the Federal Water Pollution Control Act (Pub. L. 92-500, as amended), and the Safe Drinking Water Act (Pub. L. 93-523, as amended). The goal of the Fuel Use Act is to decrease the nation's dependence upon oil and natural gas. The major impact of this legislation will be to increase the use of coal as a fuel; increased coal use will result in an increased need to dispose of ash and FGD sludge (U.S. Dep. Energy 1979). Under regulations (40 CFR 60) promulgated under recent amendments to the Clean Air Act, new coal-fired utilities will have to use FGD systems regardless of the type of coal used. Thus, these regulations will increase the need for sludge waste-storage sites. Waste-storage sites must comply with regulations restricting pollutant discharges into surface waters. The USEPA has promulgated these effluent regulations in response to the Federal Water Pollution Control Act guidelines, and standards have been promulgated (40 CFR 423) to control discharges from combustion and emission-abatement waste-handling systems in steam-electric power stations. These standards limit the volume of total suspended solids, oil, and grease which may be discharged to bodies of water. In addition, storage sites must not contaminate drinking water sources. Standards for drinking water have been promulgated by USEPA under the Safe Drinking Water Act (40 CFR 141 and 143). These standards have been used by USEPA as maximum contaminant levels for groundwater quality in determining whether solid-waste facilities pose a threat to health or environment.

SITING AND DESIGN CONSIDERATIONS

Generalized procedures for siting and design of storage facilities for coal ash and FGD sludge wastes are presented in this section. It is intended to impart to the reader an understanding of those considerations most important in the siting and proper design of impoundments and landfills for disposal of wet sludges and dry-processed wastes.

Siting Considerations

Selecting a storage site involves consideration of many individual factors, the relative importance of which varies from region to region (D'Appalonia Consulting Eng. 1975; Duvel et al. 1979). There are four categories of factors (evaluation criteria) that may influence the choice of a storage site: engineering, environmental, institutional and political, and economic (Table 38). These criteria are interdependent and in many instances overlap.

Table 38. Evaluation Criteria for Selection of Waste-Storage Sites

Engineering	Environmental	Institutional and Political	Economic
Physical size	Surface water	Legal and regulatory Local State Federal	Transportation costs
Location	Groundwater		Property-acquisition costs
Site access	Soils		Site-development costs
Topography	Air	Political and public acceptance	Processing costs
Geology	Terrestrial and aquatic ecology		Closure costs
Soils	Noise		Salvage value
	Land use		
	Scenic and aesthetic effects		
	Recreation		
	Cultural resources		

Engineering criteria. Most engineering criteria are fundamental and generally inflexible; a potential site must meet these minimum requirements to be considered suitable (Duvel et al. 1979). Principal engineering criteria are:

- Physical size--Acreage must be available to accommodate the wastes generated over the operating life of the plant, including space for ancillary facilities and an approximately 30-m buffer strip.
- Location--The site should be near the utility plant site in order to minimize transport problems.
- Site access--The site must be accessible for construction and delivery of wastes during operation.
- Topography--The site should be developable with a minimum of earthmoving, taking advantage of natural topography as much as possible and taking into consideration waste-storage plans.
- Geology--Geological hazard areas should be avoided, and the substrate must be capable of supporting necessary facilities.
- Soils--There should be an adequate depth of soil between the base of the storage area and bedrock (draft RCRA guidelines suggest a 3-m minimum).

Environmental criteria. In general, environmental criteria are as important in site selection as engineering criteria but are less rigid (Duvel et al. 1979). Often it is more feasible to make tradeoffs or mitigate adverse environmental impacts than to overcome engineering constraints. Environmental factors that should be considered are:

- Surface water--The site should be located outside the 100-year floodplain and/or coastal zones to prevent inundation. Surface waters should be avoided, or, where unavoidable, surface water should be diverted around the site.
- Groundwater--The potential for leachate seepage to groundwater should be minimized or prevented. This is particularly important where an underlying aquifer is either currently or potentially useful as a water supply. Contamination is less likely where the difference between water table elevation and bottom of the landfill or pond is large (e.g., 1.5-m minimum). The potential impact on water supplies should also be considered (USEPA draft guidelines recommend 500 ft of distance between storage sites and water supplies--surface water and groundwater). In addition, groundwater impacts would be minimized by locating away from groundwater recharge areas, particularly sole-source aquifers.
- Soils--Siting should consider capability of soils for agriculture and avoid disturbing prime agricultural soils.
- Air--Wind velocity and direction should be considered with regard to minimizing the effects of wind erosion, particularly the impacts of fugitive emissions from dry transport and storage.
- Terrestrial and aquatic ecology--The alteration or destruction of unique habitat should be avoided, and detrimental effects on rare, endangered, or commercially valuable species (resident or migratory) must be prevented.
- Noise--The impacts from transportation (truck or rail traffic), construction, and operation should be minimized.
- Land use--Areas of conflicting land use should be avoided. The storage site should have the capability for successful restoration at the end of its active use.
- Scenic and aesthetic effects--Sites should provide as little visible intrusion on the horizon as is possible.
- Recreation--Areas of recreational activity should be avoided or screened.
- Cultural resources--Unique archeological, historical, and paleontological areas should be avoided or steps taken to mitigate impacts.

Many of these criteria can be easily satisfied by selecting storage sites in isolated areas. The benefits of obtaining isolation, however, must be balanced against the cost (longer transport distance) and a higher potential for impacting fish and wildlife resources.

Institutional and political criteria. In addition to physical and environmental considerations, storage sites must meet legal and regulatory restrictions and/or requirements and be politically and publicly acceptable (Duvel et al. 1979). Expansion of existing storage sites has historically been made without difficulty because the regulatory agencies (and the public) have become conditioned to their presence. However, current attitudes are shifting toward a more rigorous analysis of new sites. Regulatory agencies that may have jurisdiction over a waste-storage site include:

- Local--Health departments, zoning commissions, and soil and water conservation districts.
- State--Energy agencies, health departments, highway departments, environmental protection agencies (or equivalents), and departments of natural resources (or equivalents).
- Federal--U.S. Environmental Protection Agency and U.S. Army Corps of Engineers.

In addition to permits required by regulatory agencies, regulations promulgated under RCRA and state solid-waste laws also place siting constraints upon waste management plans. In general, these constraints restrict sites from being located in areas of high potential for catastrophic release of wastes or in areas of high biological sensitivity. The primary effect of the regulatory constraints is to decrease the likelihood of waste constituents being dispersed into the environment.

Economic criteria. Project costs are always an important consideration in evaluating alternative sites. These include:

- Transportation--In consideration of this major cost factor, the storage site should be located close to the power plant to minimize transportation costs. Other factors to consider are the mode of waste transport (truck, rail, pipeline) and the route.

- Property acquisition.
- Site development and construction.
- Processing--Includes costs for dewatering, stabilization, or fixation.
- Maintenance of storage site.
- Closure--Reclamation and restoration costs.

Reviewing alternative sites. The recent guidelines from the Council on Environmental Quality for preparing environmental analyses (40 CFR 1502) emphasize the need for adequate site selection. The waste manager's siting decision results from the integration of all siting criteria. However, when the fish or wildlife biologist reviews the siting plan for a waste-storage facility, concern centers upon the ecological criteria. The biologist must bear in mind that the ecological factors are closely tied to the other siting factors. For example, a site located in a geologically hazardous area (e.g., floodplain) could pose a threat to fish and wildlife resources should there be catastrophic release of waste materials. In reviewing the plans, the biologist must consider those nonbiological factors that determine the ability of the waste-management operation to contain the wastes.

Existing and/or planned land uses at or in proximity to a proposed site necessitate considering numerous potential impacts that can only be evaluated on the basis of qualitative or comparative estimates. Further, the various land uses represent differing degrees of constraints regarding acceptable use of a site for waste storage. In some instances, the present or planned use of an area is an absolute or limiting land-use constraint. For example, the designation of wildlife refuges and sanctuaries essentially precludes opportunities for waste storage. The presence of state and/or federally designated threatened and endangered species, as well as critical habitat of such species, may also constitute constraints. Cultural resources of protected status include officially designated historic and archeological sites, monuments, scientific study areas, and various categories of wilderness and natural areas. Local, state, and federal forests, parks, and other types of recreation areas also will usually be unavailable for waste storage. The known presence of underlying mineral resources may also be a limiting constraint.

Evaluating impacts on land types with fewer limitations usually involves consideration of less specific or provisional use constraints. For example, the significance of impacts on agricultural lands varies in accord with the productive potential and management requirements of the land. Thus, from an agricultural viewpoint, the greater the proportion of prime and unique farmlands in a given area, the greater the land-use impact. Similar relationships exist with respect to variations in the productive potential of forestlands and rangelands. In general, wetlands and floodplains are marginal sites for waste storage because of the high degree of interconnection with other ecosystems, thereby facilitating dispersion of contaminants from the waste mass. However, development of storage facilities may cause changes in the hydroperiod or amplitude of water-level fluctuations in adjoining wetlands or floodplains. Such changes may degrade or enhance the environment depending on the intended use of these areas. The development and operation of waste-storage facilities in a proposed area may also generate land-use conflicts involving existing highways, railroads, pipelines, and/or land drainage systems. In some instances, relocations will be necessary; in others, additional development of such facilities will be required.

Aside from the preceding considerations, the imposition of another land use (waste storage) in a given area results in increased competition for use of the local land resource. With increasing competitive pressure, land-use intensity tends to increase to a level at which the quality of the area as fish and wildlife habitat is degraded. The degree to which development of a proposed waste-storage site will influence land-use intensity in adjacent areas is not readily predictable. However, the market values of local lands generally reflect the inherent productivity of soils and, among other factors, the competition for use of such lands for agriculture and various other purposes. Thus, market values can be used as a crude index for anticipating potential impacts resulting from increased competition for land. Additionally, the value of the area as fish and wildlife habitat must be evaluated. For a given area, comparatively low land values provide some assurance that imposition of waste-storage facilities will not greatly intensify other uses of adjoining areas; under otherwise comparable conditions, high land values may cause excessive exploitation of the resource. The effect of excessive exploitation on fish and wildlife resources is magnified as the habitat value of the site increases.

Design Considerations

The design procedures described herein are essentially those that have long been used to design reservoirs and/or water impoundments and to construct secure landfills (U.S. Bur. Reclam. 1973; D'Appalonia Consulting Eng. 1975; Duvel et al. 1979). The dominating factors are the properties of the construction material used and the geologic and hydrologic characteristics of the storage site. In planning, emphasis should be placed on the long-term structural stability of the storage facility design. The design of storage facilities is a major determining factor in ensuring containment of waste materials that may pose a threat to fish and wildlife resources. (See Figures 6 and 7 for pond and landfill configurations.)

Pond designs. The chosen design of a pond for the storage of wastes must be compatible with the physical conditions of the selected site, the available construction materials, and the physicochemical properties of the waste to be impounded (Duvel et al. 1979). Otherwise, the risk of dispersal of waste materials to the environment is increased.

A majority of the pond configurations will involve the design and construction of a dam or dike, commonly of earthfill, to form the impoundment (Figure 6). Most embankments constructed for the storage of wet wastes may be categorized as small to intermediate in size, and emphasis will be placed on these sizes for the purpose of this discussion. An excellent reference for embankment design is "Design of Small Dams" by the U.S. Bureau for Reclamation (1973). Some states, however, have more rigorous dam safety requirements (Duvel et al. 1979).

The design of waste-storage ponds requiring dike or dam construction is very similar to the design of water-supply impoundments (Duvel et al. 1979). As a result, the design criteria for coal waste-storage ponds is similar to that of any small dam and must provide for the safe impoundment of waste materials during all phases from construction through abandonment and successful reclamation of the site. The following general design criteria apply:

- The impoundment must have sufficient capacity and/or outlet structures to prevent embankment overtopping.
- The stability of embankment slopes must be maintained during all phases of the storage operation.
- Foundation materials must be capable of supporting the loads imposed by the embankment, including full design capacity.
- Seepage through the embankments and foundations must be controlled to prevent internal erosion and maintain embankment integrity.
- Exposed surfaces of embankments must be protected against wave, wind, and runoff erosion.

The above design criteria will prove satisfactory only if proper construction methods and field supervision and control are implemented.

Landfill designs. The primary considerations for the design of a landfill storage operation are stability and the environmental impact of failure (Duvel et al. 1979; GAI Consultants 1979). The major design elements for these considerations include development of methods to (1) reduce or eliminate contamination of surface waters and groundwaters caused by seeping leachates and surface runoff; (2) prevent landfill instability, such as landslides, slumping, and erosion; and (3) abandon and reclaim the landfill in a manner that will satisfy the preceding two criteria over the long term.

The design of the landfill will be based in part on the physicochemical properties of the waste and on the topography, geology, and hydrology of the storage site. Fixation/stabilization of sludges is necessary not only to obtain a soil-like material suitable for handling and placement in a landfill, but also to improve stability and reduce permeability and leachability of the waste. Ashes are considerably easier to handle in landfills. The geology and hydrology of the storage site play an important part in the design of a stable and environmentally acceptable landfill. Of major concern are groundwaters and surface waters within and adjacent to the landfill. The proper management of these waters is important because they are the primary means of transporting pollutants and generating leachate. The degree of contamination will depend on the amount of water that passes through or is in contact with the waste material. Secondly, the properties of the landfill foundation materials and the influence of subsurface geology (e.g., solution features and deep mining) below the storage site are important to embankment stability.

The stability of the embankment is especially important in maintenance of waste containment. The landfill design and management should include measures to prevent or mitigate the following potential hazards: (1) slope failure (landslide slumping or sloughing); (2) sloughing or shifting of the fill that would block or restrict flow, creating a temporary impoundment that could either release a hazardous floodwave after breaching the temporary impoundment or enhance seepage through the wastes; and (3) impounding of water during some stage of development, which could weaken the embankment to the point of failure. The measures that are necessary to ensure facility integrity depend on the conditions at each site. More detailed discussions of these measures can be found in Duvel et al. (1979) and GAI Consultants (1979).

Operating and Monitoring Storage Sites

Operation. In general, the method of landfill operation is similar for all sites (Duvel et al. 1979; GAI Consultants 1979). Waste is brought to the site (by truck, conveyor, rail, or other means), spread into thin layers (usually less than 2/3-m thick), and compacted for maximum density and strength. To maximize equipment utilization and to control dust or mud, the spreading is confined to as small an area as possible and the fill is placed in relatively thin, horizontal layers.

Ponds can be designed for either interim or long-term use. The purpose of some ponds is liquid-solid separation prior to removal of the solids to long-term ponds at another location for ultimate storage. The long-term ponds will eventually be drained, covered, and vegetated, or otherwise reclaimed to a beneficial use.

Redundant pumping facilities should be available for transporting waste to the pond(s) and for supernatant return. Backup pump operation should be automatic and tested routinely. Portable, gasoline-driven emergency pumps should also be available at the pond site.

Dust control measures must be employed as appropriate. Usually a simple spray truck with a spraybar at the back can control road dust. Calcium chloride crystals are effective in some cold, dry areas. For controlling dust at the working area, a periodic, fine water spray will reduce dust. Wind erosion can be reduced by various methods--e.g., planting wind breaks, using chemical sealants, and providing natural vegetative cover.

Operation and monitoring are interrelated. Monitoring results may influence future operations at a storage site.

Monitoring program. Any waste-storage area, whether pond or landfill, should be the subject of an ongoing monitoring program to (1) provide warning that the site is not being managed properly or that a malfunction has occurred, (2) satisfy the requirements of regulatory agencies, (3) determine if the design concepts were appropriate or indicate where changes are needed or when maintenance is required, and (4) ensure that the environmental acceptability and structural stability of the site are maintained (Duvel et al. 1979).

The monitoring program should be implemented before construction has started, in order to obtain base values and to record original conditions. It should be continued during operation and be maintained for some period after closure and reclamation. Comparison of the base values to ongoing monitoring results may provide a good indication of the effects of the storage operation over the active period. Monitoring should include:

- Sampling and analysis of water from monitoring wells, runoff, and underdrains.
- In-place measurement of the density of landfilled waste material.
- Pipeline leak detection.
- Maintenance of records from surveys and instrument readings in dikes or dams.
- Visual observation of general site conditions.

Sampling and analysis of water. The purpose of monitoring the groundwater (aquifer) is to determine if there has been migration of leachate from a storage area (Duvel et al. 1979). Currently, there are few explicit guidelines for the number, location, or depths of monitoring points. These parameters are dependent upon the nature of the site being monitored.

The first step in establishing a monitoring network is to determine the direction of groundwater flow, based on geologic and hydrologic data obtained during site selection. In a typical monitoring network (as shown in Figure 19), there is one sampling location upgradient at point "A", one at the storage area "B", and three at downgradient points "C". The locations shown as points "A" and "C" could be equally applicable to pond storage. The number of downgradient locations chosen should be sufficient to give a representative sampling of the area. In the case of expanding landfills, additional sampling points would be necessary as the area is enlarged. The "C" points should be located near enough to the storage area to provide "early warning" if leaching occurs.

To avoid contamination of monitoring wells, casings should be of polyvinyl chloride (PVC) rather than steel. Because single-depth sampling could miss contaminated zones that may be present, clusters of wells should be drilled to various depths. Samples can be removed by bailing or pumping.

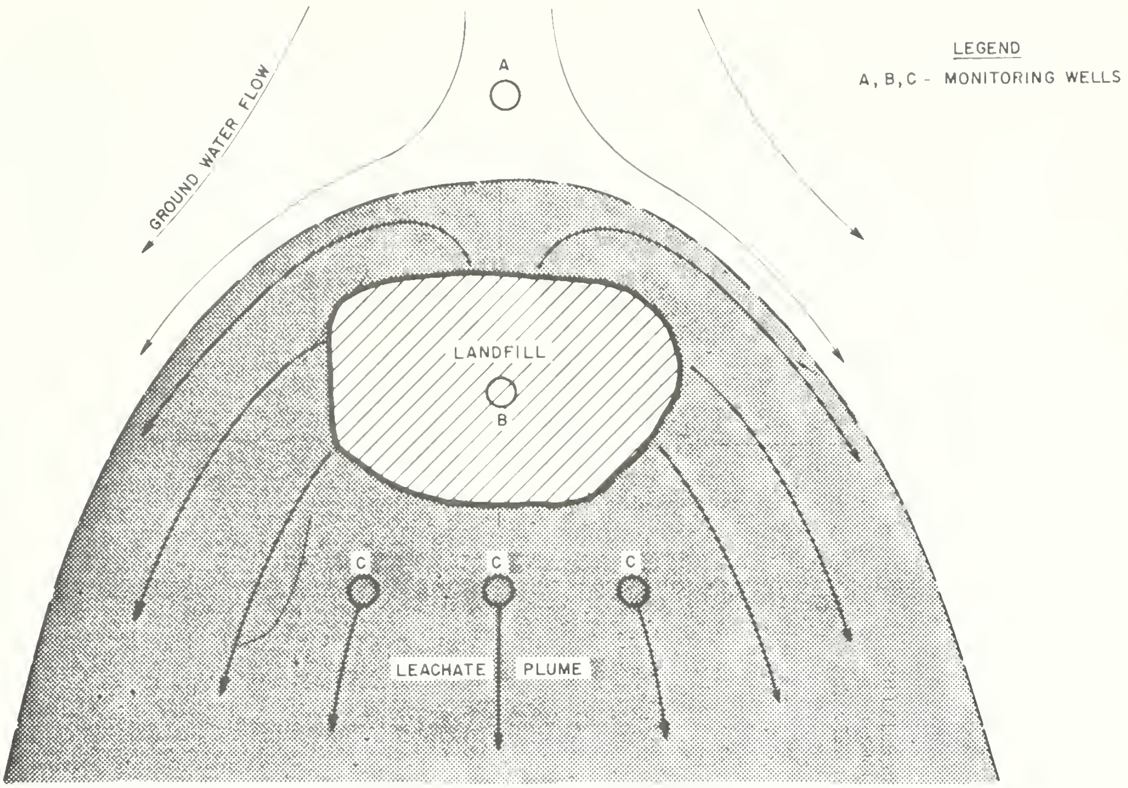

Figure 19. Typical Monitoring Network. From Duvel et al. (1979).

Runoff water from landfills should be channelled to prevent surface washing and erosion. As currently required by some agencies, runoff is directed to stilling or settling basins, and samples of influent and effluent can be taken for analysis. If not channeled to a basin, runoff can be sampled directly at the drainage facilities. If a pond or landfill is fitted with an underdrain, samples should be taken of the effluent at the point of discharge.

In practically all discussions of leachate or runoff quality, comparison to drinking water standards is made. A possible list of parameters for analysis includes: pH, specific conductance, alkalinity, acidity, As, Ba, Cd, Ca, Cr, Fe, Pb, Mg, Hg, Se, Ag, sulfate, sulfite, total dissolved solids, and chemical oxygen demand. Certain regulatory agencies may require additional or other analyses. In order to gain credence and uniformity in the analytical results, USEPA-endorsed procedures should be employed.

Physical measurements. The density of landfilled material should be periodically determined to ensure that the proper degree of compaction and optimum strength is being obtained. Typical dry bulk densities (in kg/m^3) for utility wastes are: compacted fly ash, 1120-1440; Calcilox-stabilized sludge, 530-700; and Poz-O-Tec, 1120-1440 (Duvel et al. 1979). For a particular sludge or mixture, the optimum density should be predetermined in the laboratory.

Prompt discovery of pipeline or impoundment leaks is essential in order to avoid (1) adverse impacts resulting from mobilization of waste constituents into the environment, (2) fugitive dust from dried spills, (3) damage to the adjacent area, and (4) erosion damage to containment components. Leaks may be detected by instrumental or visual means. Pipeline pump pressure should normally be logged or recorded, so that personnel can be alerted to the significance of a sudden drop in pressure and initiate an immediate check for leaks. Instrumentation may also be used to monitor impoundment integrity. Frequent walking inspections can also reveal leaks, especially those that are too small to register a measurable change in containment integrity.

Monitoring requirements for storage sites can be expected to change rapidly and radically as the regulatory agencies develop and implement regulations. This may be especially true relative to RCRA, although it is not currently known if, or to what extent, state waste management programs based on this act will affect sludge storage sites and methods. Close contact with the pertinent state agencies should be maintained in order to incorporate new developments.

LINING ASH AND SLUDGE STORAGE PONDS

For the purposes of this report, a storage pond liner is defined as any natural or synthetic material purposely placed on the inside surface of an impoundment basin to reduce seepage from the basin. Most existing coal ash and FGD sludge ponds are unlined; however, compliance with increasingly stringent local, state, and federal water quality control regulations may require that new ponds be lined (Dvorak et al. 1978). The necessity for a liner is dependent upon the properties of the ponded effluents, the quantity and chemical quality of potential leachate, the impacts of seepage, the geology and geography of the site, the availability of process water, and the regulations governing seepage. There are five major categories of liners:

1. Flexible synthetic liners (sometimes reinforced with nylon, dacron, glass fiber):
    a. Polyvinyl chloride (PVC)
    b. Polyethylene (PE)
    c. Polypropylene
    d. Butyl rubber
    e. Chlorinated polyethylene (CPE)
    f. Ethylene propylene rubber (EPDM)
    g. Chlorosulfonated polyethylene (Hypalon)
    h. Neoprene

2. Admixed materials:
    a. Asphalt concrete
    b. Soil cement
    c. Sprayed asphalt membranes
    d. Gunite

3. Soil sealants:
    a. Chemical sealants
    b. Rubber latex
    c. Bituminous sealcoat

4. Natural soil systems:
    a. Soil/bentonite
    b. Compacted clays
    c. Compacted soils

5. Stabilized wastes

These liners are briefly discussed below; more detailed characterizations can be found in the references cited at the end of this section.

Flexible Synthetic Liners

Flexible synthetic liners are the only "impermeable" liners. They are manufactured as long continuous sheets which can be sealed at the edges so that each liner exactly fits the pond. Flexible liners rely upon the earthen structure for support. Most are conditionally guaranteed by their manufacturer for 20 years. Flexible liners may be vulnerable to puncture (especially during installation), aging with exposure to sun or temperature extremes, reaction with ponded wastes, and stresses from trapped gases or groundwater. A cover of 15 to 30 cm (6 to 12 inches) of soil will protect a flexible liner from puncture by traffic.

Flexible synthetic liners vary considerably in physical properties, chemical compatibilities, installation, durability, and cost. Plastic liners are particularly popular because they are relatively inexpensive. Polyethylene (PE) was the first material to be widely used. It has since been replaced in popularity by polyvinyl chloride (PVC) which has higher strength, a good adhesive system, greater abrasion resistance, and other desirable qualities (Kays 1977). Chlorinated polyethylene (CPE) is less affected by sunlight than PE or PVC, and it is inert and does not readily react with wastes. However, because CPE also does not readily react with adhesives, plies of the liner and CPE seams do not bind well. Chlorinated polyethylene has been used for the sides of some PVC-lined ponds to take advantage of the best qualities of both liners.

Rubber liners include butyl, ethylene propylene diene monomer (EPDM), neoprene, and elasticized polyolefin. Hypalon has properties of both the rubber and the plastic liners. Butyl rubber is considerably more elastic than the plastic liners, and can resist extreme temperatures [-45° to 93°C (-50° to 200°F)] without loss of flexibility or strength. Unfortunately, it is very difficult to make seams in butyl rubber that are stronger than 60% of the strength of the material (Clark and Moyer 1974). The EPDM liners are susceptible to shrinkage when exposed to sunlight. Hypalon also tends to shrink, but this can be controlled by sandwiching Hypalon plies around reinforcing fabric. Hypalon is one of the most inert liners. Neoprene is not often used because of its cost, seldom-needed specialized properties, and poor sun aging characteristics. The polyolefin lining is unique because high-quality seams can be formed by heat welding, even when the liner is not clean, during a drizzle, or through a wide range of air temperatures. Its resistance to chemicals exceeds that of any other common lining system, and it has good resistance to sun aging (Kays 1977).

Admixtures

Materials such as gunite, asphalt concrete, and soil cement are categorized as admixed liners (Dvorak et al. 1978). They provide some structural support rigidity as well as reducing pond seepage, but they are not impermeable. The major disadvantage of rigid liners is their susceptibility to fracture under seismic, hydrostatic, thermal, and weathering stresses.

Gunite is a concrete mixture that may be sprayed on the embankment walls. It does not have construction joints and is thinner and less expensive to apply; otherwise, a gunite liner performs similarly to a concrete liner. Neither material is impermeable.

Asphalt concrete is a controlled mixture of asphalt cement and graded aggregate that is placed and compacted at elevated temperatures. It is especially well adapted to the construction of linings for all types of hydraulic structures. It may be used for the entire lining or the major part of a more complex lining. Depending on mix and placement, it may serve as an impermeable or porous layer. Properly prepared, asphalt concrete forms a stable, durable, and erosion-resistant lining (Stewart 1978).

Asphalt membrane linings (hot-sprayed type) consist of a continuous layer of asphalt, usually without filler or reinforcement. It is generally covered or buried to protect it from mechanical damage and weathering (oxidation). Its cover may be another layer of multilayer lining structure, but generally it is native soil, gravel, asphalt macadam, or other substance. Asphalt membranes are placed to thicknesses of about 0.5 to 0.8 cm (3/16 to 5/16 inch) and constitute continuous waterproof layers extending throughout the area being lined. Asphalt of special characteristics can form tough, pliable sheets that readily conform to changes or irregularities in the subgrade. Buried under a protective coating, an asphalt membrane can retain its tough, flexible qualities indefinitely. It is one of the least expensive types of liners currently available (Stewart 1978).

Soil cement may make a suitable liner for soils that are less than 5% silt and clay. Soil cement decreases erodibility and increases shear strength; however, transverse shrinkage cracks may develop as the soil cement dries. Therefore, soil cement is not frequently used as a liner.

Soil Sealants

Chemical sealants and soil additives seal the impoundment basin by filling soil interstices or by causing reactions that reduce permeability (Dvorak et al. 1978). Chemical sealants may be applied by spraying, mixing with soil, or as additions to the waste stream inflow. Chemical sealants are not always effective, due in part to soil nonhomogeneities and in part to the sealant itself.

Sodium carbonate, sodium silicate, and sodium pyrophosphate have been tested as sealants. These chemicals act to increase the sodium-to-calcium ratio in a soil, thus dispersing the soil and decreasing its permeability. Sodium carbonate has been demonstrated superior to the other two chemicals; its seal may remain effective for up to five years (Clark and Moyer 1974). Polyphosphates such as tetrasodium pyrophosphate (TSPP), sodium tripolyphosphate (STPP), and sodium hexametaphosphate (SPP) give optimal results when mixed with clay soil and compacted to 90% of maximum density (Kays 1977). Other possible chemical sealants include Zeogel (an attapulgite clay), sprayable polymer compounds, and rubber latex.

Polymeric sealants, however, do not provide structural strength. Impoundment sites that have not been compacted will still be weak following treatment with sealants. Impoundments must be compacted (Stewart 1978).

Rubber latex was studied as a sealant to control acid mine drainage (Stewart 1978). Apparently the latex seal only penetrated 25.4 cm (10 inches) of soil and could not be properly tested. Further investigations, however, may prove rubber latex to be a suitable sealant for waste impoundments (Stewart 1978).

Bituminous seal coating is used to seal the surface pores of an asphalt lining or to provide additional waterproofing. It is also considered where a reaction is anticipated between the aggregate in the mix and the liquid to be stored. There are basically two types of bituminous seals. One is simply an asphalt cement (or emulsified asphalt) sprayed over the liner surface, providing a film about 0.18 cm (1/32 inch) thick. The second consists of an asphalt mastic (25 to 50% asphalt cement, and a mineral filler such as limestone dust or a reinforcing fiber (e.g., asbestos); this mixture is generally applied at a rate of about 2.7 to 5.4 kg/m^2 (5 to 10 lb/yd^2) (Stewart 1978).

Natural Soil Systems

Soil/bentonite clay mixtures have been widely used to control excessive seepage in natural soils by decreasing their permeability. Bentonite, one of the most widely used high-swelling clay minerals, is a heterogeneous substance composed of montmorillonite and small amounts of other minerals. Bentonite has colloidal properties because of its very small particle size and the negative charge on the particles. Bentonite has the capacity of absorbing several times its weight in water and occupies a volume of many times its dry bulk volume at maximum saturation (Stewart 1978). This swollen mass fills the voids in soils that would normally permit water movement. High-swell bentonites are found in Wyoming, South Dakota, Montana, Utah, and California.

Clay liners are a special type of compacted soil liner. The clay may be borrowed from a pit near the pond site, or it may be purchased commercially. Montmorillonites, especially bentonites, are the preferred clays because of their expansive capabilities. Sodium-rich clays tend to disperse more readily than calcium- or magnesium-rich clays, and this may aid in reducing their permeability. Clay liners usually have low permeabilities; conductivities of 1.7×10^{-6} to 9.7×10^{-8} cm/s (1.8 to 0.01 ft/yr) are reported (Clark and Moyer 1974).

The level of salts found in certain wastes is often sufficient to reduce the swelling of bentonite and therefore impair its usefulness as a sealant. The use of a specially formulated form of bentonite (Saline Seal) reportedly ensures that after prehydration, the bentonite will remain swollen longer and will not deteriorate as rapidly when exposed to high salt concentrations (Stewart 1978).

Compacted soils from both the surface and subsurface in the impoundment area can be formed into a liner. Compaction decreases porosity, which in turn decreases permeability. Silty or clayey soils, tuffs, loesses, alluvium, and colluvium make some of the best compacted liners. Clean sands and gravels are poor because the permeability is only slightly decreased by compaction. Micaeous and expansive soils are difficult to compact. The greatest reduction in hydraulic conductivity is achieved when the soil is compacted in 0.15-m (6-inch) lifts to within 95% of the maximum dry density (Clark and Moyer 1974). Other than low costs, the biggest advantage of compacted soil liners is flexibility. They can withstand seismic activity and normal subgrade settlement and are usually stable in both wet and dry conditions. The primary disadvantage of compacted soil liners is their relatively high permeability compared to other types of liners. In regions with strict seepage regulations or where marked seepage effects are expected, these liners may be unsuitable.

Stabilized Wastes

Only fly ash seems to have potential value as a liner (Dvorak et al. 1978). Bottom ash is too coarse and permeable, and FGD sludges are difficult to dewater and stabilize. The average permeability of compacted (95% maximum dry density) fly ash varies between 5×10^{-5} and 1×10^{-4} cm/s, which is higher than that usually desired in a liner. Addition of a chemical sealant to fly ash could possibly reduce the permeability. The biggest advantage of this method is that it maximizes the waste-storage capacity by incorporating the waste in the pond embankments.

Compatibility of Liners and Wastes

The type of liner best suited for a given pond or impoundment containing ash and/or sludge wastes and their resultant leachates is determined by (1) allowable seepage rate, (2) availability of liner materials, (3) predicted durability and ability to resist attack from all chemical constituents, ozone, ultraviolet radiation, soil bacteria, mold, fungus, vegetation, and natural forces to which it will be exposed, and (4) cost. Although there is a lack of first-hand knowledge regarding compatibility to ash and sludge wastes, reports from various studies (Haxo 1976, 1978; Stewart 1978) on the compatibility of a variety of liners exposed to several types of wastes (industrial and municipal solid-waste leachate) provide general guidance for preliminary screening.

Haxo et al. (1979) studied flexible membrane liners relative to municipal solid-waste leachate. The reported effects of leachate immersion were small after eight months of immersion. The liners based upon chlorinated polyethylene, chlorosulfonated polyethylene, and neoprene tended to swell and soften more than the other materials. On the other hand, the polyolefins--such as polyethylene, polybutylene, elasticized polyolefin, and polyester elastomer--swelled and softened the least. Polyvinyl chloride membranes showed effects that approximated the latter. As a group, the polyvinyl chloride materials had the highest permeability to water vapor, whereas butyl rubber and elasticized polyolefin had the lowest. Permeability appeared to increase with time, probably due to swelling of the membranes with water (Haxo et al. 1979). In tests of the relative permeability of six liner materials, the order from least to most permeable was: elasticized polyolefin, polyvinyl chloride (#59), polyvinyl chloride (#11), polyester elastomer, chlorinated polyethylene, and polyvinyl chloride (#17). Liners of neoprene, chlorosulfonated polyethylene, and chlorinated polyethylene continually swelled during water-absorption tests;

however, polyethylene, polybutylene, polyester, polyvinyl chloride, and elasticized polyolefin liners reached a peak in the swell. One polyvinyl chloride liner hardened, indicating loss of plasticizer (Haxo et al. 1979).

Another study (Fry and Styron 1978; Styron and Fry 1979) on the compatibility of 18 liner materials exposed to two selected scrubber sludges was recently completed, but preliminary results are limited. The latest interim report (Styron and Fry 1979) lists and discusses the data collected after 12 months of exposing each of the 18 selected liner materials to two different FGD sludges. The report includes results of physical tests and chemical analyses at the initial stage of the study and at the end of the 12-month exposure period. A later report (expected soon) will include results of the 24-month data series. The specific liners selected for exposure testing with the two sludges are described in Table 39.

The results of the physical tests on the admix liners indicated that two of the materials, Guartec UF and M179, were obviously incompatible with both of the sludges (complete breakdown of these liners occurred). The two materials are considered unsatisfactory for sludge ponds. Five of the materials--Portland cement, lime, Portland cement plus lime, C400, and CST--exhibited increased unconfined compressive strengths. In general, the strength almost doubled; however, in the lime admix, the strength increased almost six times. The asphaltic concrete liner exhibited extensive cracks. The TACSS 020 and TACSS 025 liners suffered 5 to 25% decrease in unconfined compressive strength, indicating some degree of susceptibility to continuous exposure.

The breaking strengths of all spray-on and prefabricated membrane liners decreased with exposure time, but the percent elongation varied somewhat. Elongation increased significantly for total liner and decreased significantly for DCA-1295 and Aerospray 70. It remained essentially constant for T16, Dynatech, and Uniroyal.

The initial permeate water was sampled for chloride during the testing program. The concentration of chloride, which is not effectively attenuated by soil, is an indicator of how the sludge liquor is permeating the liners. In Tables 40 and 41, the tested liners are listed in order of increasing chloride content as determined from initial permeate analyses. The chloride in the initial permeate water samples consisted of contributions from soil pore water, material leached out of the liner, and the sludge liquor. The spray-on liners AC40 and Sucoat showed low chloride levels in the initial permeate samples from both sludges, suggesting that liquid was moving through the membrane along its entire cross section. After the 12-month exposure period, the AC40 liner had deteriorated so badly that it could not be taken intact from the test cell. The Sucoat liner had fractured completely and was unavailable for postexposure testing. Of the admixed materials, the cement/lime and CST liners showed very low chloride levels with sludges. These materials showed no evidence of local small leakage when the cells were examined after 12 months of exposure.

In summary, the data from the study of Styron and Fry (1979) suggest that Portland cement, cement plus lime, and CST when mixed with soil provide significant reduction in permeability to ash and scrubber sludge leachates. Further testing is being carried out to provide a complete picture of the usefulness of available liners in containing ash and FGD waste constituents.

CONTROLLING WIND AND WATER EROSION

Wind Erosion and Dusting

Methods for controlling wind erosion at coal ash and FGD sludge storage sites may be categorized as involving physical, chemical, and vegetative procedures. These methods basically modify the parameters that are used in the wind erosion equation. The utility of a given procedure will vary according to specific conditions of the site as well as whether the wastes are ponded or deposited as dry landfill. However, combinations of these procedures will usually be most effective in controlling wind erosion and dusting.

Physical methods. Wind barriers such as solid wood fences or snow fences, when oriented normal to the prevailing wind direction, are effective in reducing local wind velocity. The effectively protected area leeward of the barrier extends for a distance of about 15 times the height of the barrier (Woodruff et al. 1977). It may even be feasible to establish tree and/or shrub shelterbelts to control wind erosion. In some instances, temporary control can be effected by use of tillage equipment to produce, or bring to the surface, aggregates or clods large enough to resist wind force. Likewise, tillage equipment can be used to roughen or ridge the exposed surface to reduce wind velocity and trap windborne sediments. However, tillage is probably not applicable for most coal ash or FGD waste surfaces.

Many materials or substances have been used for physical stabilization of erodible surfaces. Application of water via mobile sprinkler units is the most commonly used method for controlling dust emisssions from unpaved haulageways, service roads, and other active work areas involving vehicular movement. It is also widely used as a general sitewide wind-erosion-control measure.

Table 39. Description of Liners Tested for Exposure to Two Flue-Gas-Desulfurization Sludges[a]

Liner material	Description
Admix liners	
Asphaltic concrete	Mix consisting of 1¼-cm (½-in.) (maximum) aggregate with an 11% asphalt content, compacted to 5-cm (2-in.) thickness
Cement	Type I Portland cement, applied at the rate of 10% of the dry soil weight
Cement/lime	Arbitrary combination of 4% (dry soil weight) Type I Portland cement and 6% (dry soil weight) hydrated lime (lime or calcium hydroxide [hydrated lime] is readily available and, when added to soil, reduces the volume change potential and renders the soil easier to compact)
C400	Applied at the rate of 15% of dry soil weight (C400 is a fine-ground powder produced in Japan and reportedly similar to cement with certain [unspecified] additives)
CST	Applied at the rate of 15% of dry soil weight (CST was applied at the same rate as C400 because no substantial differences were noted between the CST material and the C400 material)
Guartec UF	Applied at the rate of 4% of dry soil weight (Guartec UF is a fine powder reportedly having five to eight times the thickening ability of starch and swells to fill the soil voids; it is a highly refined gum produced by grinding the guar bean, a legume that is native to India but is now grown in northern Texas and southern Oklahoma)
Lime	Applied at the rate of 10% of dry soil weight, for comparison with Portland cement
M179	Applied at the rate of 16.5 MT/ha (45 tons/acre), about 4% of the dry soil weight (M179 is a preblended mixture of water-swellable polymers and bentonite and has been widely used as a sealant for reservoirs)
TACSS 020 and TACSS 025	Applied at a rate of 6% of dry soil weight (TACSS 020 and TACSS 025, blackish-brown transparent liquids produced in Japan, are proprietary liquid catalyzers used to adjust cure time)
Spray-on liners	
AC40	Applied at the rate of 3.4 L/m^2 (0.75 gal/yd^2) (AC40 is a refined asphalt material used for paving, industrial, and special purposes; requires a high temperature [150-200°C or 300-400°F] to flow)
Aerospray 70	Applied at the rate of 3.4 L/m^2 (0.75 gal/yd^2) (Aerospray 70 is a white polyvinyl acetate material weighing about 1100 g/L (9.2 lb/gal) and cures to form a clear flexible film; used to control erosion in areas of new vegetation)
DCA-1295	Applied at the rate of 3.4 L/m^2 (0.75 gal/yd^2) (DCA-1295 is similar to Aerospray 70 with additional plasticizers and other additives to help produce a more flexible film and increase shelf life)
Dynatech	Applied at the rate of 3.4 L/m^2 (0.75 gal/yd^2) (Dynatech is a natural rubber latex compound [designated 1-H-10 formulation No. 267])
Sucoat	Four discs, 0.95-cm (3/8-inch) thick, were tested (Sucoat is a molten sulfur product placed at high temperature [150-200°C or 300-400°F] which cures to a strong solid; a quick-setting, watertight, coating compound)
Uniroyal	Applied at the rate of 3.4 L/m^2 (0.75 gal/yd^2) (Uniroyal, a black natural latex [manufacturer designation L9241], was one of the materials that passed the traffic phase of the dust control test)
Prefabricated membrane liners	
Total liner	Applied as furnished by the manufacturer (total liner is an elasticized polyolefin about 0.51 mm [20 mils] thick)
T16	Applied as furnished by the manufacturer (T16, a chloroprene-coated nylon about 0.46 mm [18 mils] thick, is a composite material formed from a single-ply nylon fabric coated with neoprene; weighs 49.4 kg/m^2 [18.5 oz/yd^2])

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[a] Data from Styron and Fry (1979).

Table 40. Results from Liner Materials Tested on Silty Sand[a]

Liner material	Test cells for sludge A (avg Cl^-, ppm)	Liner material	Test cells for sludge B (avg Cl^-, ppm)
AC40	62	AC40	62
Sucoat	65	Sucoat	68
No liner	83	No liner	78
Total liner	102	Aerospray 70	115
Aerospray 70	109	DCA-1295	135
Uniroyal	176	Uniroyal	182
DCA-1295	190	Dynatech	190
T16	212	Asphaltic concrete	196
Dynatech	214	Total liner	202
Asphaltic concrete	460	T16	224
Sludge liquor	675	Sludge liquor	670

[a] From Styron and Fry (1979). Listed in order of increasing chloride content.

Table 41. Results from Liner Materials Tested on Clayey Silt[a]

Liner material	Test cells for sludge A (avg Cl^-, ppm)	Liner material	Test cells for sludge B (avg Cl^-, ppm)
Cement/lime	17	Cement/lime	24
CST	38	CST	75
Lime	45	No liner	79[b]
No liner	95[b]	Cement	151
Cement	142	Lime	152
TACSS 025	424	C400	462[b]
M179	437	M179	502
TACSS 020	495	TACSS 025	512
Sludge liquor	675	Guartec UF	642
Guartec UF	1117	Sludge liquor	670
		TACSS 020	679

[a] From Styron and Fry (1979). Listed in order of increasing chloride content.
[b] Single sample.

Water application in conjunction with surface compaction is particularly effective for stabilizing fine-grained materials. Emplacement of organic and inorganic mulches generally affords a greater degree of erosion control. Hay or straw crimped into surface materials by discing reduces wind velocity and traps drifting particles. Other suitable mulching materials include tree bark, corncobs, and animal manure. Commercially available matting (woodfiber, plastic, etc.) serves similar purposes. Applications of thin layers of coarse gravel, country rock, or crushed stone provide an effective mulch; these materials are particularly useful in arid areas where wind velocities are consistently high.

Chemical methods. Chemical methods involve applications of various reagents on fine-grained particles to form a surface crust. Dean et al. (1974) reported the results of tests to determine the relative effectiveness of 70 chemicals in forming an erosion-resistant crust. The tests were conducted under laboratory conditions and the substrate consisted of mill tailings. The more effective of the tested chemicals* are listed below "in order of effectiveness based on the cost in cents for the amount of reagent required to stabilize one square yard" (Dean et al. 1974):

- Coherex: a resinous adhesive
- Lignosulfonates: calcium, sodium, and ammonium lignosulfonates
- Compound SP-400, Soil Guard, and OCA-70 elastomeric polymers
- Cement and milk of lime additives
- Paracot TC-1842: a resinous emulsion
- Pamak WTP: a wax, tar, and pitch product
- Petroset SB-1: an elastomeric polymer
- Potassium silicate: an SiO_2 to K_2O ratio of 2.5
- PB-4601: a polymeric stabilizing agent
- A cationic neoprene emulsion and Rezosol, an elastomeric polymer
- Dresinol--TC 1843: an ammonium casein of tall oil pitch
- Sodium silicates: ratios of 2.4 to 2.9 SiO_2 to 1 Na_2O (calcium chloride was an effective additive to sodium silicate, resulting in reduced quantities of the latter needed for effective stabilization).

Before using any of these compounds, their potential for environmental impacts should be evaluated.

Vegetative methods. The establishment of a dense and self-perpetuating vegetative cover is one of the more effective measures for controlling wind erosion and is generally preferred for aesthetic reasons. However opportunities for establishing vegetation prior to final reclamation of a given waste-storage area will be dependent on site-specific conditions as well as the storage method. For example, disturbed areas that will not be used during active storage operations can be prepared and seeded to establish either a temporary cover crop or a permanent vegetative cover. Likewise, landfill operations can be staged such that successive portions of the storage area can be vegetated prior to final site stabilization.

Local climatic conditions and the nature of the surface materials will dictate the cultural practices (topsoiling, fertilization, and/or irrigation) necessary for establishing an effective plant cover. The implementation of other wind-erosion-control measures (applications of water, organic and inorganic mulches, etc.) is also frequently necessary for controlling sandblasting and/or burial of plant seedlings. Hydroseeding is a particularly effective practice in this respect. The practice entails blowing a slurry of wood chips, paper pulp, or similar residues with admixed seeds and fertilizer over the surface to be stabilized. The water promotes seed germination and the residues inhibit movement of surface particles by wind force.

In some instances, it may be feasible to establish vegetation directly on waste material. However, various measures will probably be necessary to modify chemical and physical properties of the wastes. In general, the wastes will be deficient in certain essential plant nutrients; concentrations of other elements or substances may be sufficiently high to inhibit seed germination and/or plant growth.

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*Reference to trade or brand names is made for identification only; no endorsement is intended.

Water Erosion

Many of the previously discussed measures for controlling wind erosion are also effective for reducing water-erosion potential. For example, tillage equipment can be used to increase the porosity, roughness, and cloddiness of exposed materials. In turn, water infiltration rates and surface-water storage capacities are increased and runoff velocities are decreased; erosion potential is decreased accordingly. In other than flat terrain, tillage equipment should be operated on the contour to provide the most effective control of water erosion.

The promotion of surface crusting by chemical stabilizers, the emplacement of organic and inorganic mulches, and the establishment of vegetation are also effective for controlling both wind and water erosion (Table 18). Effective surface crusts absorb the energy of raindrop impact, thereby preventing the detachment of surface particles. The effects of raindrop impact are also reduced by vegetation and mulches; the protection afforded varies with the density of the vegetative canopy or the completeness of the mulch layer. Vegetation and mulches also constitute obstructions that tend to reduce the velocity of runoff; the greater the obstruction, the lesser is the erosive force of the runoff.

Additional control of surface runoff at waste-storage sites can be achieved by developing various structures designed to effect one or more basic objectives as follows: increasing the surface-water storage capacity of the site, divert and spread surface runoff, and/or channelize runoff. The construction of contour terraces, at intervals normal to sloping surfaces, is an effective method for increasing surface storage capacity. The development of storage or siltation ponds serves a similar purpose. Ditches, earthen dikes, piping, and hay bales or similar organic materials can be used to temporarily divert and spread runoff, thereby controlling erosion potential until vegetation is established. Permanent structures can also be used to collect and channelize runoff. In some instances, it may be necessary to construct permanent check dams at intervals within the channel, thereby controlling gully or channel erosion and depositions of sediment downslope. Check dams may be nonporous (earthen embankment with concrete spillway) or porous (cribbed rockfills with a downslope apron) structures.

The kinds and extent of structures used for control of surface runoff will be dependent on site-specific considerations. Control measures will vary according to the storage method. Given a storage pond rather than a landfill, erosion-control measures will also entail stabilizing the inner slopes of the embankments to prevent erosion due to wave action. In some instances, the establishment of vegetation may be an effective measure; in others, riprapping may be necessary.

DISCOURAGING WILDLIFE USE OF WASTE-STORAGE PONDS

Waste storage as dry landfill will usually result in relatively limited direct contacts between wildlife and the waste materials, especially if portions of the waste surface are routinely covered as landfill operations progress. On the other hand, water and other habitat resources of storage ponds are attractive to numerous wildlife species. Chain link fences or similar barriers can be installed to exclude many terrestrial vertebrates, but storage ponds are readily accessible to both bats and birds.

Certain species of bats commonly forage over ponded water where densities of flying insects tend to be relatively high. Thus, these mammals could be adversely affected by ingesting invertebrates that metabolize or otherwise extract contaminants from the waste materials. Evidence of the potential hazard to bats resulting from bioaccumulation of contaminants from coal ash and FGD sludge wastes is not known to be documented in literature. However, were such a problem identified, methods could be implemented to limit the density of prey populations, thereby reducing the attractiveness of the waste pond as a foraging area.

General methods for repelling birds can be classified into three broad categories; i.e., biological, chemical, and mechanical. Guarino (1975) has cited some examples of biological methods as follows: decoy crops (e.g., planting attractive food crops to divert birds to adjacent areas); changes in local cultural practices (e.g., harvesting early or changing crop types); and habitat manipulation (e.g., burning roosting vegetation or thinning branches and trees in large roosts). These methods are primarily oriented toward crop protection and will probably be of limited utility for repelling birds from waste-storage ponds.

Chemical methods for manipulating bird populations are based on several differing control strategies. Accordingly, potentially effective chemicals can be differentiated as repellents, frightening or stressing agents, toxicants, or chemosterilants (Guarino 1975). The environmental costs associated with the use of some repellent and frightening agents are relatively low since less than one percent of the flocks of target bird species are killed and hazards to nontarget species are generally minor (Guarino 1975). In contrast, control programs involving avian stressing agents, toxicants, and chemosterilants entail marked depletion of bird populations, either by extensive bird kills or by inhibiting reproduction. Obviously, applications of these chemicals would be counterproductive since the objective of excluding birds from waste-storage ponds is to conserve or protect existing bird populations.

Mechanical methods for repelling birds include use of a wide variety of explosive devices such as shotguns, rifles, rope firecrackers, shell crackers, and propane and acetylene exploders. Some of these devices can be rigged to detonate automatically at timed intervals and are most effective when moved periodically. Strategic emplacement of life-sized dummies resembling common predators, as well as rigid or animated scarecrows (Maugh 1979), may be effective practices in some instances. Likewise, stationary or mobile units equipped to broadcast recorded alarm or distress calls of birds have been used for repelling birds from a given area. The use of varying combinations of the aforementioned visual and auditory stimuli frequently results in increased effectiveness of scare tactics (Mott 1975). Another alternative is to prevent access to the impoundment by covering it with netting or wiring to screen the surface liquors. In some instances, netting or trapping may be practical techniques for removing birds from a given area.

All of the biological, chemical, and mechanical techniques described above are currently being used. In general, the mechanical means are most advantageous because the majority can be readily implemented and have a low potential for harming wildlife.

The selection of a method or methods for discouraging use of a given storage pond by birds will generally depend on numerous considerations. Among others, the habitat resources and other site-specific conditions in the immediate area will provide some insight as to the scope of the problem. Bird reactions to control stimuli vary considerably; thus, the number and species of birds involved as well as whether the birds are residents or migrants will be important selection criteria. An additional consideration is the potential of the control techniques for posing a more serious problem than the use of the impoundments. If the viable control methods are likely to harm wildlife, it may be more prudent to take no action to prevent wildlife use of waste impoundments. Social factors such as public opinion and local land-use patterns may preclude using some control methods. Legal aspects of bird controls must also be considered. Most bird species are afforded protective status by one or more state and/or federal laws; thus, plans for implementing bird-control measures at a storage site should be coordinated with the appropriate state wildlife agency and various officials of the U.S. Fish and Wildlife Service, including representatives of the Animal Damage Control Division.

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RECLAMATION OF STORAGE SITES FOR COAL COMBUSTION AND EMISSION-ABATEMENT WASTES

RECLAMATION OPTIONS

One of the major problems faced by a manager of a waste-storage site is decommissioning of the area after it has served its purpose. An abandoned, unmanaged storage site may pose a hazard for fish and wildlife. Additionally, the land usually cannot be used by a diversity of wildlife unless the storage site has been rehabilitated. Methods for the reclamation and revegetation of coal ashes and FGD scrubber sludges are just beginning to be developed. To date, no large-scale effort has been made to reclaim and revegetate a coal combustion waste-storage site. It is currently unknown whether self-perpetuating plant and wildlife communities can be established at these sites, or what types of land uses for reclaimed sites will be environmentally acceptable. Reclamation experience with surface mined lands provides a framework for our discussion, but we have incorporated available information pertinent to revegetation of coal ashes and FGD sludges. The application of large amounts of fly ash to cropland and mine revegetation sites is discussed in this section as a guide to the potential revegetation of ash and sludge wastes. If site-specific limitations can be alleviated, all the options discussed here could result in habitat suitable for fish and wildlife.

Although reclamation of coal combustion wastes will probably be the most cost-effective long-term method for mitigating the impacts of the wastes on the surrounding environment, reclamation will not eliminate these impacts. The leaching of trace elements and soluble salts from the stored wastes into groundwaters will probably still occur, although at a much slower rate. If erosion of the soil mantle covering the wastes is allowed to occur, the wastes will be eventually exposed, resulting in surface water contamination and adverse effects to the surrounding wildlife and vegetation. Procedures and perhaps appropriate regulations will have to be developed to ensure minimal mobilization of waste constituents and the long-term success of reclamation.

Use of Coal Ash and Flue-Gas-Desulfurization Sludge in Agriculture

Fly ash. Due to the elemental composition of fly ash and the increasing availability of this material, it seems reasonable to assume that fly ash may be a potential source of certain plant nutrients. However, the concentrations of these nutrients present in a given fly ash sample can vary considerably (Martens et al. 1970). The increasing cost of conventional fertilizers may make fly ash an attractive alternative. Alkaline fly ashes may also be an alternative to liming for increasing soil pH.

Field and greenhouse experiments have indicated that fly ash amendment generally increases plant growth and improves agronomic properties of soil. Fly ash addition to soils has resulted in alleviation of sulfur deficiency. Yield and sulfur content of alfalfa (*Medicago sativa*), bermuda grass (*Cynodon dactylon*), turnip (*Brassica rapa*), and white clover (*Trifolium repens*) grown in pots were greatly improved by fly ash additions of 9-18 g/kg of soil by weight (Elseewi et al. 1978a); furthermore, it was concluded that fly ash-derived sulfur is as available for plant uptake as gypsum-derived sulfur. Yield increases of several crops grown in pots on either calcareous or acidic soils with fly ash additions of up to 720 g/kg of soil were attributed to increased sulfur availability (Page et al. 1979). Fly ash treatment also increased the sulfur content of Romaine lettuce (*Lactuca sativa*), Swiss chard (*Beta vulgaris*), corn (*Zea mays*), and beans (*Phaseolus vulgaris*) when grown on a variety of soil types, using a number of different ashes (Adriano et al. 1978; Elseewi et al. 1978b).

Boron in fly ash is readily available to plants (Mulford and Martens 1971); increases in the yield of alfalfa have been attributed to correction of soil-boron deficiency through field application of fly ash (Plank and Martens 1974). However, boron can also be toxic. Fly ash addition may lead to phytotoxic levels of boron in the soil unless an appropriate rate of ash application is used.

Potassium uptake by corn was increased by fly ash addition to a clay loam soil; ash-derived potassium was slightly less available to plants than potassium from KCl (Martens et al. 1970). Zinc availability to plants was increased by the addition of acidic fly ash to soil (Schnappinger et al. 1975).

Fly ash addition to soil cannot be used as a substitute for nitrogen and phosphorus fertilization; the ash is virtually devoid of nitrogen and much of the phosphorus content is quite insoluble. Phosphorus deficiencies have been observed in plants grown on ash-amended soils not supplied with additional phosphorus (Adriano et al. 1978). Another drawback to the agronomic use of fly ash results from the increase in soil pH caused by the addition of alkaline fly ash. Deficiencies of the micronutrients copper, manganese, iron, and zinc have been thought to be produced by addition of fly ash (Adriano et al. 1980).

Although selenium is not essential for plant growth and molybdenum is essential only in small quantities, the concentrations of these elements in fly ash are highly available to plants. Doran and Martens (1972) showed that molybdenum in fly ash was as available to plants as molybdenum in $Na_2MoO_4 \cdot 2H_2O$. Molybdenum concentrations in alfalfa grown in pots of ash-amended soil were in the range known to cause molybdenosis of cattle (10-20 μg Mo/g dry wt. of forage). Selenium in plant tissue increased consistently with increased fly ash addition to soil (Adriano et al. 1980). Plant-tissue concentrations of selenium that approach levels harmful to animals (4-5 μg Se/g dry wt.) have been reported in plants grown on ash-amended soils (Furr et al. 1977). The hazard to foraging wildlife from fly ash-derived selenium accumulation in plants is most severe in the western United States (Figure 20), where seleniferous plants abound. Gutenmann and Lisk (1979) showed that *Astragalus racemosus*, a selenium accumulator, concentrated selenium in its tissue in direct proportion to the amount of fly ash added to the soil. Conversely, additions of small amounts of fly ash to pastures in regions of the country deficient in soil selenium (Figure 20) might prove to be beneficial in supplying livestock with this essential element.

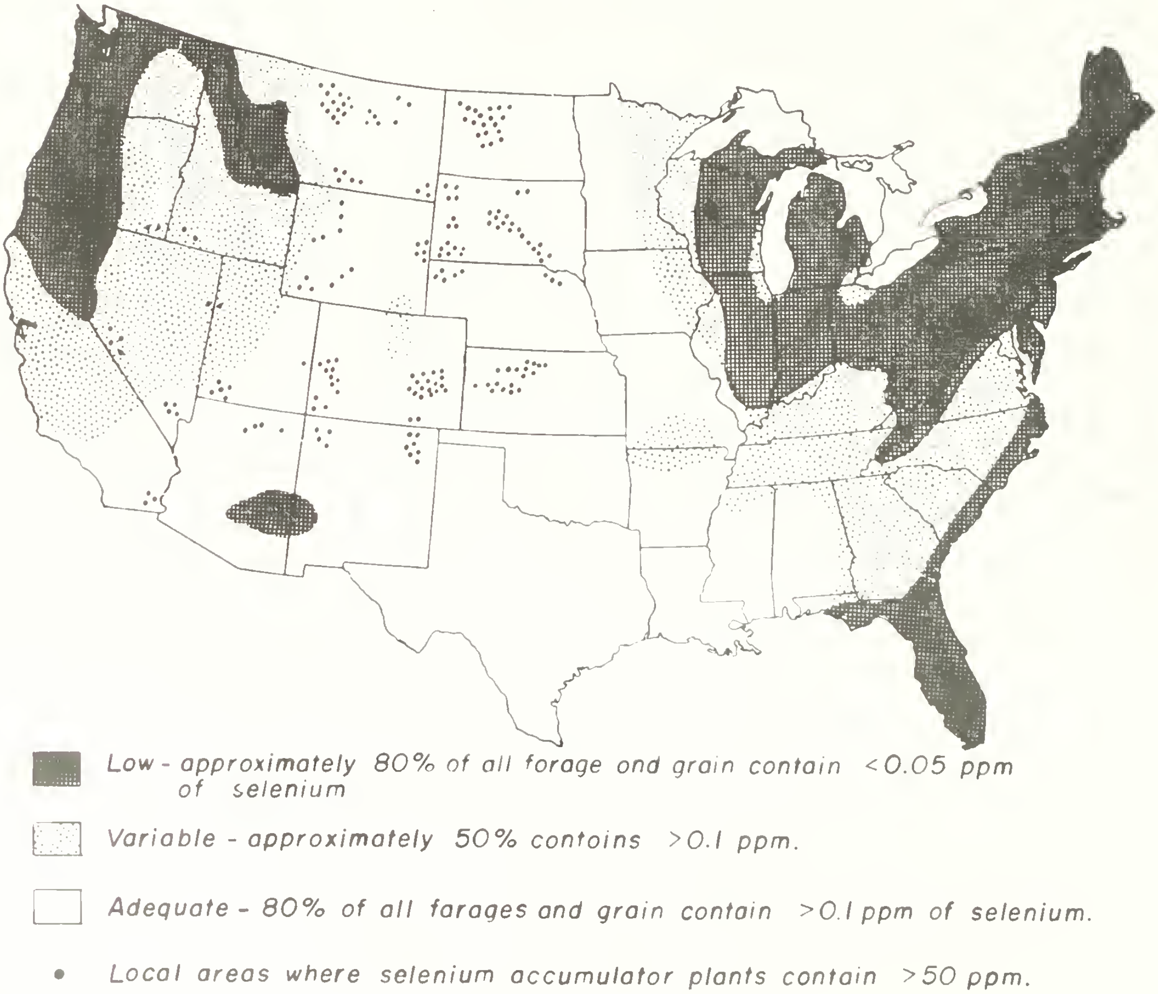

Figure 20. Geographic Distribution of Low-, Variable-, and Adequate Selenium Areas in the United States. From Kubota and Allaway (1972) (with permission, see credits).

Other nonessential trace elements (Al, As, Ba, Cs, Rb, Sr, W, and V) tend to increase in plant tissue as fly ash applications to soil are increased; however, yield reductions due to fly ash addition have only been associated with boron toxicity. The concentrations of Cd, Co, Cr, Cu, F, Ni, and Tl in fly ash (Adriano et al. 1980) have not been demonstrated to cause detrimental effects to plant growth. However, accumulation of these elements in food plants could pose a threat to wildlife or humans feeding on these plants.

The physical properties of soils can also be significantly altered by fly ash amendment. Fly ash inputs reduce the bulk density of most agricultural soils (Adriano et al. 1980). Application rates of > 25% by volume increase the water-holding capacity of most soils, but no noticeable increase in plant-available water occurs (Chang et al. 1977). Hydraulic conductivity of soils increases with small applications of fly ash, but declines rapidly as fly ash input exceeds 20% by volume for calcareous soils, and 10% for acidic soils (Chang et al. 1977). This response is thought to be caused by the pozzolanic reaction of fly ash, which cements soil particles together when wetted, thus impeding water flow. Fly ash addition also reduced the particle cohesiveness in all soils tested.

Fly ash inputs can significantly increase the salinity of the amended soil. Following a 5% addition of fly ash to Tatum silt loam soil, Mulford and Martens (1971) reported increases in the electrical conductivity of saturation extracts from about 1 to 4.4 mmho/cm; at 4 mmho/cm, the production of some crops is inhibited (Richards 1969). Weathering quickly reduces the soluble-salt content of fly ash; thus, by using only weathered fly ash for agricultural applications, salinity problems may be avoided.

Phung et al. (1978) showed alkaline fly ash to be equivalent to 20% of reagent-grade $CaCO_3$ in reducing soil acidity and supplying calcium in laboratory studies. About 105 MT/ha of fly ash was required to raise the pH of a Reyes silty clay soil from 4.1 to 6.3. Applications of 105-210 MT/ha of fly ash to this soil were not expected to cause salt injury to plants, although boron toxicity and induced phosphorus deficiency were considered possible (Phung et al. 1978).

The major limitations to large-scale use of fly ash in agriculture appear to be: phytotoxic boron levels, soil cementation, induced nutrient deficiencies, excessive soluble salt concentrations, high plant-available concentrations of selenium and molybdenum, and high pH of fly ash. However, with careful consideration of appropriate application rates, many of these problems may be avoided. The economic feasibility of fly ash use has yet to be determined, and the costs of transporting ash from storage site to field may prove to be prohibitive (Adriano et al. 1978).

A potentially important issue which has not yet been addressed in the literature is the effect of repeated fly ash applications on the physicochemical nature of soil. For example, acidic agricultural soil is typically treated with lime every four to five years. If fly ash is to be substituted for lime, as some sources suggest (Anon. 1978), significant alterations in the soil could result from the accumulation of trace elements as well as from modification of soil properties.

Flue-gas-desulfurization scrubber sludge. Due to their content of unreacted $CaCO_3$ and alkaline fly ash, certain scrubber sludges are believed to have some value as liming agents. In one of the few trials of this material as a liming agent, Terman (1978) reported that scrubber sludge was 15-40% as effective as fine $CaCO_3$ for neutralizing soil acidity. More recently, Walker and Dowdy (1980) examined (1) the use as a liming agent of scrubber sludge produced from a power plant burning low-sulfur western coal and (2) the elemental composition of barley (*Hordeum vulgare*) and rye (*Secale cereale*) grown on sludge-amended acid soils. This scrubber sludge proved adequate for raising the soil pH; however, yield reductions attributable to high plant-tissue boron concentrations occurred in both species. Elevated plant-tissue selenium concentrations were also reported. As with fly ash, the use of agronomic scrubber sludge is constrained by the presence of potentially toxic trace elements.

Use of Coal Ash and Flue-Gas-Desulfurization Sludge in Land Reclamation

Fly ash amendment of acidic coal mining wastes. Rehabilitation of coal mine spoil and refuse may be enhanced by treating these wastes with fly ash. Spoil (or overburden) is the material found lying directly above the coal seam and other subsurface materials deposited in heaps or piles during the surface mining of coal. Mine refuse consists of waste coal, rock, extraction debris, minerals, associated clays, and other wastes produced during the development and operation of underground coal mines and coal washing facilities. Typically, mine refuse was dumped in piles with steep unstable slopes. The problems caused by these waste heaps include: nonproductive use of lands, erosion and landslides, loss of aesthetic value, and pollution of air and water (Coalgate et al. 1973). The establishment of vegetation on coal mining wastes can help to alleviate these problems and recover more suitable habitat for fish and wildlife. However, the chemical and physical properties of acidic coal mine spoil and refuse often prevents natural revegetation (Table 42). These coal mining wastes can serve as a suitable medium for plant growth following amendment with lime to raise the pH of the wastes to acceptable levels (Medvick and Grandt 1976). The use of some alkaline fly ashes as a substitute for lime in the reclamation of coal mining wastes is attractive because of the availability of large quantities of fly ash and the fact that the adverse nature of one waste can be ameliorated by treatment with another (Jastrow et al. 1979).

Table 42. Properties of Coal Mine Spoil and Refuse That are Detrimental to Plant Growth[a]

Property	Effect
Acidity	High concentrations of toxic trace elements (e.g., aluminum, iron, manganese) available for plant uptake.
Compacted surface (spoil)	Plant rooting restricted, poor water penetration, drought susceptibility.
Coarse texture (refuse)	Low water-holding capacity, drought susceptibility, plant damage due to windblown particles.
Dark color (refuse)	High temperatures, low available soil moisture during summer months.

[a] Adapted from Capp and Gilmore (1973).

In addition to raising the pH of coal mining wastes, fly ash amendment results in other beneficial physical alterations of the waste material. Fly ash applications of 1220 MT/ha to abandoned acidic spoil in West Virginia resulted in a change in the particle-size distribution of the spoil from that of a sandy clay loam to a silt loam, resulting in improved tillability of the amended spoil (Adams et al. 1971). Plass and Capp (1974) examined the effect on subsurface moisture following amendment of acidic spoil (pH 3.3-3.6) with fly ash (305 MT/ha) and found that fly ash amendment increased water infiltration rates, water percolation, and spoil porosity. At this lower fly ash application rate (compared to the 1220 MT/ha of Adams et al. 1971), no change in the particle-size distribution of the spoil occurred following amendment. Fly ash amendment lightens the color of dark refuse material, reducing heat absorption and improving the microclimate of the refuse surface.

As part of a program to identify outlets for large tonnage quantities of fly ash, the U.S. Bureau of Mines (USBM) began a series of greenhouse and field experiments in 1964 to evaluate the use of alkaline fly ash as an amendment in the reclamation of acidic ($pH < 4.0$) coal strip-mine spoils and mine refuse. From this program, USBM has developed a procedure to reclaim acidic coal mining wastes using fly ash. Initially, the site is characterized by determining the acid content, pH, moisture-holding capacity, nutrient deficiencies, conductivity, particle-size distribution, and textural classification of the spoil or refuse. Once a source of fly ash has been identified, the neutralizing capacity, pH, conductivity, particle-size distribution, and concentration of plant nutrients and trace elements in the fly ash are then measured (Capp and Gilmore 1974). Based upon these data, the amounts of fly ash and fertilizer required to amend the coal mining wastes can be determined.

Following the regrading of the spoil or refuse area to a configuration that will limit erosion and landslides, sufficient fly ash is applied to raise the pH of the coal mining wastes to a level acceptable for plant growth ($\sim$ pH 5.5-7.0). Application rates of 305-405 MT/ha have typically been employed (Adams et al. 1972); however, applications of up to 2950 MT/ha also have been evaluated (Adams et al. 1971). As a point of reference, an application of 260 MT/ha is equivalent to applying a fly ash layer 2.5 cm deep. After the fly ash is spread, it is mixed into the top 15 to 30 cm of the waste material using agricultural plows and discs. When the wastes are extremely rocky or uneven, heavy construction equipment--including rototillers, rippers, and rome plows--are employed (Capp et al. 1975). Fertilizer (10-10-10 NPK) is applied at a minimum rate of 11,200 kg/ha. Prior to planting, a suitable seedbed is prepared by running a culti-packer or drag over the surface of the amended coal wastes. The amended wastes are then sown with a standard seed mixture of 'Kentucky-31' tall fescue (*Festuca arundinacea*), perennial ryegrass (*Lolium perenne*), orchard grass (*Dactylis glomerata*), redtop grass (*Agrostis alba*), and the legume birdsfoot trefoil (*Lotus corniculatus*) at a rate of 50.4 kg seed/ha.

Fly ash amendment has been used by the USBM to reclaim approximately 35 ha of spoil deposits and an additional 18 ha of refuse material (Capp et al. 1975). Although the success of these revegetation efforts has not been completely assessed, stands of vegetation have been established at most sites. Average yield of dry forage (grasses and legumes combined) from a single cutting at four fly ash-amended spoil sites in West Virginia ranged from 1.8 to 5.0 MT/ha during the years 1965-1971 (Adams et al. 1972). The annual hay yield (combined yield of three cuttings) of 7.7 MT/ha obtained from one reclaimed site (when fertilizer was applied in the spring and following each of the first two cuttings) is comparable to the yields obtained in experiments with high fertilization rates conducted at the West Virginia University Agronomy Farm (Adams et al. 1971). Visible symptoms of trace-element toxicity have generally been observed during the first growing season following fly ash amendment (Adams et al. 1972), perhaps due to boron toxicity (Terman 1978). Trace-element analysis of dry forage (grasses and legumes combined) indicates that plant tissue boron content decreases with time, probably due to leaching of boron from the fly ash/coal waste mixture into groundwater (Adams et al. 1972).

Although no other trace-element toxicity problems have been reported in USBM research, the potential for trace-element toxicity in plants and/or herbivores may limit the use of fly ash amendment in the revegetation of acidic mine refuse. Jastrow et al. (1979) reported that two commonly used revegetation species--'Kentucky 31' tall fescue (*Festuca arundinacea*) and 'Lincoln' smooth brome (*Bromus inermis*)--grown in acidic mine refuse amended with fly ash (120 MT/ha) did accumulate certain trace elements to levels that approached or exceeded concentrations reported to be associated with toxicity in some plant species. Both species accumulated high tissue concentrations of Al, Cu, Fe, Mn, V, and Zn in shoots and leaves. Parameters giving an indication of plant vigor (i.e., number of leaves and stems, width of longest leaf, and biomass) were significantly lower for plants grown on fly ash-amended refuse than for control plants grown on topsoil (Jastrow et al. 1979). The results of this short-term (60 days) pot experiment cannot be extrapolated to field conditions, but it does point out a potential problem. The addition of fly ash to acidic mine refuse could result in a substrate containing plant-available concentrations of trace elements that are potentially toxic to plants and their consumers. Without careful consideration and testing of all materials (fly ash and coal wastes) to be used at each site, this reclamation technique is not a viable disposal outlet. Further research under field conditions is warranted to determine the extent of this problem.

The fly ash-amendment technique was developed for use in the reclamation of newly deposited spoil and refuse from active coal mines as well as from abandoned mining operations (Capp et al. 1975). The Surface Mining Control and Reclamation Act of 1977 (Pub. L. 95-87) calls for the redeposition of topsoil over mine spoil, burial of mine refuse, and subsequent revegetation. In practice, this limits the use of fly-ash amendment to abandoned mind lands. The U.S. Soil Conservation Service (1979) reported that approximately 444,320 ha of abandoned lands disturbed by coal mining existed in the United States as of 1 July 1977. The vast majority of these lands (> 96%) are located in the Eastern and Central Interior coal provinces. It is currently unknown what percentage of this disturbed land could be reclaimed through fly ash amendment. If one assumes 40% of the total area disturbed (177,728 ha) to be amenable to reclamation using fly ash and an average fly ash application rate of 305 MT/ha, then over 54.2 million MT of fly ash could be disposed of in the reclamation of these areas. However, over 39.7 million MT of fly ash were collected during 1978 alone (Anon. 1979), indicating that disposal of fly ash on acidic coal mining wastes can serve as only a secondary disposal outlet. The distance between power plant and storage site is also critical. Considering the volume of fly ash required, transportation costs would become prohibitive if no suitable coal mine sites exist near the site of waste production. Economic use of limestone is usually restricted to direct trucking within 80 km (50 miles) of the quarry (Terman 1978). It can be assumed that similar restrictions would apply to the transportation of fly ash.

The option of using fly ash amendment in the reclamation of acidic mine wastes may be limited even more by factors other than the potential trace-element toxicity problems and limited number of suitable mine sites. The question of whether fly ash amendment of acidic mine wastes will allow for the development of self-sustaining plant communities on these wastes remains unanswered. This problem is critical to establishing wildlife habitat on mine spoils. To date, little effort has been made to determine the long-term success of this reclamation technique. Examination of available data indicates that problems may occur. The change in pH of fly ash-amended spoil at two sites in West Virginia over six growing seasons is shown in Figure 21 (Adams et al. 1972). The downward trend in pH of the amended spoil shows that the reserve acidity of the spoil will eventually cause the pH of the amended spoil to fall below levels acceptable for plant growth. Further treatment with fly ash (increasing the potential for trace-element toxicity problems) or another alkaline material would then be required to prevent decreases in plant production. The resulting need for long-term continuous maintenance that may be required at fly ash-amended sites will generally not meet the state and federal requirements for establishing self-sustaining communities of vegetation.

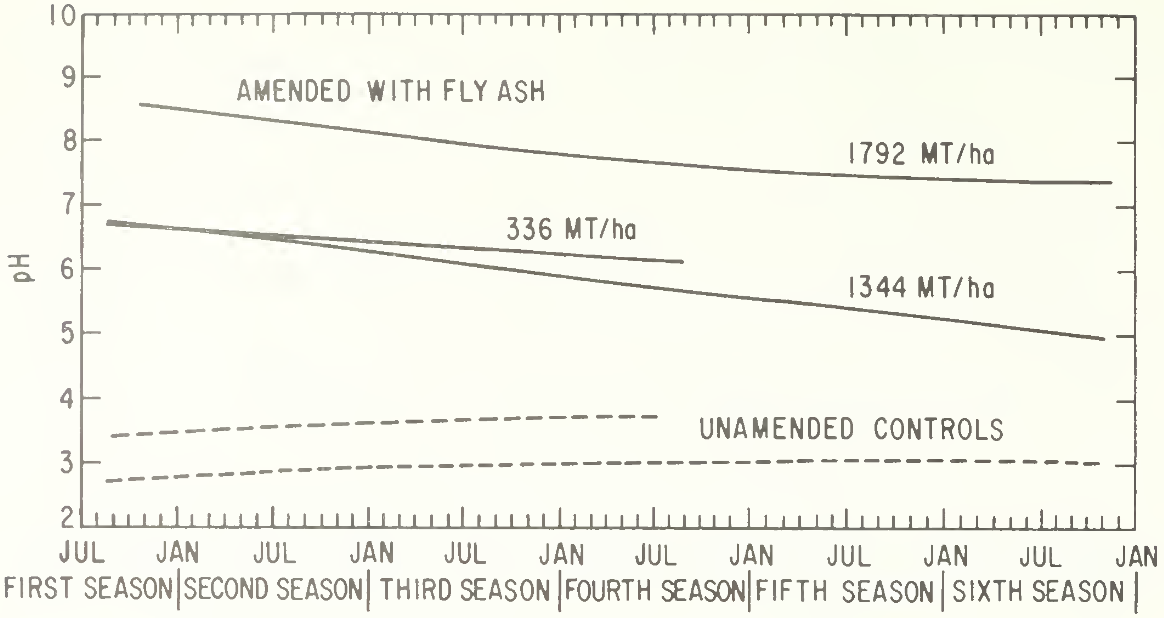

Figure 21. Changes in pH Over Six Growing Seasons of Coal Spoil Amended and Unamended with Fly Ash. Experiments conducted at two U.S. Bureau of Mines demonstration sites in West Virginia. Adapted from Adams et al. (1972).

Further research to determine the changes that will occur over time in the plant communities established on fly ash-amended mine wastes as well as changes in the physical and chemical nature of the amended wastes is needed before this technique can be endorsed as a viable reclamation method or coal ash disposal outlet.

Scrubber sludge amendment of acidic coal mining wastes. Although limestone scrubber sludge has been shown to be somewhat effective as a liming agent (40% as effective as fine $CaCO_3$), the neutralizing capacity of sludge will vary considerably with the amount of unreacted lime and the amount and buffering capacity of the fly ash present in the sludge (Terman 1978). Thus, its usefulness as an amendment to coal mine wastes or other mineral tailings may be limited. No research to determine the effectiveness of scrubber sludge as an amendment to acidic mining wastes has been conducted to date. The feasibility of using these sludges is also limited by transportation and economic constraints similar to those discussed for fly ash.

Revegetation of Storage-Site Surfaces

If revegetation of coal combustion wastes can be attained without placement of an expensive soil mantle over the wastes, the cost of storage-site reclamation will be reduced considerably. However, establishing vegetation directly upon the wastes is extremely difficult. Plant growth on fly ash or FGD scrubber sludge is constrained by: high available-boron concentrations, alkaline pH, high soluble-salt concentrations, high sulfite concentrations, "sand-blasting" by windblown particles, nitrogen deficiency, and in some instances phosphorus deficiency. In the discussion that follows, attempts to directly revegetate coal combustion wastes with a minimum of amendments will be examined to determine whether this technique can be used in large-scale reclamation of waste-storage sites. Selected plant species that may be suitable for use in this reclamation are presented in Appendix C.

Revegetation of coal ash. Despite the harsh chemical and physical nature of fly ash, natural colonization of storage sites has been reported, indicating that vegetation can be established directly upon the ash. Gonsoulin (1975) surveyed three abandoned fly ash pits near Gallatin, Tennessee, and identified 32 herbaceous species and 3 species of woody plants. The number of herbaceous species increased from 8 species in the most recently abandoned pit (6 months; pH 6.5-6.8) to 25 species in the oldest pit (8 years; pH 8.4-8.8). Dominant herbaceous species included: Pennsylvania smartweed (*Polygonum pennsylvanicum*), hairy-flowered paspalum (*Paspalum pubiflorum*), smooth brome (*Bromus inermis*), false dandelion (*Pyrrhopappus carolinianus*), white sweet clover (*Melilotus alba*), and goldenrod (*Solidago* spp.). The three woody species--cottonwood (*Populus deltoides*), sandbar willow (*Salix interior*), and black willow (*Salix nigra*)--occurred in all three pits. More recently, Skinner et al. (1978) reported that the vegetation of an "island" in a fly ash settling pond was dominated by broomsedge (*Andropogon virginicus*) and goldenrod. Camphor weed (*Hetterotheca subaxillaris*), wax myrtle (*Myrica cerifera*), consumption weed (*Baccharis halimifolia*), and several pines (*Pinus* spp.) were scattered over the site.

In England, early plant succession occurring on fly ash surfaces has been studied by Hodgson and Townsend (1973). Following initial colonization of fly ash by the moss *Funaria hygrometrica* (which can completely cover a moist ash surface within six months), a sparse cover of gray orach (*Atriplex hastata*) typically develops. As weathering of the fly ash continues, other plant species such as dock (*Rumex* spp.), coltsfoot (*Tussilago farfara*), pigweed (*Chenopodium* spp.), common mugwort (*Artemisia vulgaris*), clover (*Trifolium* spp.), meadow grass (*Poa* spp.), bentgrass (*Agrostis stonifera*), birdsfoot trefoil (*Lotus corniculatus*), and black medic (*Medicago lupulina*) appear and may form a closed canopy of vegetation over the ash surface. Eventually, the woody species gorse (*Ulex europaeus*), birch (*Betula verucosa*), and willow (*Salix* spp.) may become established. Growth abnormalities (e.g., reduced vigor, leaf chlorosis and necrosis, and extreme stunting) have been evident in most of the plant species that establish on fly ash, indicating nutrient deficiencies and/or the effects of trace-element toxicity (Hodgson and Townsend 1973). Unfortunately, surveys of vegetation growing on fly ash surfaces in the United States make no mention of the vigor of colonizing plants.

Although reports of natural colonization of fly ash surfaces are encouraging, the slow rate at which colonization occurs and the fact that colonizing plants often exhibit growth abnormalities suggest that untreated fly ash is, at best, a marginally adequate substrate for plant growth and is limited in its ability to support wildlife habitat. The factors limiting plant growth on fly ash can be generally stated as: (1) the physical and chemical nature of fly ash deposits, (2) the presence of toxic trace elements in fly ash, and (3) the concentrations of plant-available nutrients (Townsend and Hodgson 1973). Therefore, placement of a soil cover material suitable for plant growth may be required.

Physical and chemical factors influencing plant growth on fly ash. Both physical and chemical factors influence the growth of plants on fly ash. Fly ash is primarily composed of silt- and clay-sized particles, 68% having diameters of < 53 μm (Page et al. 1979). The fine texture of fly ash is in part responsible for the low permeability of the material. Low permeability enhances surface runoff, retarding the leaching of salts and trace metals and increasing susceptibility to water erosion, often resulting in unstable conditions at storage sites (Page et al. 1979).

Lateral hydraulic conductivity in fly ash deposits, especially ponded fly ash, has been reported to be much higher than vertical hydraulic conductivity (Cope 1962). The profile of ponded fly ash exhibits distinct stratification and is characteristically that of a sedimentary deposit (Townsend and Hodgson 1973). Extremely compact and impermeable layers (≥ 1 mm thick) will occur randomly throughout the ash profile, strongly influencing root development and patterns of water movement. Roots of plants established on highly weathered ponded fly ash have been shown to develop horizontally along the surface of these very compact layers (Townsend and Hodgson 1973). Cementation of the ash prevents root penetration, and the development of a surface "cap" impairs the emergence of small-seeded species (Hodgson and Townsend 1973). The problem of layering does not occur in landfilled fly ash; however, the pozzolanic nature of fly ash can retard plant growth. Subsurface hardening in landfilled ash can result in the production of drainage problems.

Wind erosion is also detrimental to the revegetation of fly ash deposits. The fine-textured ash is extremely erodible; cenospheres have a threshold wind velocity of only 19.3 km/h (Townsend and Hodgson 1973). The cultivation of fly ash surfaces greatly increases wind erosion. The resultant "sand-blasting" of plants by windblown particles during initial vegetative establishment can seriously retard growth (Goodman et al. 1973).

Although the physical properties of fly ash cannot be altered, the detrimental effect of these properties on plant growth can be minimized. By mixing soil, clay, acid shale, sewage sludge, or organic materials (e.g., straw or peat) into the surface layer of fly ash deposits, the ash can be stabilized and wind and water erosion reduced (Hodgson and Townsend 1973). The addition of these materials also can have a favorable effect on the moisture-holding capacity of the ash, can inhibit cementation, and may contribute essential nutrients. Deep cultivation of both ponded and landfilled fly ash disrupts cemented layers in the ash, improving root penetration.

The majority of fly ash produced in the United States has an alkaline pH, attributable to the very high concentration of hydroxyl (OH) ions (Figure 22) present in ash-water extracts (Page et al. 1979). High alkalinity is characteristic of fly ash derived from western lignite and subbituminous coals. A number of trace elements have a low solubility in alkaline ash extracts; iron, manganese, and zinc are among the micronutrients that become unavailable to plants at alkaline soil pH (Brady 1974). This fact may account for some of the symptoms of nutrient deficiency reported by Townsend and Gillham (1975) for ash-grown plants adequately supplied with nitrogen

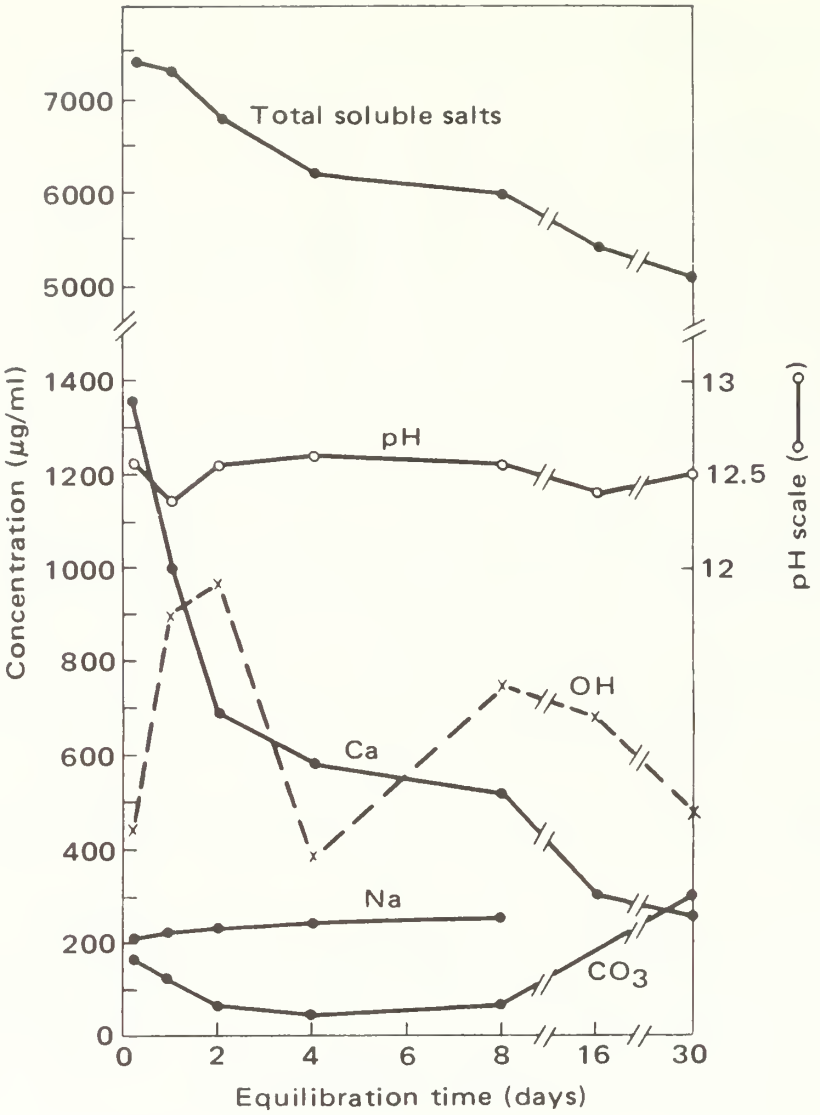

Figure 22. Total Soluble-Salt, pH, and Ion Solubility of 1:1 Fly Ash-Water Suspensions Equilibrated for 30 Days at Room Temperature in a Closed System. From Page et al. (1979) (with permission, see credits).

and phosphorus. Phosphorus availability is also very adversely affected by high alkalinity. The formation of insoluble calcium phosphates and other complexes can seriously impair the solubility of both native and applied phosphorus in alkaline soils (Brady 1974), and a similar mode of phosphorus fixation may occur in solutions containing alkaline fly ash.

The alkalinity of freshly produced fly ash (pH 8-12) is reduced through weathering, but typically stabilizes on the order of pH 8-9 (Townsend and Gillham 1975), still far above that considered optimal for plant growth. Efforts to reduce the pH of fly ash deposits still further (using applications of sulfur, ferric sulfate, or aluminum sulfate) were ineffective, nor did they increase plant growth (Hodgson and Townsend 1973). Complete neutralization of the surface 23-25 cm of a fly ash deposit might require the application of over 100 MT/ha of concentrated H_2SO_4, certainly an impractical treatment (Hodgson and Townsend 1973).

Fly ash with an acidic pH is produced in the combustion of high-sulfur coal, primarily in the eastern United States (Page et al. 1979). The pH of acidic fly ash collected in seven eastern states ranged from 4.2 to 5.9 (Furr et al. 1977). The effect of weathering on the pH of acidic fly ash deposits is unknown; no research has been conducted to determine the availability of plant nutrients in acidic fly ash. The pH of acidic fly ash can be increased through liming (Stoewsand et al. 1978). The effect on pH of mixing acidic and alkaline fly ashes has not been investigated; however, this procedure may be an attractive method for altering the pH of fly ashes. The usefulness of this technique will be limited by the distances between sources of acidic and alkaline fly ashes, and associated transportation costs.

Regardless of pH, essentially all fly ash contains concentrations of soluble salts high enough to produce osmotic potentials detrimental to plant growth. Townsend and Hodgson (1973) reported electrical conductivity of fly ash extracts ranging between 8.0 and 13.0 mmho/cm; conductivities exceeding 4.0 mmho/cm are generally considered to retard plant growth (Richards 1964). Fortunately, the deleterious effects of high ash salinity are relatively short-lived. Fly ash extracts (1:1) equilibrating over 30 days (Figure 22) showed marked reductions in the Ca and OH ion concentrations and increased CO_3 ion concentration (Page et al. 1979). As a result, total soluble salts content decreased 30% over the course of this experiment. The formation of insoluble $CaCO_3$ from a "$OH + CO_2$" reaction forming CO_3 which then reacts with Ca is thought to account for these changes. The ponding of fly ash results in a considerable decrease in soluble-salt content. Following two or three years of weathering, soluble-salt content decreases in ponded ash to harmless levels (Townsend and Hodgson 1973).

Trace elements in fly ash. The high alkalinity of fly ash/water solutions causes the solubility, and thus the availability to plants of many trace elements, to be quite low. However, a number of elements (e.g., As, B, Mo, F, Se, Cr, and V) that form anionic species remain relatively soluble in alkaline environments (Page et al. 1979). Of these, boron, molybdenum, and selenium are of most import to biotic resources. Boron and molybdenum, essential micronutrients, are toxic to plants and/or animals at higher concentrations. Selenium, which can be concentrated in plant tissue, is potentially toxic to animals.

Boron toxicity is perhaps the most important factor limiting plant establishment on fly ash. Plant-available boron in British coal ashes ranged from 3 to 150 µg B/g of fly ash with a mean of 43 µg B/g of fly ash (Townsend and Hodgson 1973). In two samples of fly ash with pH's of 4.8 and 11.2 from the southeastern United States, hot-water-soluble samples contained 22 and 50 µg B/g of fly ash, respectively (Plank and Martens 1974). Hodgson and Townsend (1973) suggested that available-boron contents of between 11 and 20 µg B/g of fly ash be considered moderately toxic to plants, with higher concentrations considered toxic.

Townsend and Gillham (1975) showed that boron was readily leached through natural weathering (Figure 23) from two British fly ashes of moderate initial boron content. Over time, the plant-available boron content of all fly ashes will decrease; however, the rate of decrease and the final concentration of available boron maintained in a fly ash deposit will be dependent upon a number of factors including: initial boron content, compaction and permeability of the ash deposit, whether the ash was landfilled or ponded (the ponding process will significantly reduce ash boron), and the amount of precipitation received at the storage site. Although the plant-available boron levels in the ashes shown in Figure 23 dropped to nontoxic levels within two to three years, 14 years were required for the plant-available boron content of another fly ash to decrease from an initial level of 216 to 20 µg B/g (Hodgson and Townsend 1973). If the rate of natural boron leaching from a specific fly ash-storage site could be determined, this information along with the initial boron content of the ash could be used to predict the time required for weathering to reduce ash boron to a nontoxic concentration.

The availability to plants of molybdenum in fly ash was shown by Doran and Martens (1972). The addition of fly ash to Groseclose silty loam increased both molybdenum uptake by alfalfa and alfalfa yield. White sweet clover (*Melilotus alba*), containing 38 µg Mo/g dry wt. after growth on acidic fly ash, was incorporated as 23.5% of the diet (dry wt.) of adult goats, newborn kids, and lambs for 173 days (Furr et al. 1978a). Although liver concentrations of molybdenum were elevated for adult goats (5.7 ± 0.1 µg Mo/g dry wt.) and lambs (6.2 ± 0.3 µg Mo/g dry wt.) over those of control animals, molybdenosis was not observed. Molybdenosis in sheep has been caused by molybdenum levels in forage of 10-12 µg Mo/g dry wt. (Gough et al. 1979). No research has been conducted to determine whether molybdenum concentrations in alkaline ash-grown plants would be harmful to animals.

Extremely high plant-tissue selenium concentrations have been reported for white sweet clover growing on acidic fly ash in New York (Gutenmann et al. 1976). When acidic fly ash-grown clover containing 66 µg Se/g dry wt. was fed as 23.5% of the diet (dry wt.) of adult goats, newborn kids, and lambs for 173 days (Furr et al. 1978a), high selenium concentrations were found in 11 body tissues, blood, goat's milk, and excreta when compared to control animals. No incidence of selenium intoxication was observed, despite the fact that the diet fed the experimental animals contained 16 µg Se/g dry wt., well within the 5-20 µg Se/g dry wt. concentration range known to cause animal poisioning (Gough et al. 1979). No gross or histologic lesions were present in any of the experimental animals. In other feeding studies, guinea pigs (Furr et al. 1975), Japanese quail (Stoewsand et al. 1978), and sheep (Furr et al. 1978b) were fed diets

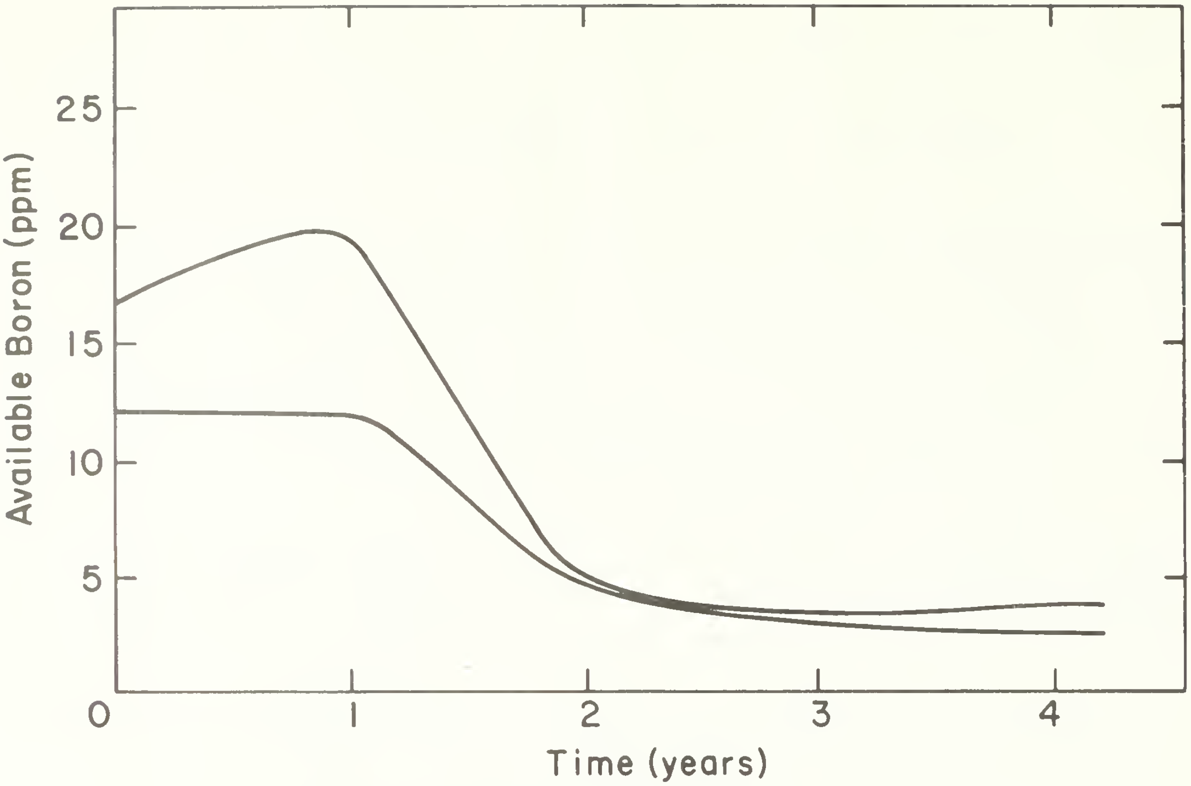

Figure 23. Variation of Plant-Available Boron with Time in Two British Fly Ashes with Moderate Initial Boron Content Exposed to Natural Weathering. Adapted from Townsend and Gillham (1975) (with permission, see credits).

which included ash-grown yellow sweet clover (*Melilotus officinalis*), ash-grown winter wheat (*Triticum aestivum*), and fly ash, respectively. In all cases, elevated selenium levels in blood and tissues were noted, but no significant toxicological or histological effects were found. Woodchucks (*Marmota monax*) inhabiting fly ash landfills exhibited elevated selenium levels in liver and lung tissue when compared to selenium levels in woodchucks collected from other locations (control animals), but again no visible lesions were found in these animals (Fleming et al. 1979).

Concentrations of rubidium were reported to be elevated in the tissues of guinea pigs (Furr et al. 1975) and in the tissues of lambs, adult goats, and newborn kids fed ash-grown sweet clover (Furr et al. 1978a). A nonessential element, rubidium is thought to partly substitute for potassium and therefore may be toxic (Gough et al. 1979). The concentrations of rubidium that will cause toxicity in animals is unknown.

The foregoing discussion indicates that animals foraging on vegetation grown on acidic fly ash may not be adversely affected by selenium and other trace metals concentrated in plant tissue. However, long-term field studies are needed of the effects of ash-grown forage on grazing animals, including cattle and wild ungulates.

Nutrition of plants growing on fly ash. Although fly ash is almost devoid of nitrogen (Tables 2 and 4), total concentrations of other essential nutrients are generally equal to or greater than those associated with typical soils. Available calcium, potassium, and magnesium concentrations in fly ash are considered more than adequate; supplemental additions of these nutrients caused small or negative growth responses in ash-grown plants (Hodgson and Townsend 1973). A considerable amount of phosphorus is present in fly ash; however, much of it is unavailable to plants. Townsend and Hodgson (1973) reported substantial yield increases following fertilization of ash-grown plants with water-soluble phosphate. The response of ash-grown plants to applied nitrogen and phosphorus is, in part, dependent upon the type of fertilizer used. Nitrate fertilizers can initially be more effective in supplying plants with nitrogen than ammonium fertilizers. Yields of white mustard grown on ponded fly ash (pH 8.5) were higher for nitrate-fertilized plants than for ammonium-fertilized plants, but the yields of a second white mustard crop grown on the same ash were approximately the same. The addition of liquid manure to fly ash has been shown to supply sufficient quantities of nitrogen, phosphorus, and potassium to support plant growth (Petrikova 1980).

Even when amply supplied with nitrogen and phosphorus, many species of plants exhibit symptoms of nutrient disorders when grown on fly ash (Townsend and Gillham 1975). Although the total concentrations of many micronutrients in fly ash are high, it appears that some nutrients are unavailable to plants. Because several micronutrients (e.g., iron, manganese, and zinc) have a low solubility in water under alkaline conditions, these nutrients as well as nitrogen and phosphorus may have to be supplied to plants growing on fly ash.

Plant establishment on fly ash. A number of studies have been conducted to determine the tolerance of plants to growth on fly ash. Among the plant types evaluated were grasses, legumes, cereals, root and vegetable crops (Hodgson and Townsend 1973; Duggan and Scanlon 1974), shrubs (Hodgson and Buckley 1975), and trees (Hodgson and Townsend 1973; Hodgson and Buckley 1975; Horton and McMinn 1977; Scanlon and Duggan 1979). These studies demonstrate that a variety of plants can be established directly upon fly ash, and they provide some information concerning growth and toxic trace-element uptake of ash-grown plants.

The emphasis of these investigations, unfortunately, has been on the growth and development of individual plant species, rather than on the development of plant communities on fly ash deposits. Study of plant community development is needed to determine whether self-sustaining communities can be established directly on fly ash. Little research has been conducted in this area, although some limited field-scale experiments have been reported. Townsend and Gillham (1975) have successfully grown timothy (*Phleum pratense*), orchard grass (*Dactylis glomerata*), and perennial ryegrass (*Lolium perenne*) for six years on both unfertilized and fertilized weathered fly ash and fly ash amended with dredged river silt. Symptoms of boron toxicity were evident in the early years of this experiment but diminished over time. In another field trial, a mixture of timothy, perennial ryegrass, and white clover (*Trifolium repens*) was seeded onto a fly ash-storage pond that had received varying applications of topsoil (0-, 8-, 15-, and 30-cm deep layers); half of each 12 × 46 m plot was then cultivated to a depth of 30 cm (to mix ash and soil), whereas the remainder was cultivated to a depth of 8 cm (Townsend and Gillham 1975). Yields on deep-cultivated ash plots without topsoil ranged from 1.7 to 2.4 MT/ha. The highest yields were obtained from shallow-cultivated plots with an 8-cm layer of topsoil over fly ash (5.6 to 6.9 MT/ha). In another series of experiments designed to establish pastureland on fly ash (Rippon and Wood 1975), dry matter yields of pure perennial ryegrass stands comparable to those discussed above were obtained when fly ash was amended with poultry manure (170 MT/ha), composted domestic refuse (1000 MT/ha), and sewage sludge (170 MT/ha). In the latter two experiments, significant yield responses occurred when plots were fertilized with nitrogen and, in some cases, with phosphorus and potassium.

It is uncertain whether the use of revegetated fly ash-storage sites for pastureland is economically or environmentally feasible, but the capability of establishing stands of vegetation on fly ash amended with fertilizer and organic materials has been proven. The longevity of these stands and the level of management required to maintain growth were not evaluated.

Discussions of the reclamation of disturbed lands and anthropogenic wastes typically do not include consideration of the role of soil organisms in functioning ecosystems. Since these organisms play a vital part in the processes of decomposition and nutrient cycling, the restoration of soil organism populations is essential to the development of self-sustaining ecosystems on reclaimed areas. Due to the high temperatures reached during coal combustion, fly ash is sterile when collected; however, microbial populations will develop through inputs of organisms from the atmosphere and from water used in ash ponding (Rippon and Wood 1975). The rate of colonization of a British fly ash by a variety of microorganisms is shown in Figure 24. One year following fly ash deposition, bacterial numbers in the fly ash were 1-10% of the numbers found in a typical soil (Rippon and Wood 1975). Cellulolytic organisms, fungi, and phosphate-splitting and denitrifying bacteria only appeared in appreciable numbers when the organic content of the ash had been increased; *Nitrobacter* (oxidizer of nitrite to nitrate) were rarely isolated.

It is known that the abundance of microorganism populations in fly ash will eventually approach that of soil (Rippon and Wood 1975), but it is unclear what effect the initially low populations of soil microorganisms in fly ash will have on plant growth during vegetative establishment. Direct revegetation of fly ash deposits has been shown to be more successful on weathered than on freshly deposited fly ash (Townsend and Gillham 1975). It is possible that enough time may elapse before reclamation for sufficient natural colonization of fly ash by soil microorganisms to occur, making inoculation of fly ash before revegetation unnecessary.

In summary, there are data to indicate that the characteristics of fly ash which are detrimental to plant growth can be ameliorated, allowing the establishment of vegetation directly on fly ash surfaces. Fertilization with nitrogen and phosphorus, and in some instances micronutrients, as well as amendment with soil or organic material (e.g., sewage sludge, poultry

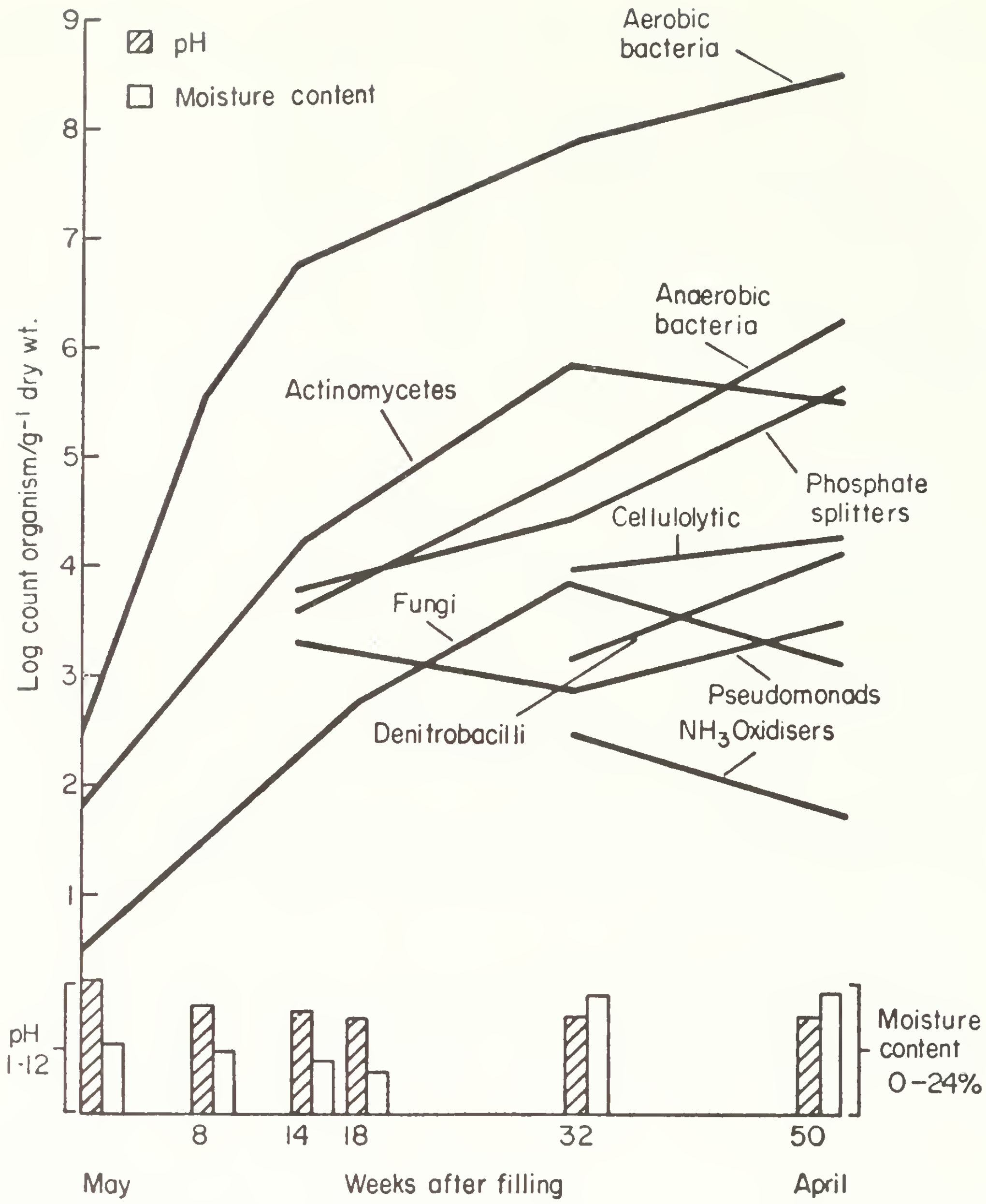

Figure 24. The Rate of Colonization by Various Soil Microorganisms of a British Fly Ash High in Boron Content. From Rippon and Wood (1975) (with permission, see credits).

manure, peat) appear to be required for plant growth. Proper amendments along with the use of plant species tolerant of the adverse properties of fly ash may allow for the successful revegetation of: (1) extensively weathered fly ash and (2) fly ash having a neutral to acidic pH. Before this technique can be endorsed for widespread use in the reclamation of fly ash deposits, further research is needed to examine the long-term survival of vegetation established on fly ash, the hazard of ash-grown forage to animals, and the costs of maintaining vegetation established on fly ash. This technique may find more use as a method of temporarily stabilizing fly ash deposits before final reclamation of the storage site is undertaken. In most cases, reclamation will require placement over the ash of a cover material capable of supporting plant growth.

Revegetation of scrubber sludge. At present, limited information is available concerning the revegetation of scrubber sludge. However, if hundreds of hectares will be needed for waste-storage ponds to contain the slurry produced by scrubbing operations (Table 17), reclamation will be required to prevent wind erosion, improve aesthetics, and return the land to other uses, e.g. wildlife habitat (Terman 1978). Direct revegetation of sludge ponds could be the most cost-effective method of reclamation, but high sulfite and boron concentrations (from entrained fly ash) coupled with the thixotropic nature of the sludge and a lack of essential plant nutrients can severely retard plant growth. Vegetation trials using grass and clover at Muscle Shoals, Alabama, indicate that boron toxicity is a primary deterrent to plant establishment (Terman 1978). The time required for the leaching of boron to nontoxic levels has not been established. However, adequate to moderately good stands of bermuda grass (*Cynodon dactylon*) and tall fescue (*Festuca arundinacea*), as well as a few reed canarygrass (*Phalaris arundinacea*) plants, have been established directly upon limestone scrubber sludge (Buchanan 1979, unpublished results).

Establishment of four tree species--black locust (*Robinia pseudoacacia*), sycamore (*Platanus occidentalis*), cottonwood (*Populus deltoides*), and autumn olive (*Elaeagnus umbellata*)--on limestone scrubber sludge has also been attempted. More vigorous growth was noted when all tree species were planted on scrubber sludge amended with sewage sludge rather than on unamended scrubber sludge (Buchanan 1979, unpublished results). Further research is required to determine whether the direct revegetation of scrubber sludge is a viable option.

Covering Wastes with Soil

Regardless of the method of storage (i.e., ponding or landfill), the reclamation of storage sites for coal combustion wastes (i.e., fly ash, scrubber sludge, and mixtures of both wastes) will primarily involve covering these wastes with a layer of soil and the revegetation of this soil mantle. The use of a soil mantle will aid in the stabilization of the waste deposit, limit water movement through the wastes, and possibly reduce the toxicity of waste constituents to the plants used in revegetation. Two types of reclamation techniques involving soil mantle placement over coal combustion wastes are: continuous reclamation and site-closure reclamation.

In continuous reclamation (Figure 25), soil is stripped from areas of a landfill being prepared for use and placed over the compacted wastes being dumped at the site. Revegetation of

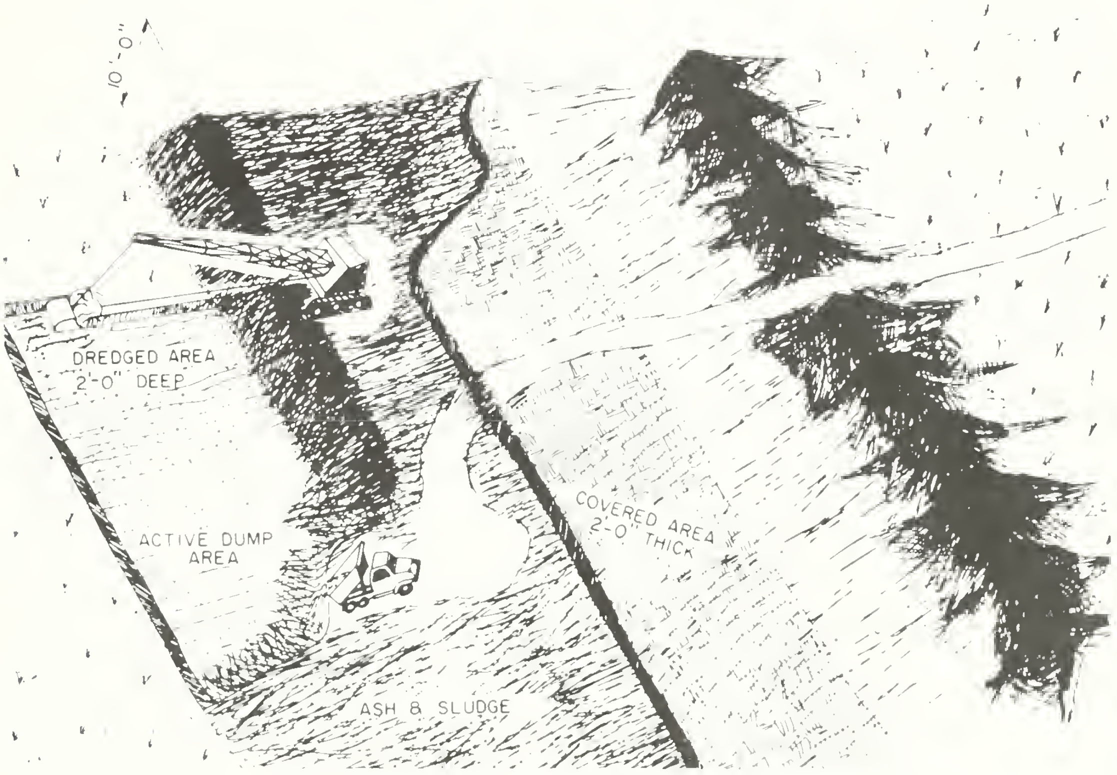

Figure 25. Diagram of Continuous Reclamation Technique for Coal Combustion Wastes. From U.S. Bureau of Land Management (1979).

the reapplied topsoil takes place soon after soil placement, and continues throughout the active life of the storage site. Due to the logistics of individual landfill sites, soil will often have to be stockpiled for future use (e.g., soil removed from the area prepared for waste deposition). The length of time soil is stockpiled should be kept to a minimum. When soil is not stockpiled or otherwise disturbed for extended periods of time, little amendment may be required to successfully revegetate this material. However, to better ensure the success of reclamation, the amount of soil amendment employed should be determined in accordance with the results of soil tests. Continuous reclamation has the added advantage of limiting the time the wastes are directly exposed to natural weathering, thus reducing wind erosion and the amounts of trace metals and soluble salts leached to surrounding surface waters and groundwaters. This reclamation technique is only applicable to landfill storage sites.

Reclamation of ponded coal combustion wastes and some landfill sites will be conducted following the active life of the site. In the case of ponded fly ash or scrubber sludge, sufficient time for the dewatering and solidification of the wastes must also be allowed. Coal combustion wastes are covered by a mantle consisting of either (1) topsoil and/or subsoil segregated for this purpose during storage-site construction or (2) soil material obtained elsewhere and transported to the site. Offsite sources of soil material include construction sites, quarries, and borrow pits (dug solely to fulfill this need). Site-closure reclamation may often require the long-term stockpiling of soil for later reapplication, a procedure detrimental to both soil organisms and soil fertility. Both stored soil and subsoil will need fertilization; in some cases, other amendments will be needed to support vigorous plant growth. Significant changes in both soil microbiology and soil chemistry are known to occur when soil is stored in large stockpiles for extended periods of time (Miller and May 1979). Subsoil typically contains lower concentrations of plant nutrients than topsoil.

Despite the emphasis placed on the burial of ash and sludge wastes in the reclamation plans of proposed waste-storage sites, few specific details of the reclamation procedures that will be employed are available. Furthermore, little research has been done to determine whether waste burial is an effective reclamation technique. Among the many questions which remain unanswered is the optimal thickness of the soil mantle required for plant growth in the various regions of the United States. The primary factor affecting soil mantle thickness will be the moisture regime of the storage site. In arid regions of the country, a very thick soil mantle will be required to sustain plant growth whereas a thinner mantle will be required in more mesic regions. The projected land use of the reclaimed storage site will affect soil mantle thickness. If the site is to be used for food crops, a deeper layer of soil may be needed to protect the integrity of the site than is needed for other uses. Soil mantle thickness not only will affect the success of revegetation (plant growth improves with increasing soil depth), but also will significantly affect the cost of reclamation. This is especially true for site-closure reclamation, when soil material may have to be purchased and transported from offsite.

Hodgson et al. (1963) studied the influence of soil cover depth over fly ash and the rate of soil fertilization on the growth and yield of four cereal crops and potatoes. The results of this small-scale field study indicated that 60 cm of soil fertilized at normal rates was required to obtain satisfactory yields. A minimum of 30 cm of soil, supplied with 1.5 times the normal fertilizer requirements of each of these crops, was needed to produce adequate yields. The high level of continuous nutrient subsidy required to sustain plant growth on 30 cm of soil is probably economically unacceptable for use in large-scale reclamation as it would require extensive long-term maintenance of the site. A 25-cm layer of subsoil applied over alkaline fly ash did not significantly improve the growth of eight woody species over the growth of these woody species on fly ash alone (Scanlon and Duggan 1979).

Dvorak et al. (1979) reported that vegetation growing on a 60-cm mantle of subsoil placed over acidic coal mine refuse was able to survive a five-week drought better than vegetation growing on 15- or 30-cm deep subsoil mantles. Because plant roots cannot grow into the underlying acidic refuse, plant-available moisture is limited to that contained in the subsoil. In the case of buried coal combustion wastes, root penetration of the wastes will probably occur, in some instances, but will be dependent upon the degree of compaction, alkalinity, and soluble salt content of the wastes. If root penetration does occur, plants will then be able to draw upon moisture held in the ash, which typically has a high moisture-holding capacity. However, if the soluble-salt content of the buried wastes is high, resulting in high osmotic pressures, plant-available moisture in the wastes may be quite low. During dry periods, high moisture content in the underlying fly ash is thought to have a beneficial effect on crops grown in soil placed over fly ash (Hodgson et al. 1963). However, a thick soil mantle--which provides greater water retention--may be required to support the vegetation established on the mantle, especially where plant-available soil moisture becomes limiting during the growing season.

On the basis of the limited available research, it appears that a 60-cm layer of soil is the minimum depth required for successful revegetation; this soil depth is often mentioned in the reclamation plans of proposed waste-storage sites. A great deal more work is needed to substantiate this finding and to determine the regions of the country where this soil depth would not be adequate.

Until further data are obtained, the use of thick soil mantles (> 60 cm) to bury coal combustion wastes may be desirable to guard against several potential problems. Root penetration of the buried wastes may allow for the translocation and transport of trace elements from the waste materials into plant tissues in sufficient concentrations to cause toxicity problems in plants and their consumers. The possibility also exists for the movement of soluble salts upward through the soil mantle, adversely affecting plant growth and further limiting the available rooting zone. If trace-element or soluble-salt toxicity problems are found to occur, thicker soil mantles may be sufficient to alleviate these problems. Although costly, the placement of an impervious clay "cap" over the wastes to retard the upward movement of trace elements and salt may be necessary. To alleviate these problems in the reclamation of retorted oil shale, a gravel, capillary water-movement prevention layer is employed (Cook 1974).

Permanent Impoundments

Generally, sites that contain coal combustion wastes should not be reclaimed to permanent impoundments. A permanent impoundment would result in the continued presence of a hydraulic head, driving seepage through the impoundment substrate which can contain constituents potentially toxic to fish and wildlife (Tables 8 and 10). Even if the impoundment is adequately sealed with a liner, the liner could fail thereby releasing leachate into the surrounding environment. In order to mitigate the potential for liner failure, continued monitoring and maintenance of the site would be required after decommissioning. Initial construction of pond underdrainage systems to collect leachate seepage would correct this problem, but continued system operation and leachate seepage treatment would be expensive.

The impounded waters over wastes could also contain a number of elements potentially toxic to biota. It appears likely that, in most areas, these waters would not be suitable for either supporting game fish or providing water for wildlife. This problem might be alleviated by placing an impermeable liner between the aquatic system and the combustion wastes. However, this liner could also fail, leading to contamination through dissolution and suspension of waste products in the water medium and subsequent contamination of the aquatic ecosystem. Reclamation of coal combustion waste sites to permanent impoundments would increase the likelihood of biota contacting the waste and is not a recommended alternative reclamation option. There might be some site-specific cases in which water quality could support biota, but an impoundment created from a storage site containing combustion wastes generally would not provide beneficial fish and wildlife habitat without extensive manipulation and monitoring of the impoundment system.

RECLAMATION SITE MANAGEMENT

Combustion waste-storage sites should be preplanned so that the design, operation, and reclamation of the site results in the maximum possible stability of the wastes during reclamation and after reclamation has been completed. The final cover material should be graded to encourage runoff and minimize infiltration; Walsh et al. (1978) recommends final slopes from 2 to 5% for municipal sludge landfills. Minimizing infiltration, however, can increase drought problems for vegetation. The stability of the material is a major determinant of the proper slope required to prevent sloughing of surface materials.

To prevent erosion, final cover material placed over combustion waste material should be managed with control practices. It is likely that extended periods of geological time could be required at some storage sites to produce soils from combustion waste materials capable of supporting viable biological communities. Thus, what currently may appear to be excessive reclamation requirements to prevent erosion could be required for long-term physical stability of storage sites as well as long-term viability of associated biological communities.

Controlling soil loss from water and wind erosion is a prerequisite to successfully establishing a vegetative cover. Final grading and drainage plans, however, are largely determined by site-specific characteristics.

POSTRECLAMATION SITE MANAGEMENT

Since no large-scale reclamation of ash and sludge waste-storage sites has yet been attempted, it is difficult to define what will be involved in managing a reclaimed storage site and impossible to identify specific management practices. For the purpose of this discussion, postreclamation site management includes all efforts required to perpetuate vegetation established on the site and maintain the physical integrity of the site. A high level of management will be required immediately following reclamation. If the reclamation plan for the storage site has been properly designed and executed, ideally the degree of site management required will diminish quickly to a low level.

The major emphasis of reclaimed waste-site management can be classified as (1) monitoring environmental and site conditions, and (2) site maintenance. Monitoring the effects of waste-storage sites on the surrounding environment has been discussed, but information concerning the condition of the site itself is also needed. Immediately following reclamation, data should be gathered describing germination and early growth of vegetation. Decisions can then be made as to the need for additional fertilization, reseeding, irrigation, and fencing of the site against grazing animals. Periodic measurement of plant density and/or plant cover over several growing seasons will indicate the success of revegetation. Additional observations should be made to estimate the suitability of the revegetated area for wildlife. At sites where ash and sludge wastes have been buried beneath a soil mantle, the effect of the underlying wastes on the mantle itself should be periodically determined by monitoring soil pH, electrical conductivity, plant-available soil moisture, and trace-element content at various levels through the mantle. Analysis of vegetation for macronutrient and trace-element content will determine the need for fertilization and whether or not trace-element toxicity problems exist. Examination of the viability of seed produced on the reclaimed site will indicate the vigor of established vegetation, as well as whether a self-sustaining plant community is being developed.

Site maintenance will include work identified as necessary by the monitoring program and other required upkeep. Repair of fences surrounding the site, maintenance and clearing of water drainage pathways, and erosion-control structures will be needed so that the reclaimed storage site can persist as a self-sustaining community.

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ILLUSTRATIVE ASSESSMENTS OF IMPACTS TO FISH AND WILDLIFE FROM FOUR MODEL STORAGE SITES

Data on four model 2100-MWe coal-fired power plants are presented to illustrate how to use the guidelines given in this report for assessing proposed coal combustion waste storage sites, management of active sites, and reclamation of former sites. Values presented for quantification purposes are approximate values and, due to rounding errors, recalculation will not result in the exact values presented here. Model plant locations and coal characteristics are presented in Table 43. Coal types reflect regionally observed variations in coal composition.

Table 43. Ash, Sulfur, and Heat Values of the Coals Utilized by the Four Model 2100-MWe Coal-Fired Power Plants

Model plant	Coal type	Ash (%)	Sulfur (%)	Heat value (Btu/lb)	Sulfur (lb/10^6 Btu)
Western	Low-sulfur[a]	6.0	0.48	8,200	0.58
Ohio River valley	High-sulfur[b]	10	3.5	11,400	3.07
Texas	Lignite[c]	10	0.8	7,705	1.04
Southeastern coastal	High-sulfur[d]	12.4	1.6	13,135	1.22

[a] Powder River, Wyoming - Anderson & Canyon & Wyodak-Anderson.
[b] Eastern Interior, Illinois - (No. 5) Harrisburg-Springfield.
[c] Gulf Coast Lignite, Texas - Wilcox Group.
[d] Southern Appalachian, Alabama - Mary Lee.

WESTERN PLANT BURNING LOW-SULFUR COAL

Plant Operations

The western plant burns low-sulfur coal, uses pulverized-coal furnaces, has a heat rate of 8980 Btu/kWh, and operates with a plant factor of 0.70. The plant uses lime scrubbers with a 90% SO_2 removal efficiency to comply with the federal SO_2 emission rate of 70% removal for raw coal with 0.58 lb SO_2 per million Btu (Figure 5). The plant uses "dry-bottom" boilers and has an ash split factor of 15% aggregate and 85% fly ash. Electrostatic precipitators with a fly ash collection efficiency of 99.5% have been installed to comply with the federal particulate emission rate of 0.03 lb particulates per million Btu of coal. The lime scrubber removes an additional 0.1% of the fly ash to achieve the federal emission rate. The ratio of $CaSO_4 \cdot 2H_2O$ to $CaSO_3 \cdot \frac{1}{2}H_2O$ in the scrubber sludge solids is 0.2 to 0.8. The scrubber sludge is 15% solids by weight.

Handling of Combustion Wastes

Ash residuals are used as fill material in the surface coal mine providing coal for the plant. Scrubber sludge is mechanically thickened to 35% solids by weight. Water from the thickening process is recycled in the scrubbing system. Supplementary scrubber system water is pumped from the Powder River. Thickened sludge is disposed of in a partially incised, diked storage pond. The storage site has been excavated to a depth of 3 m, with the excavated soil material being used in the construction of restraining dikes to allow scrubber sludge to be deposited 9.1 m deep. The above-grade restraining dike is 7.6 m high and 65.5 m wide at the base. The outer slope of the dike has a 5:1 grade, the inner slope a 3:1 grade. The storage site occupies approximately 53 ha (130 acres), including the area occupied by waste, restraining dikes, and associated pipelines and access roads. The pond is lined with clay having a hydraulic conductivity of 7.5×10^{-7} cm/s. The pond has an effluent discharge facility. As individual cells of the storage pond are filled, they will be stabilized for reclamation by natural evaporation, and the stabilized storage area will be graded, covered, and revegetated.

Description of the Storage-Site Area

Ash residuals are stored in a large upland surface coal mine 8 km (5 miles) from the plant. The sludge storage site is located in an alluvial area adjacent to the Powder River in Wyoming. The diked storage pond is 300 m or more from the river and meets applicable state and federal specifications.

The soil type of the storage pond area is Kim loam, an Entisol, with 0 to 3% slope. A soil description of Kim loam is given in Table 44.

Table 44. Characteristics of Kim Loam

Particle-size distribution, 0 - 114 cm:	
Clay (< 0.002 mm)	21 - 26%
Silt (0.05 - 0.002 mm)	39 - 42%
Very fine sand (0.1 - 0.05 mm)	23 - 26%
Fine sand (0.25 - 0.1 mm)	9 - 12%
Medium sand (0.5 - 0.25 mm)	0.4 - 0.6%
Very coarse and coarse sand (2.0 - 0.5 mm)	0.3 - 0.6%
Extractable bases:	
Calcium	14.2 - 11.4 meq/100 g soil
Magnesium	2.2 - 2.4 meq/100 g soil
Sodium	0.4 - 2.4 meq/100 g soil
Potassium	0.6 meq/100 g soil
Cation exchange capacity (base saturation = 98-100%), 0 - 114 cm	15 - 16 meq/100 g soil
Reaction pH	7.4 - 9.0
Area runoff	Slow
Erosion hazard	Slight
Hydraulic conductivity	Moderate (1×10^{-4} to 1×10^{-3} cm/s)
Available water capacity	High
Bulk density at field capacity (19%)[a]	1.45 g/mL
Depth of permanent water table surface	5 m
Depth of bedrock	9 m
Effective rooting depth	≥ 1.5 m

[a] About 7-8% of this water is unavailable to plants.

The area is currently managed for range and wildlife habitat but has potential for being managed for irrigated hay, small grain, and pasture. Dominant vegetation is the climax vegetation for the area--about 50% western wheatgrass (*Agropyron smithii*), thickspike wheatgrass (*Agropyron dasystachyum*), green needlegrass (*Stipa viridula*), Indian ricegrass (*Oryzopsis asperifolia*), American vetch (*Vicia americana minor*), winterfat (*Eurotia lanata*), and prairie-clover (*Petalostemon* spp.), and about 50% needle-and-thread (*Stipa comata*), blue grama (*Bouteloua gracilis*), Sandberg bluegrass (*Poa secunda*), junegrass (*Koeleria cristata*), silverleaf scurf pea (*Psoralea argophylla*), locoweed (*Astragalus* spp.), and hoods phlox (*Phlox hoodii canescens*). Range conditions have deteriorated slightly, allowing annual invaders [curlycup gumweed (*Grindelia squarrosa*), foxtail barley (*Hordeum jubatum*)], prickly pear cactus (*Opuntia* spp.), big sagebrush (*Artemisia tridentata*), and short grasses to appear. Wetland sedges (*Carex* spp.), plains cottonwood (*Populus sargentii*), and willow trees (*Salix* spp) are found along a small ephemeral stream which borders the area. The area around the storage site is used by pronghorn (*Antilocarpra americana*), mule deer (*Dama hemionus*), and white-tailed deer (*Dama virginiana*) as well as livestock. Desert cottontail (*Sylvilagus audubonii*), Nuttall's cottontail (*Sylvilagus nuttallii*), white-tailed jackrabbit (*Lepus townsendii*), coyote (*Canis latrans*), ring-necked pheasant (*Phasianus colchicus*), and wild turkey (*Meleagris gallopavo*) frequent the area. Forage value of the area is higher than most habitat types in the region, and similar habitats available for wildlife are in limited supply. Endangered and threatened species have not been reported in the area.

Material Requirements for the Model Plant

The coal requirement for the model plant is 6.4 × 10^6 MT/yr (7.0 × 10^6 tons/yr) (Equation 12) based on plant operation specifications and coal data presented in Table 43. Assuming that 1.1 calcium atoms are supplied to the lime scrubber for every sulfur atom removed and that the scrubber SO_2 removal efficiency is 70%, 3.7 × 10^4 MT/yr (4.1 × 10^4 tons/yr) of lime are required.

Coal Combustion Wastes

The ash generation rate is 28 ha-m/yr (220 acre-ft/yr) based on plant operation specifications, coal data presented in Table 43, and quantification procedures presented earlier (Equations 14-18). Assuming a specific gravity of 2.5 for scrubber sludge solids, the scrubber sludge generation rate is 56 ha-m/yr (450 acre-ft/yr) (Equations 20-26). Mechanically thickened sludge (35% solids by weight) is produced at a rate of 21 ha-m/yr (170 acre-ft/yr), and stabilized sludge (65% solids) is produced at a rate of 9 ha-m/yr (70 acre-ft/yr).

Consumptive Use of Water

The lime scrubbing process generates a slurry of 85% water by weight and requires 52 ha-m (420 acre-ft) of water per year supplied from the Powder River. The slurry is thickened to 65% water by weight, and water from the thickening process is recycled to the scrubbing system. Thus, 34 ha-m (280 acre-ft) of water are recycled per year, and 17 ha-m (140 acre-ft) of water are required per year. However, based on leachate seepage estimates, 9 ha-m/yr (70 acre-ft/yr) of the 17 ha-m/yr (140 acre-ft/yr) are reintroduced to the Powder River water system and the loss is 9 ha-m/yr (70 acre-ft/yr). The net loss of water is the average precipitation falling on the area, which is 38 cm on 40 ha (98 acres) or 16 ha-m/yr (130 acre-ft/yr), plus 9 ha-m/yr (70 acre-ft/yr); thus, the net water loss is 25 ha-m/yr (200 acre-ft/yr).

Loss of Habitat

The scrubber sludge storage site preempts approximately 53 ha (130 acres) of the Powder River alluvial valley floor (see the Section on Handling of Combustion Waste for this type of plant). The square storage pond is more space-effective than an elongated pond. However, a landfill would probably require less land than the pond system. Preemption of land has not been considered for ash wastes because the area has already been preempted by mining activities.

Effluent Discharge from the Storage Site

The storage site has an emergency effluent discharge structure, although no surface discharges are expected. The average annual net precipitation for the region is -81 cm and annual residual water in the storage pond, after accounting for leachate seepage and the water content of stabilized sludge, is 34 cm. Effluent from the ash wastes is not expected to substantially alter effluent assocated with the coal mine.

Runoff Dispersal of Coal Combustion Wastes

Dispersal of scrubber sludge wastes via runoff should not occur from the partially incised, diked storage pond. The length-slope factor in the Universal Soil Loss Equation is zero for the storage area; therefore, erosion loss per unit area per unit time is zero. Potential for soil loss from storage pond dikes is minimized to preserve dike integrity. Runoff dispersal of ash wastes used as mine fill is minimized by pollution-abatement procedures employed at the surface mine.

Leachate Seepage Discharge from the Storage Site

Because a permanent head of water will not develop in the pond, leachate discharge from the storage pond is estimated from Figure 16. Estimated seepage discharge from the pond is limited by the clay liner which has a hydraulic conductivity of 7.5 × 10^{-7} cm/s. Therefore, the discharge rate is approximately 6.3 m^3/day/ha (680 gal/day/acre) through an average of 40 ha (98 acres) or 250 m^3/day seepage from the storage site. Leachate from the ash wastes is not expected to substantially alter the quantity or quality of leachate associated with coal mining activities.

Fugitive Dusting

Fugitive dusting from storage pond dikes is minimized to preserve dike integrity. However, increased dusting is likely to occur from the pond as the basin surface dries. There are too many variables present to predict the intensity of sludge particulate dusting from the storage pond. Fugitive dusting from the ash wastes is not expected to substantially alter fugitive dusting associated with the coal mine.

Reclamation of the Storage Site

The successful revegetation of the scrubber sludge storage pond at the western model plant will be difficult. The principal factor limiting plant establishment in this semiarid region will be the availability of sufficient moisture to support plant growth. The deep-rooted plants native to this alluvial site receive much of their moisture requirements through subirrigation moisture, which will not be available to plants growing on topsoil placed over the dewatered sludge. Plants established on the storage site will most likely be dependent solely upon moisture contained within the soil mantle itself; it is not known whether plant roots can penetrate buried scrubber sludge and utilize the moisture held in this material. The moisture content of a typical soil mantle (0.6 m thick) will probably be insufficient to support plant growth through the dry western summer. This problem may be circumvented through the application of an extremely thick soil mantle; however, the cost of topsoil application may make this solution impracticable. Application of 0.3 m of topsoil to an acre of land in the Northern Great Plains Coal Region costs approximately $1000.00. The effective rooting depth at this site is estimated to be at least 152 cm (60 inches) deep. To apply a soil mantle of this thickness to the entire storage site would cost at least $950,000, which does not include consideration of the availability or cost of the topsoil itself.

State-of-the-art reclamation techniques used in the western United States were developed to alleviate the problems associated with strip mine spoils and mineral tailings. It is highly unlikely that these techniques can be successfully employed to reclaim scrubber sludge storage ponds without significant modifications. A considerable amount of research is needed to determine what modifications are needed, and if these areas can be successfully reclaimed at all.

Consequences to Biota

Consumptive use of water. Consumptive use of water (25 ha-m/yr or 200 acre-ft/yr) at the storage site, along with other consumptive uses of the power plant, puts increased pressure on already scarce water resources of the area. Removal of water from the Powder River during periods of low flow may result in adverse impacts to fish and wildlife.

Loss of habitat. Preemption of the land will result in an incremental loss of range and wildlife habitat (53 ha or 130 acres) that has a higher value than most habitat types in the region. Successful reclamation of coal waste-storage areas to ambient habitat conditions has not been demonstrated. Therefore, there will be a potential for decrease in those species (e.g., pronghorn, mule deer, white-tailed deer, desert cottontail, and coyote) utilizing this habitat. Even after reclamation, the site will not be as valuable for range or wildlife habitat. Although the affected area is of high wildlife value, it is not a large fraction of this habitat type that is available in the region. Thus, impacts to wildlife populations would be expected to be localized.

Effluent discharges. Surface discharges from the storage pond should not occur; however, if such discharges do occur, they should be infrequent and small. Discharges that take place are assumed to have the constituent concentrations outlined in Table 45. A moderate amount of dilution will be required to achieve acceptable estimated permissible ambient concentrations (EPC) in surface waters (Tables 30 and 45). Nickel and possibly mercury and selenium could pose problems if large, accidental discharges occur during minimum flow of the Powder River. Based on the infrequent and small nature of storage pond discharges and on the concentration of potentially toxic constituents, if the site is properly managed there should be little biological concentration and magnification of potentially toxic constituents to toxic levels. Therefore, impacts to fish and wildlife should be minor.

Runoff dispersal. Runoff dispersal from the pond storage site should result in little, if any, movement of potentially toxic combustion wastes and should have little impact on fish and wildlife.

Leachate seepage. Leachate seepage is assumed to have the constituent concentrations outlined in Table 45. Leachate seepage is expected to move at a rate of 7.5×10^{-7} cm/s through the clay liner and at a rate of 1×10^{-4} cm/s (moderate hydraulic conductivity) through the substrate beneath the liner, or 133 times faster than through the clay liner. Assuming that the substrate is 33% water by volume, leachate movement away from the site should be sufficient to dilute total constituent concentrations to EPCs in the soil (Tables 30 and 45). Even nickel should not pose a short-term problem because the USEPA estimates that dispersal and movement of

Table 45. Factors by Which Ambient Concentrations Exceed Estimated Permissible Ambient Concentrations at the Western Model Power Plant

Element	Concentration in leachate[a] (μg/L)	Water[b]	Soil or Sediment[b]
Antimony	14	0	0
Arsenic	2	0	0
Beryllium	2	0	0
Boron	2600	1	0
Cadmium	0.5	1	0
Chromium	1	0	0
Copper	31	3	0
Lead	5.6	1	0
Manganese	2	0	0
Mercury	0.5	10	1
Molybdenum	63	0	1
Nickel	50	25	3
Selenium	45	9	1
Vanadium	100	1	0
Zinc	5	0	0

[a] From Holland et al. (1975).
[b] From Equations 30 and 31.

materials through soils are attenuated by a factor of 100 below their concentrations in the leachate (Tables 36 and 37). There should be little biological concentration and magnification of potentially toxic constituents to toxic levels based on the concentration factors presented in Table 29. Therefore, little impact to biota is expected due to leachate seepage. However, in the immediate vicinity of the site, there is potential for gradual buildup of constituents (particularly nickel) in soil through physical and chemical processes to concentrations that could exceed EPC values. Leachate seepage will also provide long-term incremental additions to the soil. Additionally, there could be high background concentrations of potentially toxic constituents and, with addition of leachate seepage constituents, critical levels required for protection of fish and wildlife could be exceeded. The soil reaction pH of 7.4 to 9.0 (Table 44) could cause some potentially toxic leachate constituents to become less mobile in the substrate system. If soil attenuation of constituent concentrations approaches a factor of 100 as expected, sufficient dilution should have occurred by the time seepage reaches Powder River surface water, 300 m away, to reduce total constituent concentrations below total EPC requirements.

Wind dispersal. Wind dispersal from the pond storage site should result in minor movement of potentially toxic combustion wastes and should have little impact on fish and wildlife.

OHIO RIVER VALLEY PLANT BURNING HIGH-SULFUR COAL

Plant Operations

The Ohio River Valley plant burns high-sulfur coal, uses pulverized-coal furnaces, has a heat rate of 8980 Btu/kWh, and operates with a plant factor of 0.70. The plant uses lime scrubbers with a 90% SO_2 removal efficiency to comply with the federal SO_2 emission rate of 90% removal for raw coal with 3.07 lb SO_2 per million Btu (Figure 5). The plant uses "dry-bottom" boilers and has an ash split factor of 15% aggregate and 85% fly ash. Electrostatic precipitators with a fly ash collection efficiency of 99.5% have been installed to comply with the federal particulate emission rate of 0.03 lb particulates per million Btu of coal. The lime scrubber removes an additional 0.1% of the fly ash to achieve the federal emission rate. The sulfate to sulfite ratio in the scrubber sludge solids is 0.2 to 0.8. The scrubber sludge is 15% solids by weight.

Handling of Combustion Wastes

Ash residuals are disposed of with scrubber sludge in a diked storage pond. The diked pond is completely above-grade because of the shallow water table. Restraining dikes are 10.7 m high and 89.9 m wide at the base. The outer slope of the dike has a 5:1 grade, the inner slope a 3:1 grade. The combined ash and sludge will be deposited to a depth of 9.0 m. The square storage site (including a 30.5-m buffer zone)--with the deposited waste, restraining dikes, and associated access roads, etc.--will occupy approximately 670 ha (1700 acres) at the end of plant operations. The pond is lined with clay having a hydraulic conductivity of 5×10^{-7} cm/s. The combustion wastes are stabilized to 65% solids by weight in the storage pond by natural evaporation and by removal of excess supernatant through a controlled effluent discharge after adequate settling of suspended solids has occurred. The stabilized storage area will be graded, covered, and revegetated.

Description of the Storage-Site Area

The waste-storage site is located in an alluvial area adjacent to the Ohio River in Ohio. The diked storage pond is 300 m or more from the river and meets all applicable state and federal specifications. Levees will be constructed as needed to protect the area from flooding.

The storage pond area soil type is Huntington silt loam, a Mollisol, which is nearly level. A soil description of Huntington silt loam is given in Table 46.

Table 46. Characteristics of Huntington Loam

Particle-size distribution; 0 - 279 cm:	
Fine clay (< 0.0002 mm)	4 - 10%
Clay (< 0.002 mm)	21 - 34%
Silt (0.05 - 0.002 mm)	39 - 58%
Very fine sand (0.10 - 0.05 mm)	8 - 15%
Fine sand (0.25 - 0.10 mm)	1 - 23%
Medium sand (0.5 - 0.25 mm)	0.1 - 3%
Coarse sand (1 - 0.5 mm)	0 - 0.1%
Very coarse sand (2 - 1 mm)	0 - 0.1%
Extractable cations:	
Hydrogen	4.0 - 8.0 meq/100 g soil
Calcium	8.2 - 12.9 meq/100 g soil
Magnesium	1.5 - 3.0 meq/100 g soil
Potassium	0.17 - 0.38 meq/100 g soil
Cation exchange capacity (base saturation = 49-83%), 0 - 279 cm	13 - 24 meq/100 g soil
Reaction pH	6.2 - 6.7
Hydraulic conductivity	Moderate (1×10^{-4} to 1×10^{-3} cm/s)
Available water capacity	High (0.15 - 0.19 cm/cm soil)
Area runoff	Moderate
Depth of seasonally high water table surface	1.5 m
Depth of bedrock	10 m
Rooting zone	Deep

Approximately 50% of the area is currently used for row crops (corn and soybeans) and the other 50% was formerly cultivated but has been abandoned and is reverting to woodland. Invading species include white ash (*Fraxinus americana*), willow (*Salix* spp.), swamp-white oak (*Quercus bicolor*), and black cherry (*Prunus serotina*). Huntington silt loam is well suited for grain and seed crops, grasses and legumes, wild herbaceous upland plants, and hardwood plants making habitat that is well suited for open-land wildlife or woodland wildlife. Open-land wildlife species include bobwhite (*Colinus virginianus*), ring-necked pheasant (*Phasianus colchicus*), eastern meadowlark (*Sturnella magna*), field sparrow (*Spizella pusilla*), mourning dove (*Zenaidura macroura*), eastern cottontail rabbit (*Sylvilagus floridanus*), red fox (*Vulpes fulva*), and woodchuck (*Marmota monax*). Woodland wildlife species include American woodcock (*Philohela minor*), wood thrush (*Hylocichla mustelina*), vireo (*Vireo* spp.), scarlet tanager (*Piranga olivacea*), gray squirrel (*Sciurus carolinensis*), fox squirrel (*Sciurus niger*), gray fox (*Urocyon cinereoargenteus*), white-tailed deer (*Dama virginiana*), raccoon (*Procyon lotor*), and opossum (*Didelphis marsupialis*). Wetland wildlife are also found in the area due to the proximity of satisfactory habitat along the Ohio River. Wetland wildlife species include ducks, geese, rails, herons, shore birds, mink (*Mustela vison*), and muskrats (*Ondatra zibethicus*). Bald eagles (*Haliaeetus leucocephalus*), a threatened species, and Virginia big-eared bats (*Plecotus townsendii virginianus*), an endangered species, have been reported in the area.

Material Requirements for the Model Plant

The coal requirement for the model plant is 4.6×10^6 MT/yr (5.1×10^6 tons/yr) (Equation 12) based on plant operation specifications and coal data presented in Table 43. Assuming that 1.1 calcium atoms are supplied to the lime scrubber for every sulfur atom removed and that the scrubber SO_2 removal efficiency is 90%, 2.5×10^5 MT/yr (2.8×10^5 tons/yr) of lime are required.

Coal Combustion Wastes

The ash generation rate is 33 ha-m/yr (270 acre-ft/yr) based on plant operation specifications, coal data presented in Table 43, and quantification procedures presented earlier (Equations 14-18). Assuming a specific gravity of 2.5 for scrubber sludge solids, the scrubber sludge generation rate is 380 ha-m/yr (3080 acre-ft/yr) (Equations 20-26). The combined stabilized waste production rate (65% ash and sludge solids by weight) in diked ponds is 120 ha-m/yr (960 acre-ft/yr).

Consumptive Use of Water

The lime scrubbing process generates a slurry of 85% water by weight and requires 360 ha-m (2880 acre-ft) of water per year. Excess water from the storage pond is not recycled to the scrubbing system. However, based on leachate seepage estimates, 80 ha-m/yr (650 acre-ft/yr) of the 360 ha-m/yr are reintroduced to the Ohio River water system and 750 ha-m/yr (6100 acre-ft/yr) minus 540 ha-m/yr (4360 acre-ft/yr) of precipitation are reintroduced by surface discharges. Therefore, net consumptive water use is 360 ha-m/yr minus (80 ha-m/yr plus 210 ha-m/yr) or 70 ha-m/yr (570 acre-ft/yr).

Loss of Habitat

The waste-storage site preempts approximately 670 ha (1700 acres) of the Ohio River alluvial valley floor (see the Section on Handling of Combustion Wastes for this type of plant). The square storage pond is more space-effective than an elongated pond. However, a landfill would probably require less land area than the pond system.

Effluent Discharge from the Storage Site

Surface discharge from the pond storage site is approximately equal to the quantity of precipitation falling on the site [100 cm/yr on 510 ha (1260 acres) or 540 ha-m/yr (4360 acre-ft/yr)] plus the quantity of water discharged from the scrubbing system (360 ha-m/yr or 2880 acre-ft/yr) minus leachate seepage (80 ha-m/yr or 650 acre-ft/yr) and water retained within the combustion wastes, 35% water by weight (60 ha-m/yr or 490 acre-ft/yr); thus, the surface discharge is about 750 ha-m/yr (6100 acre-ft/yr). Evaporation from the storage site has not been quantified but would reduce the effluent discharge.

Runoff Dispersal of Coal Combustion Wastes

Runoff dispersal of coal combustion wastes should not occur from the above-grade diked storage pond. The length-slope factor in the Universal Soil Loss Equation is zero for the storage area; therefore, erosion loss per unit area per unit time is zero. Potential for soil loss from storage pond dikes is minimized to preserve dike integrity.

Leachate Seepage Discharge from the Storage Site

Because a permanent head of water will not develop in the pond, leachate discharge from the storage pond is estimated from Figure 16. Estimated seepage discharge from the pond is limited by the clay liner which has a hydraulic conductivity of 5×10^{-7} cm/s. Therefore, the discharge rate is approximately 4.2 m^3/ha (450 gal/day/acre) through an average of 510 ha (1260 acres), or 80 ha-m/yr (650 acre-ft/yr) seepage from the storage site.

Fugitive Dusting

Fugitive dusting from storage pond dikes is minimized to preserve dike integrity. However, increased dusting is likely to occur from the pond as the basin surface dries. There are too many variables present to predict the intensity of fugitive dusting from the storage pond.

Reclamation of the Storage Site

Following the placement of a 0.6-m soil mantle over the dewatered combustion wastes, it seems probable that vegetation can be quickly established on the storage area. Because of the quality of local soils, a minimum of amendment will be required for the successful development of a grassland or mixed grassland/shrub community. Due to the shallow available rooting depth, tree establishment would be difficult.

It is currently unknown what effects, if any, the underlying wastes would have on plant growth. If roots do penetrate into the wastes, the potential exists for toxicity problems in both plants and their consumers. The determination of whether trace elements are concentrated to levels hazardous to grazing animals will also help determine the acceptable future land use of the reclaimed storage area. If no toxicity problems are encountered, this area could be used as pasture or for hay production.

Consequences to Biota

Consumptive use of water. Consumptive use of water (70 ha-m/yr or 570 acre-ft/yr) at the storage site puts little pressure on water resources of the Ohio River valley. Fish and wildlife should not be impacted adversely.

Loss of habitat. Preemption of the land will result in an incremental loss of habitat associated with land managed for row crops and with land reverting to woodland (670 ha or 1700 acres). Successful reclamation of coal waste-storage areas to ambient habitat conditions has not been demonstrated. Therefore, there will be a potential for decrease in those species (e.g., bobwhite, ring-necked pheasant, eastern cottontail rabbit, woodcock, white-tailed deer, and raccoon) utilizing these habitats. Bald eagles and Virginia big-eared bats should not be directly impacted by habitat losses. Regional populations of all these wildlife are sufficiently large that there should be no threat to survival of these species in the region.

Effluent discharges. Surface discharges from the storage pond are 750 ha-m/yr (6100 acre-ft/yr) or 2.4×10^{-1} m^3/s (8.4 cfs) and are assumed to have the constituent concentrations presented in Table 47. At a flow rate of about 24 m^3/s (840 cfs), the Ohio River should provide sufficient flow to dilute total constituent concentrations to EPC in the water (Tables 30 and 47). Due to rapid dilution of the discharge, there should be little potential for biological concentration and magnification of potentially toxic constituents to potentially toxic levels based on the concentration factors presented in Table 29. There should, therefore, be little impact to area biota, including the bald eagle and Virginia big-eared bat, due to effluent discharges. However, the discharge may exacerbate existing pollution in the Ohio River system.

Runoff dispersal. Runoff dispersal from the pond storage site should result in minor movement of potentially toxic combustion wastes and should have little impact on fish and wildlife.

Leachate seepage. Leachate seepage is assumed to have the constituent concentrations outlined in Table 47. Leachate seepage is expected to move at a rate of 5×10^{-7} cm/s through the clay liner and at a rate of 1×10^{-4} cm/s (moderate hydraulic conductivity) through the substrate beneath the liner, or 200 times faster than through the clay liner. Assuming that the substrate is 33% water by volume, leachate movement away from the site should be sufficient to dilute total constituent concentrations to EPC in the soil (Tables 30 and 47). The soil concentration values presented in Table 47 indicate that there should be little biological concentration and magnification of potentially toxic constituents to toxic levels based on concentration factors presented in Table 29. Therefore, little impact to biota is expected due to leachate seepage. However, there is potential for gradual buildup of constituents in the soil (e.g., molybdenum and selenium) through physical and chemical processes to concentrations that could exceed EPC values. Leachate seepage will also provide long-term incremental additions to the soil. The soil reaction pH of 6.2-6.7 (Table 46) could cause some potentially toxic leachate constituents to become less mobile in the substrate. Additionally, there could be

Table 47. Factors by Which Ambient Concentrations Exceed Estimated Permissible Ambient Concentrations at the Ohio River Valley Model Power Plant

Element	Concentration in leachate[a] (μg/L)	Water[b]	Soil or Sediment[b]
Antimony	22	1	0
Arsenic	72	7	0
Beryllium	1	0	0
Boron	1100	0	0
Cadmium	1	2	1
Chromium	1000	20	2
Copper	13	1	0
Lead	4.3	0	0
Manganese	2	0	0
Mercury	0.3	6	1
Molybdenum	690	0	8
Nickel	50	0	3
Selenium	470	94	10
Vanadium	200	3	0
Zinc	5	0	0

[a] From Holland et al. (1975).
[b] From Equations 30 and 31.

high background concentrations of potentially toxic constituents and with the addition of leachate seepage constituents, critical levels required for protection of fish and wildlife could be exceeded. It is expected from USEPA estimates that by the time seepage reaches the Ohio River surface waters, 300 m away, sufficient dilution and attenuation should have occurred to reduce total constituent concentrations below EPC requirements.

Wind dispersal. Wind dispersal from the pond storage site should result in minor movement of potentially toxic combustion wastes as fugitive dust and should have little impact on fish and wildlife.

TEXAS PLANT BURNING LIGNITE

Plant Operations

The Texas plant burns lignite, uses pulverized-coal furnaces, has a heat rate of 8980 Btu/kWh, and operates with a plant factor of 0.70. The plant uses limestone scrubbers with a 90% SO_2 removal efficiency to comply with the federal SO_2 emission rate of 74% removal for raw coal with 1.04 lb SO_2 per million Btu (Figure 5). The plant uses "dry-bottom" boilers and has an ash split factor of 15% aggregate and 85% fly ash. Electrostatic precipitators with a fly ash collection efficiency of 99.5% have been installed to comply with the federal particulates emission rate of 0.03 lb particulates per million Btu of coal. The limestone scrubber removes an additional 0.2% of the fly ash to achieve the federal emission rate. The sulfate to sulfite ratio in the scrubber sludge solids is 0.8 to 0.2. The scrubber sludge is 15% solids by weight.

Handling of Combustion Wastes

Scrubber sludge is mechanically thickened to 50% solids by weight. Thickened sludge is mixed with ash residuals and landfilled. Following the removal of 0.6 m of topsoil from the storage site, the combined ash and sludge wastes will be deposited to a thickness of 4.6 m. The sides of the square, heaped landfill have a slope of 5:1, and at the end of plant operations, the storage site will occupy approximately 730 ha (1800 acres). The landfill site will not be lined. As the site is filled, it will be capped with a clay liner, covered with stored topsoil, and revegetated.

Description of the Storage-Site Area

The waste-storage site is located in Texas near Sam Houston National Forest. The storage site soil type is Tuckerman loam-heavy substratum, an Alfisol, with less than 0.3% slope. A soil description of Tuckerman loam is given in Table 48.

Table 48. Characteristics of Tuckerman Loam

Particle-size[a] distribution, 0 - 198 cm:	
Passing No. 10 sieve (2.0 mm)	100%
Passing No. 200 sieve (0.074 mm)	55 - 95%
Reaction pH	4.5 - 7.8
Erosion hazard	Slight
Available water capacity	Moderately high (0.10 - 0.20 cm/cm soil)
Depth of water table surface (2-6 mo/yr)	0 - 38 cm
Hydraulic conductivity:	Slow to very slow
0 - 38 cm from surface	1.4×10^{-4} to 4.4×10^{-4} cm/s
38 - 198 cm from surface	4.2×10^{-5} to 1.4×10^{-4} cm/s

[a] Silt-sized particles range from 0.05 to 0.002 mm.

The area is currently managed for loblolly pine (*Pinus taeda*) and slash pine (*Pinus elliottii*) timber and woodland grazing. The most important forage plants are sedges, which make up 80% of the herbaceous understory. Although the area is managed in part as woodland, equipment limitations are severe, plant competition is severe, and seedling mortality is severe. White-tailed deer (*Dama virginiana*), gray squirrel (*Sciurus carolinensis*), fox squirrel (*Sciurus niger*), eastern cottontail rabbit (*Sylvilagus floridanus*), and furbearers are abundant. Mourning dove (*Zenaidura macroura*) and bobwhite (*Colinus virginianus*) are abundant in woodland openings. Red wolves (*Canis rufus*), an endangered species, have been reported in the area.

Material Requirements for the Model Plant

The coal requirement for the model plant is 6.8×10^6 MT/yr (7.5×10^6 tons/yr) (Equation 12) based on plant operation specifications and coal data presented in Table 43. Assuming that 1.2 calcium atoms are supplied to the limestone scrubber for every sulfur atom removed and that the scrubber SO_2 removal efficiency is 74%, 1.3×10^5 MT (1.4×10^5 tons) of limestone are required.

Coal Combustion Wastes

Estimates of coal combustion wastes are based on plant operation specifications, coal data presented in Table 43, and quantification procedures presented earlier. Assuming a specific gravity of 2.5 for scrubber sludge solids, the scrubber sludge generation rate is 120 ha-m/yr (1010 acre-ft/yr) (Equations 20-26). Mechanically thickened sludge (50% solids by weight) is produced at a rate of 28 ha-m/yr (230 acre-ft/yr). Mechanically thickened sludge combined with ash residuals (49 ha-m/yr or 400 acre-ft/yr) (Equations 14-18) for landfill storage is produced at a rate of 78 ha-m/yr (630 acre-ft/yr).

Comsumptive Use of Water

The limestone scrubbing process generates a slurry of 85% water by weight and requires 120 ha-m/yr (950 acre-ft/yr) of water. Water from mechanically thickened sludge (98 ha-m/yr or 790 acre-ft/yr) is not recycled to the limestone scrubbing system but is discharged to a nearby stream. The average net consumptive water requirement for the scrubbing system is 20 ha-m/yr (160 acre-ft/yr).

Loss of Habitat

The waste-storage site preempts approximately 730 ha (1800 acres) near Sam Houston National Forest after 40 years of plant operations (see the section on Handling of Combustion Wastes for this type of plant). The proposed square landfill is more space-effective than an elongated landfill.

Effluent Discharge from the Storage Site

Water from the thickening process [98 ha-m/yr (790 acre-ft/yr) or 3×10^{-2} m^3/s (1 cfs)] is discharged into a nearby stream. Discharges from the landfill are discussed under the following section on Runoff Dispersal.

Runoff Dispersal of Coal Combustion Wastes

Without proper management, runoff dispersal of the wastes could occur from the heaped-landfill. The rainfall and runoff factor (R) of the Universal Soil Loss Equation is high (400). The soil erodibility factor (K) is approximately 0.15 for ash and 0.68 for Tuckerman loam--assuming that the soil has blocky structure, is very slowly permeable, and is composed of 0% organic matter, 80% fine sand and silt, and 5% sand. (See the discussion of Runoff in the chapter on Potential Impacts from Coal Ash and Flue-Gas-Desulfurization Wastes.) The support practice factor (P) can be assumed to be one. Runoff dispersal, however, cannot be quantified without values for the slope-length factor (L), the slope-steepness factor (S), and the cover and management factor (C), but can be minimized by minimizing LS and C to the extent practicable. However, an estimate of surface runoff can be derived by assuming an infiltration rate of 20% for the landfill site. Thus, the quantity of surface discharge would be 80% of the average annual rainfall of 114 cm. The surface area of the landfill site will range from 0 to 800 ha (0 to 2000 acres), depending on site age. Thus, the surface discharge could range from 0 to 730 ha-m/yr (0 to 5900 acre-ft/yr) or up to 2×10^{-1} m^3/s (8 cfs).

Leachate Seepage Discharge from the Storage Site

Leachate discharged from the landfill storage site is estimated from Figure 16. Estimated seepage discharge from the landfill is limited by the hydraulic conductivity of the ash-sludge mixture, which is assumed to be 1×10^{-6} cm/s (Figure 16). Assuming an average annual rainfall of 114 cm with 20% infiltration, the discharge rate is approximately 6.3 m^3/day/ha (680 gal/day/acre) through 0 to 730 ha (0 to 1800 acres), depending on site age.

Fugitive Dusting

Fugitive dusting from the landfill surface is likely to occur. There are, however, too many variables present to predict the intensity of fugitive dusting from the landfill site.

Reclamation of the Storage Site

Reclamation of the storage site will continue throughout the active life of the facility. Following placement of the clay cap, topsoil that was removed from a section of the site being prepared for waste deposition will be spread over the cap to a thickness of 0.6 m and revegetated. Revegetation seed mixtures employed to develop a grassland on the storage area should include locally adapted species, possibly including those occurring in the understory of the surrounding woodland. However, due to the shallow rooting depth of the soil mantle, tree establishment will probably not be possible.

Proper grading of the storage site to provide for adequate water drainage will be extremely important for successful reclamation. The presence of the rather impermeable clay cap will result in water movement along the interface between cap and topsoil. Without proper terracing and drainage channels, this subsurface water movement could result in sloughing, piping failures, and increased erosion of the soil cover on the sloped sides of the landfill (Schubert and Prodan 1979).

Consequences to Biota

Consumptive use of water. Consumptive use of water (20 ha-m/yr or 160 acre-ft/yr) during storage operations puts little pressure on water resources of the area, and fish and wildlife should not be impacted adversely.

Loss of habitat. Preemption of the land will result in an incremental loss of habitat associated with land managed for loblolly pine and slash pine timber and woodland grazing [0 to 730 ha (0 to 1800 acres)]. Successful reclamation of coal waste-storage areas to ambient habitat conditions has not been demonstrated. Therefore, there will be a potential for decrease in those species (e.g., white-tailed deer, eastern cottontail rabbit, mourning dove, and bobwhite) utilizing these habitats. There will be an incremental reduction in red wolf habitat (open woodlands), which could be detrimental to local populations. Even after reclamation, the site will not be as valuable for timber and woodland grazing or for wildlife habitat. The habitat loss, and therefore wildlife population changes, will be small compared to existing habitat and wildlife resources, but there will be incremental losses.

Effluent discharges. Water is discharged from the thickening process at a rate of 3×10^{-2} m^3/s (1 cfs) and is assumed to have the constituent concentrations presented in Table 49. Because the flow rate is small, the stream receiving the discharge must have a flow rate of only about 0.3 m^3/s (10 cfs) to sufficiently dilute total constituent concentrations to EPC in the water (Tables 30 and 49). With rapid dilution of the discharge, there should be little biological concentration and magnification of potentially toxic constituents to toxic levels based on the concentration factors presented in Table 29. Therefore, little impact to biota is expected due to normal effluent discharges. Large accidental discharges could lead to problems of toxicity from nickel, mercury, and selenium in particular. Thus, the site must be properly managed to ensure the safety of aquatic biota.

Table 49. Factors by Which Ambient Concentrations Exceed Estimated Permissible Ambient Concentrations at the Texas Model Power Plant

Element	Concentration in leachate[a] (μg/L)	Water[b]	Soil or Sediment[b]
Antimony	18	0	0
Arsenic	84	8	1
Beryllium	0.6	0	0
Boron	16,900	3	0
Cadmium	2.5	6	1
Chromium	210	4	0
Copper	31	3	0
Lead	2.7	0	0
Manganese	2	0	0
Mercury	0.5	10	1
Molybdenum	52	0	1
Nickel	15	8	1
Selenium	0.5	10	0
Vanadium	100	1	0
Zinc	25	1	0

[a] From Holland et al. (1975).
[b] From Equations 30 and 31.

Runoff dispersal. Runoff dispersal from the landfill storage site should result in minor movement of potentially toxic combustion wastes and should have little impact on fish and wildlife.

Leachate seepage. Leachate seepage is assumed to have the constituent concentrations outlined in Table 49. Leachate seepage is expected to move at a rate of 1×10^{-6} cm/s through the waste material and through the underlying substrate at a rate of 2×10^{-4} cm/s or 200 times faster than through the waste material. Assuming that the substrate is 33% water by volume, leachate movement away from the site should be sufficient to dilute total constituent concentrations to EPC in the soil (Tables 30 and 49). There should be little biological concentration and magnification of potentially toxic constituents to toxic levels based on concentration factors presented in Table 29. Therefore, little impact to biota is expected due to leachate seepage. The potential for gradual buildup of constituents in soil is not as great at this site as at the others because comparatively dilute leachate is expected. The soil reaction pH of 4.5 to 7.8 (Table 48) could cause some potentially toxic leachate constituents to become more or less mobile in the substrate. Additionally, there could be high background concentrations of potentially toxic constituents and, with the addition of leachate seepage constituents, critical levels required for protection of fish and wildlife could be exceeded.

Wind dispersal. Wind dispersal from the landfill storage site should result in minor movement of potentially toxic combustion wastes and should have little impact on fish and wildlife.

SOUTHEASTERN COASTAL PLANT BURNING HIGH-SULFUR COAL

Plant Operations

The southeastern coastal plant burns high-sulfur coal, uses pulverized-coal furnaces, has a heat rate of 8980 Btu/kWh, and operates with a plant factor of 0.70. The plant uses limestone scrubbers with a 90% SO_2 removal efficiency to comply with the federal SO_2 emission rate of 75% removal for raw coal with 1.22 lb SO_2 per million Btu (Figure 5). The plant uses "dry-bottom" boilers and has an ash split factor of 15% aggregate and 85% fly ash. Electrostatic precipitators with a fly ash collection efficiency of 99.5% have been installed to comply with the federal particulate emission rate of 0.03 lb particulates per million Btu of coal. The limestone scrubber removes an additional 0.1% of the fly ash to achieve the federal emission rate. The ratio of sulfate to sulfite in the scrubber sludge solids is 1.0 to 0. The scrubber sludge is 15% solids by weight.

Handling of Combustion Wastes

Scrubber sludge and ash residuals are stored in an above-grade, diked storage pond with an underdrain system. Underdrainage is recycled to the scrubber system. Excess supernatant is removed through a controlled effluent discharge after adequate settling of suspended solids has occurred. Restraining dikes are 7.6 m high and 65.5 wide at the base, with the outer slope of the dike at a 5:1 grade, the inner slope at 3:1. Combustion wastes will be deposited to a depth of 6.0 m. The storage site is surrounded by a 30.5 m buffer zone, and occupies 730 ha (1800 acres). The pond is lined with clay, having a permeability of 1×10^{-7} cm/s, below the underdrain system. When the storage pond is filled and stabilized to 65% solids, the storage area will be graded, covered, and revegetated.

Description of the Storage-Site Area

The waste-storage site is located on the North Carolina coastal plain. The diked storage pond is 300 m or more from the nearest stream and meets all applicable state and federal specifications.

The soil type of the storage pond area is a sandy loam, an Aquic Hapludult, which is nearly level. A soil description of sandy loam is given in Table 50.

Table 50. Characteristics of Sandy Loam

Particle-size distribution, 0 - 165 cm:	
Silt (0.05 - 0.002 mm)	25 - 55%
Clay (< 0.002 mm)	20 - 45%
Cation exchange capacity	5 - 10 meq/100 g soil
Hydraulic conductivity	Moderate (1.4×10^{-4} to 4.2×10^{-4} cm/s)
Available water capacity	High (0.12 - 0.2 cm/cm soil)
Depth of seasonally high water table	1.5 m
Depth of bedrock	15 m

Approximately 50% of the area is currently used for row crops (corn and soybeans) and the other 50% was formerly cultivated but has been abandoned and is in the early stages of old-field succession. Natural vegetation types for the area are oak-pine (*Quercus-Pinus*) and tupelo-sweet gum-bald cypress (*Nyssa* sp.-*Liquidambar styraciflua-Taxodium distichum*). Fauna of the area include eastern cottontail rabbit (*Sylvilagus floridanus*), marsh rabbit (*Sylvilagus palustris*), white-tailed deer (*Dama virginiana*), opossum (*Didelphis marsupialis*), raccoon (*Procyon lotor*), and gray fox (*Urocyon cinereoargenteus*). Bald eagles (*Haliaeetus leucocephalus*), a threatened species, and American alligators (*Alligator mississippiensis*), an endangered species, have been reported near or in a nearby small stream and large estuary.

Material Requirements for the Model Plant

The coal requirement for the model plant is 4.0×10^6 MT/yr (4.4×10^6 tons/yr) (Equation 12) based on plant operation specifications and coal data presented in Table 43. Assuming that 1.2 calcium atoms are supplied to the limestone scrubber for every sulfur atom removed and that the scrubber SO_2 removal efficiency is 75%, 1.6×10^5 MT (1.8×10^5 tons) of limestone are required.

Coal Combustion Wastes

Ash residuals will be generated at a rate of 36 ha-m/yr (290 acre-ft/yr) based on plant operation specifications, coal data presented in Table 43, and quantification procedures presented earlier (Equations 14-18). Assuming a specific gravity of 2.5 for scrubber sludge solids, the scrubber sludge generation rate is 170 ha-m/yr (1350 acre-ft/yr) (Equations 20-26). The combined stabilized waste production rate (65% solids by weight) in diked ponds is 90 ha-m/yr (730 acre-ft/yr).

Consumptive Use of Water

The limestone scrubbing process generates a slurry of 85% water by weight and requires 160 ha-m (1260 acre-ft) of water per year. Water from the storage pond underdrain system is recycled to the limestone scrubbing system. Assuming that seepage discharge from the pond is limited by the hydraulic conductivity of the ash-sludge mixture, which is assumed to be 1×10^{-6} cm/s (Figure 13) through approximately 570 surface hectares (1400 acres), the equivalent of all leachate seepage from the initial 85% water by weight and the final 35% water by weight is recycled (150 ha-m or 1200 acre-ft of water per year) to the limestone scrubbing system. The coal combustion wastes retain 43 ha-m (350 acre-ft) of water per year.

Loss of Habitat

The waste-storage site preempts approximately 730 ha (1800 acres) of coastal habitat (see section on Handling of Combustion Wastes for this type of plant). The square storage pond is more space-effective than an elongated pond. However, a landfill would probably require less land area than the pond system.

Effluent Discharge from the Storage Site

Surface discharge from the pond storage site is approximately equal to the 114 cm/yr of precipitation falling on the site (680 ha-m/yr or 5550 acre-ft/yr) plus the quantity of water discharged from the scrubbing system (160 ha-m/yr or 1260 acre-ft/yr) minus both seepage discharge to the underdrain system (170 ha-m/yr or 1410 acre-ft/yr) and water retained within the combustion wastes (43 ha-m/yr or 350 acre-ft/yr); thus, surface discharge is about 620 ha-m/yr (5050 acre-ft/yr).

Runoff Dispersal of Coal Combustion Wastes

Runoff dispersal of the wastes should not occur from the above-grade diked storage pond. The length-slope factor in the Universal Soil Loss Equation is zero for the storage area; therefore, erosion loss per unit area per unit time is zero. Potential for soil loss from storage pond dikes is minimized to preserve dike integrity.

Leachate Seepage Discharge from the Storage Site

It is assumed that the leachate seepage is collected by the underdrain system and recycled to the scrubber system. Quantification is discussed above in the section on Effluent Discharge from the Storage Site.

Fugitive Dusting

Fugitive dusting from storage pond dikes is minimal. However, increased dusting is likely to occur from the pond as the basin surface dries. There are too many variables involved to predict the intensity of fugitive dusting from the storage pond.

Reclamation of the Storage Site

Revegetation of the 0.6 m of soil placed over the dewatered wastes to a grassland plant community is feasible using presently available reclamation techniques. The problems encountered in the revegetation of this site will be basically similar to those of the Ohio model plant. Primary among these is the effect of the underlying wastes on plant growth. Although the plant community developed on the reclaimed ponds will be more susceptable to drought than the surrounding vegetation, available moisture should not be a limiting factor.

Consequences to Biota

Consumptive use of water. Consumptive use of water (43 ha-m/yr or 350 acre-ft/yr) at the storage site puts little pressure on water resources of the North Carolina coastal ecosystem. Fish and wildlife should not be impacted adversely.

Loss of habitat. Preemption of the land will result in an incremental loss of habitat associated with land managed for row crops and with land in the early stages of old-field succession [730 ha (1800 acres)]. Successful reclamation of coal waste-storage areas to ambient habitat conditions has not been demonstrated. Therefore, there will be a potential for decrease in those species (e.g., eastern cottontail rabbit, marsh rabbit, white-tailed deer, opossum, raccoon, gray fox, and red fox) utilizing these habitats. The habitat loss, and therefore population changes, will be small compared to regional habitat and wildlife resources, but there will be localized losses.

Effluent discharges. Surface discharges from the storage pond are 620 ha-m/yr (5050 acre-ft/yr) or 1.9×10^{-1} m^3/s (7 cfs) and are assumed to have the constituent concentrations presented in Table 51. The discharge enters a small stream (average annual flow 3×10^{-1} m^3/s or 10 cfs) which flows into a large estuary. The stream does not provide sufficient flow to dilute total constituent concentrations to EPCs in the water (Tables 30 and 51). However, the estuary provides sufficient volume and flow to dilute total constituent concentrations to an acceptable EPC in water. There will be a potential for biological concentration and magnification of potentially toxic constituents to toxic levels in the stream (500 m in length) before it enters the estuary based on the bioconcentration factors presented in Table 29. Although the discharge will be rapidly diluted in the estuary, biological concentration occurring in the

Table 51. Factors by Which Ambient Concentrations Exceed Estimated Permissible Ambient Concentrations at the Southeastern Coastal Model Power Plant

Element	Concentration in leachate[a] (μg/L)	Water[b]	Soil or Sediment[b]
Antimony	8.7	0	0
Arsenic	6	1	0
Beryllium	0.3	0	0
Boron	48	0	0
Cadmium	1.1	3	1
Chromium	14	0	0
Copper	15	2	0
Lead	6.3	1	0
Manganese	2	0	0
Mercury	0.3	6	1
Molybdenum	10	0	0
Nickel	46	23	2
Selenium	0.5	0	0
Vanadium	100	1	0
Zinc	17.5	1	0

[a] From Holland et al. (1975).
[b] From Equations 30 and 31.

stream could impact estuarine organisms. There could also be high background concentrations of potentially toxic constituents (particularly nickel) and, with addition of the discharge, critical levels required for protection of fish and wildlife could be exceeded. The discharge will be a long-term addition to the stream and estuarine system, and there could be biomagnification of potentially toxic trace elements in food webs leading to the bald eagle and American alligator.

Runoff dispersal. Runoff dispersal from the storage pond should result in minor movement of potentially toxic combustion wastes and should have little impact on fish and wildlife.

Leachate seepage. Leachate seepage is assumed to have the constituent concentrations outlined in Table 51. However, leachate seepage is collected by the underdrain system and recycled to the scrubber system; therefore, there should be little impact to area biota due to leachate seepage. If some of the leachate were to seep into the surrounding soil, the concentrations of waste constituents would not be expected to greatly exceed EPC values (Table 51).

Wind dispersal. Wind dispersal from the pond storage site should result in minor movement of potentially toxic combustion wastes and should have little impact on fish and wildlife.

FUTURE RESEARCH NEEDS

The technologies of flue-gas cleaning are currently in a state of evolution. Concern about the disposition of wastes from flue-gas cleaning has only recently begun to be reflected in the research community. As a consequence, the data bases for this report are highly dynamic and incomplete. There is a major absence of data linking the dispersal of potentially toxic substances from coal combustion and emission-abatement wastes into the environment and the magnitude of impacts to fish and wildlife resources.

In this section, a brief listing is presented of some of the areas that need to be addressed in future research. The list is by no means definitive and some of the needs may be currently addressed in ongoing research. The U.S. Department of Energy (1979) listed over 100 ongoing research projects in FY 1978 which addressed problems in flue-gas cleaning. These range from studies of cleaning technologies and handling options to bioenvironmental hazards from storage of cleaning wastes. The reader must maintain an awareness of new information as it appears in order to improve upon the ability to predict consequences to fish and wildlife of site-specific plans.

NATURE OF COAL ASH AND FLUE-GAS-DESULFURIZATION WASTES

1. Mechanisms of trace-element enrichment in coal ashes, in order to clarify conditions under which elements are more likely to be concentrated in wastes.

2. Factors influencing the solubility and mobility of trace elements from fly ash, particularly with respect to elemental speciation. Some of the factors to be studied should include ash age, water-ash contact time, and oxidation state within the wastes.

3. The suspected correlation between ash particle size and resistance to leaching.

4. Distribution of a particular element on the ash particles, i.e., on the outer surface, fused in the center, or evenly distributed throughout.

HANDLING COMBUSTION AND EMISSION-ABATEMENT WASTES

1. Further development of landfill and impoundment operation to minimize production of fugitive dust and seepage of leachate.

2. Comparative studies of lining materials to determine their compatibility with coal ash and FGD sludge.

3. Development of high-volume alternative uses of the wastes.

4. FGD-sludge stabilization processes.

5. The environmental safety of ocean- and mine-disposal alternatives.

6. Further development and validation of models of seepage movement.

POTENTIAL EFFECTS ON FISH AND WILDLIFE

1. Cumulative effects of seepage of trace elements from coal ash and FGD-sludge storage sites upon surrounding fish and wildlife resources.

2. Effects of seepage from waste impoundments upon the hydrologic and biologic dynamics of wetlands systems.

3. Effects of long-term exposure of fish, wildlife, and their habitats to low levels of potentially toxic trace elements.

4. Effects of wildlife use of waste impoundments and development of guidelines for discouraging such use.

RECLAMATION OF COAL COMBUSTION AND EMISSION-ABATEMENT WASTE-STORAGE SITES

1. Suitability of coal ashes and FGD sludges as media for plant growth.

2. Level of management required to establish and maintain vegetative cover on abandoned waste-storage sites.

3. Mobility of waste constituents within reclaimed ecosystems.

4. Suitability of reclaimed waste-storage sites for supporting wildlife populations and as watersheds.

5. Development of site manipulation and preparation guidelines for optimal revegetation of waste-storage sites.

REFERENCE

U.S. Department of Energy. 1979. Inventory of Federal Energy Related Environmental and Safety Research for FY 1978. Vol. II-Project Listings and Indexes. DOE/EV-0057.

ACKNOWLEDGMENTS

The authors are grateful for the guidance and financial support of the National Power Plant Team, U.S. Fish and Wildlife Service. Mr. Michael Avery and Dr. James Bennett served as Project Officers for the Power Plant Team and provided many useful comments and ideas. We appreciate the constructive comments from personnel of the Fish and Wildlife Service regional offices, the U.S. Soil Conservation Service, and Oak Ridge National Laboratory. Dr. W. Buchanan of the National Fertilizer Development Center, Tennesee Valley Authority, kindly allowed us to use some of his unpublished data.

Personnel from the Division of Environmental Impact Studies who provided us with many helpful comments and ideas were Mr. William Hallett, Dr. Anthony J. Dvorak, Dr. Berge Gureghian, Dr. Robert Goldstein, Ms. Julie Jastrow, Mr. Roger Rodiek, and Dr. Elizabeth Stull. Our special thanks are due Ms. Alice Packard for providing the information retrieval support required for our review.

Thank you all for your support.

APPENDIX A. ENGLISH/METRIC EQUIVALENTS

Multiply	By	To obtain
Acres	0.4047	Hectares (ha)
Acre-feet	1.2335×10^3	Cubic meters (m^3)
British thermal units [(Btu) thermochemical]	1.0544×10^3	Joules (J)
British thermal units/pound (Btu/lb)	2.324×10^3	Joules/kilogram (J/kg)
Cubic feet (ft^3)	0.0283	Cubic meters (m^3)
Degrees Fahrenheit (°F) - 32	5/9	Degrees Celsius (°C)
Feet (ft)	0.3048	Meters (m)
Gallons (gal)	3.7854	Liters (L)
Gallons (gal)	0.0038	Cubic meters (m^3)
Gallons/minute (gal/min)	0.0631	Liters/second (L/s)
Gallons/minute (gal/min)	6.309×10^{-5}	Cubic meters/second (m^3/s)
Inches (in.)	2.540	Centimeters (cm)
Kilowatt-hours (kWh)	3.60×10^6	Joules (J)
Miles (mi)	1.6093	Kilometers (km)
Pounds (lb)	0.4536	Kilograms (kg)
Square feet (ft^2)	0.0929	Square meters (m^2)
Square miles (mi^2)	2.590	Square kilometers (km^2)
Tons, short (t)	9.0718×10^2	Kilograms (kg)
Tons, short (t)	0.9072	Tons, metric (MT)

APPENDIX B. GLOSSARY

The technical terms selected for the Glossary are mainly terms that may not ordinarily be familiar to biologists. The definitions provided are those applicable to the subject matter of this report.

ACID MINE DRAINAGE - Acidic seepage from mines in which the spoil is high in pyrite (FeS); when oxidized in the presence of water, pyrite yields sulfuric acid.

AGGREGATE (BOILER) - That part of residual combustion solids that has fused into particles heavy enough to drop out of the furnace gas stream.

AQUIFER - A permeable unit of rock or sediment from which groundwater can be extracted. Confined aquifers are bounded on top and bottom by impermeable materials. Unconfined aquifers are bounded on top by a water table.

ASH (COAL) - The solid material remaining after coal is burned. Contains most of the mineral and inorganic material originally present in the coal.

AVAILABLE ELEMENTS (SOIL) - Chemical elements in a soil that are in a form capable of assimilation by plants. May comprise only a portion of the total amount of the element present in that soil.

BAG HOUSE - A series of filters to remove particles from the flue gases.

BERM - A bench of soil or rock built on an earthen structure. It may serve various purposes such as a dike, an encasement for a drainage system, a weight for structural stabilization of an embankment, or an erosion-control structure.

BOTTOM ASH - Dry ash from coal combustion that does not melt but is too heavy to be entrained in the flue gas. Also called cinders.

BUFFERING CAPACITY - A measure of the tendency of a soil or water to resist large changes in pH.

BULK DENSITY (SOIL) - The weight per unit volume of soil. Agricultural soils have bulk densities usually between 1.2 and 1.7 g/cm^3. A compacted clay may have a bulk density of 2 g/cm^3.

CATION EXCHANGE CAPACITY (CEC) - The relative adsorptive power of a soil for cations. Expressed as the number of milliequivalents of cations per 100 grams of dry soil.

CLARIFLOCCULATOR - A device for handling dilute suspensions to produce a relatively clear supernatant liquid (overflow) and an agglomeration of settleable or filterable solids that are withdrawn at the bottom of the device (underflow). It consists of a tank, a means for introducing the feed suspension, a drive-actuated rake mechanism for moving settled solids to a discharge point, a means for removing the thickened solids, and a means for removing the clarified liquor. Chemicals may be added to the feed to enhance the physical separation.

CLAY LINER (WASTE DISPOSAL) - A liner consisting of a compacted layer of a clay with a low hydraulic conductivity.

CONSUMPTIVE USE (WATER) - That portion of water taken into a power plant that is not directly returned to the surface water body. The water is lost through evaporation and seepage.

DEWATERING (SLURRY) - The process of removing water from a slurry. Processes include natural evaporation, centrifugation, decantation, and filtration.

ELECTROSTATIC PRECIPITATOR - A device used to remove particles from flue gases, by charging the particles electrically and collecting them on appropriate electrodes.

FIXATIVE (FOR FGD SLUDGE) - A chemical additive that is mixed with FGD sludge to give it more desirable properties for disposal. Commonly, a fixative is used to lessen the thixotropic characteristics of the sludge.

FLOODPLAIN - The portion of a river or stream valley that is periodically inundated during episodes of excessive runoff. The solid waste-disposal regulations (40 CFR, Part 257) use the term "floodplain" to refer to the 100-year floodplain. The 100-year floodplain is the area that is likely to be inundated once in one hundred years.

FLOW, AVERAGE ANNUAL - The average volume of water to pass a given cross section of a stream during a given year. Usually expressed in units such as cubic feet per second (cfs).

FLOW, 7-DAY/10-YEAR LOW FLOW - The lowest volume of flow statistically expected to pass through a given cross section of a stream during a 7-day timespan in any 10-year period.

FLUE-GAS DESULFURIZATION (FGD) - Any process used to remove sulfur (largely sulfur oxides) from flue gases.

FLUSHING TIME (IMPOUNDMENT) - The period of time required to completely replace the volume of water in an impoundment through natural processes.

FLY ASH - That portion of the coal ash carried up the flue.

FUGITIVE DUST - Particles of dust removed from a surface by the wind.

GROUNDWATER - The water contained within the pore spaces of rock or soil.

HEAT RATE - Efficiency of conversion of boiler heat energy to electrical energy--e.g., if X amount of boiler heat is needed to produce Y amount of electricity, heat rate is X Btu/Y kWh.

HEATING VALUE - Amount of heat released per weight of coal during combustion.

HIGH-SULFUR COAL - In general, coal that contains over 1% sulfur. In some instances, however, it is defined as coal containing over 3% sulfur.

HYDRAULIC CONDUCTIVITY - The rate at which water can flow through a permeable material.

HYDRAULIC GRADIENT - The change in hydraulic head over distance. Nearly horizontal flow has a very small gradient.

HYDRAULIC HEAD - The energy that allows water to flow. It consists of a pressure and a height component. Water flows from areas of higher to lower head.

IMPERMEABLE LINER (WASTE DISPOSAL) - Material placed on the bottom and sides of a waste impoundment to contain the waste material. No liner is completely impermeable, but many of the synthetic materials are relatively impermeable compared to natural earth liners.

INFILTRATION RATE (SOIL) - The rate at which water enters the surface layer of soil.

LEACHATE - Water and dissolved constituents draining out of a given column of saturated porous material such as soil.

LEACHING - The process of moving dissolved constituents (usually by water) downward through a column of porous material such as soil.

MINE-MOUTH - Operations such as coal washing and power generation carried out adjacent to the coal mine.

ORGANIC MATTER (SOIL) - The amount of plant and animal residues in a soil. Soils typically contain about 1 to 6% organic matter.

PERMEABILITY (SOIL) - The quality of a soil that enables it to transmit water or air. It is not equivalent to infiltration rate (see INFILTRATION RATE).

PERMEABILITY CLASSES (SOIL) -

	Hydraulic conductivities		
	(inches/hour)	(centimeters/second)	(meters/day)
Very slow	< 0.05	$< 3.5 \times 10^5$	< 0.006
Slow	0.05 - 0.20	$3.5 \times 10^5 - 14 \times 10^5$	0.006 - 0.023
Moderately slow	0.20 - 0.80	$14 \times 10^5 - 56 \times 10^5$	0.023 - 0.046
Moderate	0.80 - 2.50	$56 \times 10^5 - 176 \times 10^5$	0.046 - 0.289
Moderately rapid	2.50 - 5.00	$176 \times 10^5 - 352 \times 10^5$	0.289 - 0.578
Rapid	5.00 - 10.00	$352 \times 10^5 - 704 \times 10^5$	0.578 - 1.156
Very rapid	> 10.00	$> 704 \times 10^5$	> 1.156

PIPING - A progressive failure of a dike or embankment that occurs when a seepage velocity is great enough to cause internal erosion.

PLANT CAPACITY (RATED CAPACITY) - Nominal capacity for the power output by a electric generating unit, usually expressed in kilowatts or megawatts.

PLANT FACTOR - Ratio of electricity generated during a year to the electricity that could have been generated if the plant operated at nominal capacity for the entire year.

PLUME (WATER) - A stream of water that enters an existing body of water and is still distinguishable because of differences between the influent water and the receiving water in such factors as velocity, chemistry, or temperature. A plume dissipates with dilution and dispersion.

POINT SOURCE (WATER) - A single source of pollutant discharge to surface waters.

POZZOLANIC - Pertaining to a material that becomes cementlike after exposure to water.

RECLAMATION - Usually implies the restoration of disturbed land to primary production.

RUNOFF (RAINFALL) - All rainfall (and snowmelt) that does not soak into the ground, does not evaporate immediately, or is not used by vegetation. This flows down slopes and forms streams.

SCRUBBER SLUDGE (FGD) - Semisolid waste material, usually $CaSO_3$ and $CaSO_4$, resulting from the removal of sulfur oxides from flue gases using lime, limestone, or double-alkali techniques.

SEEPAGE - Any water or liquid effluent that flows through a porous medium. This term is often used to refer to the liquid lost through the bottom of a waste pond.

SLAG - That portion of the coal ash that melts to a viscous fluid at boiler operating temperatures, and cools to a glassy, angular material.

SLURRY - Any mixture of water and finely divided solids. Can refer to mixtures of coal and water (coal slurry), ash and water (ash slurry), desulfurization sludge and water (scrubber slurry), or coal refuse and water (refuse slurry).

SPLIT FACTOR - Percentage of ash that becomes entrained in flue gas as fly ash.

STEAM-ELECTRIC POWER PLANT - A power plant that generates electric power through steam-driven turbines. In commercial power plants, the fuel used to produce steam from water can be coal, oil, natural gas, or enriched uranium.

TEXTURE (SOIL) - The proportion of sand, silt, and clay in a soil. Soil texture is expressed in terms such as "sandy loam", "clay", "silty clay loam", etc.

THIXOTROPIC - Having the property of liquefying when disturbed and returning to the solid phase upon standing undisturbed.

THROW-AWAY SYSTEM (FGD) - A system in which the waste product from flue-gas desulfurization is not recycled or reclaimed, but instead is disposed of as waste.

TRACE ELEMENTS - Chemical elements that normally are present in minute (trace) quantities. Includes metals such as chromium, zinc, cadmium, and copper, and nonmetals such as selenium, boron, and arsenic.

UNDERFLOW (CLARIFIER) - The stream of coarse particles that are separated by a clarifier or cyclone (see also CLARIFLOCCULATOR).

UNSATURATED FLOW - Flow of a liquid through a porous medium in which some of the pore space is occupied by air. Unsaturated flow is usually slower than saturated flow under the same conditions.

VACUUM DISK FILTER - A continuous rotary vacuum filter made up of filter disks mounted at regular intervals around a hollow center shaft covered with a cloth filter. The device is used for dewatering sludge or solids by application of a vacuum inside the disks. A layer of caked solids (filter cake) is formed on the outer filter surface, and is subsequently removed.

WATER-HOLDING CAPACITY (SOIL) - The total amount of water capable of being held in a soil by capillary forces. Usually expressed as percent by weight of dry soil.

WATERSHED - An area, usually a valley or collection of valleys, surrounded by surface-water divides. All precipitation falling into a watershed supplies runoff to the same stream.

WATER TABLE - The surface that separates the groundwater in an unconfined aquifer (an aquifer not bounded on top by an impermeable layer) from the unsaturated zone above it (see AQUIFER).

APPENDIX C. SPECIES OF VEGETATION APPROPRIATE FOR REVEGETATING WASTE-STORAGE SITES

REVEGETATION OF FLY ASH SURFACES

The slow rate of natural colonization of fly ash-storage sites indicates that fly ash is generally a less than adequate substrate for plant growth. Included among the factors limiting plant growth on fly ash are (1) alkaline pH, (2) high soluble-salt content, (3) deficiency of nitrogen, and perhaps phosphorus, and (4) boron toxicity. Species chosen for use in the reclamation of fly ash-storage sites with a minimum of amendment should be adaptable to these conditions as well as other site-specific considerations (e.g., climate, precipitation, degree of slope). The tolerance of plants to the adverse conditions of fly ash has been shown to be correlated with their tolerance of boron or their boron requirements (Hodgson and Buckley 1975).

Limited research has been conducted to determine the tolerance of plants to fly ash conditions. Hodgson and Townsend (1973) classified grasses, legumes, and cereal, root, and vegetable crops on the basis of their tolerance to the conditions of lagooned fly ash containing 15-20 μg/g boron. The tolerance of a number of shrub and tree species to both unweathered and weathered fly ash was evaluated by measuring the growth of plants established in pots containing increasing concentrations of ash mixed with soil (Hodgson and Buckley 1975). More recently, Horton and McMinn (1977) stated that no elemental imbalances were found in seedlings of loblolly pine (*Pinus taeda*) and sweetgum (*Liquidambar styraciflua*) grown for two seasons on alkaline stoker-fed boiler ash. Both these species and sycamore (*Platanus occidentalis*) grew as well on ash as on soil (controls); however, ash-grown sycamore seedlings may have been manganese-deficient. Scanlon and Duggan (1979) evaluated the performance over three growing seasons of eight woody species planted on acidic fly ash and found that nitrogen-fixing species (*Elaeagnus multiflora ovata* and *E. umbellata*) were best adapted for use in fly ash stabilization.

Although slow, natural colonization of fly ash deposits has also been reported. Gonsoulin (1975) surveyed three alkaline fly ash pits in Tennessee and identified 35 colonizing species. Among the dominants were smooth brome (*Bromus inermis*), white sweet clover (*Melilotus alba*), cottonwood (*Populus deltoides*), and several species of willow (*Salix* spp.). Vigorous stands of yellow sweet clover (*Melilotus officinalis*) were reported growing on acidic fly ash produced by a cyclone boiler in New York (Furr et al. 1977).

The plant species listed in Table C.1 were selected as being able to grow on fly ash deposits on the basis of the above information and other data (e.g., species adapted to grow on fine-textured, alkaline limestone spoil). As the available information is quite limited, no recommendations of plants species for use in the revegetation of western ash deposits can be made. It is thought, however, that many of the listed species may be useful in the revegetation of eastern and midwestern sites. It should be noted that none of these short-term studies (i.e., no more than five growing seasons) can give any indication of the long-term survival of these plant species when grown on fly ash, or the ability of these species to form self-sustaining plant communities on ash deposits.

The importance of legumes or other nitrogen-fixing species to the plant communities developed on ash deposits is clear. Hodgson and Buckley (1975) showed that growth of sitka spruce (*Picea sitchensis*), sycamore maple (*Acer pseudoplatanus*), and poplar (*Populus robusta*) on fly ash was significantly better when grown with a ground cover of white clover (*Trifolium repens*) than when grown with no clover (control trees). As fly ash is virtually devoid of nitrogen, the amount of nitrogen supplied by nitrogen-fixing species to plant communities established in ash deposits and the efficiency of nitrogen cycling may well determine the long-term success of these developing communities.

REVEGETATION OF BURIED WASTES

The selection of plant species for use in the revegetation of buried coal combustion wastes is extremely difficult, because little effort has been made to identify species appropriate for this purpose. To date, no large-scale reclamation of these wastes has been attempted in the United States. This section can therefore only identify plant species that may be suitable for the revegetation of these wastes, based upon the performance of these species in the reclamation of other types of covered or buried anthropogenic waste. Specifically, those species used to successfully revegetate coal mining wastes and mineral tailings were considered. Table C.2 is a list of plant species adaptable to a wide range of soil pH, fertility, salinity, and other physical and environmental conditions.

Because the vegetation planted on burial sites for coal combustion wastes will not be growing directly on the waste material, species-selection criteria will be based primarily upon both the chemical and physical characteristics of the soil mantle placed over the wastes and the site-specific considerations of precipitation, topography, and climate. Roots of plants growing over buried combustion wastes will, however, be in contact with the wastes either at the interface between the soil mantle and the wastes or by root penetration into the waste material. Some tolerance to the acidic or alkaline nature of the waste material is therefore desirable. In many instances, subsoil will be used to form the mantle, requiring the use of plants adapted to harsh, low-fertility conditions. If topsoil is segregated during waste-site construction and then reapplied over the waste material, species adapted to very different soil conditions will be needed.

Table C.1. Plant Species Potentially Useful in the Revegetation of Fly Ash Surfaces

Common name	Scientific name	Species[a] origin	Region tested	Comments	Reference
Grasses					
Bahia grass	*Paspalum notatum*	I	Southern U.S.	Recommended for use in revegetation of limestone strip mining spoil in Florida; naturally revegetates alkaline spoil.	Craig and Smith (1979)
Bentgrass	*Agrostis* spp.	N/I	United Kingdom	Classified as semitolerant to growth on fly ash; *A. stolonifera* occurs naturally on weathered fly ash deposits.	Hodgson and Townsend (1973)
Bluegrass	*Poa* spp.	N/I	United Kingdom	Classified as semitolerant to tolerant of fly ash conditions; *P. pratensis* and *P. annua* occur naturally on weathered fly ash deposits.	Hodgson and Townsend (1973)
Bermuda grass	*Cynodon dactylon*	N	Southern U.S.	Recommended for use in revegetation of limestone strip-mining spoil in Florida; naturally revegetates alkaline spoil.	Craig and Smith (1979)
Italian ryegrass	*Lolium multiflorum*	I	United Kingdom	Classified as semitolerant to tolerant of fly ash conditions.	Hodgson and Townsend (1973)
Perennial ryegrass	*Lolium perenne*	I	United Kingdom	Considered to be among the most tolerant of the cultivated grasses to fly ash conditions.	Hodgson and Townsend (1973)
Red fescue	*Festuca rubra*	N	United Kingdom	Considered to be among the most tolerant of the cultivated grasses to fly ash conditions.	Hodgson and Townsend (1973)

(continued)

Table C.1. (Continued)

Common name	Scientific name	Species[a] origin	Region tested	Comments	Reference
Grasses (contd.)					
Rye	*Secale cereale*	I	United Kingdom	Classified as semitolerant to fly ash conditions; annual species may be useful as a cover crop during initial establishment of slower growing perennials on fly ash.	Hodgson and Townsend (1973)
Smooth brome	*Bromus inermis*	I	Southern U.S.	Occurs as dominant species in abandoned fly ash pits in Tennessee.	Gonsoulin (1975)
Tall fescue	*Festuca arundinacea*	I	Southern U.S.	Grown successfully on fly ash with pH 6.5-7.5.	Duggan and Scanlon (1974)
Timothy	*Phleum pratense*	I	United Kingdom	Good growth on weathered fly ash.	Townsend and Gillham (1975)
Forbs					
Hairy indigo	*Indigofera hirsuta*	N	Southern U.S.	Recommended for use during period of initial vegetative establishment on alkaline limestone strip mining spoil; naturally revegetates alkaline spoil.	Craig and Smith (1979)
Partridge-pea	*Cassia fasciculata*	N	Southern U.S.	Recommended for use during period of initial vegetative establishment on alkaline limestone strip mining spoil; naturally revegetates alkaline spoil.	Craig and Smith (1979)
Legumes					
Alfalfa	*Medicago sativa*	I	United Kingdom	Classified as semitolerant of fly ash conditions; extremely boron tolerant.	Hodgson and Townsend (1973)
Alsike clover	*Trifolium hybridum*	I	United Kingdom	Classified as semitolerant to tolerant of fly ash conditions.	Hodgson and Townsend (1973)

(continued)

Table C.1. (Continued)

Common name	Scientific name	Species[a] origin	Region tested	Comments	Reference
Legumes (contd.)					
Birdsfoot trefoil	*Lotus corniculatus*	I	United Kingdom	Occurs naturally on weathered fly ash deposits.	Hodgson and Townsend (1973)
Black medic	*Medicago lupulina*	I	United Kingdom	Occurs naturally on weathered fly ash deposits.	Hodgson and Townsend (1973)
Red clover	*Trifolium pratense*	I	United Kingdom	Classified as semitolerant to tolerant of fly ash conditions; occurs naturally on weathered fly ash deposits.	Hodgson and Townsend (1973)
Sainfoin	*Onobrychis sativa*	I	United Kingdom	Classified as semitolerant to tolerant of fly ash conditions.	Hodgson and Townsend (1973)
White clover	*Trifolium repens*	I	United Kingdom	Classified as semitolerant to tolerant of fly ash conditions; occurs naturally on weathered fly ash deposits.	Hodgson and Townsend (1973)
White sweet clover	*Melilotus alba*	I	United Kingdom, Southern U.S.	Classified as tolerant of fly ash conditions; dominant plant species on abandoned alkaline fly ash pits in Tennessee; grown successfully on fly ash with pH 6.5-7.5.	Hodgson and Townsend (1973); Gonsoulin (1975); Duggan and Scanlon (1974)
Yellow sweet clover	*Melilotus officinalis*	I	Eastern U.S.	Observed vigorously growing on acidic landfilled fly ash from a cyclone boiler in New York.	Furr et al. (1975)
Shrubs					
Autumn olive	*Elaeagnus umbellata*	I	Southern U.S.	Good early growth on acidic to neutral fly ash; no measured elemental imbalance in plant tissue.	Scanlon and Duggan (1979)

(continued)

Table C.1. (Continued)

Common name	Scientific name	Species[a] origin	Region tested	Comments	Reference
Shrubs (contd.)					
Bladder-senna	*Colutea arborescens*	I	United Kingdom	Nitrogen-fixing species; based upon greenhouse pot experiments, can be expected to establish on fly ash surfaces as a nurse crop with a minimum of amelioration.	Hodgson and Townsend (1973)
Eastern baccharis	*Baccharis halimifolia*	I	Southern U.S.	Found naturally revegetating an "island" of accumulated coal ash in settling pond; considered a pest species in some areas.	Skinner et al. (1978)
French tamarisk	*Tamarix gallica indica*	I	United Kingdom	Well adapted to saline conditions; very satisfactory establishment on fly ash in greenhouse pot experiments.	Hodgson and Townsend (1973)
Gorse	*Ulex europaeus*	I	United Kingdom	Good tolerance of weathered fly ash conditions in greenhouse pot experiments.	Hodgson and Buckley (1975)
Japanese barberry	*Berberis thunbergii*	I	United Kingdom	Classified as tolerant of fly ash conditions in greenhouse pot experiments; sensitive to high salinity of freshly collected fly ash.	Hodgson and Townsend (1973)
Many-flowered silverberry	*Elaeagnus multiflora ovata*	I	Southern U.S.	Good early growth on acidic to neutral fly ash; no measured elemental imbalance in plant tissue.	Scanlon and Duggan (1979)
Russian olive	*Elaeagnus angustifolia*	I	United Kingdom	Nitrogen-fixing species; based upon greenhouse pot experiments, can be expected to establish on fly ash surfaces as a nurse crop with a minimum of amelioration; adapted to saline conditions; reasonably boron tolerant.	Hodgson and Townsend (1973)

(continued)

Table C.1. (Continued)

Common name	Scientific name	Species[a] origin	Region tested	Comments	Reference
Shrubs (contd.)					
Smooth sumac	*Rhus glabra*	N	Southern U.S.	Naturally revegetates natural to alkaline limestone strip mine spoil in Florida.	Craig and Smith (1979)
Southern bayberry	*Myrica cerifera*	N	Southern U.S.	Found naturally revegetating an "island" of accumulated coal ash in a settling pond; naturally revegetates neutral to alkaline limestone strip mine spoil in Florida.	Skinner et al. (1978); Craig and Smith (1978)
Southernwood	*Artemisia abrotanum*	I	United Kingdom	Classified as tolerant of fly ash conditions in greenhouse pot experiments.	Hodgson and Townsend (1973)
Trees					
Black cherry	*Prunus serotina*	N	Southern U.S.	Naturally revegetates neutral to alkaline spoil resulting from limestone strip mining in Florida.	Craig and Smith (1979)
Black poplar	*Populus nigra italica*	I	United Kingdom	Consistently good early growth on both freshly collected and weathered fly ash in greenhouse pot experiments.	Hodgson and Buckley (1975)
Eastern cottonwood	*Populus deltoides*	N	Southern U.S.	Found naturally revegetating alkaline fly ash pits in Tennessee.	Gonsoulin (1975)
European alder	*Alnus glutinosa*	I	Southern U.S.	Nitrogen-fixing species; sensitive to moisture stress; poor survival rate on acidic to neutral soils--however, rapid growth of surviving individuals.	Scanlon and Duggan (1979)

(continued)

Table C.1. (Continued)

Common name	Scientific name	Species[a] origin	Region tested	Comments	Reference
Trees (contd.)					
Honeylocust	*Gleditsia triacanthos*	N	United Kingdom	Consistently good early growth on both freshly collected and weathered fly ash in greenhouse pot experiments.	Hodgson and Buckley (1975)
Horsetail casuarina	*Casuarina equisetifolia*	I	Southern U.S.	Establishes quickly on neutral to alkaline spoil resulting from limestone strip mining in Florida.	Craig and Smith (1979)
Live oak	*Quercus virginiana*	N	Southern U.S.	Naturally revegetates neutral to alkaline spoil resulting from limestone strip mining in Florida.	Craig and Smith (1979)
Loblolly pine	*Pinus taeda*	N	Southern U.S.	Fair survival rate, but good early growth on alkaline stoker-fed boiler ash.	Horton and McMinn (1977)
Sitka spruce	*Picea sitchensis*	N	United Kingdom	High tolerance and good growth when grown on fly ash under both laboratory and field conditions; extremely boron tolerant.	Hodgson and Buckley (1975)
Southern redcedar	*Juniperus silicicola*	N	Southern U.S.	Naturally revegetates neutral to alkaline spoil resulting from limestone strip mining in Florida.	Craig and Smith (1979)
Sweetgum	*Liquidambar styraciflua*	N	Southern U.S.	Good survival and early growth on alkaline stoker-fed boiler ash.	Horton and McMinn (1977)
Sycamore	*Platanus occidentalis*	N	Southern U.S.	Excellent survival and early growth on alkaline stoker-fed boiler ashes--however, may experience manganese deficiency; less successful on neutral to acidic ash.	Horton and McMinn (1977); Scanlon and Duggan (1979)

(continued)

Table C.1. (Concluded)

Common name	Scientific name	Species[a] origin	Region tested	Comments	Reference
Trees (contd.)					
Tree-of-heaven	*Ailanthus glandulosa*	I	United Kingdom	A rapidly growing species, which exhibited good tolerance of weathered fly ash conditions in greenhouse pot experiments.	Hodgson and Buckley (1975)
White poplar	*Populus alba*	I	United Kingdom	Consistently good early growth on both freshly collected and weathered fly ash in greenhouse pot experiments.	Hodgson and Buckley (1975)
Willow	*Salix* spp.	N	United Kingdom	High potential growth rate when planted as cuttings, although leaves may exhibit boron toxicity; *S. interior* and *S. nigra* found naturally revegetating alkaline fly ash pits in Tennessee.	Hodgson and Buckley (1975); Gonsoulin (1975)

[a] N = species native to United States; I = species introduced to United States (exotic); N/I = genus includes both native and introduced species.

Table C.2. Plant Species Potentially Useful in the Revegetation of Buried Coal Combustion Wastes[a]

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Grasses and Forbs				
Alfalfa	*Medicago sativa*	I	East, Midwest, West	Legume; good growth in dry regions; high boron tolerance.
Alkali sacaton	*Sporobolus airoides*	N	West	Recommended for dry regions; well adapted to moderately alkaline and saline conditions.
Bahia grass	*Paspalum notatum*	I	Southeast	Recommended for warmer climates; volunteer on alkaline limestone strip mine spoil.
Barley	*Hordeum vulgare*	I	Northeast, Southwest	Annual species; yields fast cover; good growth on alkaline and saline soils.
Bentgrass	*Agrostis* spp.	N/I	East	Semitolerant of growth on fly ash; some strains tolerant of high soil Al, Cu, Fe, and Zn concentrations.
Bermuda grass	*Cynodon dactylon*	N	Southeast, Southwest	Recommended for dry regions and saline soils.
Big bluestem	*Andropogon gerardi*	N	East, Midwest, West	Strong, deep-rooted, with short underground stems; effective in controlling erosion.
Birdsfoot trefoil	*Lotus corniculatus*	I	East, Midwest	Legume; salt tolerant; good growth on soil with pH 4.0 or greater.
Blackseed needlegrass	*Stipa avenacea*	N	West	Good for loam or heavier soils with > 33 cm precipitation per year.
Buffalograss	*Buchloë* dactyloides	N	Midwest, West	Drought-tolerant; withstands alkaline soils but not sandy ones; will regenerate if overgrazed.
Buffel grass	*Pennisetum ciliare*	I	Southeast, Southwest	Good growth on alkaline and saline spoils.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species^b origin	Region of use in United States	Comments
Grasses and Forbs (contd.)				
Canada bluegrass	*Poa compressa*	I	Northeast, Northwest	Does well on acid soils, droughty soils, or soils too low in nutrients to support good stands of Kentucky bluegrass.
Cicer milkvetch	*Astragalus cicer*	N	West	Legume; adapted to dry conditions; does well on alkaline soils.
Clover	*Trifolium* spp.	I	East, Midwest, West	Legumes; tolerant of saline and alkaline soils; adaptable to dry conditions.
Crownvetch	*Coronilla varia*	I	East, Midwest	Legume; used extensively on both moderately acid and calcareous spoils; if seeded with cover crop, may be useful in erosion control.
Deertongue	*Panicum clandestinum*	N	Northeast	Recommended for acid soils; does not compete well with other grasses.
Field brome	*Bromus arvensis*	N	Northeast, Northwest	Good winter cover plant; extensive fibrous root system; grows rapidly, easy to establish.
Flat pea	*Lathyrus sylvestris*	I	East	Legume; recommended for acid soils in cooler climates.
Foxtail millet	*Setaria italica*	I	Midwest, West	Requires warm weather during growing season; cannot tolerate drought; good seedbed preparation important.
Gramma grass	*Bouteloua* spp.	N	West	Drought-resistant species.
Indiangrass	*Sorghastrum* spp.	N	East	Good growth and vigor on some acid spoils.
Indian ricegrass	*Oryzopsis hymenoides*	N	West	Adapted to arid and semiarid regions.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species^b origin	Region of use in United States	Comments
Grasses and Forbs (contd.)				
Italian ryegrass	*Lolium multiflorum*	I	East, Midwest	Annual species; yields quick cover; adaptable to pH as low as 5.0.
Kentucky bluegrass	*Poa pratensis*	I	Northeast, Midwest	Recommended for cooler climates, moderate-pH soils.
Lespedeza	*Lespedeza* spp.	N/I	Northeast	Legumes; adaptable to a wide range of soil pH; good for erosion control.
Little bluestem	*Andropogon scoparius*	N	Northeast, Midwest	Slow to establish; good growth on moderately acid spoil.
Lovegrass	*Eragrostis* spp.	N/I	West	Recommended for dry regions; adapted to alkaline and saline conditions.
Oat	*Avena sativa*	I	East, Midwest, West	Bunch-forming; good winter cover plant; requires nitrogen for good growth.
Orchard grass	*Dactylis glomerata*	N	East, Midwest, West	Adapted to moderate-pH soils (pH 6-8); good for western, high-altitude sites.
Perennial ryegrass	*Lolium perenne*	I	East, Midwest	Highly adaptable to moderately acid and alkaline sites; can be developed for pasturelands; does well in mixtures with native grasses; good for rapid stabilization of soil and erosion control.
Prairie sandreed	*Calamovilfa longifolia*	N	Midwest, West	Tall, drought-tolerant; can be used on sandy sites; rhizomatous; seed availability poor.
Red fescue	*Festuca rubra*	N	East, Midwest, West	Grows in cold weather; remains green during summer; good seeder, widely adapatable, slow to establish.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Grasses and Forbs (contd.)				
Redtop	*Agrostis alba*	I	Northeast, Midwest	Useful for erosion control; good on extremely harsh spoil; recommended for cooler eastern climates.
Reed canarygrass	*Phalaris arundinacea*	N	East, Midwest	Highly adaptable to moderate acid sites; can be developed for pasturelands; does well in mixtures with native grasses.
Rye	*Secale cereale*	I	Northeast, Southwest	Annual species; yields fast cover for erosion control during initial vegetative establishment.
Sand dropseed	*Sporobolus cryptandrus*	N	West	Recommended for desert areas.
Sheep sorrel	*Rumex acetosella*	I	East	Root sprouting perennial; produces better cover than grasses on low-fertility soils.
Smaller seabeach grass	*Panicum amarum*	N	East	Good on very sandy, droughty sites.
Smooth brome	*Bromus inermis*	I	East, Midwest, West	Good for rapid stabilization and erosion control; fairly drought-resistant.
Switchgrass	*Panicum virgatum*	N	East, Midwest	Drought-tolerant; good growth on low-fertility soil; adaptable to wide soil pH range.
Tall fescue	*Festuca arundinacea*	I	East, Midwest, West	Shade-tolerant; does well in mixtures with other grasses.
Tall oatgrass	*Arrhenatherum elatius*	I	East, Midwest, West	Short-lived perennial bunchgrass, maturing early in the spring; less heat tolerant than orchard grass except in Northeast; good on sandy and shallow shale sites.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Grasses and Forbs (contd.)				
Timothy	*Phleum pratense*	I	Northeast	Good growth on soils with pH 5.0 or higher.
Western wheatgrass	*Agropyron smithii*	N	West	Sod-forming, spreads rapidly, slow germination; valuable for erosion control; drought-resistant.
Winter wheat	*Triticum aestivum*	I	Northeast, Midwest, Southwest	Annual species; tolerant to high salt and low moisture; may be good as cover crop during initial vegetative establishment.
Shrubs				
Big sagebrush	*Artemisia tridentata*	N	West	Adapted to growth on alkaline soils; rapid growth; effective soil stabilizer.
Black chokeberry	*Pyrus melanocarpa*	N	Northeast	Fairly good survival on acid soil.
Bladder-senna	*Colutea arborescens*	I	East	Nitrogen-fixing species; does well under alkaline conditions.
Blue paloverde	*Cercidium floridum*	N	Southwest	Drought-tolerant; will withstand alkaline conditions.
Bristly locust	*Robinia hispida* var. *vertilis*	N	East, Midwest	Nitrogen-fixing species; does well on moderate pH soil; good for erosion control.
Common matrimony-vine	*Lycium halimifolium*	I	West	Recommended for dry regions; adaptable to alkaline and saline conditions.
Coralberry	*Symphoricarpos orbiculatus*	N	Midwest	Good growth on spoil with pH 5.0-6.5.
Desert-willow	*Chilopsis linearis*	N	Southwest	Withstands cold and drought; excellent results on fertilized saline-alkaline tailings.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Shrubs (contd.)				
Elaeagnus	*Elaeagnus* spp.	I	East, Midwest, West	Adaptable to a wide range of soil pH; recommended for saline conditions in arid western climates.
Gregg catclaw	*Acacia greggii*	N	Southwest	Desert plant; with proper management, adaptable to a wide variety of soils.
Grease-wood	*Sarcobatus vermiculatus*	N	West	Adapted for growth on saline-alkaline soils in dry regions.
Hopbush	*Dodonacea viscosa*	N	Southwest	Arid, dry-country shrub; resistant to cold; excellent growth on saline-alkaline copper tailings.
Honeysuckle	*Lonicera* spp.	N/I	East, Midwest, West	Does well on moderate pH soils; poor results obtained on wet, saline-alkaline soils.
Indigobush	*Amorpha fruticosa*	N	East, Midwest	Acid-tolerant; prefers neutral to slightly alkaline soils.
Japanese barberry	*Berberis thunbergii*	I	Southeast	Tolerant of growth on alkaline soil.
Multiflora rose	*Rosa multiflora*	I	East, Midwest	Acid-tolerant; good for wildlife plantings.
Orach	*Atriplex* spp.	N/I	West	Arid, dry-country shrub; recommended for use on alkaline and saline soils; drought-resistant.
Rubber rabbitbrush	*Chrysothamnus nauseosus*	N	West	Adapted to alkaline-saline conditions; excellent growth on Arizona copper tailings.
Scotch broom	*Cytisus scoparius*	I	Northeast, Midwest	Very acid-tolerant; unable to withstand Pennsylvania and West Virginia winters.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Shrubs (contd.)				
Silver buffaloberry	*Shepherdia argentea*	N	West	Recommended for alkaline and saline conditions on wet soils.
Silky dogwood	*Cornus obliqua*	N	Northeast	Does well on moderate pH soil.
Southern arrowwood	*Viburnum dentatum*	N	East	Good survival on moderately acid spoil.
Sumac	*Rhus* spp.	N	East, Midwest, West	Eastern species are acid-tolerant; species used in West are adapted to alkaline and saline conditions in dry climates.
Tree tobacco	*Nicotiana glauca*	N	Southwest	Excellent growth on fertilized saline-alkaline tailings.
Trees				
Ash	*Fraxinus* spp.	N	Northeast, Midwest	Poor to good survival on moderate pH soils.
Arizona sycamore	*Platanus wrightii*	N	Southwest	Drought-tolerant.
Austrian pine	*Pinus nigra*	I	Northeast, Midwest	Good survival on acid sites.
Birch	*Betula* spp.	N	East	Good survival over a wide range of soil pH.
Black cherry	*Prunus serotina*	N	Northeast, Midwest	Does fairly well on acid embankments.
Black locust	*Robinia pseudoacacia*	N	Northeast, Midwest	Nitrogen-fixing species; produces fast cover; good nurse crop; excellent for erosion control; susceptible to insect attacks; good growth on alkaline overburden.
Black walnut	*Juglans nigra*	N	Northeast, Midwest	Fair survival on moderately acid soils; better growth on calcareous spoils.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Trees (contd.)				
Eastern cottonwood	*Populus deltoides*	N	Northeast, Midwest	Fast growing in pure stand on spoils with pH 4.0-8.0.
Eastern redbud	*Cercis canadensis*	N	East, Midwest	Good survival on moderately acid spoil in Illinois.
Eastern white pine	*Pinus strobus*	N	Northeast, Midwest	Tolerant to extreme acid conditions at some sites.
Eucalyptus	*Eucalyptus* spp.	I	Southwest	Drought-tolerant; adapted to dry regions.
European alder	*Alnus glutinosa*	I	East, Midwest	Good for use in erosion control; tolerant of wide range of soil pH and of high salinity.
Jack pine	*Pinus banksiana*	N	Northeast, Midwest	Superior growth on extremely acid sites.
Larch	*Larix* spp.	N	East	Acid-tolerant; requires moist soil with good drainage; some species are shallow-rooting.
Loblolly pine	*Pinus taeda*	N	East, Midwest	Superior growth on some acid-waste embankments.
Mesquite	*Prosopis* spp.	N	Southwest	Drought- and acid-tolerant.
Netleaf hackberry	*Celtis reticulata*	N	Southwest	Deep-rooting tree; very tolerant of drought and alkaline soil.
Norway spruce	*Picea abies*	I	Northeast, Midwest	Survives well on waste banks; slow early growth.
Oak	*Quercus* spp.	N	East, Midwest	Average to good survival on moderately acid spoil.
Osage-orange	*Maclura pomifera*	N	Northeast, Midwest	Grows well over a wide soil pH range; good growth on moist strip mine spoil.
Pitch pine	*Pinus rigida*	N	Northeast, Midwest	Superior growth on extremely acid soil; survives on shallow, dry, low-fertility soils.

(continued)

Table C.2. (Continued)

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Trees (contd.)				
Red pine	*Pinus resinosa*	N	Northeast, Midwest	Tolerant of low fertility and dry soils; good growth on acid spoils.
Scotch pine	*Pinus sylvestris*	I	Northeast Midwest	Hardy species on dry and infertile sites.
Shortleaf pine	*Pinus echinata*	N	Northeast, Midwest	Good growth and survival on acid sites.
Siberian elm	*Ulmus pumila*	I	Midwest, West	Recommended for dry climates; adapted for alkaline and saline conditions.
Silver maple	*Acer saccharinum*	N	Northeast, Midwest	Survival only fair on acid embankments.
Sitka spruce	*Picea sitchensis*	N	Northeast	Occasionally used on acid embankments; extremely tolerant of alkaline and saline conditions; high boron tolerance.
Speckled alder	*Alnus rugosa*	N	East, Midwest	Fast-growing; tolerant of a wide range of soil pH, and of high salinity.
Sweetgum	*Liquidambar styraciflua*	N	East, Midwest	In preliminary tests, appears to do better on neutral to alkaline soils than on acid soil.
Sycamore	*Platanus occidentalis*	N	East, Midwest	Adaptable to a wide range of soil pH; salt-tolerant.
Table mountain pine	*Pinus pungens*	N	Northeast	Slow growth; fair survival on higher acid shale.
Virginia pine	*Pinus virginiana*	N	Northeast, Midwest	Attains excellent height among conifers on some coal-waste embankments.

(continued)

Table C.2. (Concluded)

Common name	Scientific name	Species[b] origin	Region of use in United States	Comments
Trees (contd.)				
White spruce	*Picea glauca*	N	Northeast, Midwest	Good survival on acidic anthracite spoil.
Willow	*Salix* spp.	N	East, Midwest	Adaptable to a wide range of soil pH; *S. interior* and and *S. nigra* are volunteers on alkaline fly ash pits.

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[a] Data from Coalgate et al. (1973), D'Appalonia Consulting Engineers (1975), Gonsoulin (1975), Donovan et al. (1976), and GAI Consultants (1979).

[b] N = species native to United States; I = species introduced to United States (exotic); N/I = genus includes both native and introduced species.

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March 3, 1849

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Washington, D.C. 20240