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Vienna Fly Ash Disposal Site
PROBLEM DEFINITION STUDY

Final Report

Submitted to

Maryland Department of Natural Resources
Power Plant Siting Program
Tawes State Office Bldg. (B-3)
Annapolis, Maryland 21401

7 June 1982

Submitted by

Environmental Resources Management, Inc.
999 West Chester Pike
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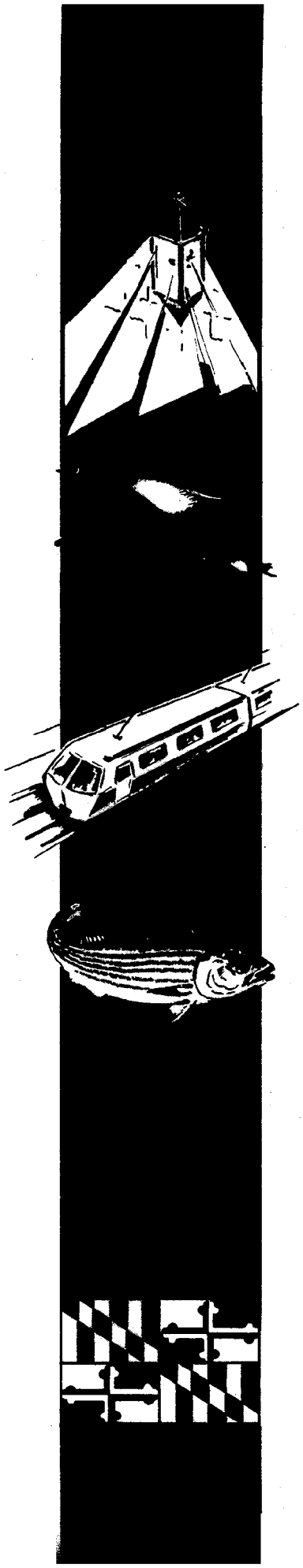
West Chester, Pennsylvania 19380



MARYLAND POWER PLANT SITING PROGRAM

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SECTION I

INTRODUCTION

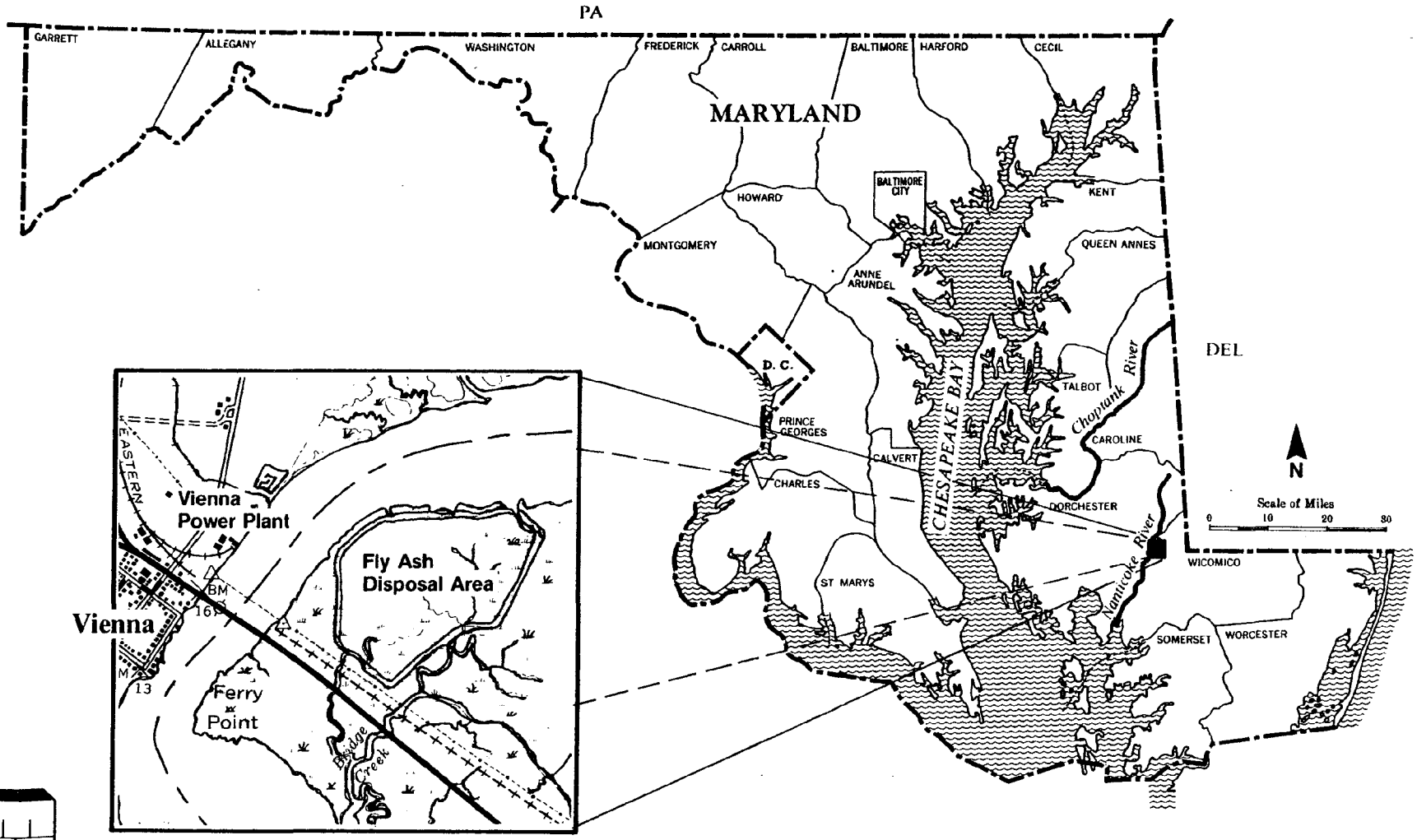
A. Background

This study was undertaken to determine if environmental contamination resulted from the disposal of fly ash in a diked disposal area. The now unused disposal area is adjacent to the east side of the Nanticoke River at Vienna, Maryland, in Wicomico County, starting about 400 feet north of U.S. Route 50 (Figure 1). The study was conducted for MD-DNR's Power Plant Siting Program by ERM, Inc., the Program's Solid Waste Studies Integrator, as part of a generalized interest on the part of the state in power plant solid waste issues.

B. Disposal History

From 1950 to 1972, Delmarva Power & Light (DP & L) operated a series of small ($\leq$ 70 MW) coal-fired generating units at their Vienna Power Station. The resultant fly ash/bottom ash from these units was first flushed into settling basins near the plant, then, from 1966 to 1972, the finer fly ash was sluiced across (under) the Nanticoke River to a diked disposal area of about 100 acres of wetland habitat adjacent to the river. This disposal site was neither lined nor covered, in conformance with standard industry practice at the time. The fly ash slurry covered about 80 percent of the disposal area when use of the site was discontinued in 1972. Water overflow was provided by two pipes through the dike in the northeastern corner of the site.

Figure 1
Vienna Site Assessment Location



The dike has since been breached at several locations permitting tidal flushing, rapid runoff, and flooding of the site during periods of high river flow. The area covered with ash has gradually revegetated since the phased conversion of the Vienna plant to oil in 1972.

The Nanticoke River is an important spawning and nursery area for several commercially important fishes, especially striped bass. The surrounding marshes are frequented by migratory waterfowl and shore birds for feeding, resting, and nesting. The Town of Vienna and a food processing plant have given consideration to using the surficial aquifer for process and drinking water. However, recent investigations of the aquatic biota near the Vienna disposal site (Mehrle et al., 1982) have indicated possible arsenic and selenium contamination problems. The current lack of containment of the ash material, the recognition of ash as a potential source of trace metal contamination, and the proximity of the disposal site to both ground water and surface water supplies necessitated an assessment of the potential environmental impacts of the ash, its leachate, and present physical and biotic conditions.

C. Study Objectives

The study objectives addressed the potential areas of impact resulting from the disposal of fly ash at the Vienna site. The study investigated the following objectives:

1. determine if fly ash-derived contaminants have affected surface and ground water quality in the surficial aquifer;

2. determine if the aquatic and terrestrial biota populating the disposal site and the surrounding area have been affected by fly ash-derived contaminants;

3. assess the implications of the above-mentioned effects on the health of the surrounding human population which utilize these resources.

SECTION II

BACKGROUND

A. Fly Ash Review

The burning of coal in power plants produces large quantities of fly ash. This ash is removed (cleaned) from the stack gases by either dry or wet methods. If wet methods are employed, the ash usually ends up in a slurry form requiring disposal in a solid waste facility. Bottom ash from the furnaces is often combined with this slurry. The Vienna power plant used a wet ash collection system and disposed of the ash in an unlined, diked disposal area in a tidal marsh, in accordance with acceptable disposal practices at the time.

As stated in a 1979 EPRI report by GAI Consultants, Inc.:

> "Fly ash is comprised of very fine particles, the majority of which are glassy spheres, scoria, iron rich fractions, and some crystalline matter and carbon. Due to its size and shape, the characeristics of fly ash are that of a high surface area to volume ratio solid that has agglomerated materials on its surface. In general, the composition of the spherical portion of the fly ash is somewhat immune to dissolution due to its glassy structure. However, on the surface of the spheres exists either easily exchangeable or adsorbed molecules which, when in the presence of a liquid, become dissolved. It is this mechanism, some researchers believe,

which ultimately produces leachate (Theis and Richter, 1979). Some of the very minute spheres may also dissolve into solution and contribute to the leachate. The elemental composition of the structure and surface material is then a function of not only the feed coal, but also the combustion sequence and method of collection."

The elements selected for investigation in this study were aluminum (Al), arsenic (As), chromium (Cr), and selenium (Se). Aluminum was chosen due to its high abundance in ash (a tracer), while the other three elements were selected because of their toxicities and potential for bioconcentration. A discussion of their potential toxicities follows in the next section. These and other trace elements have a wide range of concentration levels in fly ash as shown in Table 1. These metals chosen are also on the U.S. EPA list of priority pollutants.

Leachate from ash disposal sites is of concern due to the possibility that metals and ionic complexes, such as sulfate and nitrate, present in the ash may enter the ground water and contaminate present or future drinking water sources. Evidence is still inconclusive as to the degree of hazard of the ash materials (GAI Consultants, Inc. 1979).

The water soluble content of fly ash ranges from very little to several percent. The principal ions contained in ash leachate are calcium and sulfate, with smaller quantities of magnesium, sodium, potassium, and silicate ions present (Table 2). Free lime (CaO) accounts for part of the soluble calcium. The soluble sulfate is approximately half the total sulfate (SO_4) present in the fly ash. Fly ash leachate may be alkaline or acidic, depending on the type of coal burned.

TABLE 1

LITERATURE REVIEW OF ASH SOLIDS ANALYSES (in ppm)

Substance	Fly Ash Range	Fly Ash Avg.	Fly Ash Data Pts.
Arsenic	6 - 1,200	177	23
Barium	100 - 1,074	520.7	6
Cadmium	0.29 - 51	10	17
Chloride	-	1,000	1
Chromium	15 - 900	218.6	18
Copper	16 - 400	171	17
Fluoride	120 - 671	396	2
Iron	49,000 - 235,000	124,125	8
Lead	11 - 800	210.7	19
Manganese	100 - 1,000	389	16
Nitrate	-	85.6	1
Selenium	6.9 - 760	145	14
Silver	-	3	1
Sulfate	-	5,430	1
Zinc	50 - 9,000	1,314.3	20

Substance	Bottom Ash Range	Bottom Ash Avg.	Bottom Ash Data Pts.
Arsenic	0.5 - 18	7	14
Barium	300 - 731	481.6	7
Cadmium	0.5 - 3	1.25	12
Chloride	-	-	-
Chromium	15 - 895	213	13
Copper	12 - 300	87.2	12
Fluoride	-	10.6	1
Iron	66,000 - 211,900	116,100	9
Lead	3 - 30	13.2	11
Manganese	100 - 1,000	438.7	15
Nitrate	-	16	1
Selenium	0.08 - 20	5.45	11
Silver	-	-	-
Sulfate	-	675	1
Zinc	20 - 400	142	12

Source: D. W. Weeter and M. P. Bahor. Technical Aspects of the Resource Conservation and Recovery Act Upon Coal Combustion and Conversion Systems. Oak Ridge National Laboratory, February 1979. ORNL/OGPA-10.

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TABLE 2

LITERATURE REVIEW OF ANALYSES OF ASH POND DISCHARGES (in ppm)

Substance	Fly Ash Pond Range	Avg.	Data Pts.
Arsenic	0.01 - 1.1	0.38	3
Barium	0.2 - 0.3	0.25	2
Cadmium	0.001 - 0.037	0.019	2
Chloride	6 - 7	6.5	2
Chromium	0.02 - 0.067	0.044	2
Copper	0.02 - 2.4	0.91	3
Cyanide	-	-	-
Iron	1.44 - 630	211.12	3
Lead	0.01 - 0.91	0.33	3
Manganese	0.13 - 0.48	0.31	2
Selenium	0.002 - 0.33	0.12	3
Silver	-	-	-
Sulfate	209 - 358	283.5	2
Zinc	0.06 - 2.2	1.26	3

Substance	Bottom Ash Pond Range	Avg.	Data Pts.
Arsenic	0.006 - 0.018	0.012	2
Barium	0.1 - 0.2	0.15	2
Cadmium	0.001 - 0.003	0.002	2
Chloride	7 - 8	7.5	2
Chromium	0.009 - 0.01	0.095	2
Copper	0.041 - 0.065	0.053	2
Cyanide	-	-	-
Iron	5.29 - 5.98	5.64	2
Lead	0.02 - 0.02	0.02	2
Manganese	0.16 - 0.58	0.37	2
Selenium	-0.002 - 0.011	0.007	2
Silver	-	-	-
Sulfate	49 - 139	94	2
Zinc	0.09 - 0.14	0.12	2

Substance	Combined Ash Pond Range	Avg.	Data Pts.
Arsenic	0.005 - 0.038	0.038	9
Barium	0.1 - 0.2	0.19	10
Cadmium	0.001 - 0.005	0.002	6
Chloride	3 - 14	7.2	10
Chromium	0.004 - 0.043	0.015	10
Copper	0.01 - 0.08	0.042	10
Cyanide	0.01 - 0.05	0.03	3
Iron	0.23 - 2.3	0.8	10
Lead	0.01 - 0.025	0.014	10
Manganese	0.01 - 0.39	0.09	9
Selenium	0.003 - 0.065	0.016	10
Silver	-	0.01	1
Sulfate	59 - 156	109.7	10
Zinc	0.03 - 0.12	0.053	10

Source: Same as Table 1.

The ash samples for this study were analyzed by two methods, 1) acid digestion for total element content, and 2) leached according to the U.S. EPA EP Toxicity (45FR33127; May 19, 1980) determination procedures.

B. Metal Toxicity

1. Aluminum

Aluminum (Al) is one of the most abundant elements on earth and is a constituent of all soils, plants, and animal tissues. It does not occur naturally as the pure insoluble metal, although many of its salts are readily soluble. However, aluminum does not exist long in surface waters, as it precipitates or is absorbed as aluminum hydroxide, etc. (McKee and Wolf, 1963).

The toxicity of aluminum compounds to aquatic organisms varies for a given compound and organism (McKee and Wolf, 1963). At present, EPA has not established water quality criteria for the protection of aquatic organisms from aluminum compounds.

2. Arsenic

The principal emission source of arsenic (As) in the United States is reported to be coal-fueled power plants which produce approximately 3,000 tons of As per year (Sittig, 1980).

In natural waters As occurs predominately as soluble inorganic arsenate (+5). Elemental As exhibits relatively low toxicity because of its low solubility (Sittig, 1980).

Arsenic is known to bioconcentrate in both fresh and saltwater organisms and is toxic to invertebrates and vertebrates. In comparable studies, As toxicity (LC_{50} values) was lower in freshwater than marine organisms, with fish exhibiting less tolerance than invertebrates (Sittig, 1980). The EPA criterion for protection of freshwater aquatic life is 57 ug/l as a 24-hour average, and concentrations should never exceed 130 ug/l at any time. For saltwater organisms, the protection values are 29 ug/l and 67 ug/l, respectively.

3. Chromium

Chromium (Cr) is a commonly occurring element in the earth's crust (approximately 125 mg/kg); however, substantial concentrations are rarely found in natural waters. Industrial processes are a substantial source of Cr in natural waters, with coal combustion contributing an important fraction (Sittig, 1980).

In the aquatic environment, Cr occurs predominately as Cr^{+3} or Cr^{+6}. The oxidation and reduction of valence states of Cr are complex processes which are pH and temperature dependent. However, in general, hexavalent Cr is a strong oxidizing agent which is readily reduced to trivalent Cr which is biologically reactive with a variety of organic molecules (e.g., DNA and RNA). The uptake of Cr^{+6} by living organisms generally exceeds that of Cr^{+3} for solubility reasons (Sittig, 1980).

The toxicity of Cr is widely recognized, but toxic effects are complicated due to the existence of many Cr compounds which may contain Cr of different valences with distinct chemical, physical, and toxicological properties (Sittig, 1980). For protection of freshwater aquatic life, the EPA criterion for hexavalent Cr is 10 ug/l as a 24-hour average and should not exceed 110 ug/l at any time. For saltwater organisms the values are 25 ug/l and 230 ug/l, respectively. The value for trivalent Cr in freshwater is variable, being dependent on hardness, and no level has been established for saltwater (Sittig, 1980).

4. Selenium

The primary source of selenium (Se) in the environment is the weathering of rocks and soils; however, world-wide approximately 3,500 metric tons per year originate from human sources. Se may be an oxidizing or reducing agent, depending on its valence state which may range from -2 to +6. Se^{-2} reacts with metals to form heavy metal selenides which are extremely insoluble. The solubility of other selenium compounds may range from highly soluble to very low solubility (Sittig, 1980).

Selenium is acutely toxic to aquatic invertebrates and fishes, although toxicity is dependent on synergistic and antagonistic elements. The EPA criterion for protection of freshwater aquatic life is 9.7 ug/l as a 24-hour average, and concentrations should never exceed 22 ug/l at any time. For protection of saltwater organisms, the criterion is 4.4 ug/l over 24 hours and not to exceed 10 ug/l at any time (Sittig, 1980).

C. Trace Metal Behavior in Aquatic Systems

1. Abiotic

The behavior of trace metals which reach aquatic systems in the dissolved state is governed by the rules of coordination chemistry which deals with complex formation (Stumm and Morgan, 1970). The activity of trace metals is primarily influenced by the stability of the element-complex but is also affected by oxidation-reduction (redox) potential and pH (Brooks, 1977).

Permanent or predictable equilibrium between the available and chemically/physically bound fractions of trace elements have not been recognized. This is due to the following dynamic physical, chemical, and biological processes which continually modify the chemical activity and concentration of trace metals: temperature, salinity, solubility, water hardness, and additional biological and chemical factors.

The major portion of metal transport in rivers is through organometallic particulates, with only a small amount carried in solution. These complexes compose the class of organic acids, including humic acids, which make up the major portion of organic matter in soils and water. Humic acids have strong chelating properties for trace elements. However, in water of high inorganic mineral content, the carbonate and bicarbonate fractions become major binding agents (Dvorak et al., 1978).

In sediments, metals are associated with organic compounds and/or clays. Metals (e.g., Hg, Cd, As) are often transported in the suspended particulate fraction and later in-

corporated in the sediments in areas of deposition. In general, mobilization of trace metals in aquatic systems is greatly dependent on redox potential and pH. Alkaline systems produce precipitation of insoluble hydroxides and sulfides, while systems which are strongly acid promote solution of many metals which increase their bioavailability (Dvorak et al., 1978).

2. Biotic

The three major types of differential uptake of trace elements by biota are defined as follows: 1) bioaccumulation, which is the ability of an organism to concentrate an element above abiotic levels; 2) bioconcentration, which is the influence of size and/or age on elemental concentrations within an organism; and 3) biomagnification, which is the successive increase of trace element concentration with trophic level transfer (Dvorak et al., 1978).

Microorganisms are an important factor affecting the chemical form and concentration of trace elements in that they can convert inorganic complexes to organometallic compounds, resulting in increased bioavailability. Examples relevant to this study are the conversions of selenium to dimethyl selenide and arsenic to an organometalloid. Bacteria also bioaccumulate metals in solution with subsequent transfer of contaminants up the food chain (Patrick and Loutit, 1976).

Phytoplankton, aquatic bryophytes, and rooted aquatic macrophytes are all known to accumulate metals (Dvorak et al., 1978).

The capacity of algae to accumulate trace elements has resulted in their use as indicator organisms. The ability of aquatic plants to accumulate and maintain trace elements can be useful in determining the possibility of periodic discharges of trace metals (e.g., from storm events, etc.) when high concentrations may not be present at all times. This characteristic may prove to be important at the Vienna site.

Concentration factors for various trace elements in algae, aquatic bryophytes, and aquatic macrophytes vary with the specific plant and element (Dvorak et al., 1978). Mayes and McIntosh (1975) suggested that plants other than algae be utilized as indicators of trace metal contamination due to the high concentration factors displayed by algal species. The effects of trace metals on aquatic plants include: changes in physiology, productivity, community composition, and species abundance (Dvorak et al., 1978).

Toxicity of trace elements in invertebrates is element and species specific. Acute toxicity may not be as important to invertebrates as the sublethal effects which exhibit two types of effects: 1) reduce the fitness of the organism and 2) alter community structure and function, which is a result of the first effect. These effects manifest themselves as a reduction in the biomass, nutrients, and/or energy for transfer to higher trophic levels (Dvorak et al., 1978).

Fish take up trace elements through two primary pathways: 1) absorption through the gills and 2) ingestion (via feeding). The latter is considered to be the major source of contamination; however, trace metal concentrations are often lower

in the predator than prey. Uptake is influenced by the chemical form and water quality. In general, the toxicity of metals decreases as hardness, CO_2, and pH increase.

Direct lethal effects of trace metal contamination in fishes have been linked to failure of specific organs, e.g., the gills and liver. Sublethal effects include interference with neuro- and physiological processes which reduce fitness. A significant result of reduced fitness is increased susceptibility to predation (Dvorak et al., 1978).

In summary, trace metals usually are primarily found in the sediment, as relatively small amounts exist in the dissolved state. Trace elements are bioaccumulated by rooted macrophytes and benthic invertebrates from the substrate. Phytoplankton adsorb trace elements to their cell walls and absorb them internally. The greatest bioaccumulation is exhibited generally by the lower order consumers (e.g., grazers and detrital feeders). Fishes obtain trace metals primarily from food organisms; however, trace metal concentrations are often lower in the predator than prey, which is counter to the biomagnification theory (Dvorak et al., 1978).

SECTION III

DESCRIPTION OF STUDY AREA

A. Surface Drainage

The Vienna ash disposal site was located and described in Section I. The present surface drainage of the Vienna marsh system near the fly ash disposal site is influenced primarily by tidal and Nanticoke River currents. A single tidal stream (Bridge Creek) system drains the site and the landward shrub and forest lowlands directly east of the site (Figure 2). Aerial photographs taken for DP & L show that when the channel surrounding the disposal site was created, the northernmost headwaters of Bridge Creek were opened to the Nanticoke River (Figure 3). Three branches of the creek were impounded within the disposal site. Overflow from the disposal site, as initially constructed, was from the northeastern corner into the surrounding moat channel. Some time after fly ash disposal ceased, the dike surrounding the site was breached at a number of places along the eastern border (Figure 3) (personal communication, Mr. John Dick, DP & L, Vienna, MD).

Net flow from the site and most of Bridge Creek under existing conditions apparently is southward into the Nanticoke River. However, the northern portion of this "marsh watershed" now is tidal to the north into the Nanticoke River, especially on the ebb tide. High seasonal flood flows of the Nanticoke River can enter the disposal site in its northeastern corner and also flow through the main stem of Bridge Creek via the disposal site eastern channel (moat). Ebb tide and high water flows of the river also flow into

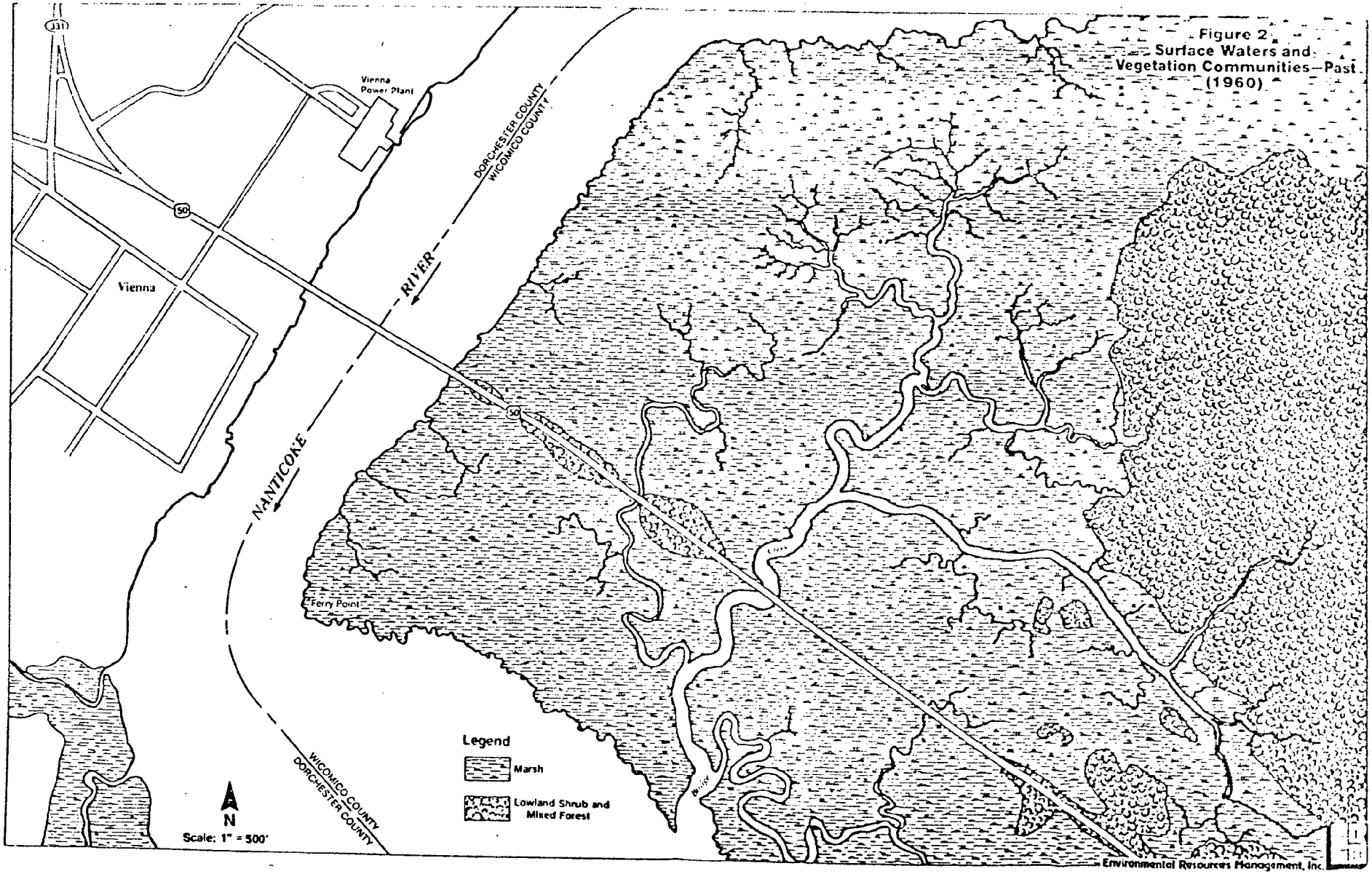


Figure 2
Surface Waters and Vegetation Communities—Past (1960)

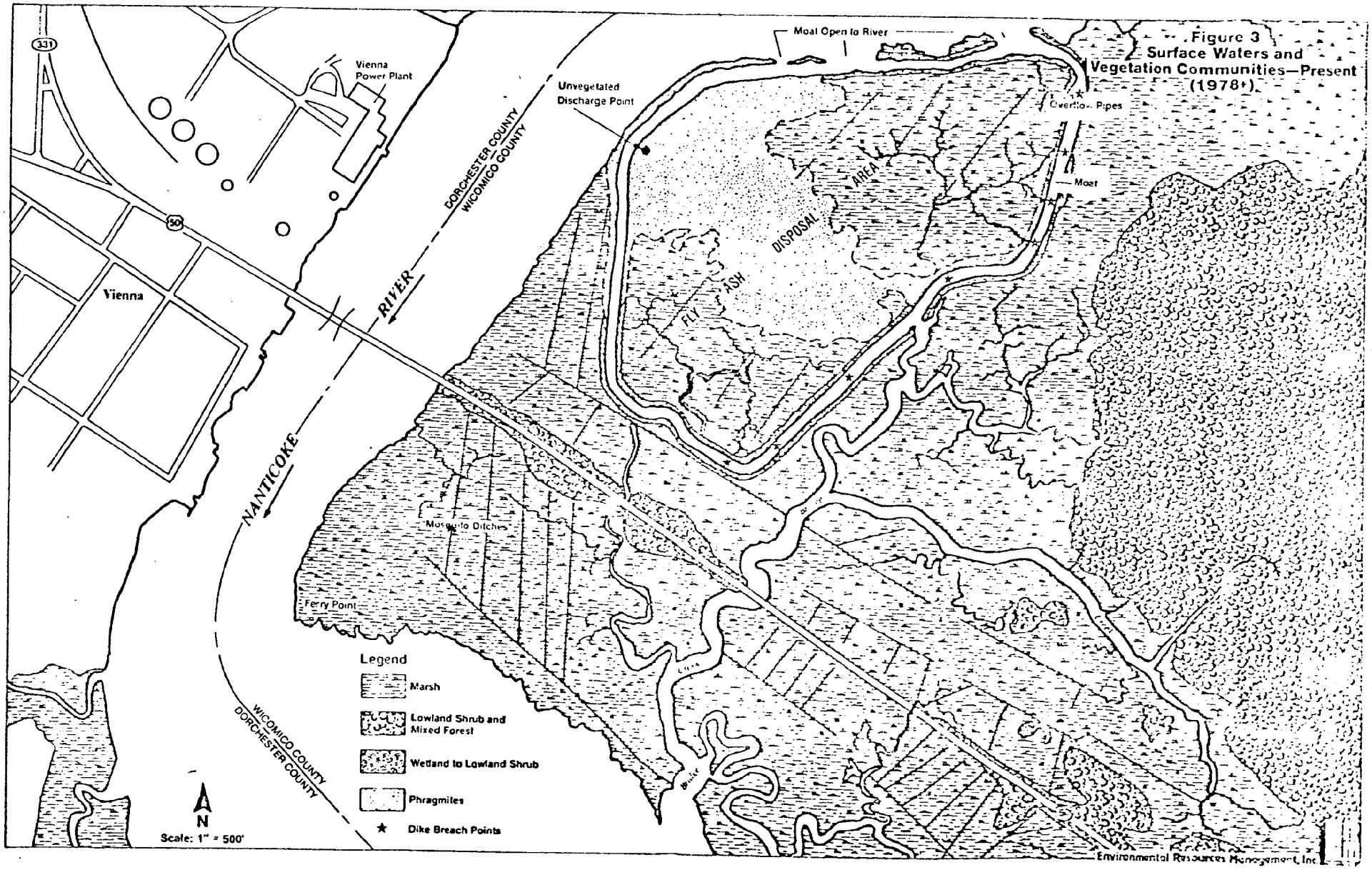


Figure 3
Surface Waters and Vegetation Communities—Present (1978+)
Moat Open to River
Vienna Power Plant
Unvegetated Discharge Point
Overflow Pipes
Moat
DISPOSAL AREA
FLY ASH
DORCHESTER COUNTY
WICOMICO COUNTY
RIVER
NANTICOKE
Vienna
331
50
Mosquito Ditches
Ferry Point
WICOMICO COUNTY
DORCHESTER COUNTY
Legend
Marsh
Lowland Shrub and Mixed Forest
Wetland to Lowland Shrub
Phragmites
Dike Breach Points
N
Scale: 1" = 500'
Environmental Resources Management, Inc.

the site channel on its western border from openings to the north. The creek channels in the disposal site are reduced to small trickles at low tide, and the site moat is not navigable with a flat bottom boat during this low water period. Local hydrological conditions are also altered somewhat by mosquito ditches put in by the Maryland Agriculture Department in the 1960's (Lesser, 1982).

B. Regional Geology

The geologic and hydrogeologic characteristics of the area determine the vulnerability of ground and surface waters to contamination. Therefore, a literature review was conducted to determine the three-dimensional hydrogeologic characteristics of the area. This report addresses only those subsurface geologic units which may be used for ground water supply in the vicinity of the site.

The disposal site under study is located in the sediments of the Atlantic Coastal Plain, a thick sequence of unconsolidated marine and fluvial deposits forming the western edge of the Atlantic continental shelf (Rasmussen et al., 1957). These sediments consist of units of sand, gravel, clay, and silt in varying proportions and thicknesses, which form a wedge thickening to the southeast. This wedge thins toward the northwest, terminating at the contact with the metamorphic rock complex of the Piedmont Physiographic Province.

The Coastal Plain sediments range in age from Lower Cretaceous through Holocene. The total thickness of the sedimentary sequence in this area is estimated to be 3,000 to 4,000 feet (Rasmussen et al., 1957).

The deepest significant geologic unit in the Vienna area is the Piney Point Fm., which occurs at about -600 feet in elevation (Mack ec al., 1971). Although it is not presently used in the area, moderate transmissivity and good water quality offer a potential for ground water development. The Piney Point is confined above by the Calvert Fm. This formation contains a sand unit called the Federalsburg Aquifer, but poor water quality precludes its use as a major water source. Overlying the Calvert is the Choptank Fm. which contains the Frederica Aquifer, a minor aquifer which is used little in the Vienna area.

Overlying the Choptank, and generally considered to be the uppermost major Coastal Plain unit in the Vienna area, is the St. Mary's Fm. (Mack et al., 1971). This formation consists of silty clay and clayey silt. Although some fine shells and sand are present, the formation functions as an aquiclude above the Choptank Fm. As such, it is not used for water supply. Its upper surface occurs from elevation -90 feet to elevation -100 feet in the area. This formation forms the top of the pre-Pleistocene southeastward dipping coastal plain deposits in the Vienna area. Its upper boundary is considered to be an erosional surface, although area borings indicate that the actual erosional surface may be in a thin veneer of overlying Manokin Fm. silts (Delmarva Permit Application, 1979). One report cites a vertical permeability of less than 1×10^{-8} cm/sec for the St. Mary's (Geraghty and Miller, 1980). Thus, leakage to the underlying Choptank aquifer is very slow.

The Salisbury Fm. is the uppermost aquifer unit in the area. It consists of Pleistocene age sediments which were deposited on the Miocene erosional surface (Mack et al., 1971). It contains two major facies. The upper unit, or Beaverdam facies, consists of lenticular light colored quartz sand and silt. The lower unit, or Red Gravelly facies, consists of red to brown sand and gravel. An upper discontinuous unit called the Walston Silt is present in some parts of the Vienna area, near the top of the Beaverdam. This fine grained unit has been defined beneath the power plant property west of the Nanticoke River. The Salisbury Fm. is used as an aquifer in the Vienna area. In some areas it is under water table conditions, and in others is confined by the Walston Silt. Limited aquifer test data indicate the transmissivity of the unit to range up to 100,000 gpd/ft in some areas. In the Vienna area, aquifer tests indicated a range from 7,800, near Vienna, to 66,000 gpd/ft toward the west (Delmarva Power and Light Company, 1978). The water quality in the Salisbury Fm. is generally good, with TDS ranging from about 600 ppm near the Nanticoke River, to as low as less than 100 ppm away from the river.

River and Tidal Marsh deposits constitute units of significance to this study. These sediments consist of alluvial point bar deposits which were laid down on the insides of meanders in the Nanticoke River, and the organic root mat of tidal marshes that overlie the channel deposits. The river deposits consist of dark brown silt with a trace of sand and infrequent lenses of clayey silt. Where the organic marsh mat has formed, the silt contains dense decaying organic matter, both root and upper plant remains.

The present surface drainage system on the coastal plain, including the Nanticoke River, is thought to have formed quickly and developed mature drainage characteristics, such as broad meanders, when sea level was considerably lower than at present. When the sea level rose, the base level of erosion rose, and the meandering rivers began to backfill their inundated estuarine channels rapidly, developing the tidal marsh system present today (Rasmussen et al., 1957). The present point bar sediments consist mainly of silts and clays (Mack et al., 1971), deposited in the low energy estuarine environment.

C. Site Geology

The study site lies on the inside of a meander in the Nanticoke River. The river channel is cut into sediments of the Salisbury Fm. East of the fly ash disposal site, in the Salisbury, Maryland, area and west of the site at the Vienna power plant, previous studies have defined a large paleochannel cut into the Miocene sediment surface, probably during the Illinoisan glaciation (Hansen, 1966). This paleochannel represented a major drainageway to the sea on the east. Although no borings are available under the fly ash disposal site itself, subsurface mapping both east and west of Vienna indicates that the center line of the paleochannel passes directly beneath the site (Hansen, 1966; and Mack et al., 1972). Thus, the deposits of the Pleistocene Salisbury Fm. beneath the site are paleochannel fill deposits. These deposits presently constitute the local shallow aquifer in the Vienna area, with the ground water discharge mainly to

the Nanticoke River. Projecting available data from the power plant area indicates that the bottom of the Pleistocene deposits occurs from about elevation -100 to -80 feet beneath the study site (Geraghty and Miller, 1980).

The disposal site occupies a tidal marsh which has developed inside a Nanticoke River meander. Thus, the sediments directly beneath the site are the Nanticoke River point bar silts and organic marsh deposits.

D. Flora

The marshes of the Nanticoke River at Vienna have been subjected to numerous disturbances (highway and transmission corridors, ditching, filling, etc.) and exist in brackish waters near the transition zone of salt to fresh water. Field survey and photo interpretation techniques were used to create the generalized vegetation community maps as per 1960 conditions (Figure 2) and 1978 to present conditions (Figure 3). The only photographic records found and available to date are from October 1960 (USGS, Reston, VA) and October 1978 (Air Survey Corp., Reston, VA) overflights. The older photo is black and white, while the latter record is in both black and white and false color infrared. Large prints of both, at a scale of 1" = 400', were acquired for purposes of this study (from Air Survey Corp.).

Dominant plant species (identified by several reconnaissance surveys) of these diverse marshes include giant cordgrass (*Spartina cynosuroids*), switch grass (*Panicum virgatum*),

marsh hibiscus (Hibiscus spp.), and some Olney three-square (Scirpus olneyi). These marshes can be classified as a Type 16 salt meadow wetland, but species now in and around and landward of the disposal site are more indicative of a Type 12 shallow fresh marsh (Shaw, 1956). Although not observed, there may be Spartina alterniflora (saltmarsh cordgrass) also present. Species such as arrow-arum (Peltandra virginica) and rice cut-grass (Leersia oryzoides) are also present, as is Baccharis halimifolia (groundsel-bush) especially on the slightly drier berms created around the disposal site. Toward the heads of the tidal creeks (inside as well as outside of the disposal area), the brackish water species are further displaced by freshwater macrophytes, such as cattails (Typha spp.), rush species, and large stands of arrow-arum.

The present ERM study (see Section V, Biological Investigations) has shown that the wetland soils inside the disposal area and on the surrounding berms have become revegetated. The berm was created by dredging up the marsh and underlying soils to enclose the disposal area. The fly ash that was slurried into the site between 1966 and 1972 buried about 80 percent of the enclosed marsh. Aerial photographs revealed that by 1978 all but a small area (about 400 sq. ft.) at the very end of the pipe was revegetated. Numerous test holes were dug throughout the site and revealed that the ash deposit forms a layer from 1 to 3 feet thick over the original marsh mat of organic matter. The ash layer is now supporting a monotypic community of Phragmites. Phragmites also covers portions of the berms, where in large part a shrub swamp or lowland open field community tends to be establishing itself. The species composition of these successional

communities on the disturbed substrates is very heterogeneous, dependent on plant dispersal mechanisms and also moisture content of sites. Any of the above mentioned species (especially those favoring drier sites) may be present, but mixtures of the following were also observed: winged sumac (Rhus copallina), tickseed-sunflowers (Bidens spp.), Aster spp. and possibly marsh Boltonia, as well as grasses, rushes, sedges, and ferns. Some of these formed dense communities inside the dike and reflected aspect dominance at least during the fall of the year.

The U.S. Route 50 roadway and embankment which crosses the marsh and the extensive mosquito ditching, throughout, undoubtedly caused many changes in the hydrologic functioning of the marsh to the north and south of the road. However, no baseline information is available that documents these changes. There has also been construction and expansion of the power line right-of-way north of and parallel to Route 50, which has involved some filling and clearing activity in the marshes and adjacent woodland. The species composition of the marsh and the filled areas has been undergoing a successional change for many years in response to these major perturbations. An artificial shrub swamp has been developing along these disturbance fill areas. Members of this community are representative of the shrub swamp forming the eastern border of the Vienna marsh which has been classified as a Type 6 shrub swamp wetland (Shaw, 1956). Also, see Appendix A.

E. Fauna

1. Aquatic

Typical of brackish water, the invertebrate diversity at the Vienna marsh is relatively low; however, most of the species found occur in abundance. Amphipods dominate the microinvertebrate population and are an important food for juvenile fishes. Other invertebrates include mud crabs, balanoid barnacles, gilled snails, ectoprocts, copepods, sea nettles, lion's mane jellyfishes, and small numbers of blue crabs (Delmarva Power, 1979).

Fishes commonly found in the Nanticoke River are listed in Table 3. The Nanticoke River above and below the Route 50 bridge is an important spawning ground for anadromous fish, many of which are important commercial and game species. The Nanticoke River is one of four principal spawning areas for striped bass in the Chesapeake Bay area. Striped bass spawn from Round Island Point upstream to Sharptown, but spawning is typically heaviest at Vienna where environmental conditions such as temperature, salinity ranges, moderate turbulence, and low turbidity are optimal (Delmarva Power, 1979).

2. Terrestrial

Vertebrates are common to the wetland marsh and shrub areas. Fifty-five species of amphibians and reptiles are reported for the Eastern Shore region. Based upon habitat requirements, approximately 20 species are believed to inhabit the Vienna marsh (Delmarva Power, 1979). A species list is provided in Appendix B.

TABLE 3

FISHES OF THE NANTICOKE RIVER
IN THE VICINITY OF VIENNA, MARYLAND

Species	Commercial/Sport Importance	Anadromous
American Eel	X	
Ladyfish		
Blueback Herring	X	X
Alewife	X	X
American Shad	X	X
Atlantic Menhaden	X	
Gizzard Shad	X	
Bay Anchovy		
Chain Pickerel		
Carp		
Golden Shiner		
Satinfin Shiner		
White Catfish	X	
Brown Bullhead	X	
Atlantic Needlefish		
Banded Killifish		
Mummichog		
Striped Killifish		
Rough Silverside		
Tidewater Silverside		
Atlantic Silverside		
White Perch	X	X
Striped Bass	X	X
Pumpkinseed	X	
Largemouth Bass	X	
Tessellated Darter		
Yellow Perch	X	
Bluefish	X	
Silver Perch	X	
Spotted Seatrout	X	
Spot	X	
Atlantic Croaker	X	
Naked Goby		
Hogchoker		

Sources: Academy of Natural Sciences of Philadelphia. 1971. Nanticoke River Surveys 1969 and 1970.

Maryland Department of Natural Resources, Fisheries Administration. Seining Records 1972-74.

Approximately 15 bird species occur in the vicinity of Vienna (Delmarva Power, 1979). Common year-round species include eagles, vultures, hawks, osprey, pheasants, owls, woodpeckers, and songbirds. Among these, pheasants and some songbirds will physically inhabit the marshes, while the others include the marshes as part of their feeding territory. Waterfowl are seasonally abundant in the Chesapeake Bay and Vienna marshes since the Bay area is the principal gathering place on the Atlantic coast during migration. Geese, ducks, and swans begin moving into the Bay region from the north in October, reach population peaks in December and January, and return northward in March (Stewart and Robbins, 1958).

Some species such as mallards, black ducks, wood ducks, and teals are typically associated with the brackish-freshwater environment found around Vienna. The fly ash site is an important roosting area for wood ducks during the fall migration. Numerous shore birds and waders also dwell in the Vienna marsh, including herons, rails, white ibis, gulls, and occasional terns. Marsh usage by these birds is variable; some may be resident breeders, transients, or stragglers (BNW Refuge, 1976).

There are some 15 species of mammals in the project area marshes and uplands (Appendix C). Many mammals which are found in wetlands will also inhabit the uplands, such as striped skunk, raccoon, and red fox. The most dominant mammal found in the Vienna marsh ecosystem is the muskrat; white-tailed deer are also abundant. The stream banks and tidally exposed banks of the channel moat around the disposal site are riddled with muskrat burrows and trails. Only one muskrat house was sighted during sampling surveys, and that was in the Nanticoke control site area. Numerous

signs of raccoon were also seen along the dike of the disposal site. Numerous signs observed at one location along the eastern dike indicated extensive raccoon usage. At this breach location they were apparently feeding on crayfish.

3. Threatened and Endangered Species

The Delmarva Fox Squirrel is believed to inhabit portions of the greater Vienna area but not the marshes. Its prime habitat is mature woodland, loblolly pine, and hardwood forests along the western bank of the Nanticoke between Marshyhope Creek and Penknife Point.

One pair of southern bald eagles is known to nest in the vicinity of Point No Point south of Vienna (Delmarva Power, 1979). These birds prefer tall trees adjacent to marshy areas. The southern bald eagle is generally declining throughout its range due to human encroachment on primary nesting areas and reduced reproduction from bioaccumulation of toxic substances.

Osprey also nest along the Nanticoke River near Vienna. Although it is not officially classified as rare or endangered, the osprey is declining in most regions and may be seriously threatened. The Chesapeake Bay has the largest breeding population known in North America.

The swamp sparrow has been reported nesting close to the ground in stands of giant cordgrass in the Vienna marsh. This species is not threatened, but it is unique in southern locations.

SECTON IV

HYDROGEOLOGIC INVESTIGATION

A. Site Reconnaissance

An initial site reconnaissance was conducted for the purpose of assessing site conditions and access. This reconnaissance revealed that site conditions were such that detailed hydrogeologic investigation could not be conducted as planned. The conditions encountered and the consequent problems in performing the work were as follows.

a) Access to the site is possible only by boat, and is further restricted during low tide by shallow water and by shallow pilings in the south and east sections of the moat surrounding the fly ash disposal area. This precluded the use of heavy equipment such as backhoes or drilling rigs on the site, as had been originally envisioned. These conditions also restricted working hours to high and mid tides, and tended to restrict water sampling to periods when tidal waters were mixed with site ground water and surface water.

b) The area containing the major portion of the fly ash deposit is covered by a thick stand of *Phragmites* reed grass. The grass obstructed vision and access, making a topographic survey of the site unfeasible.

c) The site is a shallow marsh with a hummocky vegetated surface in very soft silt. This terrain, plus the Phragmites stand, made access to the interior time consuming.

Since installation of piezometers was critical to the hydrogeologic study, the feasibility of using an all-terrain-vehicle mounted drilling rig was explored. Several drillers were contacted, but none was found who could supply a drilling rig which could perform the necessary work at the site. It was decided, therefore, that any well points or piezometers would be installed using a hand auger.

B. Methods

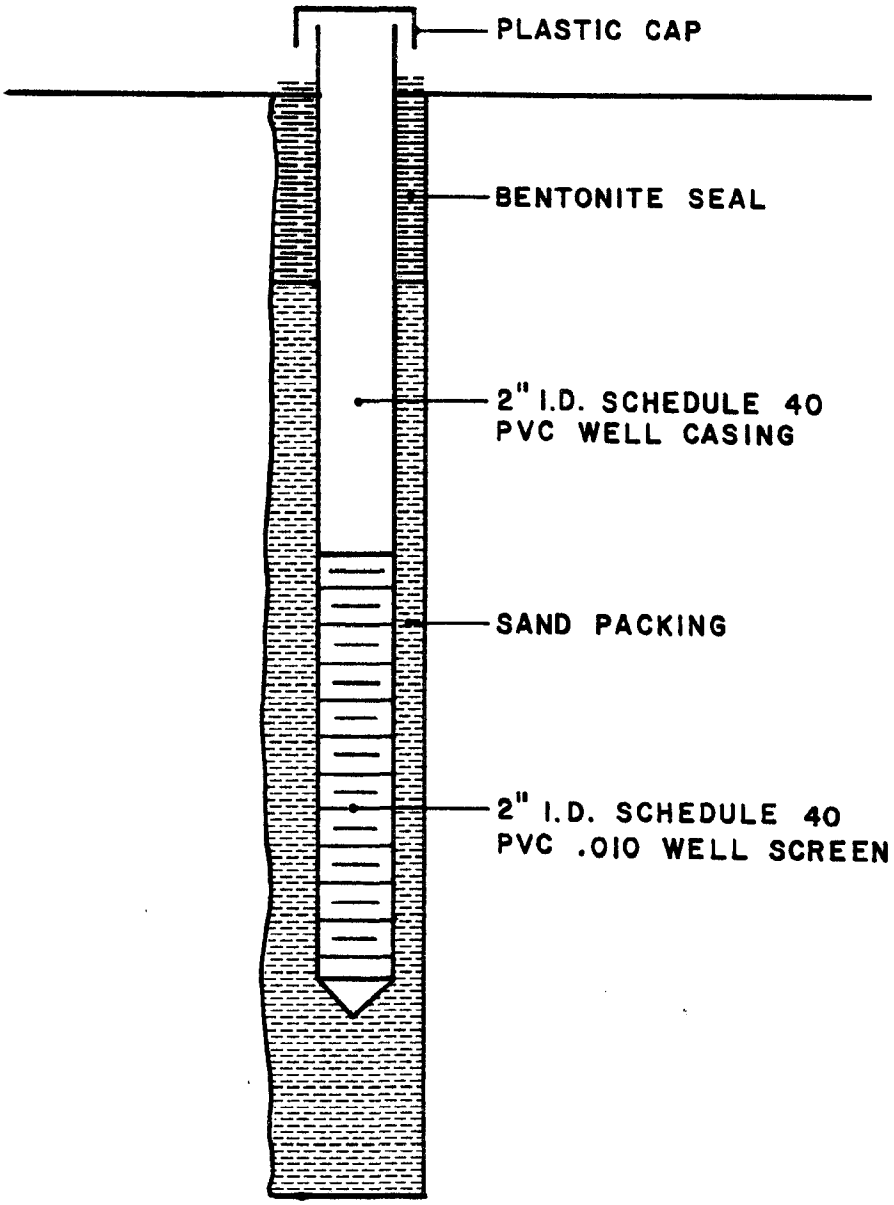
1. Fly Ash Distribution

Reconnaissance work was conducted to determine the distribution and nature of the site topography and drainage, and to determine the areal and vertical distribution of the fly ash deposits on the site. This was determined by walking the site, checking for fly ash presence and depth with a hand auger and a shovel.

2. Well Point Installation

Well Point V-1 (Figure 4) was installed by hand auger, using a 3 1/2 inch diameter bucket soil auger. An 18-foot deep boring was made, which showed the fly ash to be about 3 feet thick and to be underlain by a 9-foot thick swamp mat of

Figure 4
Well Construction at Location V-1



silt and decaying organic material. Underlying the swamp mat were alternating layers of runny, flowing dark gray silt, clayey silt, and plastic silty clay. At 18 feet a white medium grained quartz sand unit was encountered. The boring was terminated because of the length and weight of the auger string. The log of this well is included in Appendix D.

The runny silt quickly backfilled the boring to the 12-foot level. A 5-foot long, 1 1/2 inch diameter Schedule 40 PVC .010 well screen was installed in the hole. One and one-half inch Schedule 40 PVC casing extended above the surface (Figure 4). Sand packing was poured into the hole as a base for the well point, and the outer annulus was sand packed around the screen. Two feet of bentonite seal were installed at the surface. The final depth of the well was 9 feet.

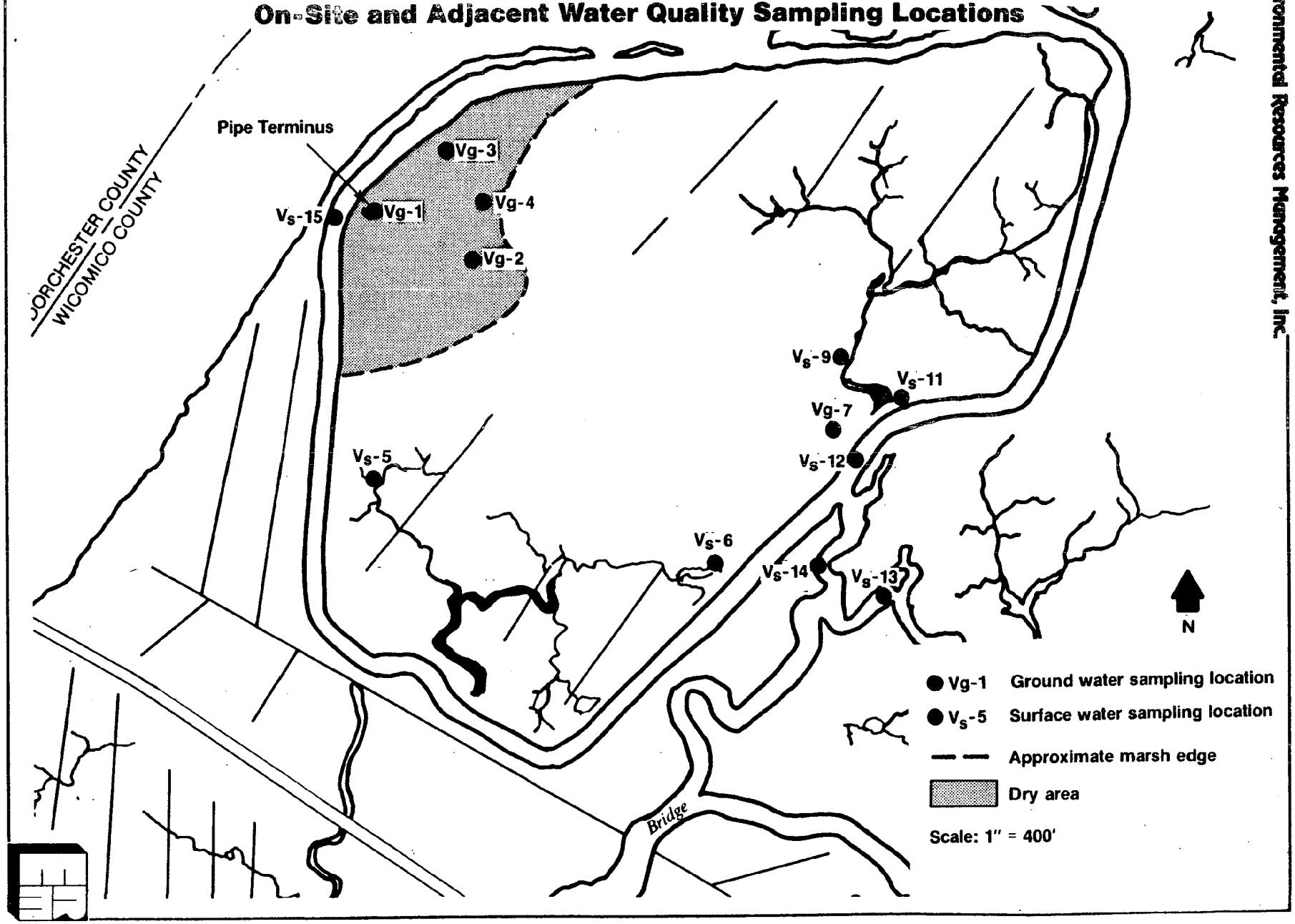
Five additional attempts to install well points elsewhere at the site were unsuccessful. The clay fraction of the sediments which held the hole open at V-1 were not encountered elsewhere on the site, and the running silts filled the borings almost to the surface. Establishing additional well points would require use of drive casing and wash equipment, and possibly mud to stabilize the Pleistocene sands for well screen installation. Since site conditions precluded the use of such methods, the hydrogeologic investigation was limited to the investigation of the shallow ground water quality and surface water quality. The one well was insufficient to provide further data on deeper ground water quality or on ground water flow system dynamics at the site.

3. Sampling

Ground water, surface water, and fly ash samples were collected from representative areas of the site. On November 19, one set of five ground water samples were collected at locations V-1, V-2, V-3, V-4, and V-7 (Figure 5). The samples were obtained from the top of the saturated zone in the limited areas where ground and surface waters are not contiguous as a marsh. The samples were obtained by hand augering into the root mat layer with a 3 1/2 diameter inch bucket auger and terminating the hole before encountering running silt. The ground water was evacuated with a hand pump and allowed to recover. A sample was obtained from the hole by bailing, and specific conductivity was measured. Two 500 ml plastic bottles were filled with ground water at each sampling location. The bottles were filled completely to minimize entrained air, packed in ice, and delivered to Lancaster Laboratories for chemical analysis. The auger borings were then deepened to locate the running silt unit.

On November 19, one set of nine surface water samples were collected on and near the site at locations V-5, V-6, V-9, V-11, V-12, V-13, V-14, and V-15. These locations are shown on Figure 5. One additional sample was collected at V-19 on December 10. Samples obtained at locations V-5 and V-9 represent surface water quality conditions within the fly ash deposits, since they directly drain the fly ash deposits. Samples V-6 and V-11 were collected to determine surface water quality leaving the site. Samples V-12 and V-15 were collected from the moat, and V-14 from Bridge Creek to determine surface water quality in the major drainageways adjacent to the site. Samples V-13 and V-19 were collected to determine natural surface water quality in the tidal marsh environment near the fly ash site, and remote from it

Figure 5
On-Site and Adjacent Water Quality Sampling Locations



(Figure 6). As with the ground water samples, two 500 ml plastic bottles were filled at each sampling location and packed in ice. Samples were collected during ebb tide to ensure that the water collected had been in contact with the fly ash site.

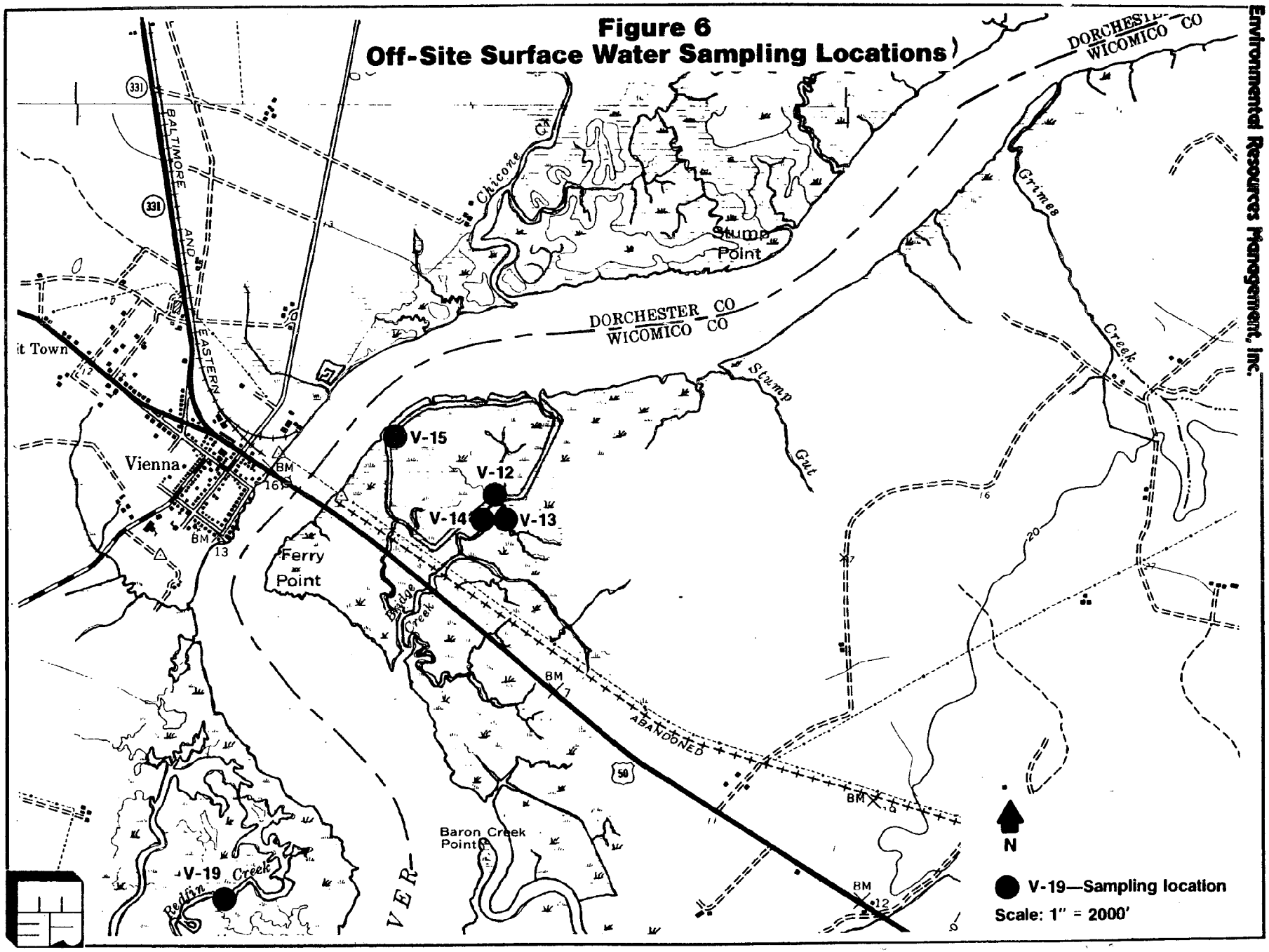
On October 6, samples Ph-6 and Ph-7 were collected near the pipe terminus at location VS-1. On November 4, 12, and 19, one set of ten fly ash samples was collected at locations VS-1, VS-3, VS-4, VS-16, and VS-17 in the dry area and at VS-8 and VS-10 in the marsh area. These locations are shown in Figure 7. At location VS-1, sample VS-1A was taken from the surface, sample VS-1B from a depth of 12-18 inches, and VS-1C from 18-24 inches. At VS-3, sample VS-3A was taken from the surface, and VS-3B from a depth of 14-28 inches. The other samples were taken from the surface. The ash samples at VS-1, 3, 4, 16, and 17 were taken with the bucket auger; the others were dug with a shovel.

4. Sample Preparation and Analyses

All of the water samples were kept on ice, and within 24 hours of collection, one of the two 500 ml portions of each sample was prepared for metals analysis. Each of these samples was filtered through a 0.45 micron filter and fixed with 5 ml of nitric acid.

Fly ash samples Ph-6 and Ph-7 were sent to Lancaster Laboratories, Inc., Lancaster, Pennsylvania, where they were analyzed for total aluminum, arsenic, chromium, and selenium. The samples were prepared by acid digestion according

Figure 6
Off-Site Surface Water Sampling Locations



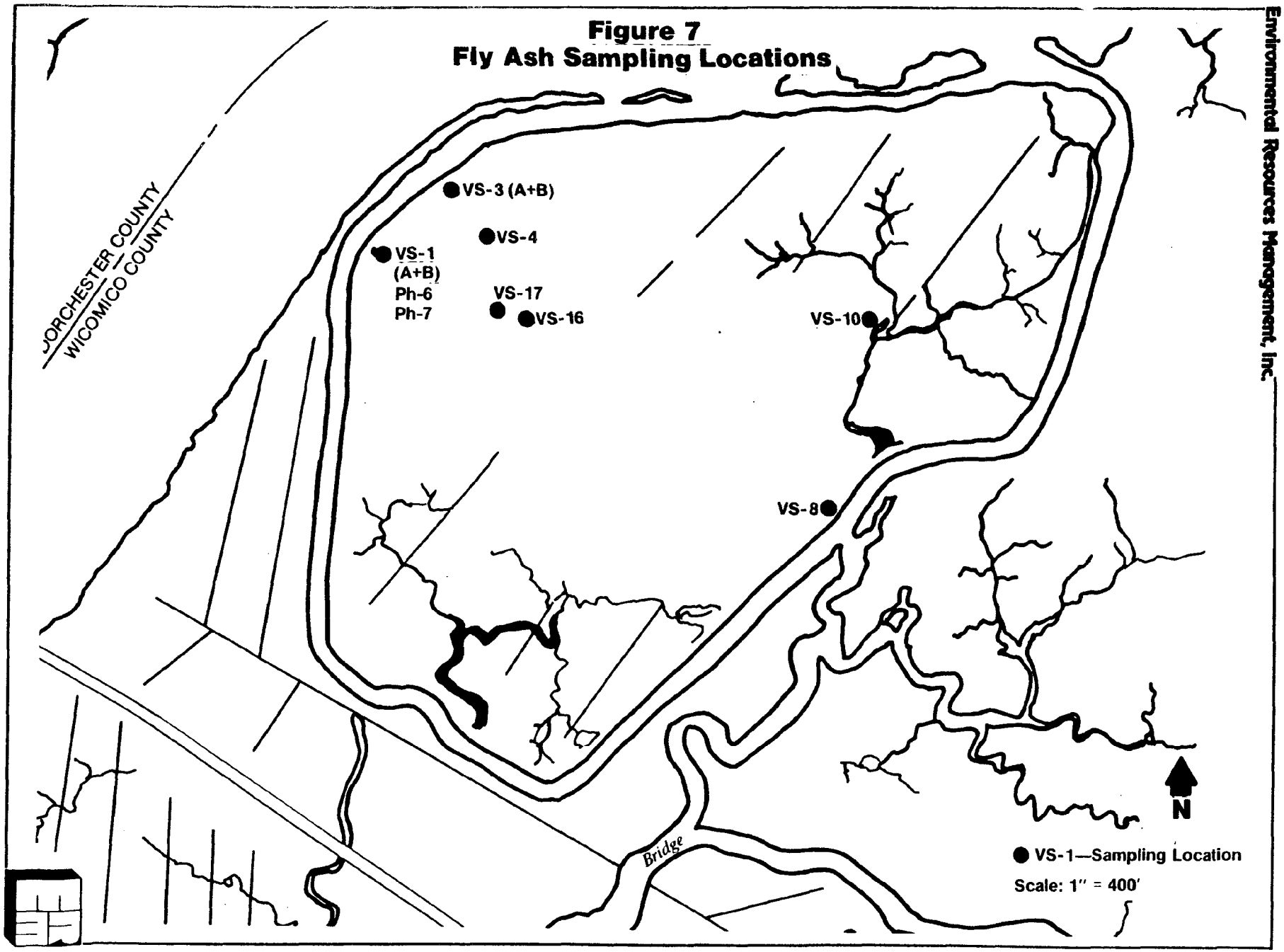


Figure 7
Fly Ash Sampling Locations

to the American Association of Analytical Chemists (AAOAC) Method 2505 (AAOAC Manual, 12 edition) and the Parr Bomb procedure (Journal of AAOAC, Vol. 55, 1972, p. 741). The other fly ash samples were split, and a sample, labeled VS-18, was composited from them. All of the fly ash and water samples were sent to Lancaster Laboratories for analysis.

In order to characterize the mobility of chemical constituents from the fly ash, and its potential environmental effects, ground water sample V-1 and composite fly ash sample VS-18 were subjected to detailed analysis. The remaining samples were held pending the results of these analyses. Sample VS-18 was leached, and the leachate analyzed according to U.S. EPA EP Toxicity determination procedures. This was done in order to leach the sample at low pH conditions to place into solution any potentially mobile heavy metals. Both VS-18 and ground water sample V-1 were analyzed for pH, total dissolved solids (TDS), chloride (Cl), sulfate (SO_4), arsenic (As), barium (Ba), beryllium (Be), boron (B), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), lead (Pb), mercury (Hg), nickel (Ni), selenium (Se), silver (Ag), and strontium (Sr). After the results of these analyses were reviewed, the balance of the samples were analyzed as follows:

- Samples V-3, V-6, V-11, V-13, and V-19 - analyzed for pH, TDS, SO_4, Cl, Ag, As, Ba, Be, Cd, Cr, Cu, Fe, Pb, Hg, Se, and Sr.

- Samples V-2, V-4, V-5, V-7, V-9, V-12, V-14, V-15 - analyzed for pH, TDS, Cl, SO_4, As, Ba, Cr, and Pb.

- Samples VS-1A, VS-1C, VS-3A, VS-3B, VS-8, VS-10 leached according to EP Toxicity Methods, and the leachates analyzed for pH, TDS, Cl, SO_4, As, Ba, Cd, Cr, Pb, and Se.

C. Results and Discussion

1. Fly Ash Distribution

The general distribution of fly ash inside the berm which surrounds the site is shown in Figure 8. The fly ash spread out across the site from the northwest corner area, where the slurry pipeline ended. It now forms a broad band trending southeast across the site to the eastern berm. A 5 to 10 inch deep root mat has formed in the fly ash, and a 1 to 2 inch thick organic layer occurs on top of most of the fly ash. The organic layer thins to nothing to the north and east. The fly ash thickness appears to be fairly uniform, at about 1 foot, except for a limited area near the end of the slurry pipeline. In this area the ash is about 3 feet thick. The fly ash deposit near the pipe terminus creates a dry area about 7 acres in extent (Figure 5). The fly ash deposit thins eastward as shown by the isopach contours in Figure 8. The entire area of concentrated fly ash deposit covers about 40 acres. With the thickness averaging about 1 foot, there are approximately 65,000 yd^3 of fly ash in the band.

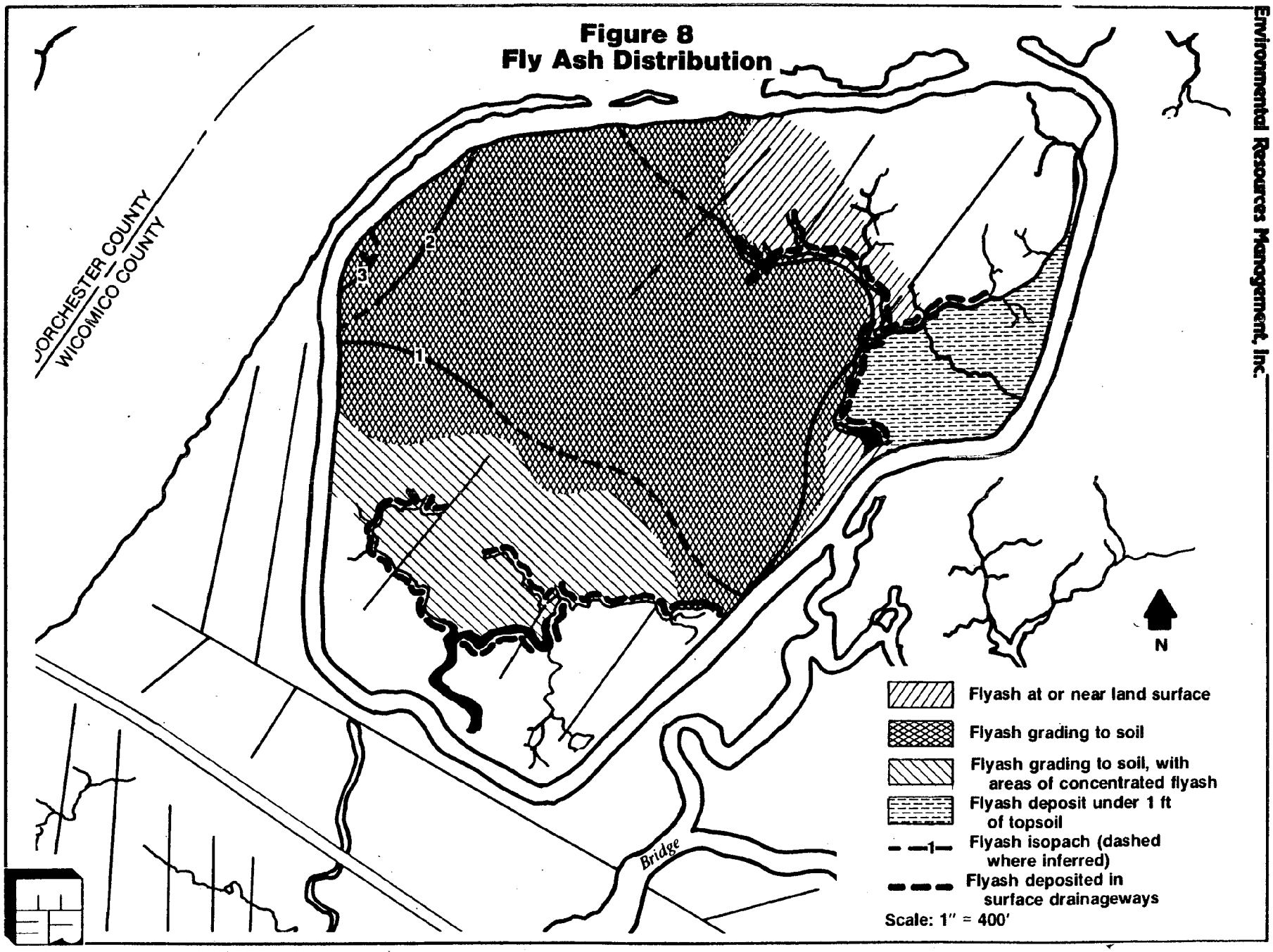


Figure 8
Fly Ash Distribution
DORCHESTER COUNTY
WICOMICO COUNTY
1
2
3
Bridge
N
Flyash at or near land surface
Flyash grading to soil
Flyash grading to soil, with areas of concentrated flyash
Flyash deposit under 1 ft of topsoil
Flyash isopach (dashed where inferred)
Flyash deposited in surface drainageways
Scale: 1" = 400'

To the northeast, the fly ash extends to the main channel of the internal site drainageway. Northeast of the stream, the fly ash is present only as a fine sandy fraction in the silty soil, quickly diminishing in percentage to the northeast. The bed of the stream consists of a thick deposit of fly ash and some bottom ash (gravelly). The thickness of this fly ash, coupled with its absence immediately to the northeast, indicates that the movement of fly ash to the northeast terminated at the stream. The fly ash fraction in the soil northeast of the stream was probably deposited during flood stages of the stream.

The small eastern piece of this northeastern section of the site, east of the main stream channel and south of the westward flowing tributary, contains a layer of fly ash buried under approximately 1 foot of silt.

In the southern part of the site, the fly ash deposit extends to the two east-west tributaries which joined to form the main channel of the south flowing original drainageway. In this section, the fly ash deposits are erratic, with fly ash present in some areas and absent in others. This is the result of erosion of the original fly ash deposit which was carried southward, into the site streams. These stream beds contain fly ash deposits, often buried under silt.

2. Surface Drainage and Topography

Before construction of the berm and the surrounding moat and disposal of the fly ash, the site was drained by two south-flowing and one east-flowing tributaries to Bridge Creek

(Figure 2). Only minor surface drainage had developed northward and westward, toward the Nanticoke. This unusual drainage pattern can also be found in other tidal marshes occupying the insides of meanders on the Nanticoke. It appears that this drainage pattern is superimposed by the incoming tides, as the tidal marsh silts are deposited.

From comparison of pre-disposal and post-disposal aerial photographs (as depicted in Figure 2 and Figure 3), it can be seen that construction of the disposal site and deposition of the fly ash caused considerable changes in the site drainage. The construction of the berm blocked off all three drainageways. The disposed fly ash filled in the stream channel of the eastward flowing stream at the center of the site and small headwater tributaries of the other streams. The main streams were also partially filled with fly ash. The fact that the two present stream channels closely resemble the original ones indicates that these channels were not completely filled by fly ash. When the berm was breached, the incoming tides followed the original stream channels once again, and the original drainage pattern was maintained. The breach of the berm at the mouth of the northernmost site drainageway allowed the natural drainage to be reestablished. However, the southern stream is still cut off by the berm. It now discharges through breaches in the eastern berm, through what were once its eastern headwater tributaries.

The disposal of fly ash created a topography which includes a higher seven-acre dry area in the northwestern corner of the site (Figure 5). From this high area, the topography

slopes gently downward in all directions into the lower swampy area. The surface drainage flows northeast and southwest from a central divide. The total relief on the site is about 5 to 10 feet.

The amount of erosion evident in the southern part of the site indicates that fly ash has been leaving the site via the stream. Tidal flushing through the breaks in the berm allows fly ash to be carried in suspension into the moat. This phenomenon has also been occurring in the northern drainageway, where small tributaries to the main channel from the central section of the site are cut into fly ash. The fly ash deposit was originally thicker in this northern area and more continuous than elsewhere on the site.

The extent of deposition of fly ash in the moat and connected surface drainageways is unknown. A fly ash deposit a few inches thick was observed in the mudflat beside the eastern berm during low tide. The black, granular character of the fly ash is very distinct from the brown silt of the mudflat, rendering such distinction possible.

3. Stratigraphy

The shallow stratigraphy at the site was determined from the auger borings. Logs of these borings are included in Appendix D. Eighty percent of the surface of the disposal area is covered by a fly ash deposit up to 3 feet thick. This deposit is completely overgrown with a stand of *Phragmites*, with a mat of organic matter at the soil/air interface. Under this fly ash deposit lies a buried marsh mat sequence, with thin interbeds of running silt. The marsh mat is composed of

dense, decaying organic matter in a matrix of clayey silt. The organic fraction ranges from greater than 50 percent near the surface to absent at a depth of about 9 to 12 feet. Approximately 7 feet of dark brown silt, containing no organic fraction, underlies the swamp mat. This unit is extremely unstable and precluded the installation of well points in most of the borings. In boring V-1, some plastic clay and clayey silt interbeds were encountered, which allowed the installation of a well point. Under the silt is a unit of subangular medium to coarse grained poorly sorted, light colored quartz sand, with some fine non-quartz gravel. Near the contact with the silt, the sand unit is very silty. The silt fraction disappears quickly downward, leaving the sand very clean. By comparison with boring logs from the Vienna power plant area across the Nanticoke River (Delmarva Power, 1979), this unit has been identified as the Beaverdam facies of the Pleistocene Salisbury Fm.

The dense root mat unit is essentially "floating" on the runny silts. During the installation of the hand auger borings, the augering activity could be felt underfoot as the marsh mat moved. The running silts were forced upward in the auger borings by the weight of the overlying marsh mat. Log decriptions of all borings are included in Appendix D. Figures 9 and 10 show a geologic cross section along the axis of the fly ash deposit.

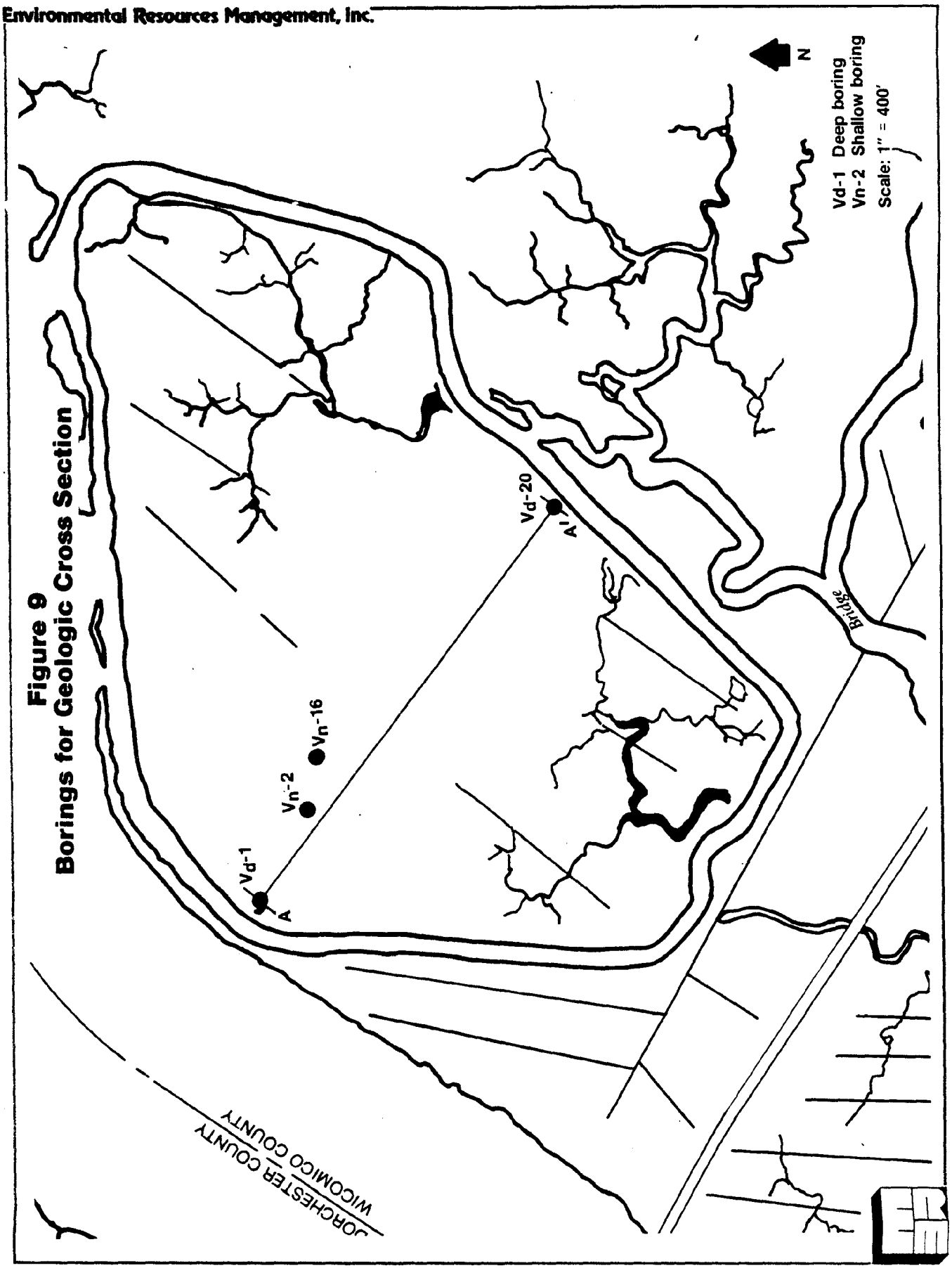
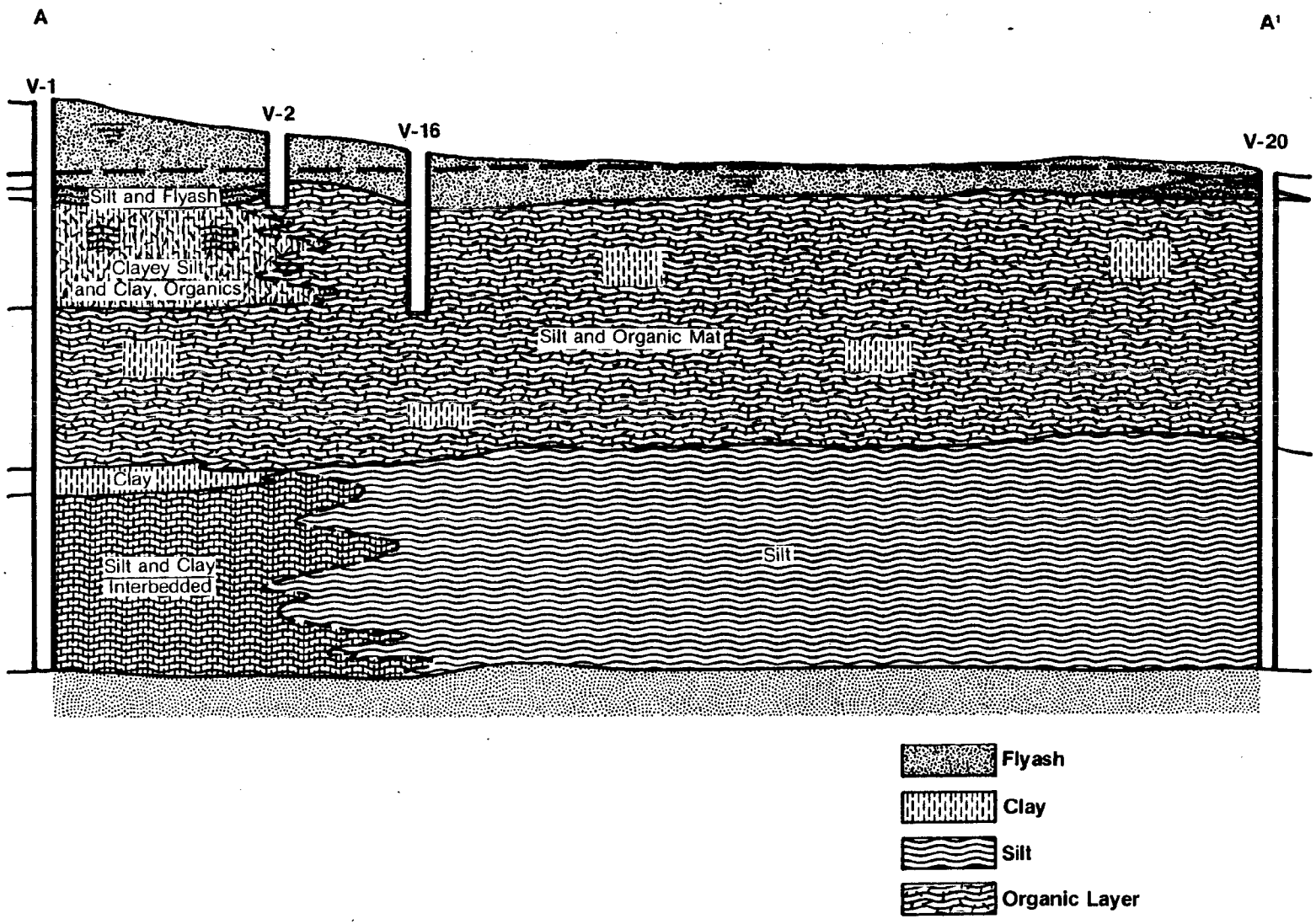


Figure 9
Borings for Geologic Cross Section

Figure 10
Geologic Cross Section—
Vienna Fly Ash Disposal Site



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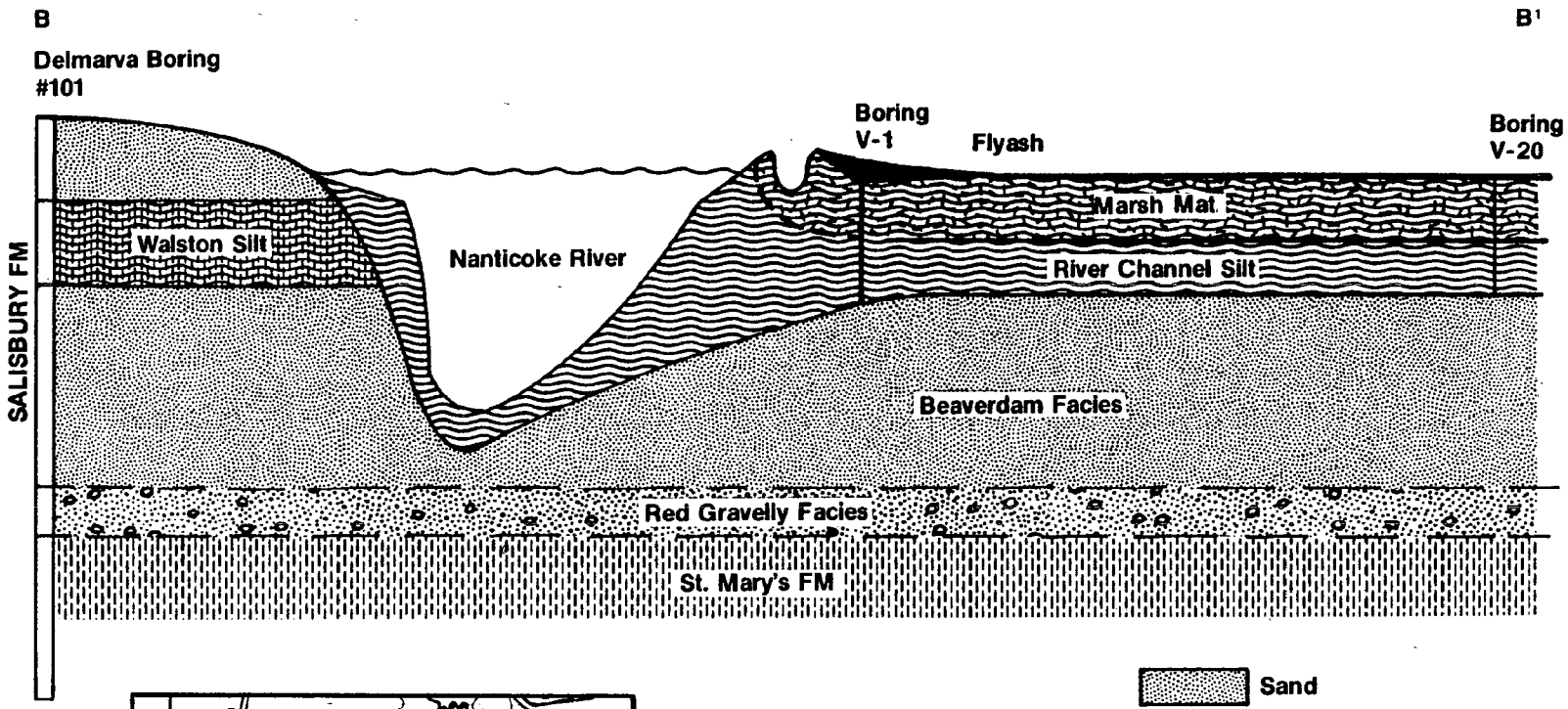
4. Hydrogeology

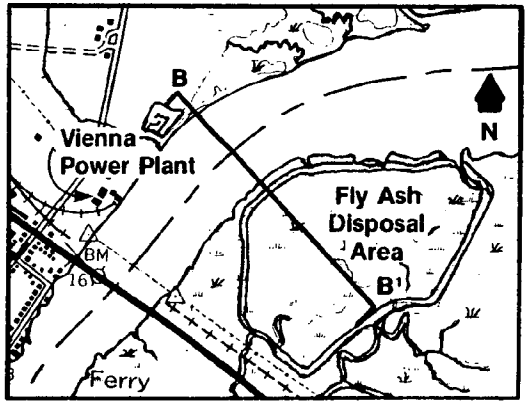
Because it was not possible to install well points and piezometers throughout the site, specific hydrogeologic conditions could not be quantified. However, certain significant conclusions can be drawn, based on the results of the hydrogeologic studies which were conducted across the river at the Vienna power plant (Geraghty and Miller, 1980) and on observation of conditions at the fly ash disposal site.

The disposal site occupies a tidal marsh which is connected hydraulically to the river and is a zone of ground water discharge. Ground water in the Pleistocene Salisbury aquifer is moving from east of the site toward the river and discharges to the river and the tidal marsh. It is probable that this ground water enters the silt deposits beneath the tidal marsh, eventually discharging to the river, mixed with tidal inflow waters. A generalized geologic cross section of the disposal site, the river, and the western bank of the river near Vienna is shown in Figure 11.

Studies at the Vienna plant site determined that the ground water in the Pleistocene aquifer tends to move horizontally toward the river (Geraghty and Miller, 1980). This is reflected in the piezometric surfaces at various depths within the aquifer. There is a slight downward head separating these piezometric surfaces, but not significant enough to effect detectable downward vectors in the vertical flow net. Slight deflection of the vertical piezometric isopleths shows a minor downward flow vector from the sands at the surface through the Walston Silt, wherever it is present. This is due to the effect of the Walston confining the underlying Beaverdam Sands in some areas. At the Vienna plant

Figure 11
Generalized Geologic Cross Section— Vienna, Maryland; Nanticoke River & Marsh System





Sand
Sand and gravel
Clay
Silt
Organic Layer

Scale: Horizontal 1" = 400'
Vertical 1" = 25'

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site, the Walston Silt occurs between elevations 0+ feet and about -25 feet. Its discontinuous areal distribution and variation in thickness do not allow it to be projected across the river without boring logs. Therefore, it is unknown whether this unit is present beneath the disposal site. Since the unit occurs mostly above -20 feet in elevation, it may be absent under the site due to either non-deposition or river erosion.

Whether leakage occurs from the Salisbury Fm. to the underlying Calvert and Piney Point aquifers is unknown. However, the presence of the thick St. Mary's Fm. aquiclude precludes major recharge of these aquifers from the surface. Therefore, local surface influences on ground water quality are unlikely to be reflected in the water quality in the deeper aquifers.

It is doubtful that the runny silt beneath the disposal site confines the underlying Beaverdam sands. It is probable that the Beaverdam ground water discharges to the tidal marsh, or that the marsh and the Beaverdam act as one unit in hydraulic equilibrium.

Since the disposal site is on a tidal marsh, the surface of much of the site is at the level of the ground water table. The dry area built up by the fly ash disposal, and other small dry areas interspersed over the site, are the only areas where ground water and surface water are not contiguous. The surface streams draining the site might suggest a ground water system with substantial discharge within the site confines. However, these streams appear to be tidal features rather than ground water discharge areas. At low tide, these streams are reduced to a trickle. Since these

streams are shallow (about 1-4 feet deep), they are developed in the swamp mat unit. Any ground water discharge from this dense unit would be very slow, as is evidenced by the failure of a ground water discharge to support low tide flows in the site streams. Therefore, the water in the streams can be considered to reflect mostly tidal inflow, with a minor contribution of waters originating on site.

Water level measurements taken in Well V-1 confirm that the ground water levels in the disposal areas are influenced by the tidal action. A water level measurement taken at high tide was 0.75 inches higher than one taken several hours later, as low tide approached. This indicates that the water beneath the marsh surface experiences regular ebb and flow as the tides change. It is not known how deep in the ground water system this effect persists. In the shallow ground water, the effect is probably a lengthened residence time of non-tidal waters in and near the fly ash.

Ultimately, the ground water flowing through the site discharges to the Nanticoke River either directly or via the moat. This discharge occurs principally through the silts directly to the river, rather than via the on-site surface streams.

Of the precipitation which falls on the 40 acres of the site where the main fly ash deposit lies, most of this is lost by runoff and evapotranspiration in the wet marsh area. The runoff has limited contact with the fly ash due to the presence of the overlying organic mat. That water which does recharge the limited dry area as ground water moves through the fly ash in the shallow saturated zone, and discharges radially outward from the dry area to the marsh.

5. Fly Ash Analysis

Results of acid digestions of fly ash samples from the area of the slurry pipe terminus are presented in Table 4. These analyses show that the fly ash in this area contains Al, As, Cr, and Se. The average Cr and Se levels here are close to average levels for fresh fly ash from 23 locations in the United States (Furr et al., 1977), as illustrated below.

Element	Literature Review Fresh Fly Ash Average	Vienna, Maryland Site Fly Ash Average
As	115.8	69.4
Cr	131.7	147.5
Se	8.4	5.9

The As levels in the fresh fly ash study showed a much greater variation around the mean than did the chromium and selenium levels. Therefore, the current arsenic level at the site may be due more to a low original concentration than to leaching. Overall, the site fly ash samples contained moderate concentrations of metals, despite the age of the fly ash and its exposure to leaching conditions.

The results of the fly ash leaching tests are presented in Table 5. Since a major concern is the effect of the disposal site on potable ground water, the analyses are compared to the U.S. Environmental Protection Agency's Interim Primary Drinking Water Standards (PDWS) and Secondary Drinking Water Standards (SDWS). Composite sample VS-18 did not leach any significant concentrations of TDS, Cl, SO_4, or metals. However, the initial pH was less than 5, and the sample required no addition of acid for the EPT analysis. Of the

Table 4

Concentrations (ppm) of Trace Metals in Upper 15 cm of Ash Material from Vienna, MD (Stations Identified in Figure 14)

Station	Al	As	Cr	Se
Ph-6	20,000	10.4	93	1.74
Ph-6	9,700	10.2	175	1.6
Ph-7	57,300	84.1	149	8.41
Ph-7	53,000	75.0	128	6.88
Mean	35,000	44.9	136	4.67

TABLE 5
FLY ASH LEACHING AND PCB ANALYSIS RESULTS
(Results in mg/l Unless Otherwise Noted)

Sample No.	Date	PCB	0.1(%)	TDS	Cl	SO_4	Ag	As	B	Ba	Be	Cd	Cr	Cu	Fe	Hg	Ni	Pb	Se	Sr
VP-1	11/4/81	< .2	.05																	
VP-2	11/4/81	< .2	.04																	
VP-3	11/19/81	< .5																		
VS-1A	11/12/81			158	1	84		<		<		<	<					.1*	<	
VS-1C	11/12/81			2080*	19	1020*		.009		<		<	.07*					<	<	
VS-3A	11/19/81			363	73	33		<		<		<	<					<	<	
VS-3B	11/19/81			251	52	72		<		<		<	<					.15*	<	
VS-8	11/19/81			337	153	39		.056*		.1		<	<					<	.007	
VS-10	11/19/81			597*	191	32		.005		<		<	<					<	<	
VS-18	Composite			183	56	46	<	<	<	.1	<	<	<	<	<	<	.2	<	<	.4
Detection Limits				na	na	na	.01	.004	40	.1	.005	.005	.05	.03	.05	.001	.05	.05	.004	na
Primary Drinking Water Standard (PDWS)							.05	.05		1.0		.01	.05			.002		.05	.01	
Secondary Drinking Water Standard (SDWS)				500	250	250														

< = less than detection limit
Blank = not run or no EPA standard
* = in excess of PDWS or SDWS

metals, Ni, Sr, and Ba leached in detectable, but insignificant, amounts. Only Ba appears on the list of Primary Drinking Water Standards used by the Environmental Protection Agency (Table 5). The 0.1 mg/l in the fly ash leachate is well below this standard.

The individual fly ash sample leachings showed a variety of results. Sample VS-1A leached 0.10 mg/l of Pb, twice its PDWS. Sample VS-1C leached 2,080 mg/l TDS, 1,020 mg/l SO_4, 0.009 mg/l As, and 0.07 mg/l Cr, all except As in excess of their standards. Sample VS-3A leached moderate amounts of TDS and Cl only. Sample VS-3B leached low TDS, Cl, and SO_4, and 0.15 mg/l Pb. Only the Pb concentration is in excess of its standard. Sample VS-8 leached moderate to low TDS, Cl, and SO_4. Arsenic was slightly above and the Se just below PDWS. Sample VS-10 leached 597 mg/l TDS, slightly in excess of SDWS. Chlorides were moderate, and As was detectable but below the PDWS. The pH's on all samples were below the minimum SDWS, with four below pH 5.

The leachability of various heavy metals from fly ash is well documented in the literature (Roy et al., 1981). Maximum leaching of many metals occurs with lower pH conditions. Thus, acidic fly ashes will tend to release metals more readily than neutral or alkaline fly ashes. The 1980 Department of Energy/ASTM study showed that leachates made acidic by the ASTM-B and EPT test methods leached As, Cr, Pb, Se, and Cd, sometimes at levels exceeding their PDWSs.

The pH levels for different fly ash leachates vary greatly (Roy et al., 1981). However, studies to date show that the

pH of alkaline fly ash leachates generally decrease significantly with age, as the calcium is leached. The low pH results for the site fly ash leachings indicate that the original fly ash was probably a neutral to moderately acidic material. Thus, the low pH conditions were probably established early in the life of the site.

The low pH of the study site fly ash leachate creates favorable conditions for the mobilization of metals. Yet the leachate analyses detected only very low levels of a few metals. Therefore, it appears probable that the fly ash has been leached of most of its available soluble metals in the years since disposal ended. This is supported by the absence of Fe in the composite sample VS-18, since Fe would conversely be expected in mg/l concentrations in fresh fly ash leachate. Since the leaching of metals appears to have progressed so far, the mobile species of metals originally present cannot be determined. Arsenic seems to be the principal metal currently leaching from the fly ash, albeit at generally low levels. From comparison of the acid digestion results (Table 4) and the leaching results from VS-1A and VS-1C, it is noted that only very limited quantities of the As, Cr, and Se remaining on site in the fly ash are mobile from the fly ash via solution in natural waters.

The leaching of Pb from fly ash Samples VS-1A and VS-3B did not appear to establish a meaningful pattern. VS-1A was collected from above the water table, and VS-3B from below it. In each case, a fly ash sample taken from the same boring did not leach detectable Pb. Therefore, it is concluded that these results represent localized occurrences of

fly ash containing leachable Pb. Other such areas may exist on the site, and Pb may have been a significant site contaminant in the past.

Sample VS-1C leached substantial concentrations of TDS and SO_4. These constituents commonly leach in concentrations in the thousands of mg/l from fresh fly ash. Since sample VS-1C was collected from a depth of 2 feet in the area of thickest fly ash deposition, the leaching test probably indicates that some constituents have been carried from the upper fly ash into the VS-1C horizon. Also, since this sample was collected above the ground water table, lack of constant contact with ground or surface water has resulted in less leaching of this horizon. For comparison, even though Sample VS-3B was collected from near the 2 foot depth, it is still low in leached constituents. This sample was in constant contact with the ground water, unlike VS-1C. Thus, the age of the site fly ash is attested to by the generally moderate to low levels of these parameters in the leachates from six of the seven site samples. Most of the fly ash deposit is in constant or almost constant contact with ground and surface water and, therefore, has been leached of most constituents.

6. Background Water Quality

Since the study site is located in a tidal marsh, background water quality reflects brackish estuarine conditions. Surface water Samples V-13 and V-19 (Table 6) show typical background values to be about 6,000 mg/l TDS, 3,200 mg/l Cl, and 400 mg/l SO_4, with detectable Ba, Fe, and Sr in both and detectable Ag in V-19 and Cu in V-13.

TABLE 6
SURFACE WATER QUALITY ANALYSIS RESULTS
(Results in mg/l, Except pH in pH Units)

Sample No.	Date	TDS	Cl	SO_4	Ag	As	Ba	Be	Cd	Cr	Cu	Fe	Hg	Pb	Se	Sr	pH
V-5	11/19/81	5970	2950	384		.005	.3			<				<			6.60
V-6	11/19/81	6510	3200	415	.01	<	.2	<	<	<	.05	.21	<		<	1.4	6.61
V-9	11/19/81	5750	3300	368		<	.3			<				<			6.63
V-11	11/19/81	6280	2700	379	.01	<	.3	<	<	<	<	1.04	<	<	<	1.4	6.54
V-12	11/19/81	7690	3550	477		<	.2			<				<			6.92
V-13	11/19/81	6250	3200	409	.01	<	.2	<	<	<	.05	.13	<	<	<	1.4	6.67
V-14	11/19/81	6530	3500	464		<	.2			<				<			7.02
V-15	11/19/81	6690	3450	446		<	.2			<				<			6.9
V-19	12/10/81	6040	3200	421	.01	<	.2	<	<	<	<	2.49	<	<	<	1.4	7.17
PDWS					.05	.05	1.0		.01	.05			.002	.05	.01		
SDWS		500	250	250													6.5-8.5

< = less than detection limit

Blank = not run

* = at or in excess of PDWS or SDWS

Note - all TDS, Cl, SO_4 in excess of SDWS, due to estuarine environment

Chemical equilibrium in sea water maintains an SO_4 to Cl ratio of about 0.13 (personal communication, Lancaster Laboratories). The values for V-13 and V-19 are 0.128 and 0.132 respectively, which reflects a sea water chemistry. This ratio is used for comparison of the water at the disposal site with background measurements from off-site marshes because, being an equilibrium condition, it should vary less between ebb and flow tidal stages than the individual parameters would. Heavy metals analyses revealed consistent small background levels of Ba (0.2 mg/l), Ag (0.01 mg/l), and Sr (0.1 mg/l). Iron concentrations were variable.

The area of the study site is all in tidal marsh, which exhibits mixing of tidal estuarine waters with ground water. The deposition of the fly ash created some dryer land, in which actual ground water could be defined. However, the rest of the tidal marshes are surface water areas, where no shallow ground water exists separately from the surface waters. Therefore, no background ground water conditions were determined.

7. Ground Water Quality

In general, the ground water quality at the site reflects a combination of fresh ground water discharge from the Salisbury Fm., estuarine tidal waters, and fly ash leachate. The ground water analyses from the most concentrated area of fly ash deposit, in the dry area, reflect the leaching of the fly ash (Table 7). The pH values range from 3.06 to 5.88, in contrast to the surface water pHs of 6.6-7.02. These pH

TABLE 7
GROUND WATER QUALITY ANALYSIS RESULTS
(Results in mg/l Except for SpC in Micromhos and pH in pH Units)

Sample No.	Date	SpC	TDS	Cl	SO_4	Ag	As	B	Ba	Be	Cd	Cr	Cu	Fe	Hg	Ni	Pb	Se	Sr	pH
V-1	11/19/81		2480	1150	711	<	.011	<	.3	.3	<	.05*	.03	210	<	<	.23*	<	1.0	5.84*
V-2	11/19/81	5200	4550*	2250*	423*		<		<			<					<			5.73*
V-3	11/19/81	4700	5660*	1050*	2780*	.01	<		<	.072	.007	.1*	1.02	56	<		.08*	<	6.1	3.06*
V-4	11/19/81	6600	5560*	1850*	460*		.013		.2			<					<			5.88*
V-7	11/19/81	4600	3550*	2150*	191		<		.3			<					<			6.05*
Detection Limit			na	na	na	.01	.004	10	.1	.005	.005	.05	.03	na	.001	.05	.05	.004	na	na
PDWS						.05	.05		1.0		.01	.05			.002		.05	.01		6.5-8.5
SDWS			500	250	250															

< = less than detection limit

Blank = not run or no EPA standard

* = at or in excess of PDWS or SDWS

results are consistent with the leaching results of the fly ash. The TDS, Cl, and SO_4 levels in the ground water samples are very high. These constituents are common in the leachates of most fly ash materials. However, the tidally induced fluctuations in Well V-1 indicate that the ground water is mixed with the estuarine water. Therefore, it is necessary to distinguish the effects of the fly ash from those of the tidal marsh environment. The following observations can be made:

a) The TDS concentrations in the ground water are lower than in the surface water. This reflects a mixing of brackish tidal water with fresh water. The ranges and means are: 2,480 to 5,660 mg/l, mean 4,360 mg/l in the ground water; and 5,970 to 7,690 mg/l, mean 6,412 mg/l in the surface water. The lower TDS values for the ground water would be expected since the precipitation recharge to the ground water in the dry area would have very low TDS concentrations, as would any discharge from the Pleistocene aquifer sands below.

b) The Cl concentrations in the ground water are lower than in the surface water; the ranges and means are: 1,050 to 5,660 mg/l, mean 1,690 mg/l in the ground water; and 2,700 to 3,550 mg/l, mean 3,114 mg/l in the surface water. Again, this is a reflection of dilution by precipitation recharge and possibly by Pleistocene ground water discharge.

c) The SO_4 concentrations in the ground water vary greatly, ranging from 191 to 2,780 mg/l, with a mean of 913 mg/l. The high values are from samples taken under the dryer part of the site and from areas where fly ash is thickest, at V-1 and V-3. The lowest value is from location V-7, where most of the fly ash has been eroded away. The SO_4 concentrations in the surface water samples range from 368 to 477 mg/l, with a mean of 418 mg/l. The SO_4/Cl ratio mean is 0.76 in the ground water samples. This is heavily weighted toward the SO_4 parameter compared to the 0.13 mean for the surface water. Rain water and the water in the Pleistocene aquifer, which constitute much of the site ground waters, are low in sulfates (Delmarva Permit Application, 1979). Therefore, the variation in the range, and the high mean of the SO_4 concentrations in the ground water, are probably due to the fly ash.

The results of the heavy metals analyses indicate that the quality of shallow ground water under the site is slightly affected by the fly ash. Sample V-1 contained Cr at the primary drinking water limit, and Pb slightly in excess of the limit. Several other constituents were detectable in concentrations of relatively minor significance. Sample V-3 contained Cr and Pb slightly above PDWS limits, in addition to other detectable constituents. As with the leaching analyses, the high Pb concentrations were limited to the V-1 and V-3 sampling locations, thus appearing to represent localized areas of Pb containing fly ash. The presence of high concentrations of Fe and Cu at these two locations may

indicate that localized elevated concentrations occur within the fly ash deposit. The other ground water samples contained no metals in significant concentrations, although a small amount of As was detected in V-4.

Comparison of these analytical results with the data on metals acid digested from site fly ash samples (Table 4) indicates a residuum of elevated levels of immobile As, Cr, and Se in the fly ash, with very low to undetectable concentrations in the ground water. Since the low pH conditions in the ground water should favor mobilization of metals, it is apparent that the mobile portion of the metals in the fly ash has been mostly leached away. The acidity of the fly ash hastened this process. It does appear that moderate levels of sulfur are still leaching from the fly ash as sulfate. This promotes the formation of sulfuric acid, which is probably responsible for the persisting low pH conditions.

Since most of the leaching of the site fly ash occurred in the past, at some time the leachate may have contained elevated concentrations of heavy metals, mobilized under the low pH conditions. Since the tidal marsh is a ground water discharge zone, the leachate is expected to have discharged via the marsh to the surface water drainage, especially during inundations of the site by floodwaters. However, since the leachates are more dense than natural waters (due to the elevated dissolved ion content), a plume of polluted ground water may have migrated vertically from the site, as a result of the density contrast. The contaminant transport model prepared for the proposed fly ash landfill at the powerplant site employs leachate density as a variable parameter (Frind and Palmer, 1980). Results indicated that

leachate density would accelerate vertical pollutant transport in the early years of the landfill. (The model was applied to the sands of the recharging Pleistocene aquifer, not the silts of the tidal marsh discharge area.) This phenomenon appears to have occurred in a ground water discharge area at a site evaluated in Tennessee (Tennessee Valley Authority, 1980). Similar density gradient induced migration at the study site would allow some metals to reach the Pleistocene aquifer. Assuming that flows through this aquifer are similar to those from the power plant site, between 0.25 and 1 foot per day (Geraghty and Miller, 1980), travel time for the eastern edge of the plume to reach the river would be from 5 to 20 years.

The existence of the St. Mary's Fm. aquiclude in this area precludes the vertical movement of significant amounts of any contaminated ground water into the underlying Calvert and Piney Point aquifers. Therefore, the fly ash disposal site poses no threat to any ground water supply wells which might use these deep aquifers.

8. Surface Water Quality

The surface water quality on and surrounding the site is typical of tidal marsh conditions as seen in samples V-13 and V-19. Sulfate to chloride ratios are typical of the .13 ratio of equilibrium conditions in sea water. No surface water sample varied from that value by more than .02, including samples V-5, V-6, V-9, and V-11, which were collected from the on-site streams. This tends to verify the observation that the streams are mainly a tidal feature, with little ground water discharge. It also tends to indicate

that the fly ash in the stream beds has been leached of most of its SO_4, Cl, and TDS by the continuous flushing of the tides.

Small concentrations of heavy metals were detected in two of the on-site samples, V-5 and V-6. A low level of As was found in sample V-5; in V-6, Cu was detected. However, Cu was also detected in background sample V-13. Other metal concentrations detected were insignificant. The off-site samples from the moat and Bridge Creek contained no metals of any significance compared to background. Thus, a small amount of As is seen in the site surface drainage, probably from the contact of the tidal inflow with the fly ash. The levels of metals in the ground water are so low that concentrations in any ground water discharge to the streams would be diluted below detection levels by the tidal waters.

Upon leaving the site, any small amounts of metals leached into surface waters from the fly ash are diluted below the detection limits.

9. Potential Effect of the Disposal Site on Ground Water Development for the Town of Vienna

One of the expressed concerns regarding the fly ash disposal site is whether or not any contaminants reaching the Pleistocene aquifer might be drawn into public water supply wells planned by the Town of Vienna, across the river. Data presented in the 1979 Delmarva permit application address the extent of the effects of developing and using a new 400 gpm

well in the Pleistocene aquifer to supply proposed Unit No. 9 with water. These data were extrapolated to possible usage of ground water by the Town of Vienna. Current water usage in the Vienna area, including Rabbit Town, (excluding any power plant usage) probably averages less than 10,000 gallons per day. Therefore, a public water supply well of approximately 100 gpm would provide adequate service to the town.

The transmissivity of the Pleistocene aquifer is about 55,000 gpd/ft in the area 1/2 to 3/4 mile west of the river, decreasing to about 16,000 gpd/ft near the river. A one year distance/drawdown curve was developed for the proposed 400 gpm Unit 9 withdrawal, based on a transmissivity of 38,000 gpd/ft. (Delmarva Power, 1979). At this withdrawal rate, the projected cone of depression developed a maximum estimated radius of 4,500 feet, assuming no recharge. If recharge is taken into account, the extreme case cone of depression radius of 4,500 feet would not develop at the proposed Unit 9 pumping rate of 400 gpm.

A public water supply well for the Town of Vienna, pumping about 100 gpm, would not develop a cone of influence which would extend the 2,000 plus feet under the river to intercept Pleistocene aquifer ground water under the tidal marsh disposal site. If located very close to the river, decreased aquifer transmissivity might produce a cone of depression which would cause induced infiltration of water from the river. Any heavy metals contribution from the disposal site to the river would be undetectable as a result of dilution with large volumes of river water.

Data available from the Vienna power plant site indicate that the ground water beneath the plant site exhibits elevated values for TDS, sulfate, and some heavy metals (Delmarva Power, 1979). The source of these dissolved constituents was not addressed in the Delmarva Permit Application but may be associated with on-site fly ash disposal and, possibly, on-site coal storage.

SECTION V

BIOLOGICAL INVESTIGATIONS

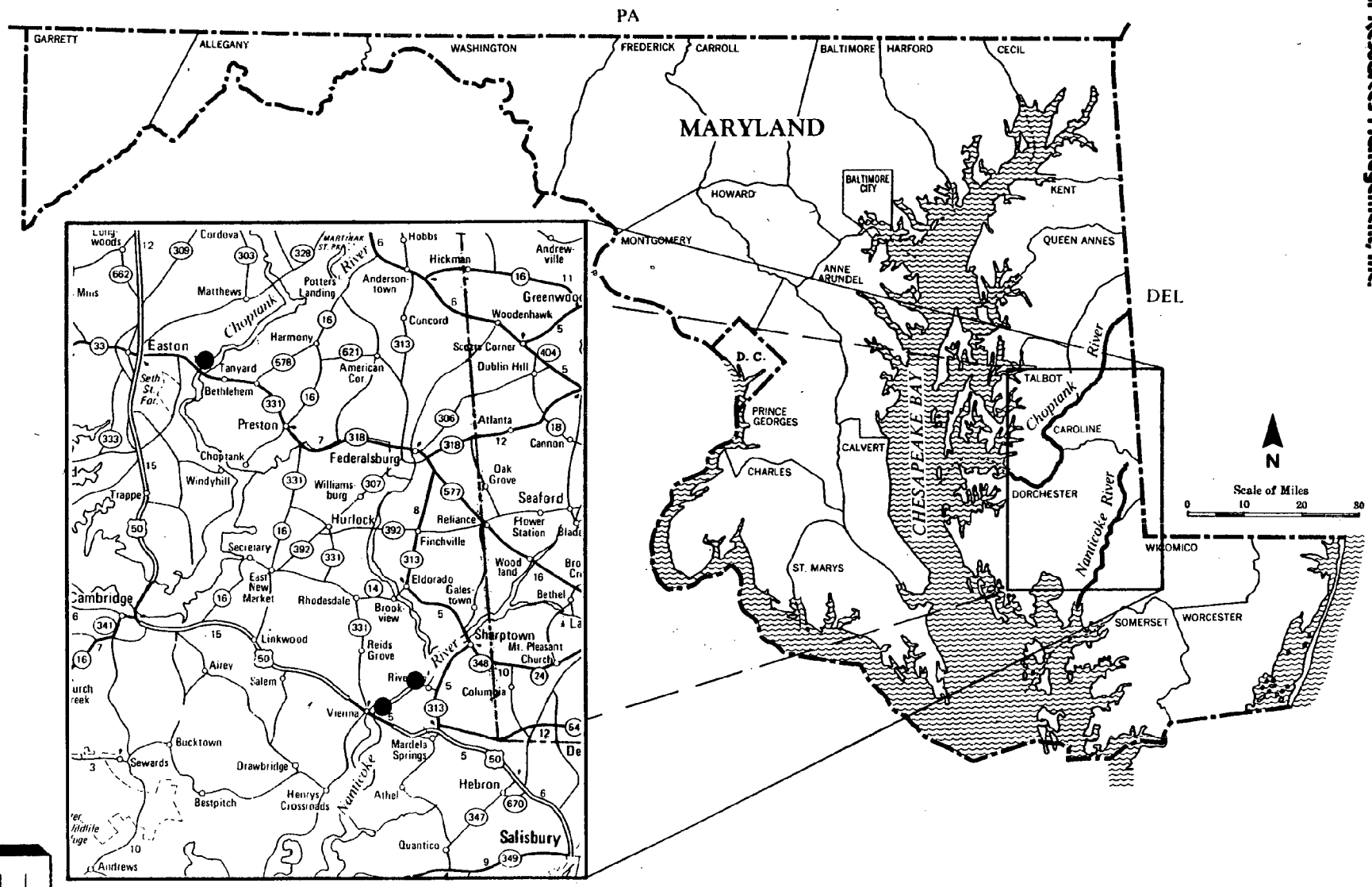
A. Field Methodology

1. Sampling Strategy and Organism Selection

The objective of the sampling design was to sample representatively the disposal site for selected indicator matrices and to be able to compare mean trace metal concentrations with similar data from reference sites. The three most important characteristics in selecting suitable reference locations were: 1) similar salinity regime in the riverine environment, 2) surrounded by a fringing marsh wetland, and 3) area removed from any potential impact of the disposal site. Sampling locations included: 1) the fly ash disposal area, 2) a reference site on the Nanticoke River approximately 2 km upstream of the disposal area, and 3) a second reference area on the Choptank River approximately 1 km above the Dover Bridge (Figure 12).

The use of these two reference locations permitted statistical comparison of disposal site parameters with background data from a site on the same river and with an ecologically similar location on a different drainage. Water chemistry parameters measured at each location to characterize the sites included: pH, salinity, specific conductance, and alkalinity. Because these factors are influenced by tidal activity, temporal variation can occur over a relatively short period. Therefore, the purpose of our measurement during a single point in time was simply to establish that the sites were essentially similar with respect to these parameters. The

Figure 12
Vienna Site Assessment & Control Locations



combined range for each parameter for the three sites is given in Table 8. For the purposes of this study, the three sites can be considered ecologically similar with respect to these basic water quality parameters.

Samples selected for examination included: 1) an aquatic emergent plant, arrow-arum, Peltandra virginica; 2) the predominant semi-terrestrial plant, Phragmites communis; 3) a benthic filter feeding clam, Rangia spp.; 4) an omnivorous fish species, mummichog, Fundulus heteroclitus; 5) a potential food fish species, white perch, Morone americana; 6) sediment samples; and 7) surface water samples. Five sampling stations were chosen for Peltandra, Fundulus, and sediment at each location. Water quality was sampled at four stations at each location. Phragmites rhizome samples were collected from the ash disposal site as this species was found to be growing directly on the ash material. White perch were collected from the Nanticoke River immediately adjacent to the disposal site to investigate trace metal levels in a representative food fish species. Sampling to determine the possible impact of the disposal area on the associated biota was conducted on 21-22 and 28-29 October 1981. Sampling stations at each of the three sites are shown in Figures 13, 14, and 15.

Sampling for Rangia spp. proved unsuccessful. Considerable effort was expended raking the substratum at various disposal site stations in an attempt to collect specimens. Although the species had been reported from the Nanticoke, only a very few specimens were collected from a single location. This lack of sampling success is consistent with the reported unsuitability of the sediment in the area for sup-

Table 8

Combined Ranges of Selected Water Chemistry Parameters as Sampled from the Three Study Locations

pH	Salinity (ppt)	Conductivity (umhos/cm)	Total Alkalinity (mg/l $CaCO$)
6.1 - 7.1	3 - 6	3,500 - 7,000	29.9 - 65.7

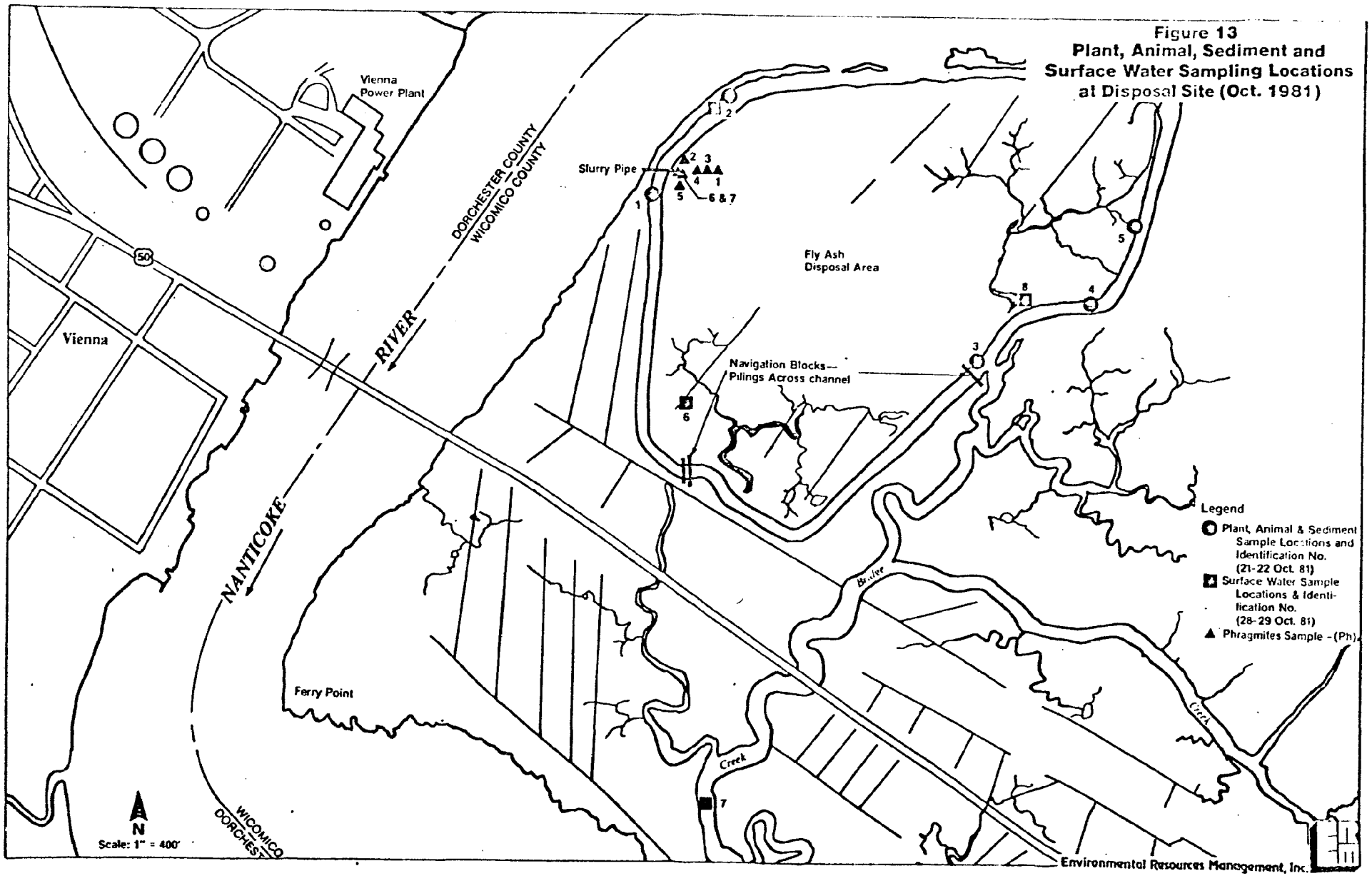


Figure 13
Plant, Animal, Sediment and Surface Water Sampling Locations at Disposal Site (Oct. 1981)

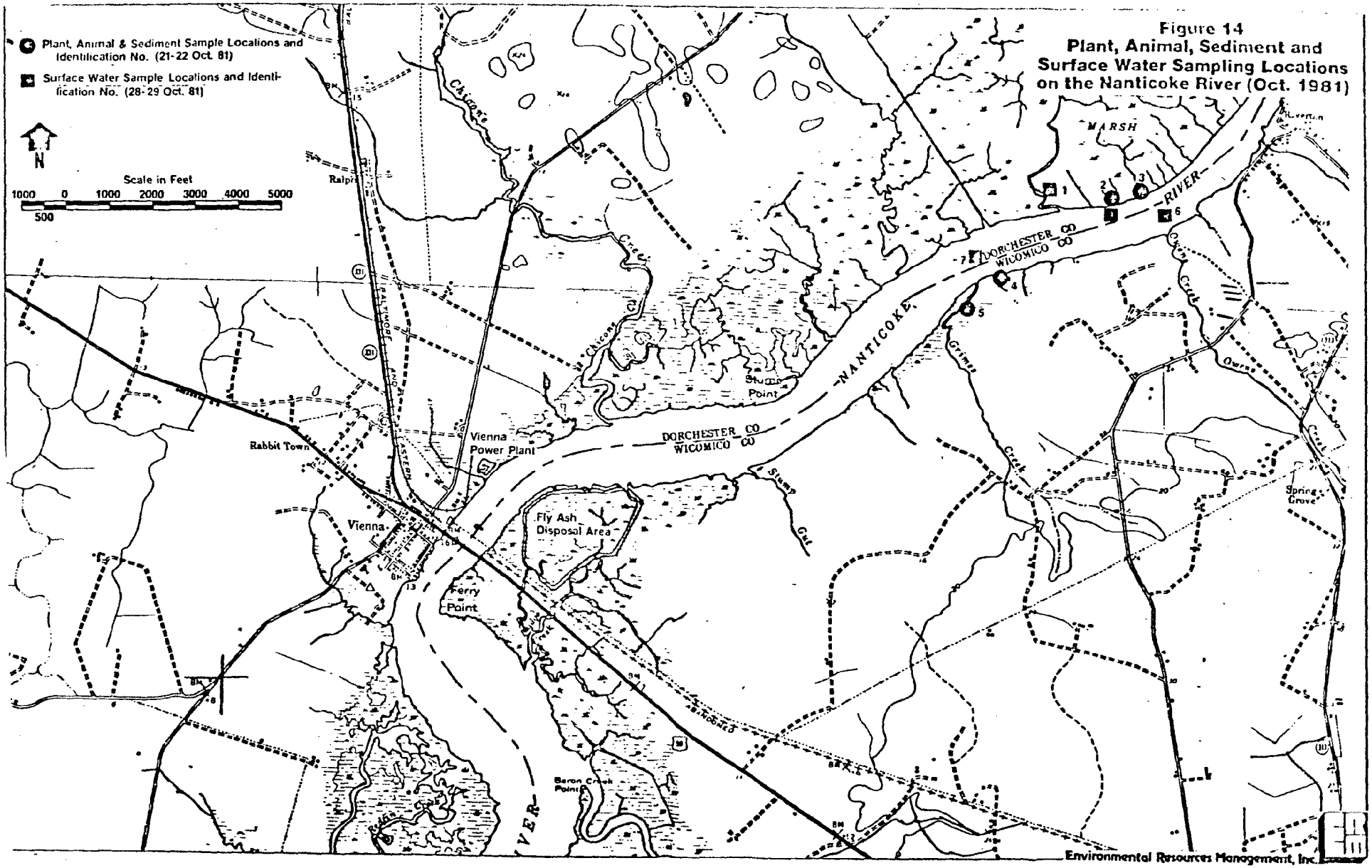


Figure 14
Plant, Animal, Sediment and Surface Water Sampling Locations on the Nanticoke River (Oct. 1981)

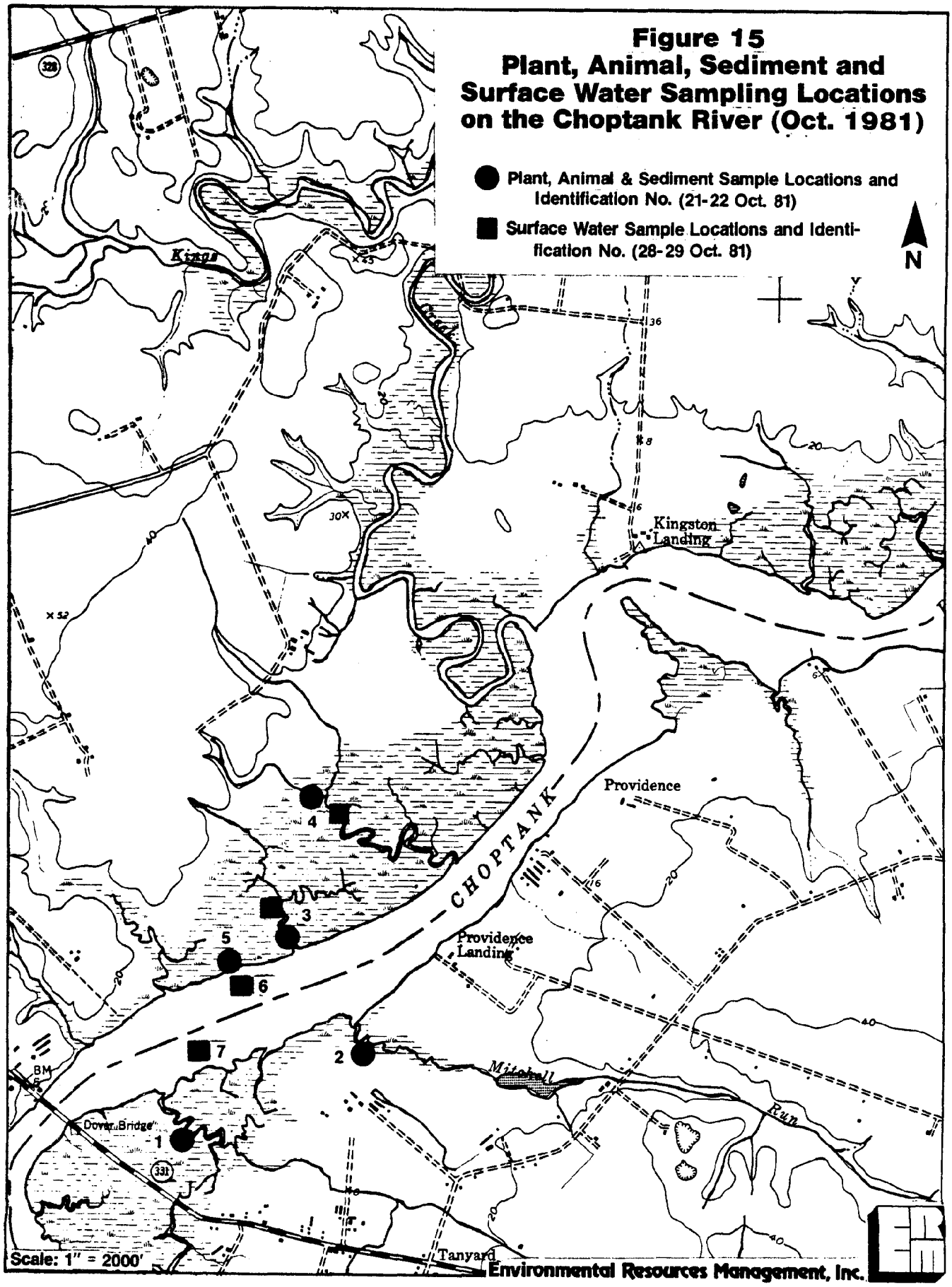


Figure 15
Plant, Animal, Sediment and Surface Water Sampling Locations on the Choptank River (Oct. 1981)

porting macrobenthic organisms (Portner, 1981). Local populations of Rangia are apparently of low density and may be broadly disjunct. Thus, the use of Rangia as a potential indicator organism in this investigation would not be practical. Other pelecypods, such as the blue mussel (Mytilus), do not occur in such low salinity (4-8 ppt) habitats.

2. Collection Program

Dip-netting was found to be an effective collection method to obtain mummichogs (killifish), Fundulus heteroclitus, and was utilized to collect all samples. Baited minnow traps proved unsuccessful. A substantial effort was made to obtain at least five specimens at each sampling station to produce a more representative composite sample; however, this was not always possible. Therefore, the number of specimens comprising a given sample is provided in the RESULTS section.

White perch, Morone americana, were collected only in the Nanticoke River adjacent to the disposal site, using fyke nets. All five specimens were prepared for analysis by removing the head, viscera, and scales to produce a sample representative of fish prepared for human consumption. However, bones were not removed from these small specimens. Scale analysis showed that all specimens were age I+. The use of fyke nets and gill nets at the control locations produced no white perch, possibly due to the fall migration of M. americana out of the sampling areas (Delmarva Power, 1979).

The root mass and fruiting bodies of Peltandra were collected and analyzed to investigate the possibility of organ-specific bioaccumulation. Each sample was composed of a minimum of one complete root mass and at least one dozen seeds from one or more fruiting bodies. The root and seed samples collected at a given station were not necessarily from the same plant because the fruit pods had all ripened and fallen by the sampling date. Therefore, individual seeds were collected, in most cases, within the immediate area of root sampling to obtain specimens produced within essentially the same microhabitat conditions.

Phragmites rhizome samples were dug from the dense root mat established in the fly ash. Each sample consisted of approximately a half meter section of rhizome with associated dormant shoots.

Sediment samples were obtained at each of the five sampling stations at all three locations. A single sample of approximately 0.3 liter was collected from the upper 0.3 m at each station.

Water samples collected from the two control locations included two samples from tributary creeks and two samples from the river (Figures 14 and 15). At the disposal site, samples were collected at four locations from the stream and channel system which drains the site (Figure 13). All water samples were collected at the end of an ebb tide period in order to obtain water which had drained off the adjacent marsh.

B. Laboratory Methodology

Tissue and sediment field samples were placed in zip-lock plastic bags, labeled, and immediately placed on ice. Before sending samples to Lancaster Laboratories, Inc. for analysis, selected larger samples of killifish, arrow-arum roots, and arrow-arum seeds were sorted into two whole-specimen samples and relabeled to obtain an estimate of within station variability. All plant samples were thoroughly cleansed of soil and then rinsed in distilled water before being sent out for analysis. Tissue and sediment samples were dried at 100°C, homogenized, and subjected to a total acid digestion prior to atomic absorption analysis.

Water samples were collected in half-liter polypropylene bottles, placed on ice, and then transported to the University of Delaware, College of Marine Studies (CMS) at Lewes, Delaware. Water samples were acidified, filtered, and analyzed using a Perkin-Elmer 603 atomic absorption spectrophotometer equipped with a graphitic furnace. Multiple injections of samples provided an estimate of analytical uncertainty.

C. Statistical Methods

Statistical analysis involved comparison of mean trace metal concentrations using Wilcoxson's two-sample test for unpaired observations (Steel and Torrie, 1960). This non-parametric test was selected as it was uncertain that the assumptions associated with applying a two-sample t-test (i.e., random samples drawn from normally distributed populations with

equal variances) were sufficiently satisfied. Therefore, the distribution-free statistical test was selected. However, for additional insight, t-tests were also conducted on all comparisons with virtually identical results. A summary of the t-test results is provided in Appendix E.

D. Results and Discussion

1. Mummichogs, *Fundulus heteroclitus*

The results of the trace metal analysis on *F. heteroclitus* collected at all three locations are shown in Table 9. Replicate samples were collected at some sample stations to obtain a qualitative estimate of the variation within a station. The replicate results indicate that substantial variability exists among fish collected in the same microhabitat. Concentrations of Al, As, and Se were significantly greater ($P < 0.05$) in the killifish collected at the disposal site than in killifish collected at the upstream control locations. Mean trace metal concentrations were also higher in disposal site specimens than in specimens collected from the Choptank River location. However, in this comparison, killifish from the disposal site possessed significantly greater ($P < 0.01$) As levels, only. Mean concentrations of the other three metals (Al, Cr, Se) were greater in the disposal site specimens (Table 9), although the differences did not prove significant ($P > 0.05$).

The results indicate that killifish collected at the disposal site had accumulated a significantly higher level of

Table 9

Whole Body Trace Metal Concentrations in Mummichogs, *Fundulus heteroclitus*

Stations	No. of Specimens in Sample (homogenized)	Per Sample Metal Concentration (ppm) Al	As	Cr	Se
V-1	0	-	-	-	-
V-2	2	138	2.2	< 2	1.15
V-3	5	1900	1.8	3	2.93
V-3	5	3150	2.9	4	2.54
V-4	5	1670	2.1	3	2.49
V-5	5	2240	1.5	3	2.28
V-5	5	1350	1.0	2	1.70
Weighted Mean*	27	1920	1.9	< 3	2.30
N-1	5	58	< 0.5	< 1	0.71
N-1	5	38	< 0.5	< 1	1.06
N-2	5	388	0.5	< 1	0.77
N-2	5	68	0.6	< 1	0.90
N-3	5	186	< 0.5	< 1	1.03
N-4	3	65	1.2	4	0.89
N-5	0				
Weighted Mean	28	139	< 0.6	< 1	0.89
Ch-1	5	1860	< 0.5	2	2.27
Ch-2	2	5100	0.8	6	3.19
Ch-3	5	651	0.5	< 1	1.94
Ch-3	5	189	< 0.5	1	1.48
Ch-4	3	697	< 0.5	2	1.79
Ch-5	5	474	< 0.5	1	1.63
Ch-5	5	1200	< 0.5	2	1.64
Weighted Mean	30	1139	< 0.5	< 2	1.89

*Means were numerically weighted by the size of the sample.

trace metals than fish taken approximately 2 km upstream on the Nanticoke River. Furthermore, the differences in metal concentrations were approximately an order of magnitude for Al and As and more than double for Se, resulting in highly significant concentration differences ($P < 0.01$). If it is assumed that the killifish populations are highly localized and, therefore, are primarily influenced by the environmental quality of the immediate area, the results suggest that a localized source of contamination may exist near the disposal site which is affecting this resident fish species. However, compared to the Choptank River specimens, only the As concentration would suggest that a potential source of contamination exists near the disposal site. Although comparison with the Nanticoke reference station specimens suggests a possible pollution source at the disposal location, the greater similarity among other metals between the disposal site specimens and Choptank River reference specimens imposes the need for cautious interpretation of the results. However, there is some indication of a source of contamination on the Choptank River above the reference station which likely influenced the samples (pers. comm. MD Dept. of Health).

2. White Perch, *Morone americana*

The results of trace metal analyses on white perch taken from the Nanticoke River near the disposal site are given in Table 10. Unfortunately, the data cannot be compared to similar information for the two control locations because sampling efforts failed to produce white perch at the controls. Levels of As and Cr were below detectable limits. The Al concentration was also relatively low. The notably high Se

Table 10

Trace Metal Concentrations in
White Perch (Prepared in the Round)
from the Nanticoke River Near Vienna Disposal Site

Metal	Tissue Concentration (ppm)
Al	5
As	< 0.5
Cr	< 1
Se	1.40

value may be largely attributable to the fact that a portion of the skeletal system was included in the tissue sample, and bone is a known accumulation site for trace metals (Dvorak et al., 1978). These results are similar to those reported by Guthrie and Cherry (1976) in examination of stream vertebrates and invertebrates receiving fly ash settling-basin effluent. Mean annual concentrations of virtually all trace elements were reportedly lowest in fish with the exception of Ca and Se. Also consistent with this evidence are the findings of Mehrle et al. (1982) on contaminants of east coast striped bass, Morone saxatilis. Se and As were shown to accumulate in Nanticoke River fish from the riverine environment. Furthermore, Se residues tended to increase with the age of the fish.

This evidence suggests that Morone spp., and possibily other Nanticoke River fishes, are capable of bioaccumulating Se from the riverine environment. However, several factors

The U.S. EPA (1980), recognizing the potential for bioconcentration, utilizes an average daily per capita consumption of 6.5 g of freshwater and estuarine fish and shellfish to estimate daily Se intake. The U.S. EPA also uses a 50 to 200 µg per day dietary intake of Se as being safe and adequate for adults. Using the recommended 6.5 g fish per day consumed, multiplied by the average fish tissue Se concentration of 1.4 µg/g, results in a value of 9.1 µg per day from consumption of Morone spp. captured at the Vienna site. This value is relatively low compared with the recommended EPA guidelines stated above.

Several factors warrant cautious interpretation: 1) the inclusion of bone tissue in our white perch samples prevents comparison to EPA guidelines for safe consumption, 2) our data cannot be compared to results from the two control locations because sampling efforts did not produce the same species (white perch), and 3) the migratory nature of Morone spp. in this area complicates the determination of the source(s) and mechanism(s) of possible contamination.

3. Sediment

The results of the sediment analyses are shown in Table 11. No significant mean concentration differences ($P < 0.05$) were found between the Nanticoke reference site and the disposal site. The Choptank site was significantly higher ($P < 0.05$) in Al than the disposal site, and the disposal site was significantly higher ($P < 0.05$) in As than the Choptank site. The order of elemental abundance in sediment samples was the same for all three sites (i.e., Al> Cr> As> Se) and typical of concentration levels found in endogenous soils (Dvorak, 1978). This order of trace metal abundance was also found in the fly ash from the disposal site (Table 4). However, comparisons of the mean trace metal concentration levels between the fly ash samples and sediment samples from the Vienna disposal site (Table 11) were significantly different ($P < 0.05$) for Al and Cr. The Al concentration was greater in the sediment than the fly ash; conversely, the Cr level was greater in the fly ash. Although the As and Se levels were substantially greater in the fly ash samples, the differences did not prove significant ($P > 0.05$). These differences between the fly ash and sediment at the disposal site suggest that the constituency of the sediment (in the areas sampled) is not predominantly fly ash material.

Table 11

Trace Metal Concentrations in Sediment Samples of Tidal Streams and Disposal Site Moat

Station	Al (% dry wt.)	Metal Concentration (ppm) As	Cr	Se
V-1	6.1	9.05	64	0.6
V-2	6.4	7.84	60	0.62
V-3	5.9	9.7	67	0.37
V-4	5.7	9.0	68	0.38
V-5	5.8	11.8	65	< 0.25
Mean	6.0	9.48	65	< 0.44
N-1	6.1	8.03	70	0.72
N-2	5.8	9.6	68	0.4
N-3	6.0	7.78	64	0.61
N-4	6.1	7.38	63	< 0.25
N-5	6.0	10.2	67	0.42
Mean	6.0	8.6	66	< 0.48
Ch-1	6.9	6.69	73	0.50
Ch-2	6.5	6.67	75	0.66
Ch-3	7.0	8.61	80	0.35
Ch-4	6.7	6.58	59	< 0.25
Ch-5	7.2	6.7	69	0.36
Mean	6.9	7.05	71	< 0.42

4. Arrow-arum, Peltandra virginica

The results of the trace metal analyses on the roots and seeds of P. virginica are shown in Tables 12 and 13, respectively. Statistical comparisons of the rhizome metal levels revealed significant differences ($P < 0.05$) between the Vienna and Choptank sites for As and Se. Arsenic levels were greater at the disposal site, while Se levels were higher at the Choptank site. Similar comparisons of metal concentrations in the seeds revealed a significantly greater ($P < 0.01$) level of As in fruiting bodies from the disposal site than from the Choptank River location. No significant differences ($P < 0.05$) were found between the disposal site and the Nanticoke reference site for root or seed sample comparisons.

Comparisons of trace metal levels were made between the roots and seeds collected at each location to examine the possibility of organ-specific bioaccumulation. The roots possessed significantly higher ($P < 0.05$) levels of both Al and As than the seeds for all three locations. Mean concentrations of Cr were also substantially higher in the roots than in seeds; however, statistical significance could not be determined because Cr levels in all seed samples were below detectable values (Table 13). Comparison of Se levels between roots and seeds revealed that mean concentrations were not significantly different ($P > 0.05$).

Organ-specific bioaccumulation is further evidenced by comparing concentration ratios. The ratio of mean metal concentration in roots and seeds (Tables 12 and 13) to mean concentration found in the sediment (Table 10) is provided in Table 14. The results indicated substantial differences

Table 12

Trace Metal Concentrations in
Roots of Arrow-arum, *Peltandra virginica*

Station	Metal Concentration (ppm)			
	Al	As	Cr	Se
V-1	1820	2.13	6	< 0.05
V-2	632	5.43	4	0.07
V-2	403	3.09	3	< 0.05
V-3	705	1.50	2	< 0.05
V-3	1180	2.77	4	< 0.05
V-4	595	1.35	2	0.18
V-5	268	0.55	< 1	0.16
Mean	800	2.40	< 3	< 0.09
N-1	1260	1.47	5	0.06
N-1	1520	2.05	5	0.14
N-2	2400	4.81	16	< 0.05
N-3	391	1.09	4	< 0.05
N-4	386	0.49	2	< 0.05
N-5	384	0.32	3	< 0.05
N-5	344	0.30	7	< 0.05
Mean	955	1.50	6	< 0.06
Ch-1	575	0.35	3	0.12
Ch-2	505	1.28	3	0.22
Ch-3	850	0.65	3	0.16
Ch-4	734	0.80	2	0.18
Ch-4	782	0.75	3	0.16
Ch-5	345	0.33	2	0.11
Ch-5	310	0.28	2	0.21
Mean	586	0.63	3	0.17

Table 13

Trace Metal Concentrations in Seeds of Arrow-arum, *Peltandra virginica*

Station	Metal Concentration (ppm)			
	Al	As	Cr	Se
V-1 & V-2	34	0.15	< 1	0.12
V-3	20	0.18	< 1	0.10
V-3	108	0.27	< 1	0.22
V-4	46	0.13	< 1	0.13
V-5	14	0.14	< 1	0.10
Mean	44	0.17	< 1	0.13
N-1	36	0.15	< 1	0.21
N-2	46	0.51	< 1	0.11
N-2	28	0.10	< 1	0.07
N-3	< 2	0.10	< 1	< 0.05
N-4	262	0.22	< 1	0.17
N-5	11	0.10	< 1	0.12
N-5	9	0.14	< 1	0.09
Mean	< 56	0.19	< 1	< 0.12
Ch-1	3	< 0.05	< 1	0.09
Ch-2	23	< 0.05	< 1	0.06
Ch-3	53	0.10	< 1	0.11
Ch-4	24	0.08	< 1	0.22
Ch-4	57	0.11	< 1	0.30
Ch-5	25	0.10	< 1	0.16
Mean	31	< 0.08	< 1	0.16

Table 14

Plant:Soil Concentration Ratios

Element	Concentration Ratio[a]	*P. virginica* Concentration Ratios		
		Roots	Fruit	Location
Aluminum		0.013	0.0007	Vienna
		0.016	0.0009	Nanticoke
		0.008	0.0004	Choptank
Arsenic	0.14	0.25	0.02	Vienna
		0.17	0.02	Nanticoke
		0.09	< 0.01	Choptank
Chromium	0.02	0.05	< 0.02	Vienna
		0.09	< 0.02	Nanticoke
		0.04	< 0.01	Choptank
Selenium	1.0	< 0.2	< 0.30	Vienna
		0.13	0.25	Nanticoke
		0.4	0.38	Choptank

[a]From Dvorak et al. 1978
This is a generalized approximation of the ability of plants to accumulate trace elements similar to the method employed by Hodgson. 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.

between seeds and root concentration ratios for Al and As and, to a lesser extent, for Cr. The values for Se showed no clear trend. These findings are essentially identical to the results of the statistical comparisons.

In general, organ-specific bioaccumulation in P. virginica appears to be consistent with the reported trend of roots> stems and leaves> fruits or seeds (Dvorak et al., 1978). This finding is generally encouraging since the root tissue is less likely to be used as a food source than the fruit or seeds, thereby reducing the potential for biomagnification in animals. However, as these plant parts decay, these metals may be available in the habitat for recycling or transport.

5. Phragmites communis

Samples of Phragmites rhizome were taken at six stations from the ash disposal site (Figure 13). The results of the trace metal analyses are shown in Table 15. Mean relative concentrations found in the rhizomes reflected the relative concentrations of the four metals in the upper 15 cm of the ash material (Table 4).

Relative concentrations of the elements were Al> Cr> As> Se. This order of elemental abundance is also characteristic of endogenous soil concentrations (Dvorak, 1978) and is commonly observed in plant tissue.

Table 15

Trace Metal Concentrations in
Rhizomes of *Phragmitis communis*

	Metal Concentration (ppm)			
Station	Al	As	Cr	Se
Ph-1	55	0.25	5	0.09
Ph-2	442	0.42	25	0.19
Ph-3	235	0.95	8	0.28
Ph-4	320	0.55	9	0.07
Ph-5	100	0.30	8	0.20
Ph-6	119	0.69	7	0.16
Mean	212	0.53	10	0.17

6. Surface Waters

The results of the surface water analyses are provided in Table 16. The average Se concentrations at all three locations exceeded the EPA criterion of 22 ug/l for protection of freshwater aquatic life. This finding resulted in a check of the original analysis utilizing a new Se lamp. The second set of Se results were quite similar to the first (Table 17). A review of sample preparation revealed that the CMS laboratory had acidified the samples before filtering, which is opposite of the standard EPA method for analysis of dissolved metals (U.S. EPA, 1979). Therefore, metals tied up in the suspended fraction would have been at least partially dissolved upon acidification to pH 2. Thus, the results of the analyses would be higher than the actual dissolved trace metal concentrations.

The utility of these data is limited, especially in regard to Se. The fact that these Se concentrations were greater than EPA guidelines for dissolved Se does not necessarily mean that the ambient dissolved Se concentration is a problem. The results of the surface water analyses conducted during the hydrogeologic phase of this study, in which samples were not acidified first, indicate that dissolved Se concentrations are within the prescribed EPA guidelines at the disposal site (Table 6).

The recommended standards for protection of aquatic organisms for Al, As, and Cr are 63 ug/l, 130 ug/l, and 100 ug/l, respectively (Sittig, 1980). The observed mean concentrations of these metals, even though artificially increased by the processing error, were within these guidelines. This is further evidenced by the results of the

Table 16

Trace Metal Concentrations in Water Samples
from the Three Study Locations

Station	Metal Concentration (ppb)			
	Al	As	Cr	Se
V-6	100.0	39.7	< 1.2	30
V-7	55.6	54.1	1.4	21
V-8	44.4	37.4	< 1.2	29
V-2	47.2	34.4	1.6	34
Mean	61.8	41.4	< 1.35	28.5
N-1	55.6	31.5	< 1.2	26
N-2	50.0	37.1	< 1.2	34
N-6	58.3	32.7	1.4	31
N-7	65.0	33.5	< 1.2	27
Mean	57.2	33.7	< 1.25	29.5
Ch-3	61.1	39.3	< 1.2	58
Ch-4	50.0	35.6	< 1.2	58
Ch-6	61.1	40.7	< 1.2	57
Ch-7	61.1	43.2	< 1.2	57
Mean	58.3	39.7	< 1.2	57.5

Table 17

Second Analysis of Selenium Concentration (ppm) in Waters from the Three Study Locations

Choptank Control		Nanticoke Control		Vienna Site	
Station	Metal Conc.	Station	Metal Conc.	Station	Metal Conc.
Ch-3	0.044	N-1	0.044	V-6	0.038
Ch-4	0.044	N-3	0.047	V-7	0.031
Ch-6	0.045	N-6	0.053	V-8	0.031
Ch-7	0.050	N-7	0.048	V-9	0.040
Mean	0.046		0.048		0.035

Analytical Uncertainty ± .015

disposal site surface water tests presented in the hydrogeologic section (Table 6), wherein As and Cr values were virtually all below detectable limits. Therefore, the concentrations of the four trace metals in the surface water near the disposal site appear to be within the recommended EPA standards for protection of aquatic organisms.

It is generally difficult to identify a pollution source (i.e., food, seawater, and/or sediments) on the basis of field surveys because field studies cannot easily discriminate inputs from multiple sources; also, the differing ability to accumulate pollutants within a population (e.g., size, age, microhabitat) may obscure relationships between the source and body burden. Furthermore, accumulation may occur rapidly over brief periods such as storm events. These discrete episodes can be difficult to chacterize, but they are likely mechanisms at Vienna, as increased runoff from the disposal site and bankfull tributary erosion would suspend greater amounts of ash material in the runoff. However, the bioavailability of this material would be uncertain. Most investigations of metal uptake have found food and particulates to be a more important source of metals than water (Swartz and Lee, 1980).

The results of this study indicate that the dissolved metals in the surface water at the disposal site are probably not a source of contamination for the affected Fundulus population. This investigation revealed that the primary mechanism(s) of migration of heavy metals from the disposal site was through erosion and physical transport in the site tributaries to the surrounding environment. This process identifies the

primary pathway of physical dispersal; however, the mechansim(s) by which metals are entering the Fundulus population are unknown. The two primary routes for uptake of trace metals by fishes are: 1) passive or active absorption through the gills and 2) ingestion. Ingestion of contaminated food items is generally considered to be the major source of trace elements accumulated in fishes (Dvorak et al., 1978).

Evidence indicates that plants accumulate high concentrations of pollutants which eventually become available to consumers as particulate detritus or dissolved organic matter (Dvorak et al., 1978). Benthic invertebrates are the principal consumers of contaminated plant detritus. Although no comprehensive theory has adequately described trace metal cycling through aquatic systems, largely due to the inherent variability in the natural environment, the greatest bioaccumulation/concentration factors generally are found among sediment and detrital feeders. Trophic transfer of trace metals to higher order consumers in aquatic systems generally results in lower trace-element concentrations in predators than in their prey (Dvorak et al., 1978). This lack of obvious structure and unidirectional flow through the food chain may limit, but does not preclude, biomagnification. However, these differences among trace metals and organisms in their potential for biomagnification may complicate determination of how F. heteroclitus bioaccumulate contaminants near the disposal site.

SECTION VI

CONCLUSIONS

Having completed this problem definition phase of the study; which included field reconnaissance, fall season sampling, and limited ground water data and laboratory analyses; we draw the following conclusions:

A. Hydrogeological

1. The fly ash deposit on the tidal marsh disposal site east of the Nanticoke River contains As, Cr, and Se.

2. The shallow ground water system shows local contamination by sulfates, and a few detectable heavy metals slightly above PDWS.

3. The local contamination in the shallow ground water system at the site has no significant effect on surface water quality leaving the site.

4. The majority of the leachable components of the fly ash on the tidal marsh have apparently been leached away, and the fly ash now constitutes no significant threat to ground water quality.

5. Unknown concentrations of sulfates and heavy metals may have leached into the ground water between the time of fly ash deposition and the present.

6. If contaminants reached the Pleistocene aquifer, they would probably discharge to the Nanticoke River and be diluted below detection limits.

7. The major avenue for migration of heavy metals leaving the site is via erosion and physical transport of the fly ash from the site drainageways to the moat and natural drainageways.

8. Indications are that the tidal marsh disposal site would pose no threat to ground water quality at a well which might be developed in the Pleistocene aquifer by the Town of Vienna, west of the river.

B. Biological

1. The organism which exhibited the greatest differences in concentration of the four trace metals among the three sampling locations was the mummichog, *Fundulus heteroclitus*. Mean trace metal concentrations were greater in disposal site specimens in all comparisons with the reference locations (Table 18) with five of the eight comparisons being statistically significant ($P < 0.05$). These results suggest that the *Fundulus* sampled at the disposal site and the Choptank site may be affected by localized sources of trace metal contamination. The Choptank site was later confirmed by the Maryland Department of Health as being affected by an unauthorized waste discharge.

Table 18

Summary of Wilcoxson's Test Results

	Vienna: Choptank				Vienna: Nanticoke			
Matrix	Al	As	Cr	Se	Al	As	Cr	Se
Mummichogs	N.S.[1] >[3]	*[2] >	N.S. >	N.S. >	* >	* >	N.S. >	* >
Arrow-Arum Roots	N.S. >	* >	N.S. <	* <	N.S. <	N.S. >	N.S. <	N.S. >
Fruits	N.S. >	* >	N.S. ?[4]	N.S. <	N.S. <	N.S. <	N.S. ?	N.S. >
Sediment	* <	* >	N.S. <	N.S. ≥	N.S. =	N.S. >	N.S. <	N.S. ≤

[1]N.S. = not significant

[2] * = significant difference ($P < 0.05$)

[3]inequalities indicate which of the mean values was greater

[4] ? = all values were below detectable limits

2. Arrow-arum exhibited organ-specific bioconcentration for Al, As, and Cr. Root tissue levels of Al and As were significantly greater ($P < 0.05$) than in seeds for each location. Mean Cr levels were also substantially greater in the root tissue. Se levels exhibited no clear trend. Examination of concentration ratios between roots and seeds with metal concentrations in the associated sediment revealed similar evidence of organ-specific bioaccumulation.

3. Rhizome samples of Phragmites collected on the ash site exhibited relative metal concentrations similar to the top 15 cm of ash in which they had grown (i.e., Al> Cr> As> Se). This same elemental constituency is typical of endogenous soils and common in root tissue.

4. Analysis of sediment samples from the three locations showed only As to be consistently greater at the disposal site (Table 11) compared to the two reference locations. However, only one As comparison proved statistically significant ($P < 0.05$). The order of trace metal abundance in sediment was similar with fly ash results (i.e., Al> Cr> As> Se). However, statistical comparison of actual concentrations in sediment and fly ash from the disposal site revealed significant differences ($P < 0.05$) which suggest that the sediment samples collected near the disposal site were not predominantly fly ash.

5. Analysis of surface water indicated atypically high Se values in one set of samples. However, the laboratory technique used in preparing the samples is believed to

be the causative factor. All other metal concentrations were within recommended EPA guidelines for protection of aquatic organisms.

6. Comparisons of As levels in samples collected at the disposal site with those collected at the control locations revealed that the disposal site samples possessed greater As concentrations in seven of eight comparisons (Table 18). Furthermore, concentrations of As in mummichogs, arrow-arum roots and seeds, and sediment collected at the disposal site were significantly greater ($P < 0.05$) than As levels found in Choptank River collections (Table 18). Since the principal emission source of As in the U.S. is thought to be coal-fired power plants (Sittig, 1980), this element may prove to be the best "fingerprint" for identifying the potential environmental impact of the abandoned ash site.

SECTION VII

RECOMMENDATIONS

The above conclusions lead us to the following recommendations, which would best be implemented as a multi-phased program, each succeeding step dependent on the findings of the previous one. The biological and hydrogeological steps can be followed concurrently.

A. Hydrogeological

1. The amount of fly ash presently leaving the site via the surface drainageways should be determined to assess whether control measures are needed to prevent the release of fly ash from the site.

2. The amount of fly ash in the shallow sediments of the moat and adjacent drainageways should be defined and the levels of heavy metals determined to assess whether or not sediment contamination is present, and whether or not remedial measures are needed.

B. Biological

1. Spring, summer, and fall sampling of mummichogs, from the disposal site and reference locations, to confirm tissue metal concentrations and determine food habits (potential metal sources).

2. Sampling of predominant taxa of food organisms of mummichogs to determine concentrations of metals and their potential as a source of contamination.

3. Further sampling of stream sediments and/or suspended particulates in the surface waters to determine heavy metal content and their role as potential sources of contamination.

4. Research possible higher and/or lower pathways among trophic levels to further clarify the impact of the Vienna site on area biota and to what level toxicity may be a significant concern to organisms on or off the site.

SECTION VIII

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SECTION IX

APPENDICES

APPENDIX A

CLASSIFICATION OF WETLANDS AND DEEPWATER HABITATS OF THE VIENNA MARSHES

APPENDIX A

CLASSIFICATION OF WETLANDS AND DEEPWATER HABITATS OF THE VIENNA MARSHES

1)

System	Estuarine
Subsystem	Intertidal
Class	Unconsolidated Shore (clams)
Subclass	Mud and Nonpersistent
Water Regime	Regularly Flooded
Water Chemistry	Mixohaline
Soil	Mineral

2)

System	Estuarine
Subsystem	Intertidal
Class	Emergent Wetland (arrow-arum)
Subclass	Nonpersistent
Water Regime	Regularly Flooded
Water Chemistry	Mixohaline
Soil	Mineral

3)

System	Estuarine
Subsystem	Intertidal
Class	Emergent Wetland (Phragmites, Big Cordgrass - dominant) (Saltmeadow Cordgrass - subdom.)
Subclass	Persistent
Water Regime	Irregularly Flooded and Regularly Flooded
Water Chemistry	Mixohaline
Soil	Mineral

4)

System	Estuarine
Subsystem	Intertidal
Class	Scrub-Shrub Wetland (Baccharis, Dogwood)
Subclass	Broad-leaved Deciduous
Water Regime	Irregularly Flooded
Water Chemistry	Fresh to Mixohaline
Soil	Mineral

APPENDIX A

CLASSIFICATION OF WETLANDS AND DEEPWATER HABITATS
OF THE VIENNA MARSHES
(cont'd)

5)	System	Riverine
	Subsystem	Tidal
	Class	Streambed
	Subclass	Mud and (Sand)
	Water Regime	Regularly Flooded
	Water Chemistry	Mixohaline to Fresh
	Soil	Mineral

6)	System	Riverine
	Subsystem	Tidal
	Class	Unconsolidated Shore and Emergent Wetland (arrow-arum)
	Subclass	Mud and Nonpersistent
	Water Regime	Regularly Flooded
	Water Chemistry	Fresh to Mixohaline
	Soil	Mineral

According to: Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of Wetlands and Deepwater Habitats of the United States. FWS/OBS-79/31. U.S. Dept. of Interior. Wash., DC. 103 pp.

APPENDIX B

PROBABLE AMPHIBIANS AND REPTILES IN THE VIENNA AREA

APPENDIX B

PROBABLE AMPHIBIANS AND REPTILES IN THE VIENNA AREA

Species*	General Habitat: Marsh-Wetlands	General Habitat: Uplands**
Eastern Tiger Salamander *Ambystoma tigrinum*		X
Northern Dusky Salamander *Desmognathus fuscus*		X
Red-backed Salamander *Plethodon cinereus*		X
Eastern Mud Salamander *Pseudotriton montanus*		X
Northern Two-lined Salamander *Eurycea bislineata*		X
Eastern Spadefoot Toad *Scaphiopus holbrooki*		X
Bullfrog *Rana catesbeiana*	X	
Green Frog *Rana clamitans*	X	
Southern Leopard Frog *Rana sphenocephala*	X	
Wood Frog *Rana sylvatica*	X	X
Eastern Narrow-mouthed Toad *Gastrophryne carolinesis*	X	X
American Toad *Bufo americanus*	X	X
Fowler's Toad *Bufo fowleri*	X	X
Green Tree Frog *Hyla cinera*	X	X

APPENDIX B
(continued)

Species*	General Habitat: Marsh-Wetlands	General Habitat: Uplands**
Northern Spring Peeper *Hyla crucifer*	X	X
Northern Cricket Frog *Acris crepitans*	X	
New Jersey Chorus Frog *Pseudacris triseriata*	X	X
Common Snapping Turtle *Chelydra serpentina*	X	
Eastern Mud Turtle *Kinosternon subrubrum*	X	
Spotted Turtle *Clemmys guttata*	X	
Eastern Bog Turtle *Terrapene carolina*		X
Eastern Painted Turtle *Chrysemys picta*	X	
Red-bellied Turtle *Chrysemys rubriventris*	X	
Northern Fence Lizzard *Sceloporus undulatus*		X
Five-lined Skink *Eumeces faciatus*		X
Broad-headed Skink *Eumeces laticeps*		X
Ground Skink *Lygosoma laterale*		X
Northern Water Snake *Natrix sipedon*	X	

APPENDIX B
(continued)

Species*	General Habitat: Marsh-Wetlands	General Habitat: Uplands**
Eastern Ribbon Snake *Thamnophis sauritus*	X	
Eastern Garter Snake *Thamnophis sirtalis*	X	
Northern Brown Snake *Storeria dekayi*		X
Northern Red-bellied Snake *Storeria occipitomaculata*		X
Eastern Hog-nosed Snake *Heterodon platyrhinos*		X
Southern Ring-necked Snake *Diadophis punctatus*		X
Eastern Worm Snake *Carphophis amoenus*		X
Northern Black Snake *Caluber constrictor*	X	
Rough Green Snake *Opheodrys aestivus*	X	X
Corn Snake *Elaphe guttata*		X
Black Rat Snake *Elaphe obsoleta*		X
Eastern King Snake *Lampropeltis getulus*	X	X

APPENDIX B
(continued)

Species*	General Habitat: Marsh-Wetlands	General Habitat: Uplands**
Coastal Plain Milk Snake *Lampropeltis triangulium*		X
Southeastern Scarlet Snake *Coccinea spp.*		X
Northern Copperhead *Agkistrodon contartrix*		X

Sources: Conant, R. 1958. A Field Guide to Reptiles and Amphibians. Houghton Mifflin, Boston.

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* Subspecies nomenclature omitted.

** Includes moist and/or aquatic upland habitats, such as springs and small streams.

APPENDIX C

PROBABLE MAMMALS OF THE VIENNA AREA

APPENDIX C

PROBABLE MAMMALS OF THE VIENNA AREA

Species	General Habitat	
	Marsh-Wetlands	Uplands
Opossum *Didelphis marsupialis*	X	X
Least Shrew *Cryptotis paroa*	X	
Shorttail Shrew *Blarina brevicauda*	X	X
Eastern Mole *Scalopus aquaticus*		X
Little Brown Myotis *Myotis lucifugus*		X
Eastern Pipistrel *Pipistrellus subflavus*		X
Red Bat *Lasiurus borealis*		X
Big Brown Bat *Eptesicus fuscus*		X
Raccoon *Procyon lotor*	X	X
Mink *Mustela vison*	X	
River Otter *Lutra canadensis*	X	
Striped Skunk *Mephitis mephitis*	X	X
Red Fox *Vulpes fulva*	X	X
Gray Fox *Urocyon cinereoargenteus*		X
Woodchuck *Marmota monax*		X

Source: Paradisco, J.L. 1969. "Mammals of Maryland." U.S.D.I., Bureau of Sport Fisheries and Wildlife. North American Fauna Number 66.

APPENDIX D

BORING LOGS

Environmental Resources Management

Drilling Log

Project Maryland PPSP Owner

Location Vienna, Maryland W.O. Number

Well Number V-20 Total Depth 16.5 ft. Diameter 3.5 in.

Surface Elevation Water Level: Initial 5 in. 24-hrs.

Screen: Dia. Length Slot Size

Casing: Dia. Length Type

Drilling Company Drilling Method Hand Auger

Driller Log By M. Hewitt Date Drilled 12/10/81

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
				Dark brown silt and dark gray to black fine fly ash.
1				
2				
3				Organic root and vegetal mat in dark brown clayey silt matrix. Plant remains graded in and out, with runny silts.
4				
5				
6				
7				
8				
9				Dark brown very runny silt.
10				
11				
12				
13				

Environmental Resources Management

Drilling Log

Project Maryland PPSP Owner

Location Vienna, Maryland W.O. Number

Well Number V-1 Total Depth 19 ft. Diameter 3.5 in.

Surface Elevation Water Level: Initial 24-hrs. 2.25 in.

Screen: Dia. Length Slot Size

Casing: Dia. Length Type

Drilling Company Drilling Method Hand Auger

Driller Log By M. Hewitt Date Drilled 11/17/81

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
1				Dark gray to black fine fly ash, with roots and dark brown silt from 34-38 inches.
2				
3				
4				Organic root and vegetal mat in dark brown clayey silt to silty clay matrix. Plant remains grade in and out, with runny silts.
5				
6				
7				
8				Dark brown very plastic clay with trace silt.
9				
10				Alternating dark brown very runny silts and clayey silts to silty plastic clays.
11				
12				
13				

Environmental Resources Management

Drilling Log

Project Maryland PPSP Owner

Location Vienna, Maryland W.O. Number

Well Number V-7 Total Depth 7 ft. Diameter 3.5 in.

Surface Elevation Water Level: Initial 5 in. 24-hrs.

Screen: Dia. Length Slot Size

Casing: Dia. Length Type

Drilling Company Drilling Method

Driller Log By Date Drilled

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
				Organic root and vegetal mat in dark brown silt matrix, with some fly ash.
1				Dark brown silt and fly ash, with some roots.
2				Dark brown silt and roots with some fly ash.
3				
4				Organic root and vegetal mat with dark brown clayey silt matrix.
5				
6				
7				Dark brown very runny silt.

Page ____ of ____

Environmental Resources Management

Drilling Log

Project Maryland PPSP Owner

Location Vienna, Maryland W.O. Number

Well Number V-4 Total Depth 7.5 ft. Diameter

Surface Elevation Water Level: Initial 1.5 in. 24-hrs.

Screen: Dia. Length Slot Size

Casing: Dia. Length Type

Drilling Company Drilling Method Hand Auger

Driller Log By M. Hewitt Date Drilled 11/19/81

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
1				Two-inch root mat underlain by dark gray to black fine fly ash.
2				
3				
4				Organic root and vegetal mat in dark brown silty clay to clayey silt matrix.
5				
6				
7				
8				

Environmental Resources Management

Drilling Log

Project: Maryland PPSP Owner: _____

Location: Vienna, Maryland W.O. Number: _____

Well Number: V-3 Total Depth: 28 in. Diameter: 3.5 in.

Surface Elevation: _____ Water Level: Initial: 7 in. 24-hrs.: _____

Screen: Dia. _____ Length _____ Slot Size _____

Casing: Dia. _____ Length _____ Type _____

Drilling Company _____ Drilling Method: Hand Auger

Driller _____ Log By: M. Hewitt Date Drilled: 11/19/81

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
1				Two-inch root mat, underlain by dark gray to black fine fly ash.
2				
3				Organic root and vegetal mat in dark brown clayey silt matrix.

Environmental Resources Management

Drilling Log

Project Maryland PPSP Owner

Location Vienna, Maryland W.O. Number

Well Number V-2 Total Depth 32 in. Diameter 3.5 in.

Surface Elevation Water Level: Initial 13 in. 24-hrs.

Screen: Dia. Length Slot Size

Casing: Dia. Length Type

Drilling Company Drilling Method Hand Auger

Driller Log By M. Hewitt Date Drilled 11/19/81

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
1				Two-inch root mat, underlain by dark gray to black fine fly ash.
2				Organic root and vegetal mat, in dark brown clayey silt matrix.
3				Dark brown very runny silt.

Environmental Resources Management

Drilling Log

Project Maryland PPSP Owner

Location Vienna, Maryland W.O. Number

Well Number V-1 Total Depth Diameter

Surface Elevation Water Level: Initial 24-hrs.

Screen: Dia. Length Slot Size

Casing: Dia. Length Type

Drilling Company Drilling Method

Driller Log By Date Drilled

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
14				
15				
16				
17				Dark brown very runny silt.
18				
19				Silty fine white quartz sand, well sorted.

Environmental Resources Management

Drilling Log

Project Maryland PPSP Owner

Location Vienna, Maryland W.O. Number

Well Number V-20 Total Depth Diameter

Surface Elevation Water Level: Initial 24-hrs.

Screen: Dia. Length Slot Size

Casing: Dia. Length Type

Drilling Company Drilling Method

Driller Log By Date Drilled

Sketch Map

Notes

Depth (Feet)	Graphic Log	Well Construction	Sample Number	Description/Soil Classification (Color, Texture, Structures)
14				
15				
16				Very silty medium to coarse grained poorly sorted quartz sand; subangular; silt fraction grades out within 2 feet. Some fine non-quartz gravel.
17				

Page 2 of 2

APPENDIX E

SUMMARY OF t-TEST RESULTS

APPENDIX E

SUMMARY OF t-TEST RESULTS

	Vienna: Choptank				Vienna: Nanticoke			
Matrix	Al	As	Cr	Se	Al	As	Cr	Se
Mummichogs	N.S.[1] >[3]	*[2] >	N.S. >	N.S. >	* >	* >	* >	* >
Arrow-Arum Roots	N.S. >	* >	N.S. <	* <	N.S. <	N.S. >	N.S. <	N.S. >
Fruits	N.S. >	* >	N.S. ?[4]	N.S. <	N.S. <	N.S. <	N.S. ?	N.S. >
Sediment	* <	* >	N.S. <	N.S. ≥	N.S. =	N.S. >	N.S. <	N.S. ≤

[1] N.S. = not significant

[2] * = significant difference ($P < 0.05$)

[3] inequalities indicate which of the mean values was greater

[4] ? = all values were below detectable limits

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