DEPARTMENT OF NATURAL RESOURCES
MARYLAND GEOLOGICAL SURVEY

Kenneth N. Weaver, Director

REPORT OF INVESTIGATIONS NO. 18

SOLID-WASTE DISPOSAL IN THE GEOHYDROLOGIC ENVIRONMENT OF MARYLAND

by

Edmond G. Otton

Prepared in cooperation with the Geological Survey
United States Department of the Interior

1972

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COMMISSION
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MARYLAND GEOLOGICAL SURVEY

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SOLID-WASTE DISPOSAL IN THE GEOHYDROLOGIC ENVIRONMENT OF MARYLAND

By Edmond G. Otton

ABSTRACT

In 1920 about 0.7 million tons of urban solid wastes were disposed of in Maryland. By 1970 the quantity of these wastes had increased to 3.2 million tons, and by 1980 it is predicted that disposal of about 4.5 million tons will be required. Solid wastes are disposed of by open dumping, sanitary landfilling, and incineration. The use of sanitary landfills and the abandonment of open dumping by Maryland towns is recommended by the Maryland Department of Health.

As Maryland is in a humid region having about 43 inches of precipitation annually, sufficient moisture passes through sanitary landfills to dissolve or otherwise carry along various metals, chemicals, bacteria, and other undesirable materials, collectively called "leachate." The water may also trigger the formation of such gases as methane and carbon dioxide. Under certain conditions the leachate may enter surficial aquifers at or near the base of the landfill. Rates of leachate infiltration range widely depending on the nature of the earth materials and the available water. Infiltration rates in undisturbed soils have been reported to range from almost nil to about 5 inches per hour.

Chemical analyses of landfill leachate in other states indicate that abnormally high concentrations of total hardness, chloride, nitrate, sulfate, and iron are especially common. Bacteriologic analyses of coliforms in leachate show very high colony counts, values in the range of 1.1×10^6 to 10.7×10^6 per 100 milliliters having been reported from simulated landfills in West Virginia.

In this report, Maryland is divided into five terrane types on the basis of the hydrologic characteristics of the land as related to solid-waste disposal. Terrane I, shale, siltstone, and sandstone of the Appalachian province, appears to be moderately amenable to solid-waste disposal, with certain limitations. Degradation of the underlying ground water by leachate is least likely in areas underlain by shale or other impervious material, but leachate generated at such sites may enter nearby streams unless adequate precautions are taken. Some abandoned coal strip mines appear to be amenable to disposal of solid municipal wastes.

Terrane II, valleys underlain by limestone or marble, is somewhat less amenable than terrane I to solid-waste disposal because of the common development of solution channels and crevices in the underlying rocks. Thus, direct pollution of the underlying ground-water reservoirs is possible. Parts of terrane II, where the residuum or alluvium overlying the limestone is sufficiently thick or impervious to retard or inhibit vertical movement of pollutants, may be more amenable to solid-waste disposal than other parts.

ABSTRACT (Cont.)

Susceptibility of any terrane to disposal of solid wastes is, of course, also dependent upon the design of the landfill—the use of relatively impervious cover material, etc.

Terrane III, crystalline silicate rocks of the Piedmont, is suitable for sanitary landfills where a relatively thick zone of saprolite lies well above the water table. Where the saprolite is thin and the rocks are extensively fractured, the possibility of pollution of the ground water by leachate is increased. Terrane IV, upland Coastal Plain deposits, may be locally suitable for the protection of ground waters from leachate where the fill site is underlain by clayey strata well above the local water table. Local groundwater pollution is likely where sand and gravel underlies the surface at a fill site. Terrane V, low-lying Coastal Plain deposits, offers protection from leachate pollution of deep-lying artesian aquifers because of the thick and extensive impervious strata that normally underlie the land surface. However, ample leachate production is expected in filled swamps, rivers, and estuaries due to the high water table prevalent in these areas. Such leachate will, of course, be diluted by the native water.

Surface waters in all five terranes are subject to degradation by landfill leachate. Some protection can be afforded by designing the landfill to minimize the generation of leachate or by collecting and treating the leachate before permitting it to enter a surface watercourse.

INTRODUCTION, PURPOSE, AND SCOPE

Increasing attention is being given to the disposal of solid-waste materials generated by our expanding, consumer-oriented civilization. The problem of disposing of this growing volume of material is becoming more acute, especially in the urbanized areas, because of the limited land available for this purpose, and because of the general unesthetic and undesirable nature of open dumps, ordinary waste piles, or even the modern sanitary landfill. Regardless of what form it is in, solid waste deposited on the land surface becomes a part of the land-water environment and, hence, is an appropriate field for review and investigation by the geohydrologist.

The purpose of this report is to review the geohydrologic factors involved in the disposal of solid wastes, and especially to describe: 1) how the disposal of solid waste in certain geohydrologic environments might result in degradation of nearby ground and surface waters by leachate[1]; and 2) how other, more suitable, geohydrologic environments might result in less degradation of the ground and surface waters. A wide spectrum of geohydrologic environments exists within Maryland. This report will describe some of them in relation to their use as sites for solid-waste disposal. It is hoped this report will be useful for urban planners, municipal officials, engineers concerned with solid-waste disposal, interested citizens, and all who share a concern for the problems of both solid-waste disposal and pure, undegraded water resources.

ACKNOWLEDGMENTS

Acknowledgment is given for the assistance and advice of the staffs of both the U.S. Geological Survey and the Maryland Geological Survey. Discussions with William Shields, Chief, and other staff members of the Solid Wastes Division, Maryland Department of Health, have been especially valuable in providing the writer with background material regarding the solid-waste disposal problem in the State. Edward Newell, of the Solid Wastes Division, was especially helpful in providing data on the Frostburg demonstration sanitary landfill.

Miss Catherine Manly drafted the illustrations and Mrs. J. Elizabeth Smith typed the report. Stuart W. McKenzie made all the field bacteriological analyses and many of the field chemical analyses for the site studies for this report.

The cover photograph was provided by the Environmental Protection Agency.

NATURE OF THE SOLID-WASTE DISPOSAL PROBLEM

The nature of the solid-waste disposal problem and the types of waste currently being generated in the United States have been summarized by the American Public Works Association in a recent publication of the Public Administration Service (1966, p. 12). Table 1 is adapted from that agency's report.

Types of Waste

The proportionate quantities of the various types of residential wastes generated in the United States annually are shown in table 2:

[1] With reference to solid-waste disposal, leachate is a water-derived fluid draining from a landfill which has acquired certain chemical and physical properties as a result of its passage through the fill materials.

Table 1. Solid wastes by kind, composition, and sources.

Kind		Composition	Sources
Solid Wastes (Refuse)	Garbage	Wastes from preparation, cooking, and serving of food; market wastes; wastes from handling, storage, and sale of produce	Households, restaurants, institutions, stores, markets
	Rubbish	Combustible: paper, cartons, boxes, barrels, wood, excelsior, tree branches, yard trimmings, wood furniture, bedding, dunnage	
		Noncombustible: metals, tin cans, metal furniture, dirt, glass, crockery, minerals	
	Ashes	Residue from fires used for cooking and heating and from on-site incineration	
	Street refuse	Sweepings, dirt, leaves, catch-basin dirt, contents of litter receptacles	Streets, sidewalks, alleys, vacant lots
	Dead animals	Cats, dogs, horses, cows, chickens	
	Abandoned vehicles	Unwanted cars and trucks left on public property	
	Industrial wastes	Food-processing wastes, boiler-house cinders, lumber scraps, metal scraps, shavings	Factories, power plants
	Demolition wastes	Lumber, pipes, brick, masonry, and other construction materials from razed buildings and other structures	Demolition sites to be used for new buildings, renewal projects, expressways
	Construction wastes	Scrap lumber, pipe, other construction materials	New construction, remodeling
	Special wastes	Hazardous solids and liquids: explosives, pathological wastes, radioactive materials	Households, hotels, hospitals, institutions, stores, industry
	Sewage-treatment residue	Solids from coarse screening and from grit chambers; septic-tank sludge	Sewage-treatment plants; septic tanks

Table 2. Average composition of residential wastes in the United States, in percent by weight of total wastes (from Am. Public Works Assoc., 1968, p. 42).

Type of waste	Percent of total wastes
Paper	60.0
Food	8.5
Glass and ceramics	8.0
Metallics	8.0
Plants and grass	6.5
Plastic products	3.5
Furniture and boxes	1.5
Construction wastes	1.0
Textiles	.5
Dirt and vacuum-cleaner catch	.5
Rubber products	.4
Leather products	.4
Household and garden chemicals	.4
Paints, oils, and varnishes	.3
Miscellaneous (liquids, special wastes, etc.)	.5

The types of solid wastes available for disposal are similar throughout the United States, but in recent years the amount of garbage, or putrescible refuse, has decreased owing to the trend toward installation of garbage grinders in many homes and restaurants. The decrease in the production of garbage in municipal waste is also due to the increased consumption of pretrimmed and ready-to-cook frozen foods (Maryland State Department of Health, 1966, p. III-1). The increased use of throw-away paper, plastic, and glass containers in the past few years has resulted in a comparative increase in this type of refuse. Variations in types of waste have been observed between industrial and residential communities, and within a community owing to seasonal causes; for example, more grass and shrub cuttings are collected during the summer months (Am. Public Works Assoc., 1968, p. 27). Differences in the living standards of the inhabitants of different communities appear to generate slightly greater quantities of wastes in the more affluent communities.

Some communities incinerate their wastes, a practice that reduces the volume of material to be disposed. Table 3 (Rampaceck and Sullivan, 1968, p. 46) shows the nature of incinerator waste from Washington, D. C. The proportion of paper and organic material is greatly reduced in comparison to pre-burned waste (table 2).

Table 3. Composition of residue from incinerators in metropolitan Washington, D. C.

Item	Percent
Glass	43.8
Tin cans	17.5
Ash	15.4
Unburned paper and charcoal	8.3
Mill scale and small iron	6.8
Massive iron	3.5
Nonferrous metals	1.4
Stones and bricks	1.3
Ceramics	.9
Iron wire	.7
Partially burned organics	.7

Quantities of Waste

For the United States as a whole it is reported that three major sources contribute about 1.5 billion tons of solid wastes per year (Environmental Science and Technology, 1970, p. 384). These sources and the quantities contributed by each are as follows:

Source	Million tons per year	Percent of total
Mineral	1,126	76
Urban	256	17
Industrial	110	7

The large quantity of mineral wastes includes mine tailings and quarry wastes, and the industrial wastes include all types of solid-waste materials, such as plastics, wood, and fabrics, which have little or no recoverable value. Of course, some industrial waste becomes part of the urban waste, so its proper classification is difficult.

According to the Maryland State Department of Health (1966, p. I-1) the quantities of refuse disposed on a per capita basis in Maryland have increased substantially during the past 50 years and will presumably continue to increase during the future. The reported and estimated quantities are:

Year	Pounds per day per capita	Tons per year per capita	Tons per year (total in millions)	Population[1] (millions)
1920	2.75	0.50	0.7	1.45
1966	4.2	.77	2.9	3.75
1970	4.5	.83	3.2	3.84
1980	5.5	1.00	4.5	4.54

[1] Population estimates for 1970 and 1980 from Maryland State Department of Planning, Newsletter, v. 16, no. 7, July, 1963.

Based on a consultant's recent report (Black and Veatch, 1967, p. m), the total refuse from the District of Columbia was reported to be 5.6 pounds per capita per day in 1965, a quantity somewhat higher than the 1966 Maryland quantity of 4.2 pounds per capita per day. However, the report indicates that for the seven Maryland and Virginia counties composing the remainder of the Washington Metropolitan Region only 3.25 pounds of refuse per capita per day were collected for disposal in 1965. A simple average of the two estimates for the Washington region is 4.4 pounds per capita per day, a value in fair agreement with the reported Maryland estimate.

Methods of Disposal

Several methods of refuse disposal exist, but each method must involve to some degree the land, water, and air, that is—the total environment. Refuse disposal on the land is probably the oldest method in use, although its development as a scientific method of disposal in the United States is probably relatively recent (Committee on Sanitary Landfill Practice, 1959, p. 1). The common methods of refuse disposal are identified and discussed briefly:

1) *Open dumping*—This method usually consists of dumping the solid wastes in a pit, quarry, swamp, or open pile. Little or no attention is paid to the problems of nuisance or public health that may result, and insects, rodents, odors, and blowing trash are common sources of nuisance or health hazards. Modern sanitary engineering practice condemns the use of open dumps.
2) *Sanitary landfill*—This method of solid-waste disposal evolved from the practice of open dumping as an attempt to develop an economical method of disposal that eliminates most of the undesirable features of open dumps. The Committe on Sanitary Landfill Practice (1959, p. 1) defines a sanitary landfill as follows:

   "Sanitary landfill is a method of disposing of refuse on land without creating nuisances or hazards to public health or safety by utilizing the principles of engineering to confine the refuse to smallest practical area, to reduce it to the smallest practical volume, and to cover it with a layer of earth at the conclusion of each day's operation or at such more frequent intervals as may be necessary."

   The several methods of sanitary landfill construction, land requirements, and other factors involved in site selection are described later in this report.
3) *Composting*—Composting is a method of disposal whereby the organic components of the refuse can be broken down by bacterial action to a compost, a material similar to the organic parts of soil. Although this method of waste disposal yields a useful end-product, the undecomposable components of the refuse, such as metal, rubber, glass, and plastic, must be removed. After removal of these constituents, the remaining material is usually shredded, chopped, and placed into mounds where decomposition takes place. Because of the limited market for the compost and the large volume of noncomposted materials that must subsequently be disposed of, composting has not been widely adopted by cities in the United States.
4) *Incineration*—Incineration of solid wastes is a well known and commonly used method of disposal. Modern large-capacity incinerators are relatively expensive, both to build and operate. An improperly designed or operated incinerator can result in excessive smoke and air pollution. Also, as incinerators reduce the volume of wastes material by only about 80 percent, a method of disposal must be found for the remaining 20 percent of the waste (Maryland State Department of Health, 1966, p. V-7). Nevertheless, where land for disposal purposes is scarce or extremely costly, incinerators are being used. Washington, D. C., Baltimore and Salisbury, Md., and Montgomery County, Md., all use incinerators. It is reported that the capacity of the District of Columbia's incinerators is inadequate to handle the load of solid wastes generated and that the use of sanitary landfills is also required (Black and Veatch, 1967, p. h).
5) *Disposal at sea*—The method of disposing of solid wastes by hauling them out to sea in barges and dumping them has been used by coastal cities. In recent years this practice has met with disapproval. The Maryland Department of Health (1966, p. V-14) reports that, as a result of a suit by New Jersey in 1933, the United States Supreme Court prohibited the city of New York from dumping ". . . any garbage or refuse, or other noxious, offensive or injurious matter, into the ocean, or waters

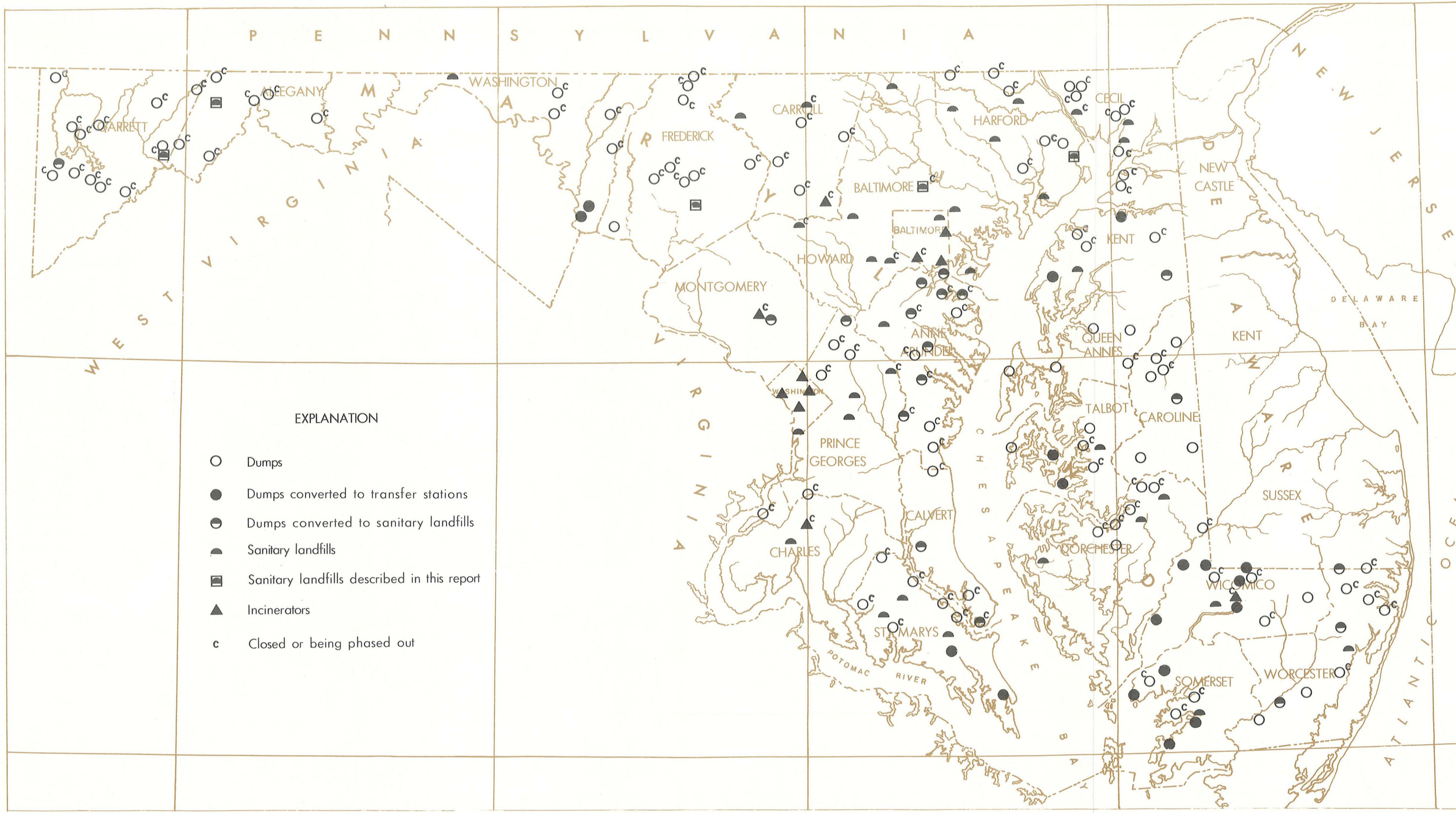


Figure 1. Map showing solid-waste-disposal facilities in Maryland and the District of Columbia as of 1977.

of the United States off the coast of New Jersey, and from otherwise defiling or polluting said waters and the shores and beaches thereof . . . ". However, it is reported that as of 1968 about 4 million tons of sewage sludge were being dumped off New York Harbor (Council on Environmental Quality, 1970, p. 5).

Although dumping at sea might appear to be an economical alternative method of waste disposal, it has several drawbacks, not the least of which is the likely pollution of an area of the ocean. It is reported in a recent article in Environmental Science and Technology (1970, v. 4, no. 5, p. 388) that as of 1968 some 48 million tons of solid wastes were disposed of at sea in the United States.

SOLID-WASTE DISPOSAL IN MARYLAND

According to the Maryland State Department of Health (1966, p. IX-1) a wide range of disposal methods were in use in the State. The location and methods of solid-waste disposal in the counties of the State are described as follows:

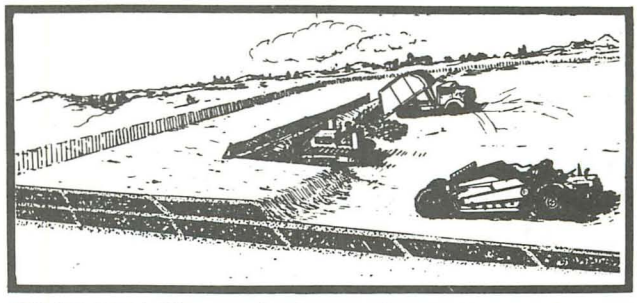
> "Most counties use a wide range of disposal methods. Six counties have sanitary landfills, 12 counties have landfills, 20 counties have open dumps, and 4 counties have incinerators. Of the 155 refuse disposal sites surveyed and classified by county sanitarians, 18 were sanitary landfill, 30 were landfills, 101 were open dumps, and there were 6 incinerators."

Subsequent to the preparation of the 1966 report, it has been reported that, as a result of the urging of the Maryland Department of Health, the number of open dumps in the State has decreased. Many of the dumps have been converted to sanitary landfills, and some of them have been covered or closed, being replaced by sanitary landfills at the same site or at other sites. The policy of the Department of Health has been to request local governments to convert open dumps to sanitary landfills or to a more advanced disposal method such as incinerators.

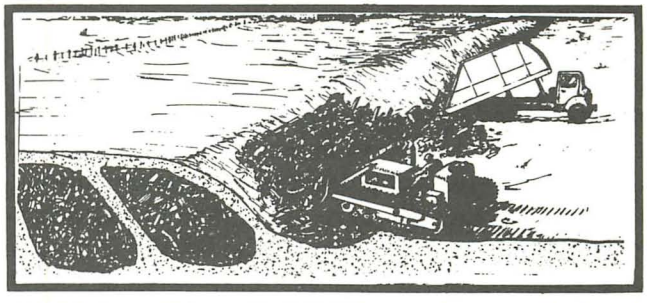
The locations of sanitary landfills, incinerators, and refuse dumps in Maryland are shown in figure 1. Although reasonably complete, the map may not show some of the older abandoned sites or some of the small private dumps.

Methods of Operation

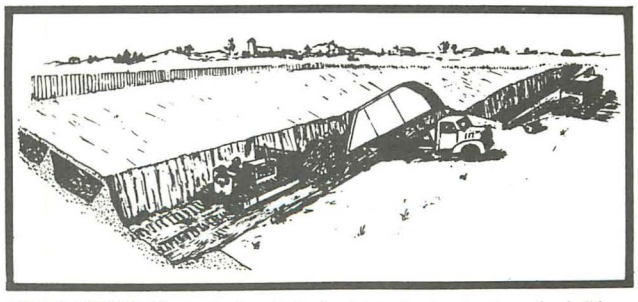
A sanitary landfill is normally constructed using one of three general methods—the area method, the trench method, or the depression method (Committee on Sanitary Landfill Practice, p. 7). The area method (fig. 2) is used generally in flat



AREA METHOD. A bulldozer spreads and compacts the waste. The scraper (foreground) is used to haul cover material at the end of the day's operations. Note finished cells.



RAMP VARIATION. Solid wastes are spread and compacted on a slope. The daily cell may be covered with earth scraped from the base of the ramp or trucked from elsewhere.



TRENCH METHOD. The truck deposits its load into the trench where the bulldozer spreads and compacts it. Dragline digs next trench, using earth for covering first.

Figure 2. Sketches showing details of area and trench methods of constructing a sanitary landfill.

areas where the waste is mounded, compacted, and covered by suitable earth material hauled in from a source area not a part of the landfill site. Where tidal marshes are used as landfill sites, the cover material has sometimes been obtained by dredging in a nearby estuary. A completed landfill is made up of a series of cells consisting of compacted waste surrounded by earth. In constructing each cell, a fill of some convenient width is started at one limit of the disposal area and progressively lengthened until it reaches the other limit of the area, or to some intermediate position predetermined by the plan of operation. A suitable earth cover is placed on the refuse as the operation proceeds. Compaction is aided by the collection vehicles and bulldozers operating on top of the fill (fig. 2).

The depth of compacted fill in a cell varies, but 7 to 8 feet is commonly advocated (Committee on Sanitary Landfill Practice, p. 9) and a thickness of final cover material of at least 2 feet is recommended.

The trench method of sanitary landfilling is similar to the area method except that a trench is excavated in the ground to provide a site for depositing the refuse. The material removed from a subsequent trench is used as cover material for a trench being filled with refuse (fig. 2). Normally the trench method is not used in landfills where the water table lies at shallow depths.

Depression landfills are those situated in natural or manmade sags in the topography. Thus, ravines, sand and gravel pits, and quarries would fall in this category. Commonly the total depth of compacted refuse in a depression-type landfill will be greater than in the area-type landfill. The unit-cell method of construction of the landfill is used in a manner similar to that used in the area or trench method. Ravines are usually filled by starting at the upper end and working downslope. If the surficial soil is suitable, cover material may be obtained from the upper sides of the ravine walls.

Factors in Site Selection

Several factors are involved in site selection of sanitary landfills. These are discussed in some detail in the report by the American Public Works Association (1966, p. 91-96). These factors are reviewed briefly as follows:

1) *Land requirements*—Sufficient land should be available to meet the expected space requirements for the population to be served. A rule of thumb is that about 1 acre of land is required for the solid wastes generated by 10,000 persons in 1 year, if the depth of filling is to be limited to about 7 feet. Of course, where greater depths of fill are plannned a larger quantity of waste could be handled per acre.
2) *Character of the land*—The land at the site should not be so rocky or swampy that equipment might be damaged or bogged down when filling operations are attempted.
3) *Availability of cover material*—A suitable and adequate source of cover material should be available at the site or at an economical haul distance from the site. The ideal soil for a sanitary landfill operation should contain 50 to 60 percent sand and about 20 to 25 percent silt and clay, in roughly equal proportions (Committee on Sanitary Landfill Practice, p. 5).
4) *Proximity to residences and industry*—Locating a sanitary landfill close to residences, commercial establishments, or industrial plants can and frequently does result in strong objections by the concerned parties. This is largely because of the association of the sanitary landfill practice with the older, open-type garbage dump, and the public's fear that the use of land for this purpose will depreciate its value as well as the value of the surrounding land. Nearby residents and landowners have a proper concern for possible nuisances and health hazards. The American Public Works Association (1966, p. 93) recommends that, if the landfill is to contain garbage, it should be located not closer than 0.25 mile from the nearest residence, factory, or store, unless unusual circumstances make a closer location both desirable and acceptable.
5) *Accessibility*—The site should have two or more access roads, so that if one road is temporarily unusable the site is still accessible. In metropolitan areas the road plan should be such that the haul trucks can avoid heavily traveled commercial or residential sections.
6) *Future land uses*—The planned improvement of a site by filling and covering is said to be one of the major economic advantages of the sanitary-landfill method of

disposal. Many communities have turned completed landfills into parks, athletic fields, and play areas. Others have used these sites to extend airport runways or as a locale for industrial buildings (American Public Works Association, 1966, p. 94). However, construction of residences on former landfills is usually discouraged, and the construction of large buildings usually is permitted only after careful engineering studies. Some sanitary landfills and dumps are situated on swampy flood plans of rivers. The proponents of locating a landfill or dump on such a site often indicate that after the completion of the fill, the land will be elevated, less subject to flooding, and a greater economic value. However, flood control is essentially a space-allocation problem and, under natural conditions, flood plains and swamps provide natural channel-storage areas for surplus water (Northeastern Illinois Metropolitan Area Planning Commission, 1963, p. 27). If these areas are filled they are no longer available for flood storage and the space needed to accommodate flood waters will be obtained by higher flood stages and increased upstream flood heights. This may result in flooding of upstream areas not formerly subject to floods.

7) *Zoning regulations*—It is reported by the American Public Works Association that many cities in the United States (60 percent of 742 cities studied) are restricted from acquiring sanitary-landfill sites by their zoning regulations. However, the Michigan Supreme Court has ruled that a sanitary landfill is not an ultimate land use and therefore cannot be prohibited on the basis of zoning laws (American Public Works Association, 1966, p. 95).
8) *Pollution of water resources*—The selection of a site for a sanitary landfill should take into consideration those geohydrologic factors that will minimize pollution of ground- or surface-water resources. The geohydrologic factors include climate, topography, drainage, soils, and geology. The remainder of this report is concerned with these elements. The complex interaction of these factors at the waste-disposal site determines the extent and degree of pollution that will occur.

EFFECT OF SOLID-WASTE DISPOSAL ON THE GEOHYDROLOGIC ENVIRONMENT

One of the major changes in the land-water environment resulting from solid-waste disposal in a sanitary landfill is the actual change in the topography of the land resulting from the fill operation. In fact, this element is considered to be one of the advantages of the sanitary landfill over other methods of waste disposal. The report of the Committee on Sanitary Landfill Practice (1959, p. 3) discusses this matter as follows:

> "As its name implies, a completed landfill project will raise the previous elevation of the ground unless large quantities of earth are disposed of elsewhere. Swampy areas, ravines, abandoned borrow pits, and low-lying areas are thus natural locations for a sanitary-landfill operation. A well planned program may result in the reclamation of such lands for many beneficial community projects. Completed landfills have been used for golf courses, parks, parking lots, levee improvements, airports, runway extensions, and, in some instances, for building sites."

A golf course was built on a completed landfill near Cockeysville in Baltimore County (Shields, 1969, p. 4) and a park and a zoo were constructed on a landfill along a stream in Salisbury, Maryland (Wicomico County).

Some of the possible changes in the land-water environment resulting from solid-waste disposal on the land are:

1) Elimination of flood plains or swamps along stream valleys or near estuaries. The elimination of these features may affect the flow characteristics of the watercourse, as well as affecting the local biota (destruction of breeding grounds, for example).
2) A change in the permeability of the landfill cover relative to the preexisting soil, possibly resulting in a decrease or an increase in overland flow from the site. If an impervious cover is used, an increase in overland flow from the site may result.
3) A possibly significant change in the chemical and physical character of the ground water issuing from the landfill.

In summary, landfilling may change preexisting rates of runoff and infiltration in the locale of the fill. Infiltration and runoff, important elements of the hydrologic cycle, are discussed briefly in the following section of the report.

The Hydrologic Cycle in Maryland

Briefly, the hydrologic cycle is the earth's gigantic water-circulating system operated chiefly by the energy of the sun. The elements of the hydrologic cycle may be expressed by the hydrologic equation, which, in simplified form, is:

$$P = R + ET \pm S_{(GW)}$$

P = precipitation (source of nearly all the water)

R = runoff (that part of precipitation that appears in surface streams)

ET = evapotranspiration (water returned to the atmosphere)

$S_{(GW)}$ = change in ground-water storage (the result of infiltration of precipitation to the ground-water reservoir)

Although landfills, dumps, or incinerator waste piles may locally change some of the above components of the hydrologic cycle, their chief effect is to degrade the chemical and bacteriologic quality of ground or surface water draining from them. Water from precipitation on a properly earth-covered landfill might be expected to run off relatively unchanged in chemical, physical, and bacteriologic character except in those elements that it would normally acquire from surface soil or cover. However, some degree of perviousness exists in nearly all types of cover material, and some precipitation will thus infiltrate into the fill material. Changes in the chemical, physical, and bacteriologic character of this infiltrating water will depend on the rates of infiltration, the time the water is in contact with the fill materials, and the nature of the materials. The infiltrating water may be expected to be degraded in quality. Where the base of a sanitary landfill lies at or near the top of a preexisting saturated zone, the upper limit of this zone may rise and cause a more or less permanent spring or seep zone to develop at the base of the fill.

Maryland lies in the humid eastern part of the United States, and precipitation is normally adequate to provide sufficient water to replenish the

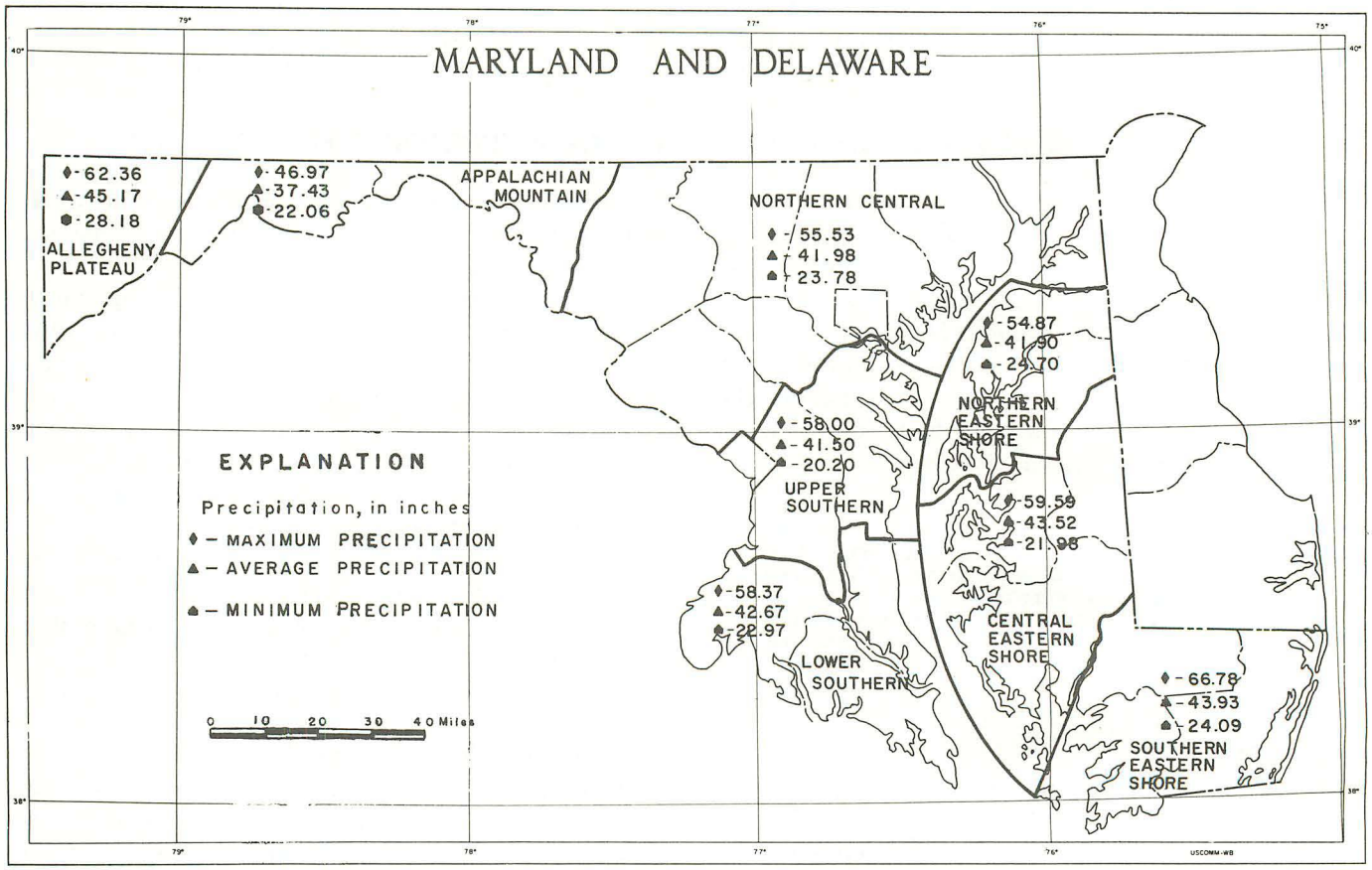


Figure 3. Map showing average, maximum, and minimum annual precipitation for each of Maryland's climatic divisions for 1929-68.

seasonal soil-moisture deficiency and to recharge the ground-water reservoirs or aquifers several times during a normal year. The average precipitation in the State ranges from slightly more than 45 inches in the Allegheny Plateau to slightly more than 37 inches in the adjoining Appalachian Mountain division (See fig. 3). The 8-inch difference in average annual precipitation between the two climatic divisions of the State is attributed by Schweisow and others (1970, p. 23) to the "rain shadow" effect on the prevailing moisture-laden westerly winds as they pass eastward over the higher elevations of the Allegheny Plateau.

The lowest annual precipitation (22.06 inches) was recorded in the Applalachian Mountain division, and the highest (66.78 inches) was recorded in the southern Eastern Shore division. Thus, the extreme values in the entire State differ from one another by more than 44 inches. The disposition of the precipitation according to components of evapotranspiration and runoff are given in table 4, compiled from studies of the hydrologic cycle in four Maryland stream basins.

The annual precipitation in the four basins ranged from 41.4 to 43.6 inches, and the evapotranspiration ranged from 25.2 to 30.9 inches (61 to 71 percent of the precipitation). Total annual runoff ranged from 12.6 to 17.1 inches (29 to 40 percent of precipitation) and ground-water runoff, or effective long-term recharge to the ground-water reservoirs, ranged from 7.4 to 11.3 inches (or 17 to 27 percent of the precipitation). Actually the ground-water recharge was somewhat greater than the values given above because the method of estimation did not account for the water that entered the ground-water reservoirs but became ground-water evapotranspiration instead of streamflow leaving the basin.

The major significance of the hydrologic budget with regard to this study is to indicate that, for Maryland as a whole, at least 10 inches of precipitation infiltrates annually into the surficial aquifers. This quantity of water amounts to about 174 million gallons per year per square mile, or about 470,000 gallons per day per square mile. Because of imprecision in the method of obtaining this figure, an average value of long-term ground-water recharge for the entire State might be on the order of 500,000 gallons per day per square mile. In some instances, more than 10 inches of precipitation may infiltrate into a sanitary landfill, as Hughes and others (1969, p. 21) report a ground-water recharge rate of 15 inches per year

Table 4. Elements of the hydrologic cycle in four river basins in Maryland.

River basin	Annual precipitation		Annual evapotranspiration		Total runoff		Ground-water runoff or effective recharge	
	inches	percent of precipitation	inches	percent of precipitation	inches	percent of precipitation	inches	percent precipitation
Beaverdam Creek near Salisbury 1/	41.4	100	25.2	61	14.9	36 2/	10.7	26
Patuxent River basin 3/	43.6	100	28.0	64	15.6	36	7.4 4/	17
Little Gunpowder Falls basin 5/	42.6	100	25.5	60	17.1	40	11.3	27
Rock Creek basin 6/	43.5	100	30.9	71	12.6	29	8.5	20
Mean (rounded)	43	100	27	63	16	37	10 7/	23

1/ From Rasmussen and Andreasen, 1959, p. 98.
2/ During the 2-year period of the study 3 percent of the precipitation became ground-water storage.
3/ From Crooks, O'Bryan, and others, 1967, Sheet 2.
4/ Maximum value in range given by Crooks, O'Bryan, and others, Sheet 2.
5/ From Dingman and Ferguson, 1956, p. 48.
6/ From Dingman and Meyer, 1954, p. 39.
7/ 1.3 inches of estimated runoff added to Beaverdam Creek for use in computation of mean of four basins to adjust for loss to ground-water storage.

at a landfill site in northern Illinois where the average annual precipitation was 33 inches. Ground-water recharge is significant with regard to solid-waste disposal, as it indicates the possible rates of accretion of water to sanitary landfills and refuse dumps and the corresponding potential for leachate generation from them.

Owing to local variations in soil and rock type, vegetal cover, topographic slope, and the effect of the works of man, a wide range occurs in the rates of infiltration and recharge from one locality to another, even within comparatively small basins. Where the surficial materials are clay or shale, rates of infiltration to the ground-water reservoir will be slow. Such areas are characterized by flash floods, rapid rates of runoff, and the drying up of streams during droughts of even brief duration. The storage coefficient of clay or shale is very small, commonly less than 1 or 2 percent, and such terranes have little stored water to maintain the flow of streams between precipitation events.

Degradation of the Hydrologic Environment by Solid Wastes

A landfill may be considered a source of heterogeneous unwanted material subject to decomposition or attack by water, air, and bacteria. In a humid region, decomposition results in leachate and both odorous and nonodorous gases that tend to degrade the water and air environment. Leachate is generated as a result of the infiltration of precipitation or overland runoff into the landfill, or by the movement of ground or surface water through the landfill, where its base may lie below the zone of saturation.

It has been reported that at a landfill site in Great Britain, a year's rainfall of 25 inches resulted in 10 inches of percolation into a landfill (Ministry of Housing and Local Government, 1961, p. 11). The particular landfill reported upon had an 18-inch soil cover and a flat surface.

Nearly every hydrologically oriented study of refuse dumps or sanitary landfills in humid regions indicates that leachate is generated in the fill where a sufficient quantity of water comes in contact with the fill materials. Experience in Illinois in level topography has shown that a ground-water mound may build up within a large soil-covered refuse pile (Hughes, 1967, p. 14). Discharge of leachate from a mound of this type may be radial and could result in the degradation of the water in several watercourses.

Some leachate may emerge from the landfill into a nearby stream or ditch, where it is subject to dilution and attenuation. However, it may also travel for considerable distances through the ground before ultimately discharging into streams or other water-filled depressions. During its travel through the ground it will be diluted by the native ground water. The movement of leachate-degraded ground water will be in response to the laws of ground-water flow, but the rates of movement may be quite slow. However, as such water cannot be readily observed at the surface it may remain undetected unless it enters a nearby well or issues from a spring. The possibility of pollution of wells or springs in the vicinity of landfill sites is obvious and pollution of the shallow aquifers has been demonstrated at landfills in northern Illinois (Hughes and others 1969, p. 22, p. 30, and p. 36) and elsewhere. It is, therefore, pertinent to describe briefly the nature of ground-water flow systems in Maryland.

Movement of Ground Water

A ground-water flow system involves the progressive movement of water through the earth. The energy causing the movement of the water is potential energy, or the energy of position (gravitational energy). The source of all nonsaline ground water in Maryland is precipitation. Precipitation on topographically high sites possesses energy driving the water to seek a lower position. Water entering the ground on hilltops and hillsides moves through the earth materials toward valleys or other depressions, where it discharges from the ground to become streamflow. Although both recharge and discharge occur throughout an area, ground-water recharge tends to predominate on the hilltops and hillsides, and ground-water discharge tends to predominate on the valley floors and other topographically low sites.

Downward-moving water will fill the interconnected voids or pores in the rocks, resulting in a condition of saturation. The upper surface of the zone of saturation is the water table. The water table in the surficial aquifers is generally highest in the late winter and early spring and lowest in the late summer and fall, fluctuating in annual seasonal cycles. It fluctuates in response to changes in the rate of downward replenishment, in response to changes in the rate of withdrawal by means of wells, and in response to the water demands of plants whose roots penetrate below the

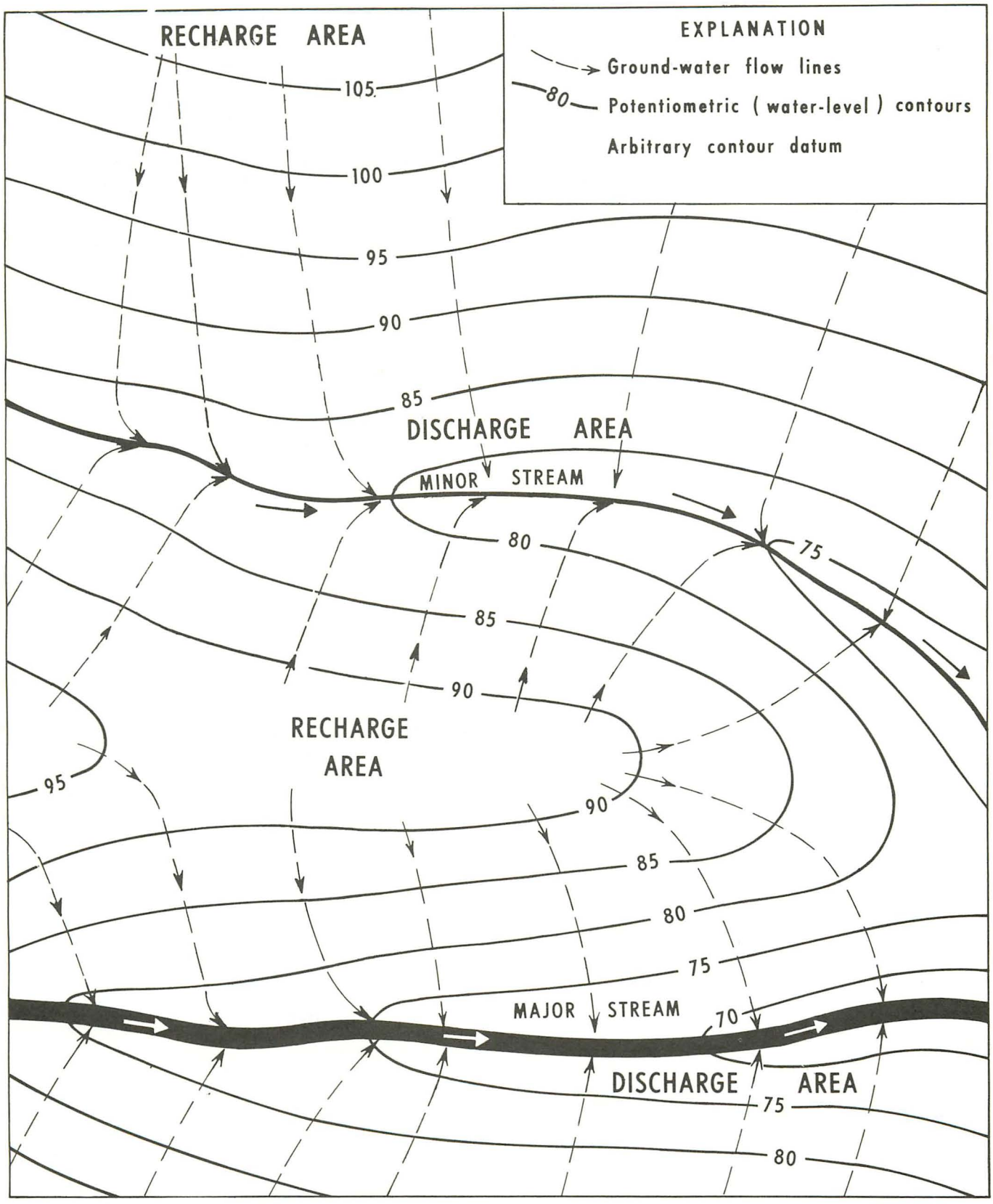


Figure 4. Map showing idealized ground-water recharge and discharge areas, potentiometric contours, and ground-water flow lines in an area drained by a major stream and a minor tributary stream.

water table. The surface may also change as a result of upward capillary migration of water from the saturated zone to replace water used by plants whose roots do not penetrate below the water table. The zone above the saturated zone has been termed the zone of aeration, and the moisture in it has been termed "suspended water."

As mentioned earlier, normally the hill tops can be considered as ground-water recharge areas and the stream valleys as ground-water discharge areas. Figure 4 shows an idealized map configuration of the position of the water table throughout a small area drained by a major stream and a minor tributary stream. The area is assumed to be underlain by homogeneous earth materials possessing rather uniform hydraulic properties. The map shows the potentiometric contours and the ground-water flow lines, which cross the contours at approximately right angles. The flow lines show the direction and path of movement of a particle of water moving through the flow system toward a nearby stream.

Figure 5 shows in the vertical dimension the idealized pattern of ground-water flow into a

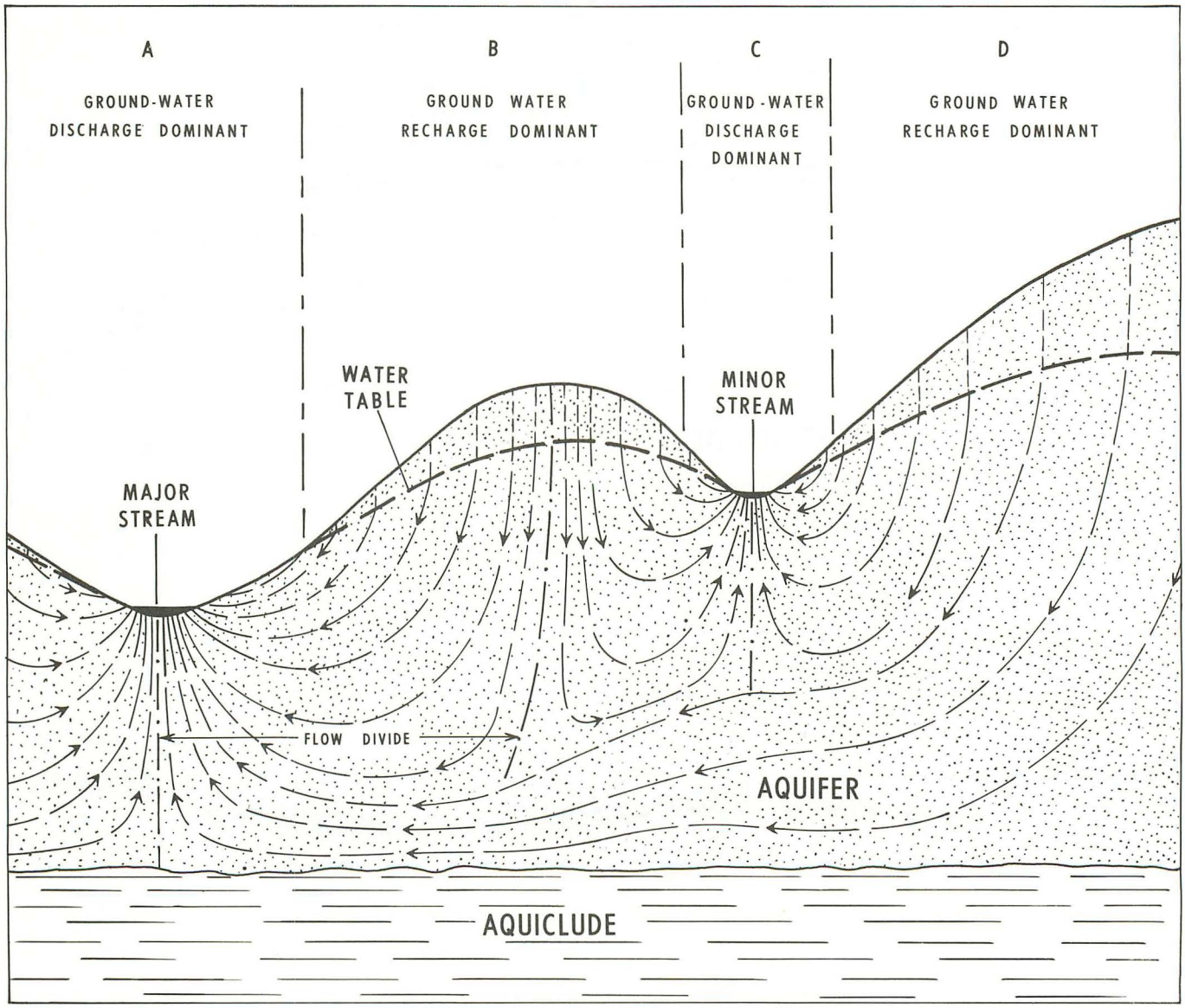


Figure 5. Schematic section across a major and a minor stream valley showing idealized pattern of ground-water flow lines in an aquifer inderlain by a confining layer.

major and a minor tributary stream. Figure 5 is based on a situation similar to that portrayed in figure 4, except that here the aquifer is underlain by a confining layer, assumed to be sufficiently impervious to deflect the flow lines in the manner indicated. The figure shows that, under the conditions depicted, ground water will flow from the adjacent hillsides into both the major and the minor stream. Some water, however, will flow directly into the major stream from the high upland area on the right side of the figure. Thus, even in this idealized, relatively simple case, a complex system of ground-water flow cells will exist. The figure emphasizes the importance of the local geology and topography in determining directions and rates of ground-water flow.

In the real world an almost infinite number of variations in the flow pattern shown in figure 5 can occur as a result of differences in the local geologic framework. An example of one possible variation is shown in figure 6, where the topographic situation is identical to that in figure 5, but the underlying geologic framework differs. In this instance, an aquiclude occupies the interval between the level of the major stream and the minor stream and lies to the right of the major stream. As a result of this circumstance two aquifers are present, but only the upper aquifer receives direct recharge from precipitation. Artesian conditions prevail in the lower aquifer, whose source of recharge is from a distant area to the right. In reality few rocks are perfect confining layers, so that some vertical leakage may

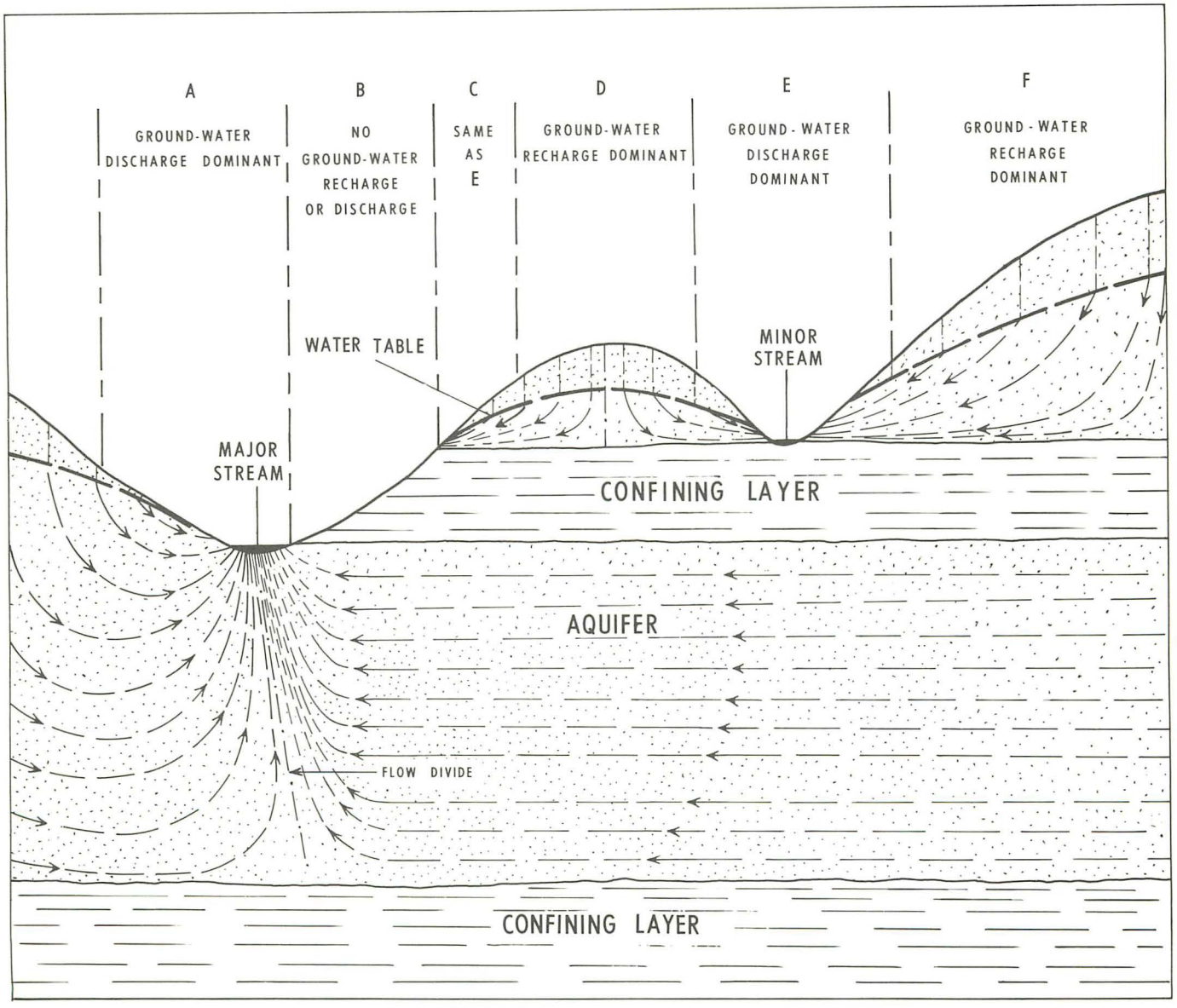


Figure 6. Schematic section across a major and a minor stream valley showing pattern of ground-water flow where a shallow confining layer separates two aquifers.

take place through these layers to provide at least minimal recharge to the artesian system. With regard to the problem of ground-water pollution, figure 6 shows how a water-borne pollutant entering the ground-water flow system of the surficial aquifer in the vicinity of the minor stream could be prevented from entering the flow system of the deeper aquifer by the presence of the aquiclude or confining stratum.

The movement of water into a landfill or into the soil beneath a landfill is governed by the infiltration capacity of the soil, including in this term the disturbed earth material used as a cover for the landfill cells. *Infiltration capacity* is defined as the maximum rate at which a soil in any given condition is capable of absorbing water (Wisler and Brater, 1959 p. 104). The infiltration capacity of landfill cover, however, is affected to a degree by compaction by the earth-moving equipment. It is difficult to evaluate this factor. Infiltration capacities of uncompacted soils are characterized by extreme variability, both from one place to another and from one time to another at a given location. The chief factor causing a downward movement of water through the soil is gravity. Water moves through the soil as moisture until it reaches the zone of saturation. The moisture content of the soil also governs its infiltration capacity. If the soil is quite dry at the beginning of precipitation, the wetting of the top layer creates a strong capillary potential beneath the surface, which supplements gravity in causing infiltration. However, an infiltration-retarding effect may be caused by the swelling of colloids in the soil when it is initially wetted. Colloidal swelling tends to reduce the infiltration capacity during the initial period of rainfall. Also, the energy imparted by the falling raindrops may mechanically compact a soil at the surface and somewhat reduce its infiltration capacity. Additionally, fine particles at the surface may be carried downward into the soil pores, where they are deposited in the coarser voids, producing clogging. Slope of the soil surface is an important factor governing its infiltration capacity. Steeper slopes promote greater runoff and tend to lessen infiltration.

Vegetal cover is an important factor governing the infiltration capacity of a soil. The presence of a dense vegetal cover, such as grass or forest, tends to promote infiltration, probably because the cover increases the looseness of the soil and promotes retention of moisture. In fact, some investigators believe the effect of cover may be a more important determinant of infiltration capacity than soil type (Wisler and Brater, 1959, p. 106).

Temperature affects the infiltration capacity of the soil. During warm seasons the viscosity of the water is less, and infiltration rates are somewhat greater than in cold seasons. Furthermore, during the winter, rain may fall on ground having a temperature below freezing, resulting in an ice crust being formed at the ground surface that effectively retards infiltration.

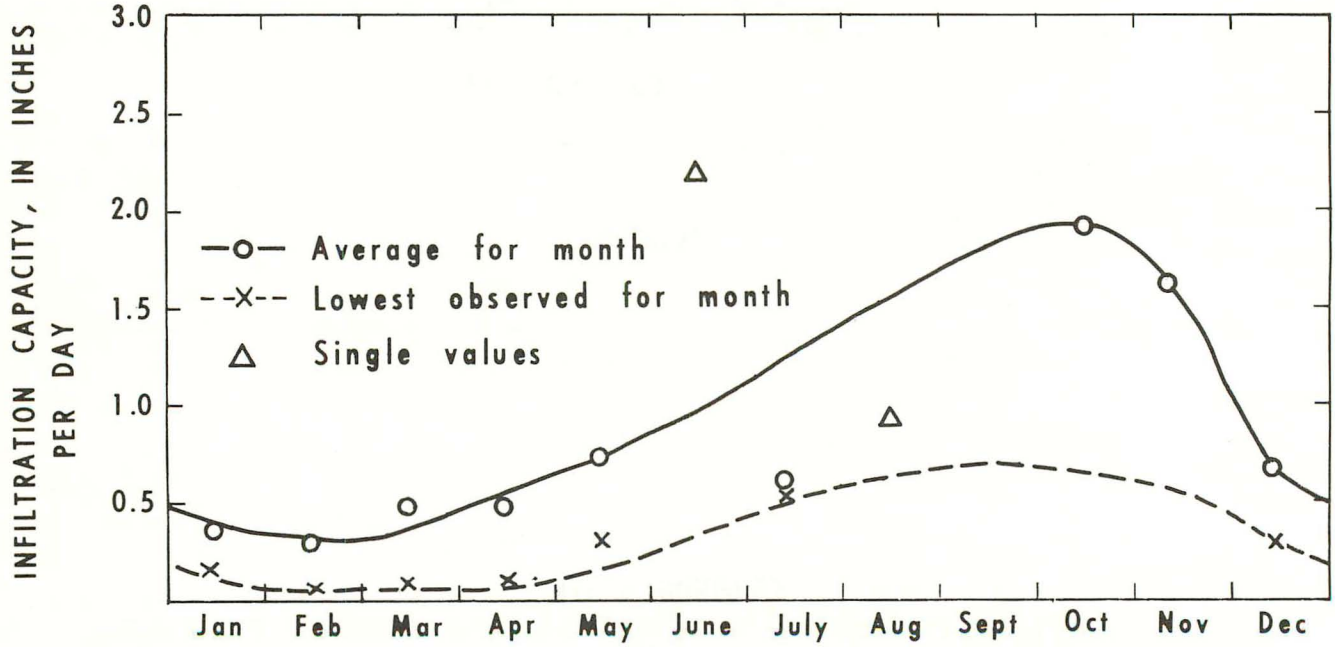


Figure 7. Graph showing seasonal range in infiltration capacity of soils in a small watershed in Illinois.

Table 5. Infiltration rates for different soils as measured by infiltrometer rings in third hour of a wet run (after Free, Browning, and Musgrave, 1940).

Soil type	Porosity (percent)		Infiltration rate		
	Total	Non-capillary (specific yield)	Inches per hour	Feet per day	Centi-meters per hour
Gravelly silt loam 1/	54.9	28.1	4.96	9.92	12.60
Clay loam	61.1	36.3	3.98	7.96	10.11
Silt loam	57.0	32.0	2.09	4.18	5.31
Sandy loam	49.6	26.3	1.93	3.86	4.90
Clay (eroded)	54.3	28.7	1.78	3.56	4.52
Sandy clay loam	48.8	27.7	1.42	2.84	3.61
Silty clay loam	50.8	24.3	.72	1.44	1.83
Stony silt loam	59.7	32.6	.55	1.10	1.40
Fine sandy loam	41.5	24.2	.55	1.10	1.40
Very fine sandy loam	49.6	23.4	.51	1.02	1.29
Loam	45.7	17.2	.50	1.00	1.27
Sandy clay	42.9	16.9	.05	.10	.13
Heavy clay	57.8	27.0	.02	.04	.05
Light clay	47.0	19.8	.00	.00	.00
Clayey silt loam	49.4	17.6	.00	.00	.00

1/ Loam is a soil consisting of a relatively even mixture of sand, silt, and clay size particles. Sand particles have a mean diameter of 2.0 to 0.05 mm, silt particles a diameter of 0.05 to 0.005 mm, and clay particles less than 0.005 mm.

Animal and insect borings, as well as decayed roots, provide passageways through the soil and consequently may increase its infiltration capacity.

The above discussion indicates that the various factors affecting infiltration capacity are not constant throughout the year. Commonly, the soil-moisture content reaches a maximum in the spring and a minimum in the fall. The effects of vegetal cover and temperature vary seasonally. The seasonal range in infiltration capacity, measured during an 18-month period, is shown in figure 7, where the data were obtained from a study of a small watershed at Edwardsville, Ill. (Horner and Lloyd, 1940, p. 538). The figure shows that infiltration is significantly lower in the winter than in the late summer and early fall in an area climatically similar to Maryland.

The preceding discussion indicates that soils may range widely in their infiltration capacities, both spatially and temporally. Furthermore, the various soil types differ one from the other in their average infiltration rate. Table 5, shows that the infiltration rate of the least permeable soil, a clayey silt loam, was 0.00 foot per day. The most permeable soil, a gravelly silt loam, had an infiltration rate of 9.92 feet per day. The total porosity of the soils ranged from about 41 to 61 percent.

Physical and Chemical Character of Waste Materials

Physical Composition of the Waste: Analyses of the physical and chemical character of municipal solid wastes provide some insight into the nature of the materials that can degrade the water resources. The relatively few analyses available express the results in different classifications making the comparison of the wastes from one area with those from another difficult. Table 6 shows the composition of municipal waste from Chicago during a 2-year period. The data show that paper was the major constituent of the waste, on both a weight and volume basis. Garbage, a possible major source of bacterial pollution, was a minor constituent of the Chicago waste.

Settlement and Compaction: The settlement and compaction of a sanitary landfill depends on the kind of refuse deposited, the degree of compaction achieved by the heavy equipment, and the proportion of earth material used in covering the refuse. The American Public Works Association (1966, p. 126) reported that in 58 cities fill depths ranged from 4 to 20 feet and half the cities reported only slight settlement after deposition of the wastes. However, the remainder of the cities reported that the amount of settlement ranged from 1 to 25 percent after 2 years. In New York City it is reported that initial compaction of the raw refuse from the trucks is about 50 percent; that is—3 cubic yards of truck material compacts to about 1.5 cubic yards of fill material. Further compaction occurs with time as a result of decomposition and void-filling in the refuse. As a result, an initial 3 cubic yards of truck refuse is reduced to about 1 cubic yard of fill material (Amer. Pub. Works Assoc., 1966, p. 128). Experience in New York indicates that at least 90 percent of the total compaction occurs within 2 to 5 years after deposition of the fill.

Table 6. Average composition of municipal waste from Chicago, Ill., by weight and volume (American Public Works Assoc. 1966, p.47).

Constituent[1]	Percent by weight	Percent by volume
Paper	56.5	74.9
Ashes	18.7	6.0
Metal and glass	14.8	13.3
Grass	9.6	8.8
Garbage	4.8	2.9

[1] Analyses of 54 samples collected during 1956-58. Each sample consists of one truckload of refuse weighing approximately 3 tons. Average density of samples 13.6 lbs per cu. ft. Percentages are weighted average; not all constituents were determined for each sample.

Landfill Temperatures: Temperatures within a landfill may be expected to be somewhat higher than in the surrounding earth materials. The rise in temperature results chiefly from the decomposition of the refuse. In fact, the temperature within a fill may be a good measure of the rate at which decomposition of the materials is taking place. Temperature as high as 71°C at a depth of 3 feet has been reported in New York 6 days after emplacement of the refuse. Subsequently, during a 30-day period, the temperature at this depth dropped to 50°C (New York University, 1940, p. 35). The temperature range in a 6-foot-deep fill at Monterey Park, Calif., is reported to have been from 29°C to 50°C during an 18-month period (Committee on Sanitary Engineering Research, 1959, p. 45). Similar measurements in a landfill at Riverside, Calif., showed a temperature range of 29°C to 51°C during a 17-month period. The

maximum temperature was observed 2 days after emplacement of the fill (California State Water Pollution Control Board, 1954, p. 56). Leachate-polluted water issuing from a landfill might have a somewhat higher temperature than normal unpolluted water, at least early in the filling period.

Chemical Composition of the Waste: Few data are available on the chemical character of refuse, and the character of the refuse varies from one site to another as well as at different places within a single site. Obviously, if industrial wastes are incorporated in urban wastes these will affect the chemical character of the material. Table 7 gives the results of a series of analyses of refuse from a landfill at Mandan, N. D. It is not known if these data are representative of many other landfills in the United States, as most investigators report the chemical character of the leachate from a fill rather than the chemical character of the waste material itself.

Decomposition and Leaching

With regard to the decomposition of landfills the following information is provided by the American Public Works Association, 1966 (p. 128):

> "Decomposition of landfills depends on many factors, including permeability of cover material, depth of burial, rainfall, moisture content and putrescibility of the refuse, and degree of compaction.
>
> In Richmond, Va., and in New Orleans, La., refuse in landfills has reportedly been decomposed in three to four years. As a general rule, however, climatological and operational conditions are such that much slower decomposition may be expected.
>
> Garbage and rubbish are made up principally of carbohydrates, fats, and proteins. The elements of carbon, oxygen, hydrogen, nitrogen and sulfur are most prevalent in organic refuse. Refuse in sanitary landfills decomposes slowly through the anaerobic process of decomposition. Microorganisms of the soil and the refuse are responsible for this decomposition. The organic acids of decomposition tend to pickle refuse, especially in deep fills. After many successive breakdowns through the feeding and growth of microorganisms and the action of their enzymes, the end-products are gases and humus material. The gaseous end-products are represented by the following diagram:"

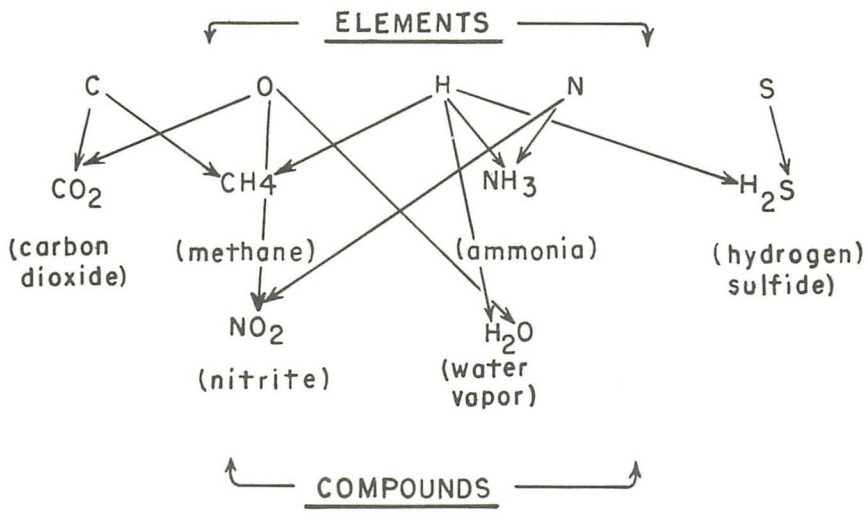


Table 7. Analyses of samples of material from a sanitary landfill at Mandan, N. D. (from Weaver, 1961, p. 105)

Constituent	A Average of 10 samples of refuse ready for burial collected monthly during period Feb. to Dec. 1950	B Sample after 1 year of burial 1950	C Sample after 2 years of burial 1951	D Approximate percent change column A to C + = gain - = loss
Total solids (percent)	44.9	37.4	66.1	+ 47
Volatile solids (percent)	22.5	21.9	21.5	- 1
Fixed solids (percent)	22.4	25.4	44.6	+ 50
pH	6.5	6.0	6.6	---
Acidity to phenolphthalein as $CaCO_3$ (mg/l)	2,239	1,853	146	- 93
Alkalinity to M.O. as $CaCO_3$ (mg/l)	13,130	727	2,300	- 82
NH_4 as N (mg/l)	289	320	163	- 44
Total N as N (mg/l)	5,357	3,118	2,000	- 63
Oxygen consumed (mg/l)	270,970	208,000	230,000	- 15
5-day BOD (mg/l)	77,050	30,000	23,500	- 70

Some of the gases escape from the fill by upward migration through the cover, and some dissolve in percolating water migrating downward and laterally into the ground-water flow system or into streams. The dissolved gases are further involved in the complex chemistry of the landfill and the leachate generated by it, but a discussion of this subject is beyond the scope of this report.

Methane and carbon dioxide are the principal gases produced during the decomposition of refuse in a fill. Studies of a Seattle landfill (Dunn, 1957, p. 790) indicate that the greatest amount of gas was produced from refuse that was 6 months to 2 years old where the refuse was in layers 20 to 40 feet deep, or where there were two or more layers about 8 feet thick. It was also observed that gas production declined as the moisture content of the refuse decreased. Carbon dioxide dissolves readily in water, forming carbonic acid, which, in turn, dissolves calcium and magnesium carbonates (limestone and dolomite) and increases the hardness of the leachate. Other cations in solution may

Table 8. Range and average of chemical constituents and properties from a leachate bin at Riverside, Calif. (from California State Water Pollution Control Board, 1954, p. 49).

Constituent	Minimum (mg/l except for pH)	Maximum (mg/l except for pH)	Average of 35 samples (mg/l except for pH)
Calcium carbonate ($CaCO_3$) as $CaCO_3$	650	8,120	3,633
Alkalinity, as $CaCO_3$	730	9,520	4,629
Calcium (Ca)	115	2,570	1,047
Magnesium (Mg)	64	410	181
Sodium (Na)	85	1,805	940
Potassium (K)	28	1,860	959
Iron (Fe), total	6	305	110
Iron (Fe), ferrous	2	93	24
Chloride (Cl)	96	2,350	1,314
SO_4	39	730	248
Phosphate (PO_4)	.2	29.0	7.0
Ammonia as N (nitrogen)	.2	895	437
BOD	81	33,100	10,850
pH	5.6	7.6	6.5

also contribute to the hardness of the leachate.

Chemical Character of Leachate: As would be expected, the chemical character of leachate varies widely, depending on the character of the refuse, the character of the earth material used as a cover, and on climatic or meteorologic factors. Relatively few data are in the literature regarding the nature of refuse leachate. However, one of the first detailed studies in the United States of leaching from a sanitary landfill was made by the University of Southern California at Riverside (California State Water Pollution Control Board, 1954, p. 41). The test apparatus consisted of a 3 x 4 x 14-foot bin incorporated into a landfill and equipped with an underdrain for the collection of leachate. The bin was filled with a representative

load of mixed domestic refuse and during a period of about 18 months, 1,800 gallons of water (90 inches) were added to the bin. The source of the added water was a local well. Table 8 shows the minimum, maximum, and average values of the constitutents analyzed:

The California report (p. 48) has the following comments regarding the study of leachate from the bin at Riverside:

> "Initial leach samples collected after the application of 320 gallons of water were dark green in color, and had an extremely offensive odor. The color of subsequent samples varied from green to amber, while the odor remained unchanged. It was only over the last six months that the leach samples were but slightly colored and had little odor.
>
> The concentration of both organic and inorganic components was very high in the first sample of leach and continued to rise for approximately five weeks, after which a decline in concentrations began. The decline was due to the fact that the rate of decomposition to soluble compounds had reached an equilibrium with the rate of leaching. At the end of the first six months, the concentration of most ions had dropped to about one-half the maximum values. The reduction of ion concentrations continued until the advent of hot, summer weather. During the months of July and August, when air and bin temperatures were highest, concentrations of sodium, potassium, calcium and chloride increased markedly. With the exception of chloride, there then followed a generally steady decline until the concentrations reached values which have been fairly consistent over the last six months. Chloride concentrations fluctuated considerably.
>
> The magnesium and sulfate concentrations did not undergo a similar sharp increase during the summer months, but steadily declined to values which have been consistently low over the last eight months.
>
> Inorganic phosphate showed a temporary decrease, which was to be expected on the basis of synthesis of organic phosphates by actively multiplying bacterial cells. At the same time, there was a considerable development of acid, as shown by the decrease in pH.
>
> Approximately one year elapsed before the leach became alkaline.
>
> The initial increase in concentration of organic matter shown by the peak BOD of 33,100 mg/l represents the conversion of complex and insoluble organic compounds into more simple substances by the action of microorganisms. These decomposition products still had a high BOD after the first ten months. A very large drop in BOD then occurred, and within three months' time values as low as 375 mg/l were being found. At the conclusion of the study, the BODs had dropped to 81 mg/l.
>
> The relatively slow reduction of inorganic components—even such soluble ions as sodium and chloride—shows that there is a large degree of physical holdback of the inorganic substances. Of special interest is the consistently high ratio of potassium to sodium.
>
> In general, the ion concentration reached a maximum of 10 to 20 times the concentration found in the applied water. Ammonia nitrogen and phosphorus, which are not usually present in unpolluted water, were always present in the leach in extremely high concentrations because of their presence as essential components of living organic material. The initial high iron content is attributable to the acidity, and in all probability to complex formation with the organic matter."

A more recent study to determine the character of leachate from simulated landfills of various thickness is reported by Quasim and Burchinal (1970, p. 371-379). This experiment consisted of analyzing the leachate from three concrete cylinders filled with municipal refuse from Morgantown, W. Va. The cylinders were 3 feet in diameter and filled with ground refuse to depths of 2.6, 6.4, and 10.1 feet. Metals, cans, bottles, and large pieces of wood were removed from the refuse before it was ground and placed in the cylinders. The simulated fills were covered with a 1-foot layer of soil, and approximately 46 inches of water were applied to the cylinders during a 163-day period. The quantities of leachate recovered from the cylinders ranged from 20 to 37 inches. Table 9, which follows, shows the chemical analyses of the constituents extracted from the three simulated landfills:

Regarding the results of the leaching experiment, the authors state:

> "As water moved through the refuse beds, a transfer of materials from the solid phase to the liquid phase occurred, with the material being retained in thin liquid films surrounding the particles. Since the fills were being leached continuously during the operating period, the rate of decomposition controlled the rate of extraction of materials; however, early surges of water through the beds seemed to produce a washing action, which extracted the weakly absorbed material already present on the solid particles. The initial decrease in the concentrations of leaching materials indicated this phenomenon. The second gradual increase followed by a prolonged decline in concentrations resulted from the release of decomposition materials.
>
> The leach samples showed a slight increase in pH during the first two weeks, followed by a rapid decrease. The minimum pH of 5.3 was attained in 79, 135, and 93 days in cylinders A, B, and C, respectively, indicating the accumulation of short-chain fatty acids. The bacterial population was affected adversely by the pH change and accumulation of end products. The highest bacterial populations in the three cylinders corresponded to the occurrence of maximum pH values after which populations followed a constant decline, similar to that of pH. Decreases of 91.7, 96.9, and 97.3 percent occurred in bacterial populations in cylinders A, B, and C, respectively, by the end of testing period. . . .
>
> Ammonia and organic nitrogen compounds were present in extremely high concentrations in the leachate due to their presence as components of living organic material. Peak values of organic nitrogen occurred in the initial samples. As time progressed, organic nitrogen gradually was converted to ammonia nitrogen. It took about 10 weeks to attain maximum concentration of ammonia nitrogen. . . .
>
> Sulfate in leaching waters came from ash, in organic matter, and organic sulfur produced by microbial decomposition of amino acids.
>
> High iron content in the leach from simulated landfills is attributed to the low pH. Maximum concentrations of 175, 546, and 860 mg/l in cylinders A, B, and C occurred after 65, 107, and 51 days, respectively . . .

Table 9. Maximum concentrations and total quantities of constituents extracted from three simulated landfill cylinders at Morgantown, W. Va. (from Quasim and Burchinal, 1970, p. 377).

Constituent and property	Maximum concentration (mg/l except for pH)			Quantities extracted during period of test (pounds)		
	Cylinder A 1/	Cylinder B 2/	Cylinder C 3/	Cylinder A	Cylinder B	Cylinder C
Acidity ($CaCO_3$)	1,800	2,960	3,510	1.38	2.44	2.69
Alkalinity ($CaCO_3$)	10,630	16,200	20,850	9.54	15.95	15.21
BOD (5 day, 20°C)	14,760	26,940	33,360	12.84	23.70	23.59
Calcium (Ca)	3,030	4,080	2,790	2.66	3.74	2.48
Chloride (Cl)	951	2,000	2,310	.78	1.32	1.54
Hardness ($CaCO_3$)	7,600	13,100	10,950	7.57	11.60	8.05
Total iron (Fe)	175	546	860	.15	.50	.71
Magnesium (Mg)	176	387	420	.17	.33	.33
Ammonia nitrogen (N)	473	756	1,106	.40	.74	.86
Organic nitrogen (N)	288	664	1,416	.22	.50	.64
Phosphate (PO_4)	154	147	128	.12	.04	.10
Potassium (K)	1,050	2,535	3,770	1.07	1.88	3.04
Sodium (Na)	584	1,428	1,439	.53	.98	.90
Solids (organic)	11,385	32,500	32,250	12.58	23.40	23.59
Solids (total)	21,140	49,800	59,200	22.34	39.18	38.48
Sulfate (SO_4)	615	1,002	768	.51	.72	.53
Tannin and lignin (tannic acid)	389	756	1,120	.37	.75	.97
pH	6.3	6.48	5.88	--	--	--

1/ Cylinder A; total thickness of fill and earth cover = 3.6 feet; quantity of leachate recovered = 37 inches (163 gallons).

2/ Cylinder B; total thickness of fill and earth cover = 7.6 feet; quantity of leachate recovered = 31 inches (135 gallons).

3/ Cylinder C; total thickness of fill and earth cover = 11.6 feet; quantity of leachate recovered = 20 inches (87 gallons).

Leach samples were rich in sodium, potassium, calcium, magnesium, and chloride. There seemed to be considerable mechanical holding even in the case of inorganic ions such as chloride and sodium. Time versus concentration curves for sodium, potassium, and magnesium were very similar, increasing to a peak value within two to four weeks and then decreasing gradually. Initial leach samples had the highest chloride concentrations. Calcium curves for three cylinders reached a maximum value after about eight to nine weeks. Total hardness followed essentially the same pattern as calcium. Nearly 80 percent of the hardness in leach samples is attributable to calcium hardness alone."

On the basis of their study, Quasim and Burchinal (1970, p. 379) concluded that the concentrations of leachate produced from landfills were not directly proportional to the depth of refuse fill, but leachates from a deeper fill should be more concentrated than those from a shallower fill. Although the Quasim and Burchinal experiments are informative, they may be atypical, as the fill material used did not contain metal objects, glass bottles, or wood. Furthermore, analysis of the hydrologic cycle (table 4) suggests that the quantity of water which could infiltrate landfills in most of the eastern United States would be less than half the 45 inches of water used in the Morgantown experiments.

Bacteriologic Character of Leachate: Few published data are available concerning the bacteriologic characteristics of landfill material, but the bacterial count is assumed to be very high, as bacteria are chiefly responsible for the decomposition within the fill. Weaver (1961, p. 105) indicates that the coliform count at one landfill (site not mentioned) averaged 740,000 per gram of refuse.

Consulting engineers (Green Associates, 1969, p. 7) reported that bacteriologic analyses of five samples of pond water receiving leachate from a landfill near Glen Burnie, Md., showed a coliform density described as "too numerous to count." These samples were collected during August 1969.

Quasim and Burchinal (1970, p. 377), in the previously described simulated landfill study at Morgantown, W. Va., report that the maximum standard bacterial plate count in leachate from the three landfill cylinders ranged from 1.1×10^6 to 10.7×10^6 per 100 milliliters, extremely high values. However, these authors report the MPN[1] of the leachate from the three cylinders decreased markedly during the 163-day period of their test. (See Quasim and Burchinal, 1970, fig. 10, p. 376.)

These reported coliform densities may be compared with the recommended standards of the Technical Advisory Committee on Water Quality Standards (U.S. Federal Water Quality Admin., 1968). For water used for recreational use other than swimming the subcommittee of the Technical Advisory Committee recommends the following limits for fecal coliforms (1968, p. 10):

"In water designated for recreation uses other than primary contact recreation, the Sub-committee recommends that the fecal coliform content as determined by either multiple-tube fermentation or membrane filter techniques, should not exceed a log mean of 1,000/100 ml, nor equal or exceed 2,000/100 ml in more than 10 percent of the samples."

For primary contact recreational waters the Committee recommends the following (p. 12):

"Fecal coliforms should be used as the indicator organism for evaluating the microbiological suitability of recreation waters. As determined by multiple-tube fermentation or membrane filter procedures and based on a minimum of not less than five samples taken over not more than a 30-day period, the fecal coliform content of primary contact recreation waters shall not exceed a log mean of 200/100 ml, nor shall more than 10 percent of total samples during any 30-day period exceed 400/100 ml."

The microbiologic water criteria for public surface-water supplies recommended by the Committee are as follows (1968, Table II-I, p. 20):

	Permissible criteria	Desirable criteria
Coliform organisms	10,000/100 ml	100/100 ml
Fecal coliforms	2,000/100 ml	20/100 ml

Bacteriological sampling results are discussed further in the section of the report describing the sampling of water near three Maryland landfills.

Significance of Field Analyses of Water: During the course of this study field analyses were made of certain chemical and physical properties of waters suspected of being leachate degraded. The description and significance of these analyses are described below.

Total hardness is a measure of the alkaline earth cations in solution, although Hem states (1959, p. 146) that "hardness is a property not strictly allocable to any one constituent. . . . Total hardness supposedly represents the effects of all the substances which react with soap." Hardness is readily determined with a field analysis kit. Studies in Illinois, California, and other states have shown that landfill leachate is characterized by excessive hardness. Hardness of a sample of leachate from a site in Illinois has been reported in excess of 10,000 mg/l (Hughes, Landon, and Farvolden, 1969, p. 85).

[1] Most probable number (MPN) of coliform organisms in a water sample is the density most likely to produce a particular result. A number of portions of different size of the water sample are examined for the presence of coliform organisms. It is assumed that variations follow a probability curve and that the MPN can be determined from a complex mathematical expression (for the formula used refer to Clark and Viessman, 1965, p. 247).

Chloride is present in all natural waters, although in many areas the amounts are small. Precipitation may contain 2-3 mg/l or more of chloride. Thomas and Heidel (1969, p. 7) indicate that in Maryland the concentration of chloride in water used for public water supplies is commonly less than 10 mg/l. The chloride content of raw water in several public water supplies in Maryland cities and towns is (from Thomas and Heidel, 1969):

Place	Source	Chloride content (mg/l)
Baltimore City	Loch Raven and Prettyboy reservoirs	8.9
" "	Liberty reservoir	12
Bel Air	Winters Run	6.2
Cambridge	Drilled wells	6.1
Cumberland	Evitts Creek	2.7
Easton	Drilled wells	2.7
Frederick	Monocacy River	8.3
Hagerstown	Potomac River and mountain springs	17

Chloride in concentrations of several hundred milligrams per liter is not harmful to human beings or most animals, but concentrations in streams or wells in excess of 20-30 mg/l suggest that degradation of the water may be occurring, at least in most areas of Maryland.

Leachate from landfills is characterized by high chloride content. Hughes and others (1969, p. 93) report that two samples of ground water beneath the Woodstock landfill in northern Illinois have a chloride content of about 2,400 mg/l, or about 200 times the normal chloride content of water in the aquifer. Andersen and Dornbush (1967, p. 465) report the mean chloride content of water samples from three shallow wells near the landfill at Brookings, S. D., was 320 mg/l. These authors consider analyses of chloride content a most useful guide to degradation of water by leachate from a landfill. They make the following statement in this regard (p. 466):

> "The chloride ion determination was the most sensitive parameter used. Wells in the center of the refuse area showed an increase of over 50 times the chloride concentrations found in the native water. The chloride ion is perhaps the most suitable indicator of leaching from a refuse dump because: it is easily measured, it is not readily absorbed by soil formations, is not altered by biological processes, and considerable quantities may be present in refuse."

Dissolved oxygen is readily measured in the field by means of a portable meter. The concentration of dissolved oxygen (D.O.) in water is a significant index of its sanitary quality. Waters high in organic material, such as sewage wastes, are usually low in dissolved oxygen. Dissolved oxygen is also of significance in evaluating the corrosiveness of the water. The solubility of oxygen in water is dependent in part on the temperature of the water and in part on the concentrations of other gases as well as dissolved solids such as chloride. Waters percolating through refuse piles (or sanitary landfills) may be expected to be depleted of dissolved oxygen by the organic matter in the refuse (California State Water Pollution Control Board, 1961, p. 61). However, after leaving the landfill the water may be subject to reaeration, especially in flowing streams, and its dissolved-oxygen content will normally rise.

Specific conductance is a measurement of the electrical conductance of water. Chemically pure water has a very low electrical conductance, but the presence of dissociated ions in solution renders the solution conductive. Conductance increases as the temperature rises; therefore, to make measured values comparable they must be referred to the same temperature. Commonly, analyses by the U.S. Geological Survey are referred to a standard temperature of 25°C (77°F). The conductance increases approximately 2 percent for each increase of 1°C in temperature (Hem, 1959, p. 38). Specific conductance is a rough measure of the dissolved-solids content and provides qualitative evidence of an increase in the mineralization of water.

The pH, or negative logarithm of the hydrogen-ion activity in moles per liter, is a quantitative expression of the acidity or alkalinity of water. Acid water has a pH of less than 7.0 and alkaline water has a pH of more than 7.0. A change of one unit represents a 10-fold increase in activity; that is, water having a pH of 5.0 is 10 times more acidic than water whose pH is 6.0.

Travel of Pollutants Through Earth Materials

Pollutants moving through saturated earth materials may be expected to travel at the rates of movement of water, or, in some instances, at slower rates. Pollutants leached from sanitary landfills that enter the ground-water flow system will move at rates governed by the permeability (K) of the materials, their effective porosity (p) and the prevailing hydraulic gradients (i).

The basic formula for computing ground-water velocity is:

$$V = \frac{Ki}{p}$$

where:

V = mean velocity, in feet per day
K = hydraulic conductivity, in feet per day[1]
i = hydraulic gradient, expressed as a decimal fraction
p = effective porosity, in percent

A further refinement of the above formula can be made by applying a temperature correction for the change in the viscosity of water at different temperatures. However, this refinement is probably not warranted, as most velocity computations involve significant assumptions regarding both permeability and the effective porosity of the materials, which probably have a greater error factor than that involved in correcting for temperature differences. For instance, Meinzer and Wenzel (1942, p. 449) indicate that the most permeable natural earth material tested was more than 450 million times more permeable than the least permeable material. Even for relatively permeable materials a wide range exists in this property.

Effective porosities of most materials transmitting water are commonly in the range of 10 to 50 percent. Hydraulic gradients under undisturbed conditions seldom are greater than 0.05, and normally are in the range 0.01 or less to 0.05. Of course, in the vicinity of a pumping well, hydraulic gradients are large, and ground-water velocities may be several times greater than they are at distance from the well. Thus, porosities of earth materials range within relatively narrow limits, hydraulic gradients generally are not large, but permeabilities may range over many orders of magnitude. Because of the wide range in permeability of earth materials, accurate prediction of the velocity of movement of ground water is difficult. However, where ground-water velocities have been measured, based on the time required for a chemical substance to travel from one place to another, they range from 0.04 to 13 feet per day. The source and locale of this information follows:

1) Pollutants from sanitary landfill at Brookings, S. D. (Andersen and Dornbush, 1967, p. 461).
   Earth material: Glacial outwash (Pleistocene) consisting of sand, gravel, silt, and clay
   Hydraulic gradient: not given
   Rate of movement: < 1 to 3 feet per day
   Distance of travel: 1,200 ± feet
2) Saline ground water at Derby, Colo. (Petri, 1961, p. 120).
   Earth material: not given
   Hydraulic gradient: not given
   Rate of movement: 13 feet per day
   Distance of travel: 4 ± miles
3) Du Page County landfill, Ill. (Hughes, Landon, and Farvolden, 1969, p. 109)
   Earth material: glacial sand and gravel (Pleistocene)
   Hydraulic gradient: 0.0012 to .0097
   Rate of movement: 0.04 to 0.27[1] foot per day
   Distance of travel: 600 ± feet
4) Chrome waste at Douglas, Mich. (Deutsch, 1963, p. 26)
   Earth material: glacial sand and gravel (Pleistocene)
   Hydraulic gradient: not given
   Rate of movement: 1 foot per day
   Distance of travel: 1,000 ± feet
5) Garbage liquor at Indianapolis, Ind. (Calvert, 1932, p. 267)
   Earth material: river alluvium and glacial outwash
   Hydraulic gradient: not given
   Rate of movement: 2 feet per day
   Distance of travel: 500 + feet
6) Sanitary landfill at Riverside, Calif. (California State Water Pollution Control Board, 1954, p. 72)
   Earth material: Holocene and Pleistocene alluvium
   Hydraulic gradient: 0.003
   Rate of movement: 4.9 feet per day[1]
   Distance of travel: 900 + feet

Dilution and dispersion are effective mechanisms for reducing the concentration of pollutants in both surface and ground waters. The flow of water in streams is normally turbulent; hence, the mixing of polluted water with native water may occur rapidly. However, ground-water flow is laminar—that is, it occurs essentially along parallel lines. Hence, mixing and dilution tend to be more gradual. Furthermore, the rocks themselves may act upon the polluted water and change its character. Permeable sand and gravel may effectively filter out most bacteria within relatively short distances. Clay may reduce the concentrations of some of the objectionable constituents (chiefly

[1] To convert hydraulic conductivity to permeability in Meinzer units multiply by 7.48 (gallons per cubic foot).

[1] Computed from measurements of permeability, porosity, and hydraulic gradients measured at the site.

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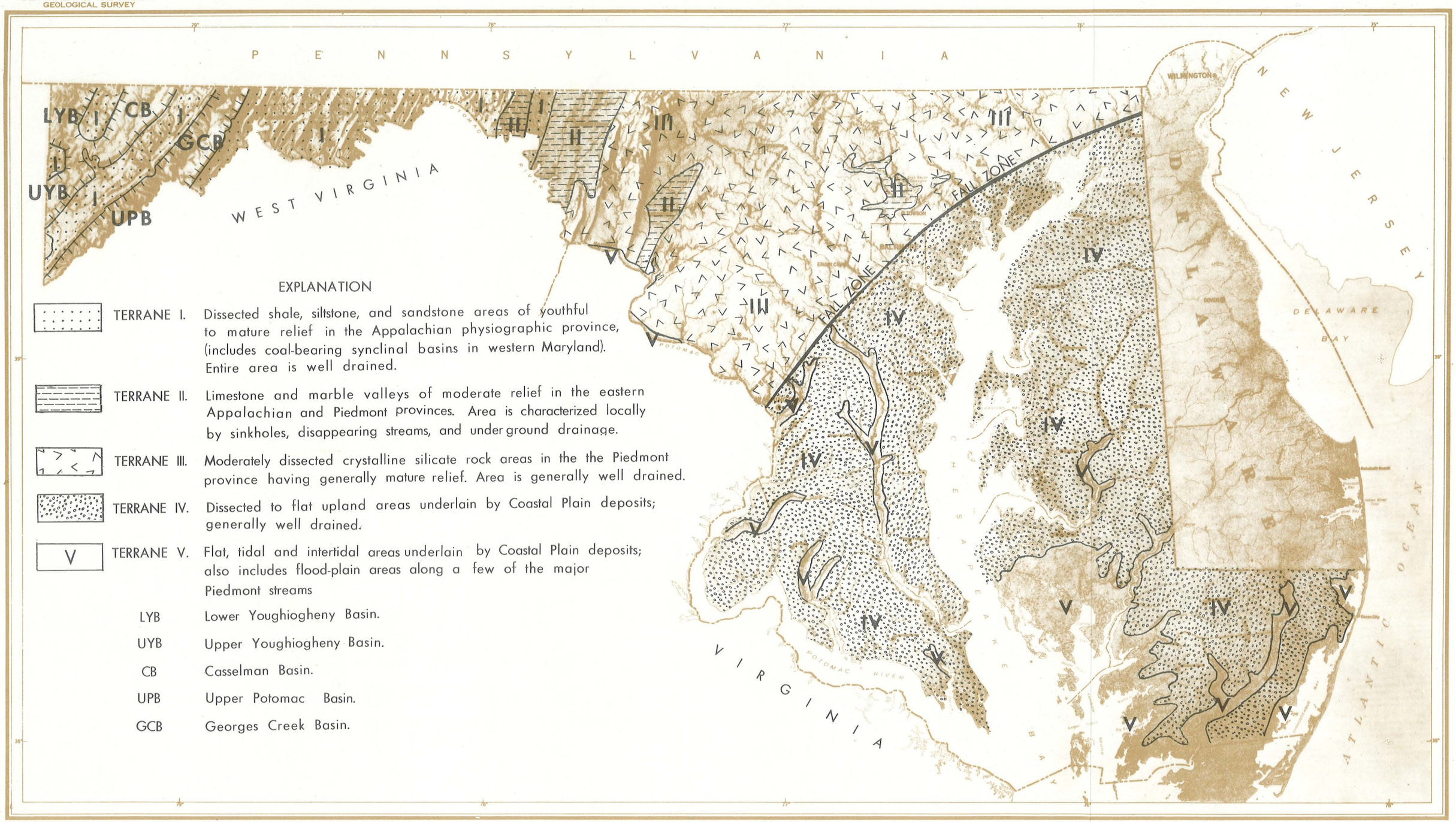
GEOLOGICAL SURVEY



EXPLANATION

TERRANE I. Dissected shale, siltstone, and sandstone areas of youthful to mature relief in the Appalachian physiographic province, (includes coal-bearing synclinal basins in western Maryland). Entire area is well drained.

TERRANE II. Limestone and marble valleys of moderate relief in the eastern Appalachian and Piedmont provinces. Area is characterized locally by sinkholes, disappearing streams, and underground drainage.

TERRANE III. Moderately dissected crystalline silicate rock areas in the the Piedmont province having generally mature relief. Area is generally well drained.

TERRANE IV. Dissected to flat upland areas underlain by Coastal Plain deposits; generally well drained.

V TERRANE V. Flat, tidal and intertidal areas underlain by Coastal Plain deposits; also includes flood-plain areas along a few of the major Piedmont streams

LYB Lower Youghiogheny Basin.

UYB Upper Youghiogheny Basin.

CB Casselman Basin.

UPB Upper Potomac Basin.

GCB Georges Creek Basin.

Compiled, edited and published by Geological Survey
1927 North American datum
Datum is mean sea level

Lambert conformal conic projection
Standard parallels 33° and 45°

Figure 8. Map of Maryland showing terranes designated in this report according to their geologic, topographic, and hydrologic characteristics relevant to solid-waste disposal.

cations) in landfill leachate through ion exchange, although the net result of the process may leave the dissolved-solids content about the same as it was before passage through the clay. Recent studies in northeastern Illinois (Hughes and others, 1969, p. 101) suggest that some clay layers may act as semipermeable membranes that restrict the passage of dissolved ions, although the applicability of this mechanism to landfill leachate has not yet been demonstrated.

TERRANES AS WASTE-DISPOSAL SITES

The approximately 155 solid-waste disposal sites in Maryland (fig. 1) are distributed throughout the range of geologic formations and physiographic regions represented in the State. This diversity also represents a wide range in geohydrologic conditions, and, thus, many different mechanisms exist within the State for possible degradation of the ground and surface waters by leachate from dumps and sanitary landfills. With regard to general geohydrologic conditions, the State is classified into five major terrane types. The term terrane as used here is "an area or region considered in relation to its fitness or suitability for some specific purpose" (American Geological Institute, 1962, p. 495). The classification herein presented is imprecise, and some localities may be classified only with difficulty, but the scheme is useful for the purposes of this report. The designation of the five terranes is:

Terrane I. Dissected shale, siltstone, and sandstone areas of youthful to mature relief in the Appalachian physiographic province (includes coal-bearing synclinal basins in western Maryland); entire area well drained.

Terrane II. Limestone and marble valleys of moderate relief in the eastern Appalachian and Piedmont provinces; area characterized locally by sinkholes, disappearing streams, and underground drainage.

Terrane III. Moderately dissected crystalline silicate rock areas in the Piedmont province having generally mature relief; area generally well drained.

Terrane IV. Dissected to flat upland areas underlain by Coastal Plain deposits; area generally well drained.

Terrane V. Flat, tidal, and intertidal areas underlain by Coastal Plain deposits; also includes flood-plain areas along a few of the major Piedmont streams.

Figure 8 shows Maryland divided into the five terranes described above.

Terrane I—Shale, Siltstone, and Sandstone of the Appalachian Province

Terrane I is characterized by well-drained, dissected, sedimentary rocks commonly having low primary permeability and porosity. In the shale and siltstone movement of ground water is slow, except in crevices or fractures, or near the land surface where weathering has resulted in increased permeability. The sandstone is somewhat more permeable than the shale and siltsone, but in the unfractured condition many of the sandstone beds are poorly permeable. It is believed that most of the water movement in the sandstone occurs along fractures and joint planes.

The coal-bearing strata (of Pennsylvanian age) in Maryland are in Terrane I. These strata occupy five northeast-southwest-trending structural basins or troughs in Garrett and Allegany Counties (Vokes, 1957, p. 109). There are reported to be several hundred abandoned coal strip mines in the two major coal basins, the Upper Potomac basin and the Georges Creek basin. At present (1970) a few of these mines are being used as sanitary landfills, and the possibility exists that other abandoned strip mines will also be used for this purpose in the near future.

Abandoned strip mines may be considered suitable sites for sanitary landfills because: 1) the extraction of the coal has created a depression on the hillside that may be filled without the expense

of additional excavation; and 2) the strata below the coal may be sufficiently impervious that leachate from the fill cannot move downward to pollute underlying aquifers directly; 3) in some instances, the chemical nature of the fill material may be such that the acidic water normally draining from abandoned mines may be partly neutralized by the fill material; and 4) the physical configuration of the mine may facilitate collection and treatment of the leachate, if that should be desirable or necessary.

Waagé (1950, p. 46) illustrates the character of a typical sequence of four lithologic zones associated with each coal bed. The sequence is:

4. Shale zone (top): Gray to black shales of fresh-water, brackish-water, or marine deposition. Some of the fresh and brackish-water shales are silty or sandy; some are carbonaceous or pyritic, or have ironstone concretions. The marine shales are usually calcareous, and in the more western exposures contain thin beds of fossiliferous limestone or of limestone concretions.
3. Coal zone: Coal, cannel coal, "bone," or highly carbonaceous shale.
2. Underclay zone: This is the most variable part of the sequence. It may be a simple zone of dark-gray silty clay, or a complex zone of plastic, semi-plastic and flint clay, claystones, fresh-water argillaceous limestones and marls. In the Conemaugh and Monongahela Formations this zone often contains red clays.
1. Sandstone zone: This zone consists of coarser clastic materials ranging from siltstone to conglomerate; fine to medium-coarse sand is most characteristic. Pebbles in the conglomerates are predominantly quartz.

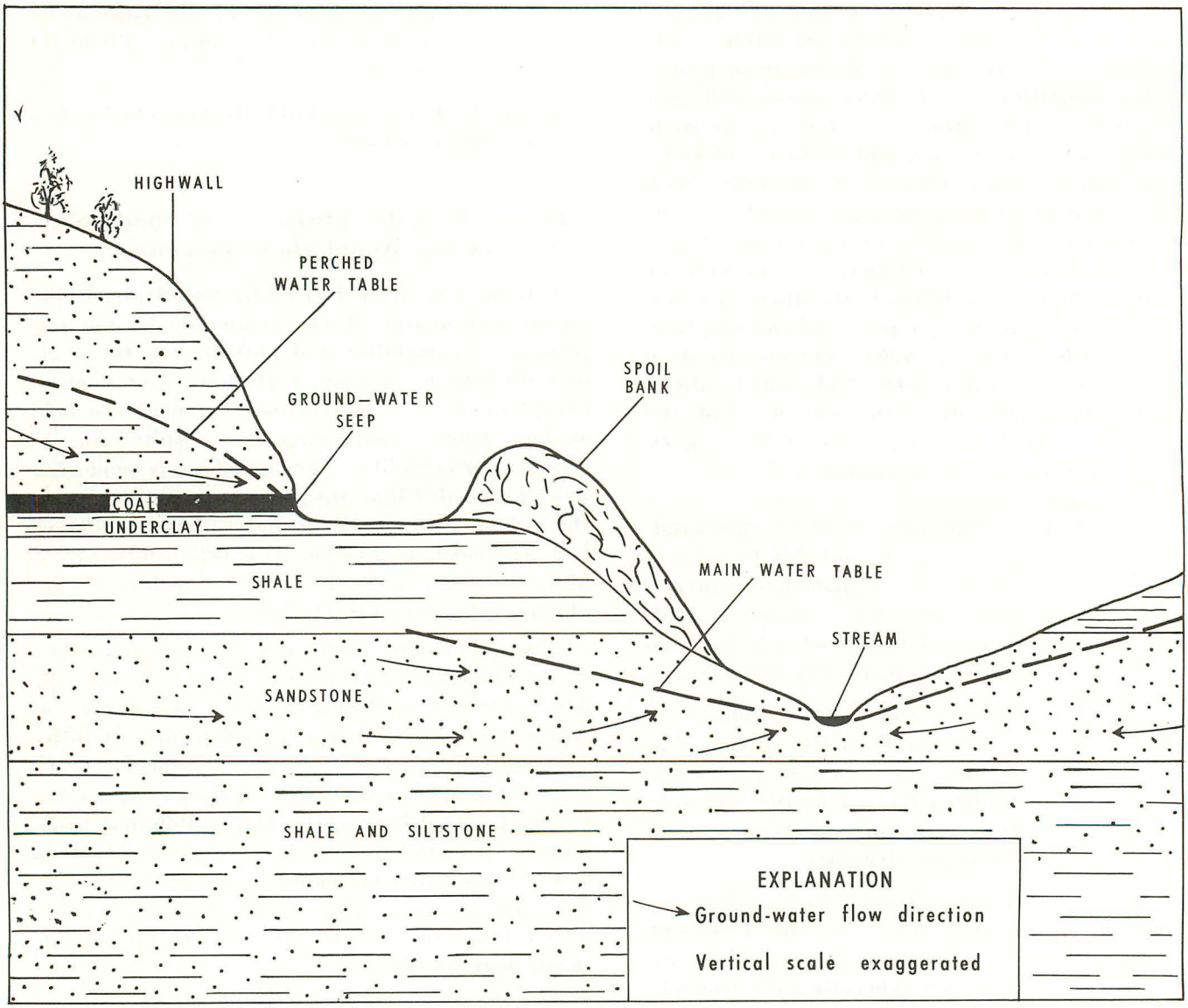


Figure 9. Schematic section across an abandoned coal strip mine showing possible geohydrologic conditions.

The presence of underclay and shale below and above the coal should retard or limit the direct downward migration of leachate generated from a sanitary landfill in a strip mine. Figure 9 shows an idealized geologic section across an abandoned strip mine in western Maryland before its conversion to a landfill operation. Commonly, the base of the coal bed is marked by a seep zone or several small springs. During extended droughts these may dry up, or the flow from them will be significantly reduced, thus sporadically reducing their potential for leachate generation. Unless provision is made for drainage by the mine operator, the trough between the coal bed and the spoil bank many contain small shallow ponds of acid mine water. As is shown in figure 9 the general direction of ground-water movement is from the hill above the coal vein to the small stream in the right center of the figure.

Figure 9A shows the geologic section across the same strip mine after it has been converted to a sanitary landfill and covered with a few feet of spoil-bank material. An impervious cover over the landfill will retard or diminish the direct percolation of precipitation or runoff into the fill, but the cover cannot prevent the fill material from becoming saturated with ground water in mines where ground-water seeps existed before use of the mine as a sanitary landfill. Now the ground water issuing from the coal seam may percolate through the landfill. Unless adequate drainage is provided, water can saturate the fill and result in

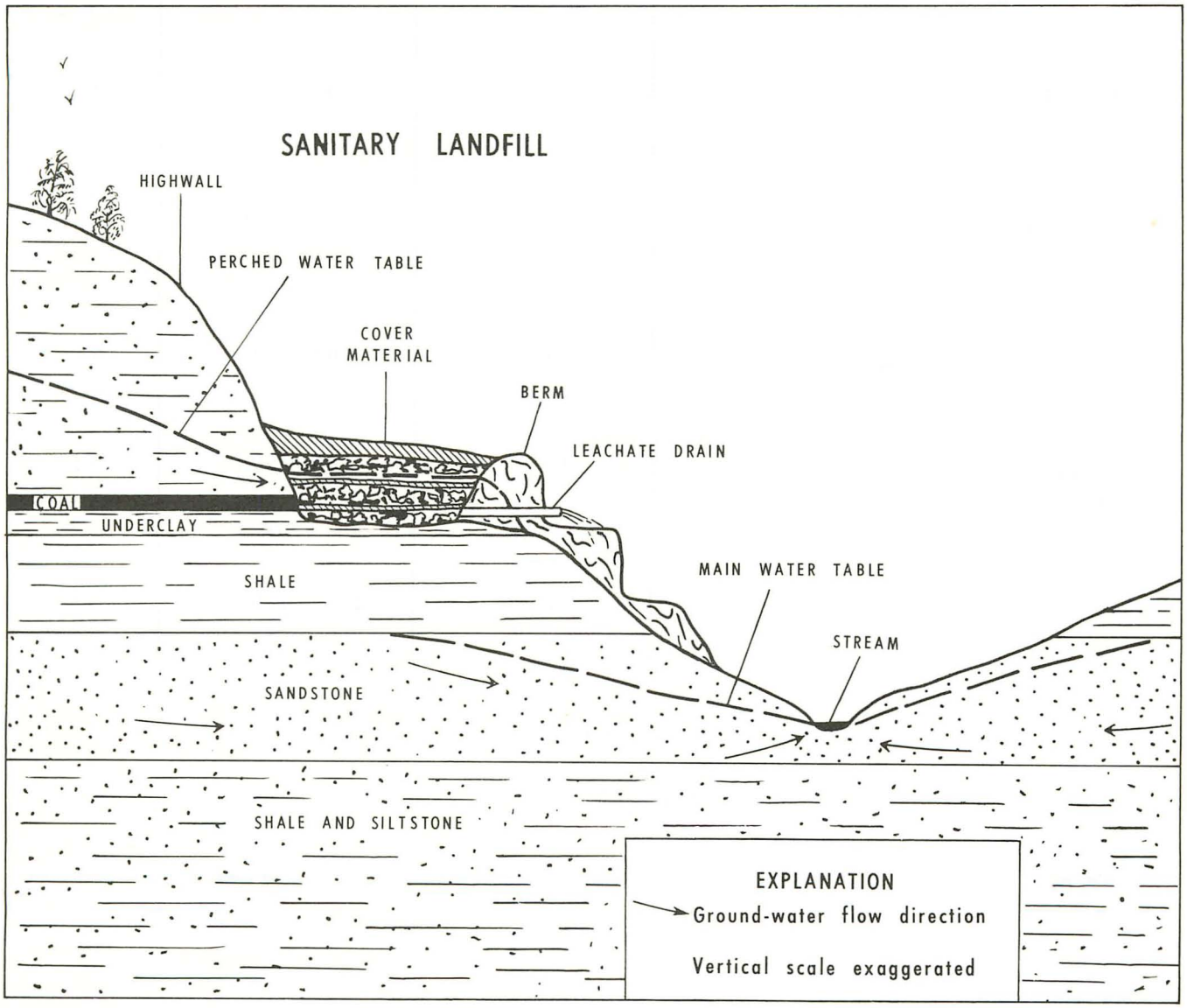


Figure 9a. Schematic section across an abandoned coal strip mine converted to a sanitary landfill showing possible modification of local geohydrologic conditions.

Table 10. Chemical analyses of water during a 3-1/2-year period from the Lewis spring near the Frostburg demonstration sanitary landfill.

Constituent or property 1/	Sample of February 20, 1967 (mg/l except for pH)	Sample of July 28, 1970 (mg/l except for pH)	Range in values during period covered (mg/l except for pH)	Number of analyses
Iron (Fe)	0.0	0.0	0.0 - 0.7	13
Chloride (Cl)	4.0	1.0	1.0 - 9.0	13
Nitrate as N	0.7 2/	3.4	0.06 - 3.4	13
Total solids (mg/l)	120	150	104 - 340	14
Hardness as $CaCO_3$	92	113	88 - 270	14
pH	4.3	7.6	4.3 - 7.9	12

1/ All analyses by the Maryland Dept. of Health.

2/ Analysis from sample of March 16, 1967.

the generation of leachate from the fill material as well as acid water from the coal seam. Figure 9A shows a leachate drain that can be installed to prevent excessive saturation of the fill material. Water issuing from the drain could be treated to remove undesirable constituents if this were economically feasible. In the absence of a leachate drain the water may move perpendicular to the plane of the section and emerge at a lower outlet along the stripped area. The ultimate disposition of the leachate from this source is usually the nearest ditch or small stream.

Frostburg Demonstration Landfill (Allegany County): An abandoned strip mine in the Georges Creek basin in western Maryland has been used as a sanitary landfill since April 1967. Part of the financing of the landfill operation has been by the U.S. Public Health Service to the Maryland Department of Health as a demonstration project to show whether abandoned coal strip mines could be used effectively for the disposal of solid wastes. The former mine is located approximately 1 mile west of the town of Clarysville and about 1.3 miles southeast of the campus of Frostburg State College in Allegany County. The mine was about 1,900 feet long, 50 feet wide at the bottom, and about 110 feet wide at the top. Its depth ranged from 30 to about 44 feet.

The site is underlain by the Monongahela Formation of Pennsylvanian age. In Maryland this unit consists of 240 to 270 feet of sandstone, shale, siltstone, limestone, and coal. It is not known for certain which coal was mined, but it is presumed to be the Lower Sewickley coal (also known as the Tyson coal). The strata in the walls of the mine consist chiefly of black and gray shale and thin beds of silty sandstone. The coal at the site is approximately 2 feet thick, but is nearly everywhere covered with slump material.

In the summer of 1966 the floor of the mine was reported to be dry, but in March 1967 (shortly before beginning the fill operation) about 5 feet of standing water was present in the mine (Shields and Barstow, 1967, p. 5). As it was concluded that this water was chiefly the result of runoff into the mine from rainfall and melting snow, a diversion ditch was constructed around the upper edge of the mine to retard hillslope runoff from entering the pit. A drainage ditch was also constructed to remove the standing water from the mine before beginning the landfill operation, but intermittent ponding occurred during the operation.

By the summer of 1970 the Frostburg landfill had been in operation for more than 3 years, serving as the disposal site for the solid wastes from nearly all of Allegany County, including the city of Cumberland (1970 population, 29,084).

To determine the extent of possible ground- and surface-water degradation resulting from the landfill operation, collection of a series of samples from nearby water sources was begun in February and March 1967 by the Division of Solid Wastes of the Maryland Department of Health. These sources are: a privately owned drilled well approximately 1 mile west of the landfill; a spring, known as the Lewis spring, approximately 0.5 mile east of the landfill and at a lower elevation; and Braddock Run, a tributary of Wills Creeek, at a point along Maryland Route 55 about 0.7 mile east of the landfill. In addition to these waters, chemical analyses were made of the water in an intermittent pond at the lower end of the abandoned strip-mine used for sanitary landfilling.

Only a few water samples were collected from the privately owned well, and these were collected mainly during the initial period of landfill operation. Because of the lack of information concerning the construction of the well and the nature of the strata penetrated by it, the chemical analyses of the well samples are not reviewed here.

Chemical analyses of selected important constituents from Lewis Spring fail to show evidence of pollution of the water by leachate during a 3½-year period. These analyses are summarized in table 10, which follows:

Chemical analyses of selected important constituents were made during the period January 1968 to August 1970 from Braddock Run at a site east of the Frostburg landfill. These data, in table 11, do not show any significant degradation of this stream as a result of leachate issuing from the landfill. However, since little information is available regarding possible dilution of such leachate by the water in Braddock Run, the chemical analyses are inconclusive, by themselves. If future studies were to be made of the geohydrology in the vicinity of this landfill, it would be desirable to measure the flow of nearby streams concordantly with the collection of water samples for chemical analyses. An additional refinement to evaluate the pollutional capability of the landfill would involve the introduction of a tracer substance into the

Table 11. Chemical analyses of water during a 2-1/2-year period from Braddock Run near Md. Route 55 in the vicinity of the Frostburg landfill.

Constituent or property 1/	Sample of January 15, 1968	Sample of August 18, 1970	Range in values during period covered	Number of analyses in range
	(mg/l except for pH)	(mg/l except for pH)	(mg/l except for pH)	
Iron (Fe)	9.0	9.0	0.0 - 12	11
Chloride (Cl)	2.5	1.5	2.5 - 208	11
Nitrate as N	1.9	.1	.04 - 1.9	11
Total solids	754	1,040	684 - 1,040	10
Hardness as $CaCO_3$	438	569	399 - 569	11
pH	6.1	6.8	3.7 - 6.8	11

1/ All analyses by Maryland Dept. of Health.

landfill and the subsequent monitoring of nearby streams, springs, and wells to determine if and when the tracer arrived at these sources. Possible tracers include potassium iodide, sugar, and rhodamine dyes.

The only chemical data available from the Frostburg landfill indicating that leachate-degraded water issues from the fill are a series of analyses of water from an intermittent pond at the lower end of the abandoned strip mine which is being filled. These analyses, summarized in table 12, show that the iron content of the water increased several hundredfold during a 3-year period, chlorides increased about 30 times what they were initially, and total solids increased by a factor of 13. Analyses of the common cations, trace ele-

Table 12. Chemical analyses of water from a pond at lower end of the strip-mine near Frostburg prior to and during a 3-year period of landfilling.

Constituent or property 1/	Sample of March 16, 1967 (mg/l except for pH)	Sample of April 27, 1970 (mg/l except for pH)	Range in values during period covered (mg/l except for pH)	Number of analyses in range
Iron (Fe)	0.3	100	0.2 - 2,250	10
Chloride (Cl)	6.0	192	.5 - 231	9
Nitrate as N	.1	3.0	.1 - 3.0	9
Total solids	248	3,316	208 - 7,058	9
Hardness as $CaCO_3$	106	--	106 - 2,310	9
pH	7.1	5.3	3.7 - 7.9	9

1/ All analyses by Maryland Dept. of Health.

ments, and complex organic substances in the water are not available.

The relatively small number and kind of chemical analyses of water available from the few monitoring sites does not indicate that significant degradation of the ground or surface waters has taken place in the vicinity of the Frostburg landfill, but the number and location of these sampling sites is inadequate and non-optimal. The Maryland Department of Health is currently (1970) drilling monitoring wells near the landfill and has established a more extensive chemical-quality monitoring system than the earlier system described here.

Westernport Demonstration Landfill (Garrett County): During September 1968 refuse-disposal operations were started on a sanitary landfill in an abandoned strip mine at Franklin Hill in eastern Garrett County. The site is located along Aaron Run, a south-flowing tributary of the Savage River, and is about 2 miles west of the town of Westernport and nearby communities. It is known as the Westernport landfill. The fill operation and experimental work undertaken at this site have also been financed in part by funds from the U.S. Public Health Service through the Solid Wastes

Division of the Maryland Department of Health.

Franklin Hill is underlain by the Conemaugh Formation of Pennsylvanian age, which consists of siltstone, sandstone, coal, shale, and underclay. In Garrett County the Conemaugh is from 825 to 925 feet thick and is divided into an upper and lower member by the Barton Coal, the seam formerly mined at the demonstration site.

During early 1969 two experimental landfill cells were constructed at the lower end of the strip mine. These cells, 100 feet long and 16 feet wide, were underlain by a 1-foot layer of crushed stone underdrain and a 2-foot layer of crushed stone cover. Acid mine water draining from the floor of the strip mine was pumped intermittently over the surface of the landfill cell and allowed to drain through the fill material, where it was collected by the underdrain in a concrete sump pit.

Chemical and bacteriologic analyses were made of the mine-drainage water before and after its passage through the fill material. It was considered possible that the refuse in the landfill cells might decrease the acidity of the mine water. At the time of preparation of this report analyses of the Westernport leachate were not available. Currently (1970) the Franklin Hill strip mine is being continued as a sanitary landfill, but no additional leaching experiments are being made at the demonstration cells.

Terrane II—Limestone and Marble Valleys

Areas underlain by calcareous rocks constitute a distinct type of terrane having lithologic and hydrologic characteristics that distinguish them from other terranes. Because of the greater solubility of calcareous rocks in comparison with siliceous rocks, areas underlain by limestone commonly are the sites of valleys of gentle to moderate relief, at least in humid regions.

Limestones (including dolomites) occur in Maryland chiefly in three localities, the Hagerstown Valley (or the Great Valley), the Frederick Valley, and a group of smaller valleys in central Baltimore County. Minor areas underlain by calcareous rocks occur in eastern Frederick and central Carroll Counties and in the vicinity of Cumberland in western Maryland. It is estimated that about 500 square miles in Maryland are underlain by limestone or marble (L. J. Nutter, oral commun., 1970).

As a result of weathering, erosion, and deposition, the limestones are mantled with saprolite, residuum, and colluvium, but all three types of material may not be present at any one place. Saprolite and residuum are formed in place, but colluvium is usually derived by the processes of creep or slope wash from nearby topographically higher sites. The three types of material are sometimes designated by the general term "weathered zone" where they have not been identified separately.

The weathered zone is quite variable in thickness. In the Hagerstown Valley, Slaughter (1962, p. 72) reports that the weathered zone in limestone strata ranges in thickness from 0 to 127 feet and averages 33 feet, based on the casing records of about 250 water wells. In the marble valleys of central Baltimore County, Dingman and Ferguson (1956, p. 20) report that the depth of weathering ranges from 6 to 222 feet and averages about 42 feet, based on the records of 44 wells. The extreme reported depth of 222 feet is probably the result of a well being situated along a deeply weathered fault plane.

Distinctive soil types develop in areas underlain by calcareous rocks. Knowledge of the physical and hydraulic properties of these soils is important in evaluating limestone terranes for their water-pollution hazard with regard to sanitary landfills and refuse dumps. The major soil series in the limestone valleys have been mapped by the U.S. Dept. of Agriculture (1959 and 1960). The soils consist of fine sandy loam, cherty silt loam, gravelly loam, and clay. Drainage characteristics range from well drained to poorly drained, and the reported infiltration rates range from 0.06 to 6.3 inches per hour.

Throughout a large part of the limestone terrane the strata are complexly folded, faulted, and structurally disturbed. Thus, the dip of the strata may range from a few to 90°. Beds of shale or siliceous limestone separate the more soluble, purer limestone strata from one another. Furthermore, the thickness and distribution of the overlying residuum is irregular, and in extensive areas in the Hagerstown Valley the residuum appears to have been largely removed by erosion. This frequently results in complex patterns of solution channels in the limestones.

Hydraulically, the residuum behaves quite differently from the underlying creviced limestone. The residuum may significantly retard the movement of water, and it may effectively filter bacteria and fine suspended material from the water. Figure 10 shows a schematic section across dipping limestone and shale beds where a sanitary landfill has been established not far from a stream. As depicted in the section, ground water is moving from the hilltop beneath the landfill and is discharging into the nearby stream. If the soil underlying the landfill is sufficiently thick and impervious, some protection from pollution might be afforded the underlying ground water. Wells or springs located on the opposite side of the stream might be safe from pollution unless pumping (in the case of a well) diverts ground water from beneath the landfill. It would appear that, because of the dipping shale layer, any wells located on the extreme left side of the section would be safe from pollution. The section, of course, does not necessarily represent any real situation, but illustrates the nature of the possible hazard that can be created by the unwise location of a landfill.

Sanitary Landfill near Texas (Baltimore County): In August 1966 sanitary-landfill operations were begun on a 55-acre tract of land approximately 0.5 mile north of the community of Texas in central Baltimore County. By July 1970 the Texas landfill was filled to about 75 percent of capacity (Charles Farley, Baltimore County Dept. Public Works, oral commun., 1970). The rate of

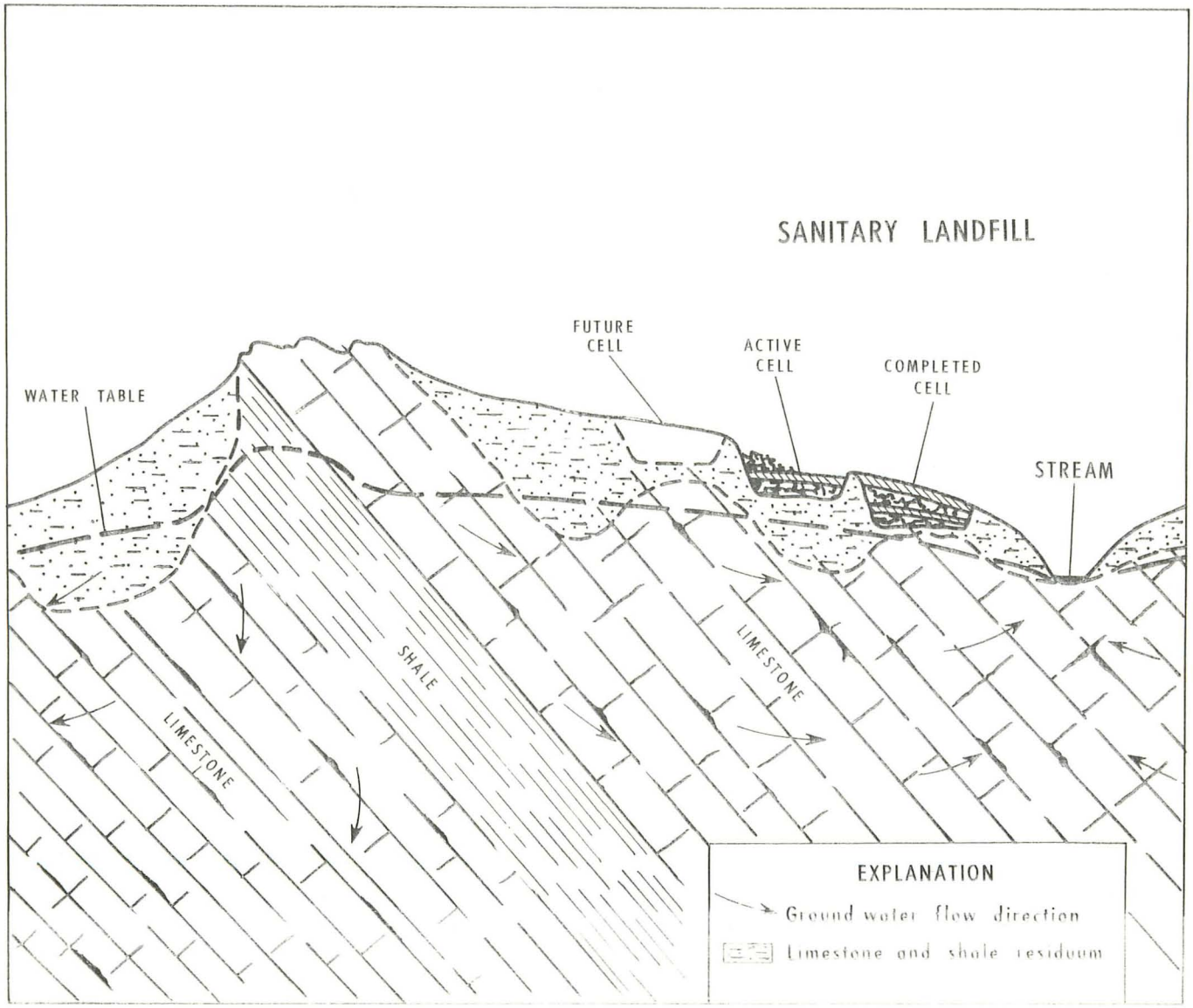


Figure 10. Schematic section showing a sanitary landfill in an area underlain by tilted limestone and shale.

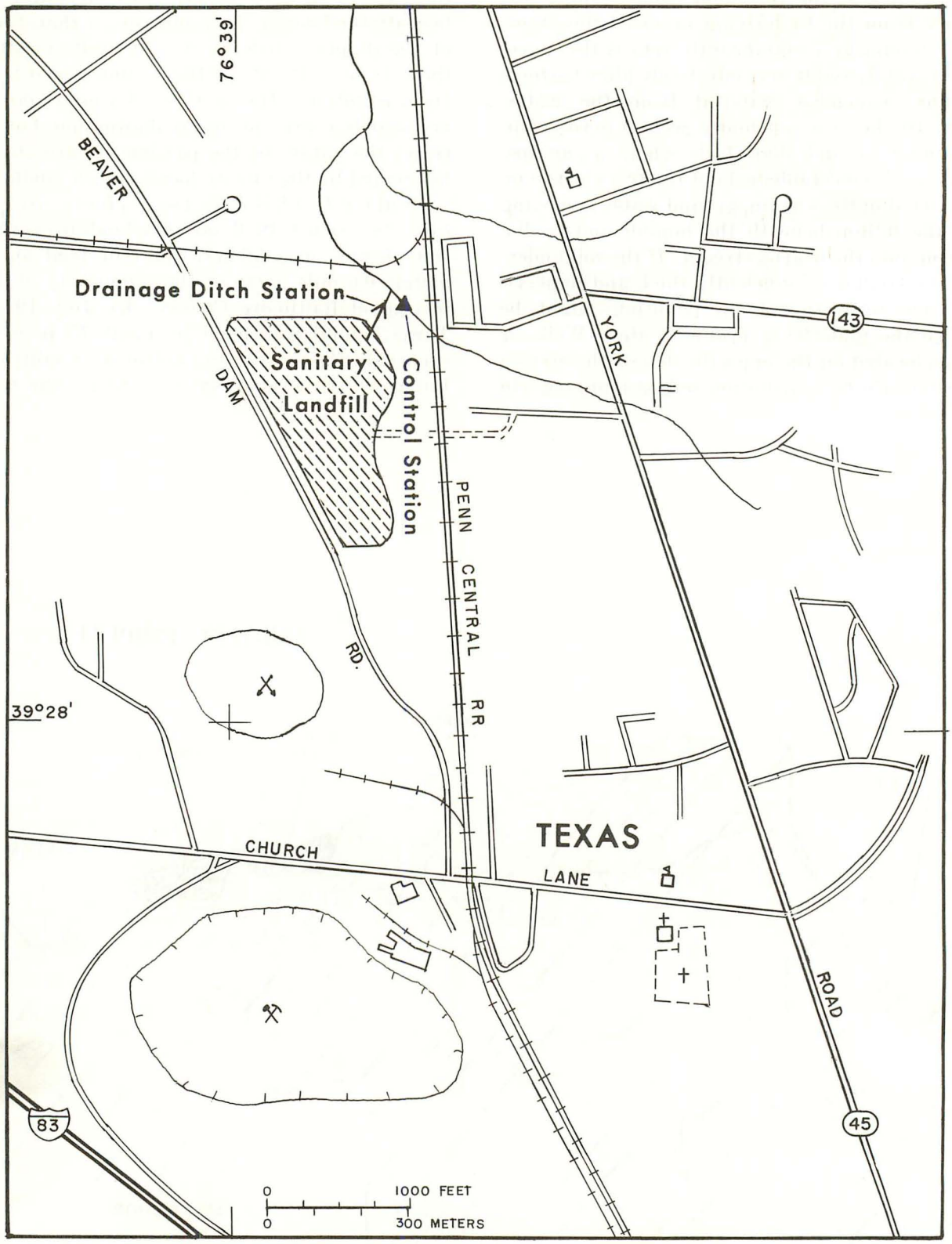


Figure 11. Map showing location of the sanitary landfill and water-quality monitoring sites at Texas, Md.

filling was about 600 tons per day.

This landfill occupies the site of a series of sand pits formerly used as a source of dolomite sand. The sand is derived from the weathering of a phlogopite-calcite metadolomite rock association in the Cockeysville Marble of Precambrian(?) age (Choquette, 1960, p. 1,039). Ten borings by the Baltimore County Department of Public Works made before the use of the site for a landfill show that the soil and saprolite ranged from 7 to 28 feet thick and had an average thickness of 15 feet.

The Cockeysville Marble is about 750 feet thick. At Texas it forms a 0.5- to 2-mile wide and 3- to 4-mile long north-trending valley. The marble has been extensively quarried at a large operation immediately west of Texas, and there are other smaller abandoned marble quarries in the Cockeysville-Texas area. The marble is highly metamorphosed and is complexly folded and faulted.

Solution features, such as sinkholes and tunnels developed by circulating ground water, are not prominent in the Cockeysville. However, at a few places underground drainage channels exist. In 1961 a cavernous zone was found in the marble about 3.5 miles southeast of Texas during the digging of a soil-percolation test hole. The cavern or tunnel at this place must be several hundred feet in length, as a tracer dye poured in the well reportedly emerged at Loch Raven Reservior after a few hours. The test hole was about 0.4 mile from the reservoir.

Examination of the Texas landfill early in December 1969 showed that water was issuing from a narrow drainage ditch at the north end of the site. About 75 feet north of the active fill area the ditch empties into Parks Run, a small tributary of Beaverdam Run. Figure 11 shows the location of the Texas landfill and the water-quality monitoring sites discussed in this report.

Monitoring was begun on December 3, 1969, to determine what differences might exist in the chemical and bacteriologic character of the water from the landfill and the water from the receiving stream (Parks Run) at an upstream control point (fig. 11). The waters were analyzed for total hardness, chloride, dissolved oxygen, specific conductance, and total organic carbon. The pH and temperature were also measured. An approximate index of the bacteriologic character of the water was obtained by counts of the total coliform group and fecal coliforms.

The pH of 10 samples of water from the landfill drainage ditch (fig. 11) ranged from 7.0 to 7.6. The slightly alkaline pH is due chiefly to the effect of the dolomite sand used as a cover material for the landfill.

Hardness of water results chiefly from the presence of ions of calcium and magnesium. The total hardness of water from the drainage ditch ranged from 390 to 1,200 mg/l in nine samples analyzed.

Chloride content of the drainage ditch water, in five samples, ranged from 157 to 198 mg/l. The chloride content appeared to be progressively increasing during the period of measurement from June to November 1970.

Seven discharge measurements of flow in the drainage ditch during 1970 ranged from 11.5 to 36 gpm and averaged about 23 gpm. Most of the measurements were made using a V-notch weir installed in the ditch. These measurements indicate that an average of about 30,000 gallons per day flowed from the ditch into Parks Run.

Tables 1 and 1A (Appendix) show the results of periodic chemical and bacteriologic analyses of water from the Texas landfill drainage ditch and from the nearby control stream (Parks Run). Table 13, which follows, summarizes the results of these data.

Table 13. Comparative mean values of selected constituents and properties of water from the Texas landfill drainage ditch and from Parks Run, a nearby control stream. 1/

Constituent or property	Landfill drainage ditch		Control stream (Parks Run)	
	Mean value	Number of measurements	Mean value	Number of measurements
Total hardness as $CaCO_3$ (mg/l)	880	9	309	6
Chloride (Cl) (mg/l)	181	5	49	4
Dissolved oxygen (O_2) (mg/l)	5.3	12	8.9	8
Specific conductance at 25°C	1,463	12	590	8

1/ Analyses by U.S. Geological Survey.

The data in table 13 show that a significant difference existed in the total hardness of the water issuing from the Texas sanitary landfill compared with that from Parks Run, the control stream. The landfill water had hardness nearly three times that of Parks Run. Chloride content, based on five analyses, was nearly four times greater in the water from the landfill than in the water from the control stream. Dissolved-oxygen content was significantly lower in the water from the landfill than in the water from the control stream. Specific conductance of the water from the landfill was more than double that of the control stream. Thus, the water issuing from the landfill was degraded, but the analyses made do not indicate the existence, nature, or concentrations of specific toxic or highly deleterious substances that were leaching from the landfill. It would be appropriate to analyze for the possible presence of such substances in the landfill discharge.

Analyses for coliform bacteria in the water of the landfill drainage ditch during the period December 1969 to November 1970 ranged from 320 per 100 ml (milliliters) to 86,000 per 100 ml (see Appendix, Table 1). The range of coliforms in the control stream during the period from late April 1970 to November 1970 was 1,000 per 100 ml to 39,000 per 100 ml. The U.S. Federal Water Quality Administration (1968, p. 20) has suggested that for public-water supplies from surface sources the permissible limit of coliforms is 10,000 per 100 ml and the desirable limit is 100 per 100 ml.[1]

Fecal coliform bacteria in the drainage-ditch water ranged from 25 per 100 ml to 6,000 per 100 ml during the same period covered by the coliform analyses. The U.S. Federal Water Quality Administration (1968, p. 20) has suggested that for public-water supplies from surface sources the permissible limit of fecal coliforms is 2,000 per 100 ml and the desirable limit is 20 per 100 ml.

There are several possible sources of fecal coliforms in the drainage-ditch water, but the exceptionally high counts during January and February 1970 (6,000 and 5,400 per 100 ml) are believed to be due to several hundred sea gulls at the landfill during these months. The gulls apparently migrate from the Chesapeake Bay during midwinter when ice cover in the upper Bay interferes with the normal food-gathering activities of the birds.

[1] Microbiologic limits are monthly arithmetic averages based upon an adequate number of samples. Total coliform limit may be relaxed if the fecal coliform concentration does not exceed the specified limit.

Terrane III—Crystalline Silicate-Rock Areas of the Piedmont

Terrane III consists of crystalline silicate rocks of the Piedmont physiographic province. The land surface is undulating to hilly or rugged and generally well drained. The Piedmont province extends from the Fall Zone on the east to Catoctin Mountain on the west and includes about 2,500 square miles, or one-fourth of the area of Maryland (Vokes, 1957, p. 56). The Piedmont is underlain chiefly by ancient igneous and metamorphic rocks consisting of gneiss, gabbro, phyllite, slate, schist, serpentine, and granite. Marble and related carbonate rocks also occur in the Piedmont, but these rocks have been included in Terrane II because of their distinctive hydrologic and geologic characteristics. In places in the Monocacy and Potomac River valleys, sedimentary rocks of Triassic age, chiefly shales and sandstones, occur in troughs surrounded by much older crystalline rocks. However, for the purposes of this report, these rocks have been included in the discussion of the Piedmont area.

The crystalline rocks extend to great depths, but most ground water in them occurs in fractures, crevices, and joint planes in the uppermost few hundred feet. The rocks weather to form a mantle of saprolite. The thickness of the saprolite is greatly influenced by the climate. A warm, humid climate favors the development of a thick and extensive zone of saprolite. Saprolite grades from a true soil at the surface to unaltered (fresh) rock at depth. The thickness of saprolite in the Maryland Piedmont ranges from a few feet to more than 100 feet and averages about 45 feet. Some differences exist in the thickness and nature of the saprolite derived from various rock types. Commonly the phyllite and phyllitic slate is mantled by a relatively thin zone of saprolite and true soil. Certain rocks, such as gabbro, diabase, and marble, weather to residual boulders irregularly distributed throughout the saprolite. The presence of large numbers of such boulders in the weathered zone may create a problem where it is planned to use the weathered rock (subsoil) as a source of landfill cover material.

Figure 12 is a schematic section across a sanitary landfill in crystalline-rock terrane. The geologic and hydrologic situation depicted is only one of a large number of possible situations. The illustration shows an active landfill cell and a com-

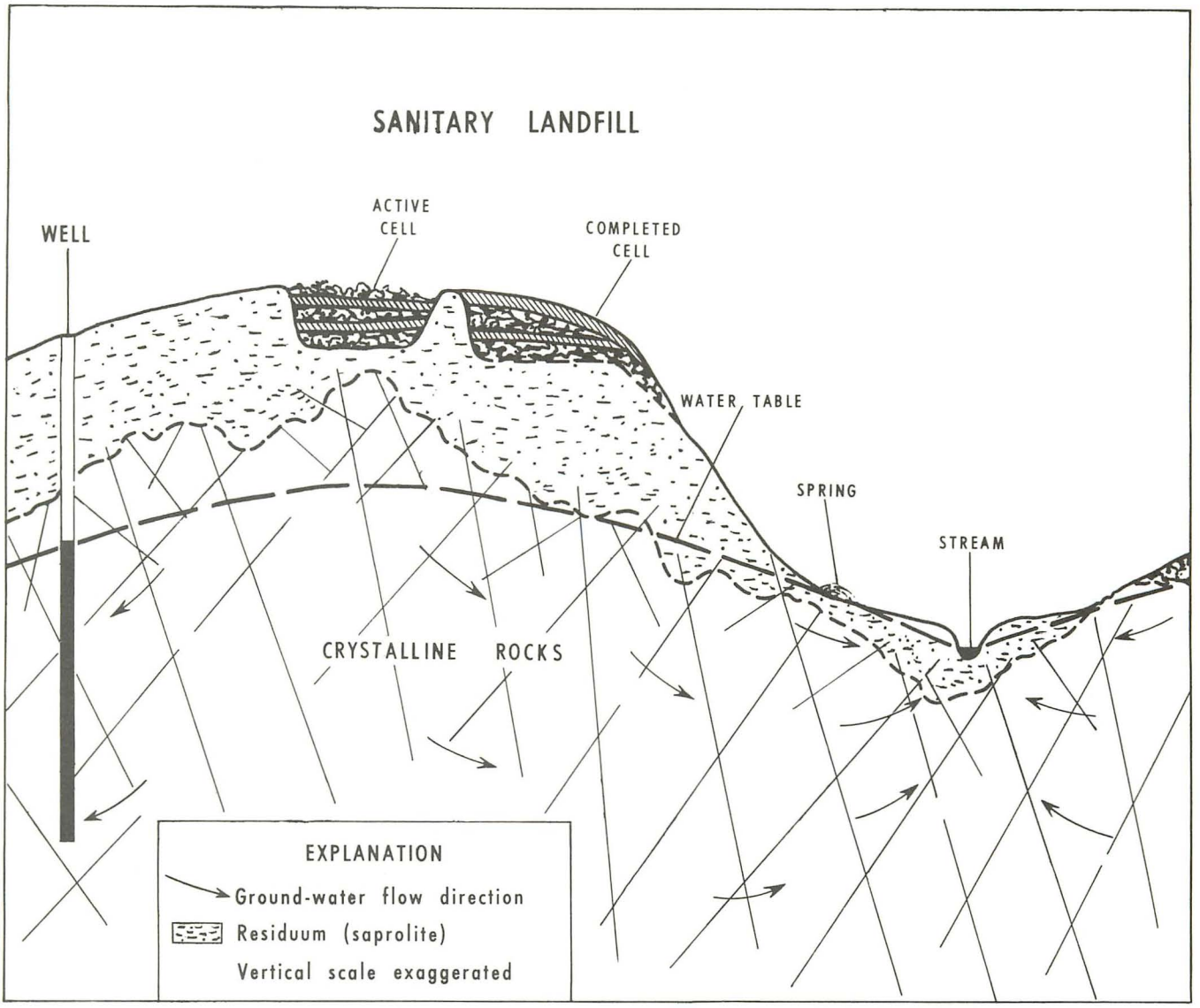


Figure 12. Schematic section across a sanitary landfill in crystalline-rock terrane showing hypothetical movement of ground water.

pleted cell situated near a hilltop underlain by fractured crystalline rocks. As depicted, both cells are well above the water table, whose position fluctuates seasonally in response to ground-water recharge. Leachate from the landfill can percolate down through the saprolite and enter crevices and fractures in the underlying rocks. Water in the hypothetical well, spring, and stream might be subject to degradation by the leachate. The permeability and thickness of the subsoil beneath the fill determines whether or not the ground water underlying the landfill is degraded. It is possible that a thick but moderately permeable subsoil might effectively filter most bacteria derived from the refuse and might alter the composition of the leachate significantly, possibly by removal of certain compounds present in colloidal from, or by ion exchange. However, many of the ionic constituents will pass unchanged through the underlying earth materials, subject only to dilution and attenuation by the deeper ground water. This statement applies also to subsoil in terranes other than crystalline rocks.

Reichs Ford Road Sanitary Landfill (Frederick County): Sanitary-landfill operations were begun during August 1969 on a tract of land approximately 3 miles southeast of Frederick, Md. (1970

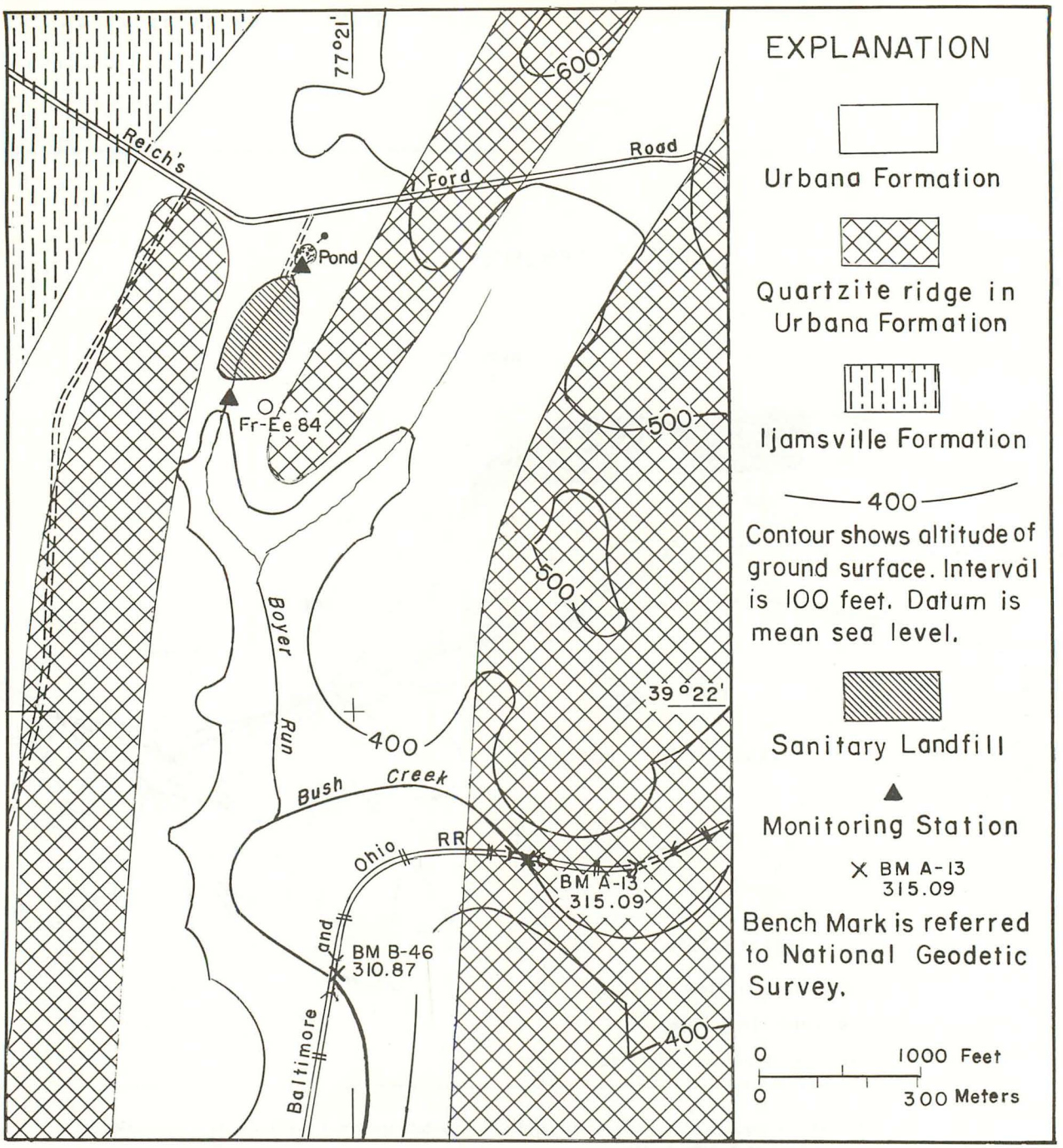


Figure 13. Map showing geology, location of the sanitary landfill, and water-quality monitoring stations in the vicinity of Reichs Ford Road near Frederick.

population 23,463). The landfill, serving the Frederick area, lies immediately south of Reichs Ford Road and north of Bush Creek, a tributary of the Monocacy River. The landfill occupies a small valley drained by the west branch of Boyer Run, a south-flowing tributary of Bush Creek (fig. 13). The landfill site was formerly the Paul W. Boyer farm. By the summer of 1970 the rate of filling was reported to be about 100 to 125 tons per day (W. Barstow, Maryland Department of Health, oral commun., 1970).

The landfill is underlain by the Urbana Formation[1] of probable Precambrian age. This formation extends to great but unknown depths and consists of hard, dense, dark-gray to green seri-

[1] Geologic nomenclature is that of the Maryland Geological Survey and may not be the same as that of the U.S. Geological Survey.

cite-chlorite phyllite, metasiltstone, and quartzite. Thin lenses of impure marble and calcareous phyllite occur locally. The geology of the site is shown by Jonas and Stose (1938), and more detailed geologic information has been furnished by Jonathan Edwards, Jr., of the Maryland Geological Survey from his unpublished map. The mapping by Edwards is shown in figure 13.

Before acquisition of the site for the purpose of a landfill, 11 borings were made to determine the depth and nature of the weathered rock and soil. G. G. Gill, engineer of the firm that made the borings, stated (written commun. to R. J. Rothenhoefer, City Engineer of Frederick, Md., 1967) :

> "The borings revealed overburden soils which are principally clayey silts extending to depths of anywhere from 2.5 to 12 feet below the existing ground surface. Below this point was encountered a very indurated material that has undoubtedly been formed in place through weathering of the underlying rock. . . ."

The surface soil at the landfill site is shown on the soils map of Frederick County (U.S. Dept, of Agriculture, Soil Conservation Service, 1960, Sheet 40) to be a Manor channery[1] and gravelly loam (MaDZ and MaEA). With regard to the engineering characteristics of soils of this type the following statement is made (p. 122) :

> "Somewhat excessively drained, shallow to very shallow, upland soils developed in residuum from highly micaceous schist and phyllite. . . . The entire profile is rapidly permeable to very rapidly permeable. Bedrock lies at a depth of 3 to 5 feet."

The upper limit of the landfill is about 75 feet below a small spring-fed pond a short distance south of Reichs Ford Road (fig. 13). Before landfilling began, a 24-inch-diameter concrete underdrain was laid to carry the overflow from the pond beneath the landfill. A layer of graded soil and earth material was placed on top of the underdrain. By July 1970 the fill had been extended to the end of the underdrain, and plans had been made by Frederick County to extend the underdrain an additional 600 feet. Beginning in May 1970 a series of analyses were made to measure total hardness, chloride, dissolved oxygen, specific conductance, temperature, pH, and total and fecal coliforms. Analyses were made of water collected from a point along the west branch of Boyer Run below the landfill outfall and, for control purposes, from the small spring-fed pond above the landfill.

[1] *channery* is defined as containing thin and flat fragments of limestone, sandstone, or schist having a diameter of less than 3 inches and lengths up to 6 inches (U.S. Dept. of Agriculture, 1951, p. 214).

The results of these analyses are in tables 2 and 2A (Appendix). Table 14, which follows, summarizes the chemical data obtained:

Table 14. Comparative mean values of selected constituents and properties of water from the west branch of Boyer Run below the Reichs Ford Road sanitary landfill and from the spring-fed pond above the landfill 1/

Constituent or property	West branch of Boyer Run below landfill		Spring-fed pond above landfill	
	Mean value	Number of measurements	Mean value	Number of measurements
Total hardness as $CaCO_3$ (mg/l)	224	7	68	6
Chloride (Cl) (mg/l)	30	6	9.4	5
Dissolved oxygen (O_2) (mg/l)	3.9	7	9.2	6
Specific conductance (micromhos @ 25°C)	457	7	157	6

1/ Analyses by U.S. Geological Survey.

Mean total hardness (table 14) is more than three times greater in the samples from Boyer Run below the landfill than in those from the spring-fed pond above it. Chloride content is also greater in the same proportion. Neither group of samples is high in chloride; the taste threshold of chloride is said to be about 250 mg/l. As might be expected, the dissolved-oxygen content of water below the landfill is significantly reduced, averaging about 3.9 mg/l compared with 9.2 mg/l from the pond. Specific conductance of samples from below the landfill is nearly three times as great as in samples from the control pond.

Counts of coliform bacteria (table 2, Appendix), based on six samples from the stream sampling point below the landfill, ranged from 200 to 4,400 per 100 ml. The coliform count of five samples from the pond above the landfill (table 2A, Appendix) ranged from 150 to 6,600 per 100 ml. This suggests that total coliforms may decrease between the upper and lower sampling points, but the data are inconclusive in this regard. Counts of fecal coliforms below the landfill, based on six samples, ranged from 5 per 100 ml to 230 per 100 ml. Fecal coliforms from the control pond above the landfill ranged from 30 to 900 per 100 ml. The

bacteriologic data, of itself, is inconclusive but does not indicate any significant increase in coliform bacteria below the landfill.

Five discharge measurements made below the landfill between early June and November 1970 ranged from 12 to 29 and averaged about 16 gpm. Thus, during this period, an average of about 23,000 gallons per day flowed down the west branch of Boyer Run to join water from the east branch and thence flow into Bush Creek, a tributary of the Monocacy River. Water from Boyer Run is significantly diluted after joining Bush Creek, a much larger stream.

It is emphasized that the entire monitoring program, both bacteriologic and chemical, is somewhat qualitative and does not show whether the leachate from the landfill may contain toxic or hazardous substances in a sufficient concentration to be a cause for concern. It would appear desirable to analyze the effluent water below the landfill for the presence of hazardous inorganic and organic substances.

Terrane IV—Dissected Upland Areas of Coastal Plain Deposits

Terrane IV, dissected to flat upland areas underlain by Coastal Plain sedimentary deposits, is well drained, except for the valley bottoms of the major streams, which commonly are swampy, filled with alluvium, and have a water table only a few feet below the land surface. Near the hilltops, depths to the water table may be several tens of feet, especially in areas of permeable sandy subsoil.

Terrane IV includes most of the area of Southern Maryland, most of the northern five counties of the Eastern Shore, and the interior areas of the remaining Eastern Shore counties (fig. 8). Throughout much of the area the relief commonly is in the range of 50 to 100 feet or less. However, in places, for example, near Piscataway Creek immediately south of Washington, D. C., the local relief is slightly more than 200 feet. (See Piscataway 7½-minute U.S. Geol. Survey quadrangle map.)

The Coastal Plain deposits, consisting of stratified sand, gravel, silt, and clay and various mixtures thereof, contain several aquifers commonly separated from one another by intervening relatively impervious layers. The impervious layers, which may range in thickness from a few tens of feet to a few hundred feet, function to protect the deeper underlying aquifers from pollution by surface sources. A common situation in a Coastal Plain locality is shown in figure 14, where ground water in the deeper sand occurs under artesian, or confined conditions, because of the presence of a thick overlying clay. In contrast, ground water in the shallow upper sand occurs under water-table conditions. This aquifer may be readily polluted by downward moving water in the unsaturated zone. Thus, septic-tank effluent, sanitary-landfill leachate, or other wastes may pollute the shallow aquifer. As depicted in figure 14, the movement of the shallow ground water is toward the nearby estuary. Well B (fig. 14) could be polluted by leachate from the sanitary landfill only if its cone of depression intersected the area beneath the landfill. On the other hand, water from well A very likely would remain unpolluted because the intervening clay would be an effective barrier to the movement of ground water into the lower sand. In actuality, the clay stratum depicted in the figure might be a sandy silt or a sandy clay, functioning only as a partial barrier to the vertical movement of ground water. Nevertheless, as previously stated, such material might effectively filter many harmful biological substances. The problem is extremely complex and the discussion given here is somewhat simplified.

Mullins Sanitary Landfill near Aberdeen (Harford County): Landfill operations were begun during December 1969 on a 102-acre tract of land, known as the Mullins property. This land is about 0.5 mile east of Aberdeen, Md., and south of both U.S. Route 40 and the Penn Central Railroad tracks. The landfill is owned by Harford County and serves the Aberdeen and Havre de Grace areas. It has been reported that the rate of filling is about 100 to 130 tons per day (W. Barstow, Maryland Dept. of Health, oral commun., 1970).

Although this landfill does not ideally fit the designation of Terrane IV, it is well drained, slightly dissected, and is underlain by a few tens of feet of Coastal Plain deposits. It is the site of a former sand and gravel pit and lies in the Coastal Plain province immediately south of the Fall Zone. The generalized geology of the site is shown in figure 15. Most of the fill is being made in formerly excavated land at altitudes from less than 10 to about 30 feet. The major stream draining the site is Swan Creek, which apparently changes from tidal to nontidal water along a curving reach

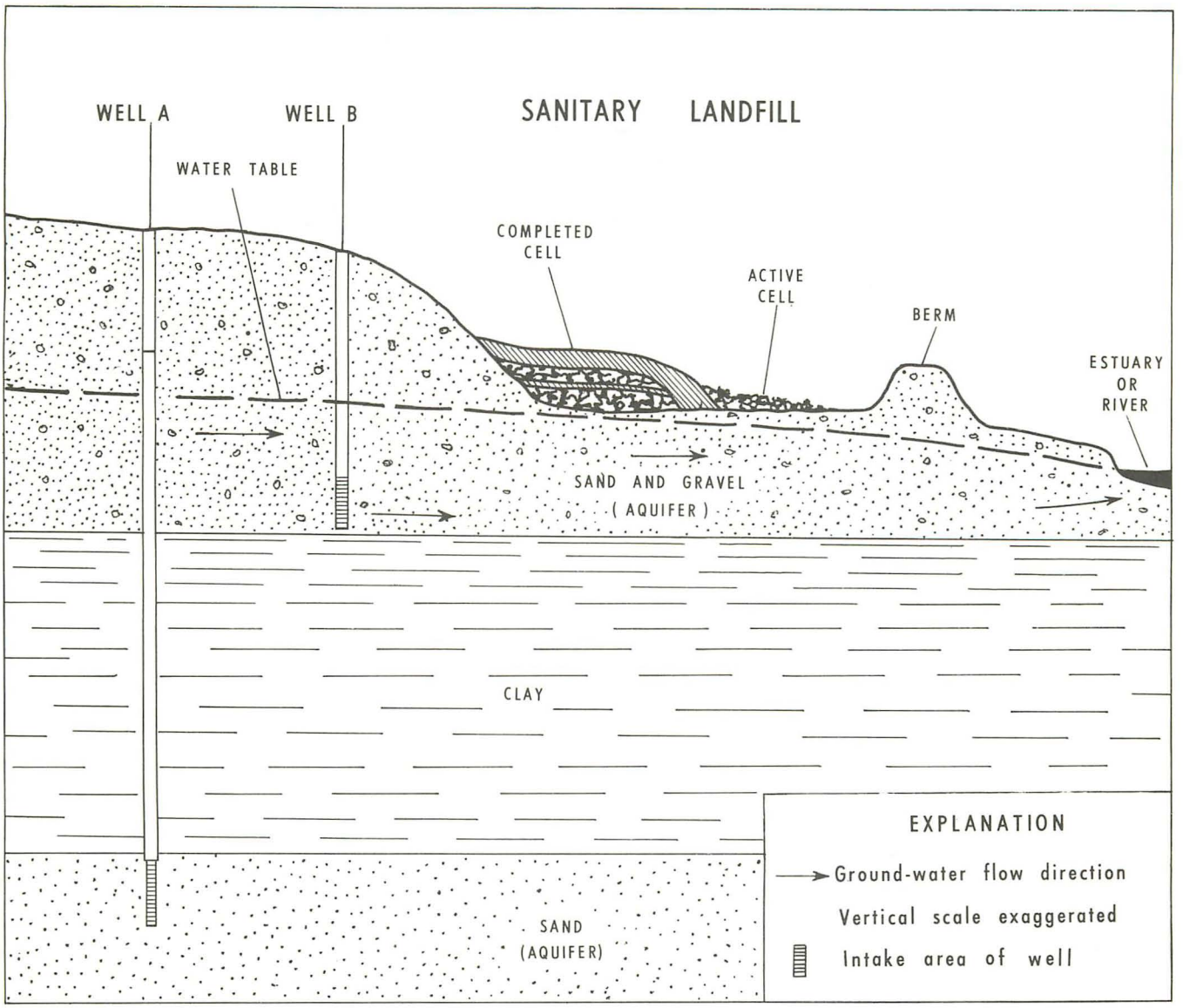


Figure 14. Schematic section in Coastal Plain terrane showing a sanitary landfill overlying both a shallow and a deep aquifer.

just west of the landfill (fig. 15). Before the acquisition of the Mullins property by Harford County for use as a landfill, 18 auger borings were made by a private contractor to show the nature of the underlying material, which consists chiefly of brown sand and gravel, brown and gray silty clay, and clayey sand. The maximum depth augered was about 20 feet. At one hole refusal was encountered at a depth of 3 feet, which may have been the result of the auger bit striking a large boulder, or possibly a ledge of crystalline rock. The Soil Conservation Service (U.S. Dept. of Agriculture, 1969) reports that the following soil types having slight limitations for a sanitary landfill are present: Elsinboro slit loam, Elsinboro gravelly sandy loam, and Matapeake silt loam. The following soils at the site have severe limitations for sanitary-landfill purposes: Comus loam, Keyport silt loam, Elkton silt loam, and Tidal marsh soil.

No visible evidence of leachate was observed at the Mullins landfill when it was first examined during March 1970. By August of that year the first and easternmost cell was completed and filling had been started on the adjacent cell. Examination of the toe of the completed cell showed the characteristic black leachate flowing into a 4- to 5-foot deep water-filled trench about 35 feet long and 10 feet wide. No overflow was visible from the trench at the time, but the water in it was visibly degraded by leachate. No chemical analyses were

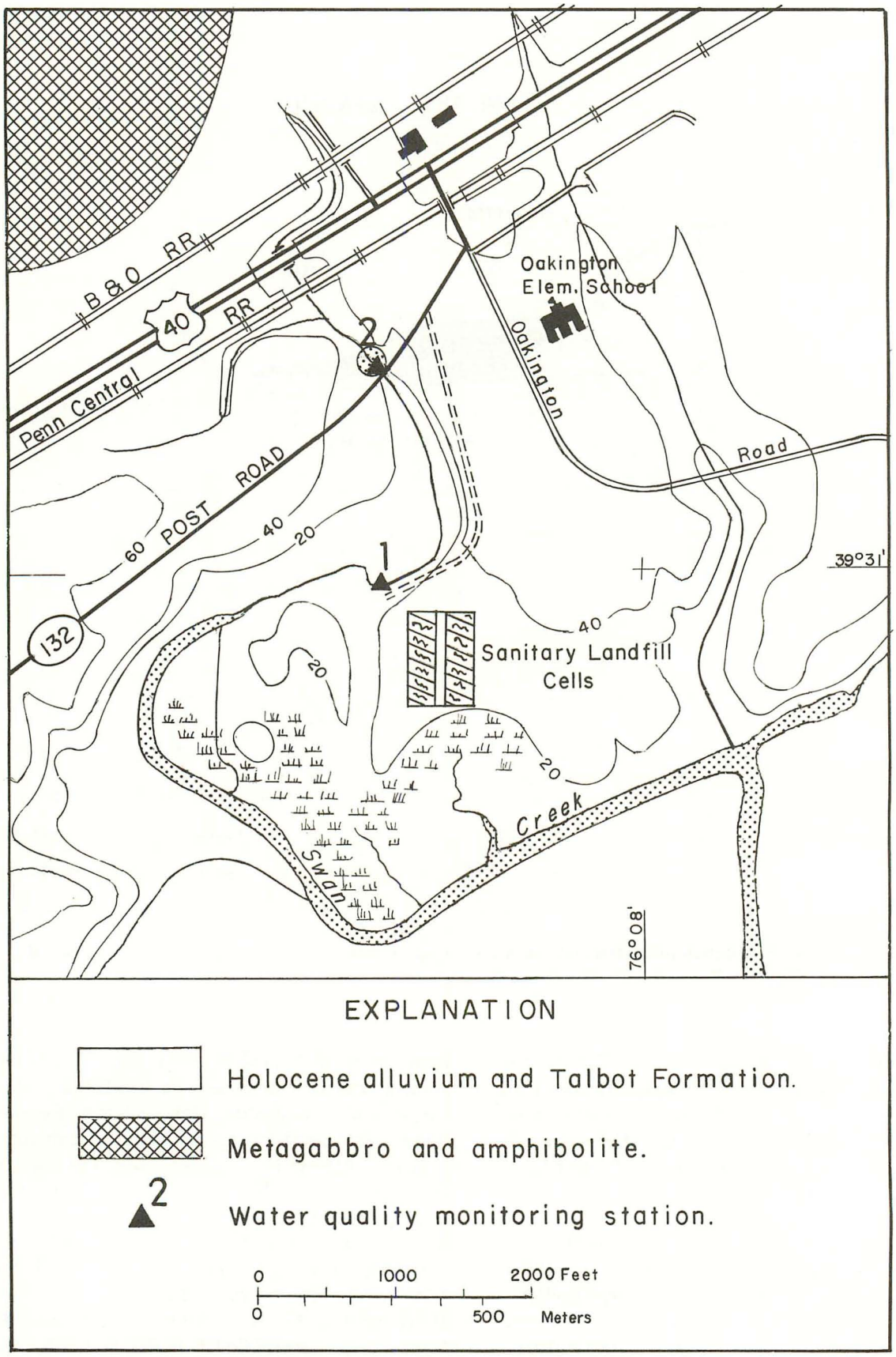


Figure 15. Map showing geology, location of the Mullins sanitary landfill, and water-quality monitoring stations near Aberdeen, Md.

made of the water in this trench.

Based on the possibility that leachate might be moving through the underlying earth materials and entering Swan Creek, two chemical-quality monitoring stations were established in March 1970 to sample the creek water. The first of these (Station 1, fig. 15) is downgradient from the nearest landfill cell at a place where it was believed likely that percolating leachate might be detected. Station 2, selected for control purposes, is on Swan Creek, where it is crossed by Maryland Route 22 (fig. 15). This site is 0.2 mile upstream from the nearest active fill area.

Starting on March 19, 1970, a series of analyses were made to measure total hardness, chloride, dissolved oxygen, specific conductance, pH, temperature, and total and fecal coliforms in the water from Swan Creek at Stations 1 and 2. These results are given in tables 3 and 3A (Appendix) and are summarized in table 15.

Table 15. Comparative mean values of selected constituents and properties of water from two monitoring stations on Swan Creek near the Mullins sanitary landfill.1/

Constituent or property	Swan Creek opposite landfill (Station 1)		Swan Creek at Md. Route 22 (Station 2)	
	Mean value	Number of measurements	Mean value	Number of measurements
Total hardness as $CaCO_3$ (mg/l)	50	7	49	6
Chloride (Cl) (mg/l)	15	4	16	4
Dissolved oxygen (O_2) (mg/l)	9.4	6	9.7	6
Specific conductance (micromhos at 25°C)	150	7	154	7

1/ Analyses by the U.S. Geological Survey.

Of particular interest in this series of measurements were the anomalously high pH values (8.4 and 8.9) measured on March 19 and April 28, 1970. Subsequent investigation of the source of the alkaline stream water by the Maryland Department of Water Resources revealed that alkaline solid wastes were being leached at an upstream site within the small drainage basin. A warning to the offending firm was issued by the State agency, and the alkalinity of the stream water was significantly reduced.

The data in table 15 show no significant difference between the samples from the two sites in the mean total hardness, chloride, or dissolved-oxygen content, and specific conductance.

Total coliforms in the water from Swan Creek (Appendix, table 3 and 3A) in seven samples near the landfill ranged from 800 to 8,200 per 100 ml. These may be compared with a range of 2,600 to 7,800 per 100 ml in six samples from the control station above the landfill. Fecal coliforms in seven samples opposite the landfill ranged from 40 to 3,200 per 100 ml compared to a range of 84 to 3,400 per 100 ml in six samples from the control station. Thus, neither the chemical nor the bacteriologic data show that leachate from the Mullins landfill is causing measurable degradation of the waters of Swan Creek, at least in the reach between the monitoring stations. However, because of the magnitude of possible dilution, it is likely that leachate could mix with the waters of the creek and not be detected on the basis of the measurements made. For example, the flow of Swan Creek was about 1.8 cfs (cubic feet per second) on September 22, 1970, and was about 16.0 cfs on July 8, 1970. The mean of four measurements is about 7 cfs; a flow of 7 cfs is equivalent to about 4.5 million gallons per day. Thus, if the landfill were contributing as much as 10,000 gallons per day of leachate-polluted ground water to the stream, and thorough mixing occurred before sampling, the leachate-polluted ground water would be diluted by a factor of more than 450 to 1 at streamflow rates of 7 cfs in the reach near the landfill.

Terrane V—Flat, Intertidal, Low-lying Areas of Coastal Plain Deposits

Terrane V, consisting of flat, intertidal, low-lying areas underlain by Coastal Plain sedimentary deposits, includes large areas of the Eastern Shore counties of Dorchester, Somerset, Worcester, and smaller areas of other tidewater counties (fig. 8). In localities typical of this terrane the altitude is not more than 10 feet above mean sea level, and in many localities it is less than 10 feet. Typical examples of this terrane are shown on the Crisfield, Md., and Assawoman Bay, Md.-Del. 7½-minute U.S. Geological Survey quadrangle maps.

Crisfield is a small seaport in the southeastern

part of the State about 4 miles north of the Virginia boundary. The maximum land-surface altitude in the entire Crisfield quadrangle is only 7 feet above mean sea level. The outstanding physiographic features of this area are the many swamps, small islands, and generally poor drainage. Although not shown on the topographic map, the shallow water table nearly everywhere lies only a few feet below the land surface. The Assawoman Bay quadrangle presents a similar appearance except that it includes part of Ocean City, built on a barrier beach along the ocean. Much of the land in the quadrangle is swampy, although drainage has been improved by the construction of numerous tidal canals. The highest point in the Maryland part of the quadrangle is a hilltop on St. Martins Neck, only 16 feet above sea level. Interestingly enough, this locality is the site of an old cemetery, indicating that the residents used the land in this manner in one of the few places where the water table was at sufficient depth to permit below-ground burial.

Because the water table lies nearly everywhere close to the land surface, few, if any, dry pits or excavations exist in Terrane V for use as sanitary landfills. However, swampy land is abundant and in places, appears to offer an economical and handy disposal site, providing hydrologic and ecologic factors could be ignored. Under such conditions, swampland has been "reclaimed" by using it for sanitary-landfill purposes, and has been made available for numerous other uses by man when the filling was completed. This has been a common practice in many cities in the United States. It is reported that large areas in and around New York City, New Orleans, Seattle, Baltimore, Richmond, and many other cities have been "reclaimed" by sanitary landfilling (American Public Works Assoc., 1966, p. 111 and 112). An extensive area, known as the "Patapsco flats," in the southwestern area of Baltimore is currently (1970) being filled with incinerator residue and nonputrescible refuse. Although biologists have indicated that the filling or "reclaiming" of swamps may destroy the breeding grounds for birds and the habitat of small fish and other wildlife, discussion of this subject is beyond the scope of this report. The term *reclaiming* is, however, in common use among engineers, land developers, and others.

It is obvious that filling of swamps places the refuse in direct contact with water and must result in the generation of leachate. With regard to degradation of ground-water supplies by leachate from refuse-filled swamps, it would seem that, while such degradation is likely under most circumstances, it has not been reported because: 1) few wells are drilled in the immediate vicinity of sanitary landfills; 2) swamps are commonly discharge areas of ground water, and the movement of water is from the underlying aquifers into the swamps; 3) tidewater swamps may be underlain by shallow aquifers containing brackish water, which is not normally used as a source of water supply; and 4) if a ground-water supply in a coastal area is needed (near a landfill site), the supply probably would be obtained from a deep-lying artesian aquifer normally protected from surficial pollution by one or more confining layers overlying the aquifer. These layers are relatively impervious and would retard or prevent direct vertical movement of a pollutant.

It is possible that pollutants derived from landfills in swamps in Terrane V may be present in the nearby waters in sufficient concentration to be readily detected. The rise and fall of the water surface due to tidal action, as well as tidal currents, tend to disperse these pollutants in surrounding waters. No water-quality analyses were made at or near sanitary landfills in Terrane V. However, it would appear this is an area toward which additional research effort should be directed.

SUGGESTED CRITERIA FOR THE PROTECTION OF WATER RESOURCES FROM THE DELETERIOUS EFFECTS OF LANDFILL LEACHATE

Review of the available literature and analysis and study of the information given in this report indicate that there are means by which the water resources of a locality may be protected, at least to a degree, from degradation by landfill leachate. The first of these involves the classification and segregation of waste materials, possibly in the manner suggested by the California Department of Water Resources (1969, p. 73). The agency proposes the following classification:

Group 1. General industrial wastes with significant water-pollution potential (includes liquid and (or) soluble industrial wastes and toxic industrial

ashes)

Group 2. Household and commercial refuse and rubbish (includes cans, metals, paper products, plastics, shrubbery, lawn cuttings, wood, hair, hide, bones, small dead animals, garbage, etc.)

Group 3. Solid inert wastes (includes earth, rock, bricks, concrete slabs, glass, plaster, slag, asphalt paving fragments, etc.)

Each proposed landfill site is then classified on the basis of the local geology, hydrology, and topography into three major classes, or categories. These are:

Class I. Waste-disposal sites designated as Class I provide the maximum degree of protection for ground water (surface water not mentioned). The following criteria are used: 1) The proposed site is entirely underlain by non-waterbearing earth materials. If underlying rock is highly fractured, grouting of fractures would be required before a site could be designated as Class I; 2) fill area is not in hydraulic continuity with an underlying ground-water body; 3) topography and nature of the earth materials in vicinity must be such that they will permit construction of diversion ditches or drains to divert all runoff from the landfill. Class I sites may receive waste from Groups 1, 2, and 3.

Class II. Some degradation of ground water may be expected at sites in this category. These sites may be ravines, sand or gravel pits, or open areas. Generally they are underlain by rocks having low transmissivity. Where the site is underlain by a confined aquifer, the confining layer should have sufficient thickness to mitigate a direct hydraulic connection between the fill material and the aquifer. The existence of active or inactive wells near the site is a factor to be considered in classification of a particular site. Again, no runoff should be permitted to enter the site and the local topography and earth materials must be such that diversion ditches can be constructed. However, if leachate that discharges to a surface-watercourse is generated, it should be collected and discharged to a sewer, storm drain, or stream channel in such a manner that receiving waters are not adversely affected. Group 2 and 3 wastes are permitted at Class II sites.

Class III. Sanitary landfill sites that afford little or no protection to water resources are designated as Class III. Class III sites have no geologic or hydrologic requirements. The only topographic requirement is that if the site is on the edge of a stream, the stream channel should have adequate channel capacity to prevent erosion and removal of the landfill materials. Class III sites are designated to receive only Group III wastes.

Proposed landfill sites may be modified to provide some protection against degradation of the ground and surface waters. Where the earth material underlying the landfill is highly permeable sand, gravel, or fractured rock, a clay or asphalt lining material may be used to create an artificial barrier to the vertical and lateral movement of leachate (California Dept. of Water Resources, 1969, p. 75). The landfill cells may then be designed to collect the leachate at a designated outflow point, where it may be treated or hauled away for disposal. Sheet-plastic barriers underlying and overlying the fill have been proposed and have been tested in California. However, plastic sheets are subject to puncturing by sharp objects in the fill and by the action of the bulldozer working back and forth across the fill as it compacts and covers the refuse.

Diversion ditches or drains placed along the upper part of the sloping walls of a ravine or depression used as the site of a landfill are recommended to divert runoff from entering the landfill.

Sanitary landfills should not be located in the vicinity of wells or proposed wells where these will tap a shallow aquifer subject to direct surface recharge; or, conversely, wells should not be located in the vicinity of a sanitary landfill that directly overlies a shallow aquifer. Sanitary landfills located in valleys adjacent to rivers or estuaries may result in minimal ground-water degradation, as such sites are the loci for ground-water discharge. However, landfills near streams must inevitably contribute leachate to the waters of the stream. Of course, the degree of hazard then becomes a matter of the nature and quantities of the leachate generated, the dilution to which it is subject, and the downstream uses to which the stream water may be put.

With regard to the lithology of earth materials underlying a proposed landfill site, unfractured shale and clay of sufficient thickness should afford the maximum protection to an underlying ground-water reservoir. As coal strip mines are commonly underlain by clay and shale and as these mines are

frequently well above the local drainage base level, they probably are one of the safest environments in which to dispose of solid wastes, insofar as the potential for degradation of ground water is concerned.

Channeled and creviced limestone probably offers the least protection to the ground-water reservoir, especially where the residuum is thin or lacking. Fractured crystalline rocks also offer little protection to the ground-water reservoir where the saprolite (or weathered zone) is thin or lacking. However, a thick band of saprolite or residuum, extensively distributed over the underlying rock, probably affords a moderately high degree of protection to the ground-water reservoirs because of its low permeability. Furthermore, residuum and other fine-grained materials may effectively filter out both bacteria and colloidal-size pollutants present in the percolating leachate.

Leachate generated from landfills located on permeable sand and gravel in the Coastal Plain may be expected to percolate downward and mix with the underlying ground water, subject to dilution and attentuation with increased distance from the fill. However, where clayey strata directly underlie the fill, leachate probably will move laterally to the nearest point of surface discharge to mix with the surface waters. Coastal Plain aquifers are, in many places, protected from surficial pollution, regardless of its source, by the presence of extensive confining layers lying between deep aquifers and the source of contamination at or near the land surface. Leaky or corroded well casings, however, offer a possible means of vertical pollution of a deep aquifer, where otherwise favorable geologic conditions exist for its protection.

The previous discussion emphasizes the point that geologic and hydrologic conditions are seldom simple and that a wide variety of environmental conditions may be found at localities only short distances from one another. The evaluation of the geologic and hydrologic conditions at any proposed landfill site is a task for the experienced professional, and such a person must draw heavily upon his experience and judgment to draw valid conclusions and to make sound recommendations to protect the water resources.

SUMMARY AND CONCLUSIONS

Nearly 1,500 million tons of solid wastes of all kinds are generated each year in the United States. About 17 percent of the wastes is classed as urban, 7 percent as industrial, and 76 percent as mineral. This report is concerned mainly with the relation of urban-waste disposal practices and the geohydrologic environment.

The quantity of solid urban wastes generated in Maryland has increased significantly during the past 50 years. It is estimated that by 1980 these wastes will amount to 5.5 pounds per capita per day, or a total of about 4.5 million tons per year.

Solid urban wastes are disposed of by open dumping, sanitary landfilling, incineration, composting, and disposal at sea. The practice of sanitary landfilling is recommended by many health officials and sanitary engineers, as it is relatively inexpensive, and when properly done, largely eliminates the objectionable hazardous and unesthetic features of the open dump.

The solid-waste disposal practices of open dumping and sanitary landfilling will, at least locally, influence the land-water environment in a humid region. Elements of the hydrologic environment are governed by the hydrologic cycle. Studies of the cycle in four river basins in Maryland show that precipitation averages about 43 inches per year, evapotranspiration averages about 27 inches, and runoff about 16 inches. The ground-water portion of runoff (or effective recharge) averages about 10 inches per year. Ground-water recharge is significant with regard to solid-waste disposal, as it indicates the possible mean rates of water accretion to sanitary landfills and refuse dumps and the potential for leachate generation.

Hilltops and upland localities are, in general, sites where ground-water recharge predominates over ground-water discharge. Stream valleys and other depressions are localities where ground-water discharge predominates over recharge, except where the natural situation might be altered by pumping from wells. Infiltration capacities of soils and other earth materials vary widely, and the infiltration capacity of a soil is markedly affected by the type and density of the vegetal cover. High runoff rates and low infiltration rates are

typical of clay or shale, especially where plant cover is thin or lacking. Sandy soils in level or undulating topography generally have the highest infiltration and recharge rates.

Analyses of leachate from experimental simulated landfills in West Virginia showed a range in total hardness from 7,600 to 13,100 mg/l. Chloride ranged from 951 to 2,310 mg/l, organic nitrogen from 473 to 1,106 mg/l, and total iron from 175 to 860 mg/l. The pH of the leachate ranged from 5.8 to 6.3. These data have only approximate transfer value in relating them to sanitary landfills in the field.

Published rates of movement of leachate-degraded and otherwise polluted ground water in California, Colorado, Indiana, Michigan, and South Dakota ranged from 1 to 13 feet per day. Total reported travel distances through the ground for the polluting substances ranged from 600 feet to 4 miles. The earth materials through which the pollutants moved were glacial outwash and river alluvium.

With regard to considerations of solid-waste disposal, five major terrane classes are recognized in Maryland. These terranes are characterized by somewhat distinctive geologic and hydrologic conditions, which affect their hydrologic suitability as solid-waste disposal sites. The terrane designations are: Terrane I, dissected shale, siltstone, and sandstone in the Appalachian province; Terrane II, limestone and marble valleys in the eastern part of the Appalachian province and in the Piedmont; Terrane III, crystalline silicate rocks of the Piedmont; Terrane IV, upland areas underlain by Coastal Plain deposits; and Terrane V, flat, intertidal, low-lying areas underlain by Coastal Plain deposits.

The Frostburg and Westernport demonstration landfills in the coal-bearing strata of western Maryland indicate the feasibility of these sites for sanitary landfills. Chemical and bacteriologic analyses have been made of waters from a well, a spring, and an intermittent pond near the Frostburg landfill. Evidence of water pollution has been found only in the intermittent pond lying within the active landfill area.

A sanitary landfill was started in 1969 on a tract of land about 3 miles east of Frederick, Md., at a site underlain by crystalline rocks. In 1970 key chemical constituents in the water were analyzed monthly from the underdrain below the landfill. For control purposes, similar constituents were analyzed in water from a small spring-fed pond above the landfill. These analyses show a significant increase in chloride, total hardness, and specific conductance for the water from the underdrain when compared with water from the spring-fed pond.

A 55-acre sanitary landfill near Texas in Baltimore County occupies the site of a former sand pit. A monitoring program was set up to analyze periodically key chemical constituents in water issuing from the landfill. The chemical analyses show a significant increase in chloride, total hardness, and specific conductance between the control stream and the landfill effluent.

Landfill operations were begun in late 1969 on the 102-acre Mullins Landfill on Swan Creek near Aberdeen. As no water was observed issuing directly from this landfill, a series of chemical analyses of key constituents were made at a point on Swan Creek opposite the fill and at a control point a short distance upstream. These chemical data show no significant differences between the two sampling points. The admittedly limited data do not indicate that ground water of degraded quality is entering the stream from the landfill in sufficient concentrations to be detected.

It would appear desirable to establish an adequate chemical-quality and bacteriologic monitoring program at the three sites described in this report for a period of time to encompass at least the completion of the filling at each site. After each site is completed and abandoned or used for other purposes, there should occur a gradual dieout of the leachate constituents derived from organic decomposition of the fill materials. However, the inorganic constituents probably will continue to be leached for many years.

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APPENDIX

TABLES

APPENDIX

Table 1. Chemical, physical, and bacteriologic analyses of water from the drainage ditch at the Texas, Md. sanitary landfill

(Analyses by the U.S. Geological Survey)

Date of collection	Measured discharge (gpm)	Chemical and physical character						Bacteriologic character		Remarks
		Total hardness as $CaCO_3$ (mg/l)	Chloride (Cl) (mg/l)	Dissolved oxygen (O_2) (mg/l)	Specific conductance (micromhos @ 25°C)	pH	Temperature (C°)	Coliforms (colonies per 100 ml)	Fecal coliforms (colonies per 100 ml)	
Dec. 3, 1969	--	--	--	9.3	950	7.3	4.5	1,100	140	Water clear.
Dec. 18, 1969	--	--	--	7.1	980	7.4	1.0	2,600	160	Clear.
Jan. 14, 1970	--	--	--	5.2	1,300	7.0	1.0	33,000	6,000	Color resembles sewage; turned pale gray in bottle.
Feb. 11, 1970	--	700	--	6.9	1,400	7.1	4.0	19,000	5,400	Color slightly gray; faint septic odor.
Mar. 19, 1970	--	390	--	6.3	1,700	7.4	8.0	320	25	Faint septic odor.
Apr. 28, 1970	35 1/	1,200	--	5.9	1,700	7.2	12	1,400	100	Water pale gray; faint odor.
May 26, 1970	26 2/	860	--	3.8	1,600	7.1	17	10,000	3,500	Water light brown; faint sewage odor.
June 17, 1970	36 3/	920	157	3.5	1,600	7.1	16	4,600	2,300	Water light gray; faint odor; sludge worms.
July 8, 1970 4/	13.5	855	173	4.5	1,150	7.0	17	6,200	1,900	Water clear; fresh sand in ditch.
Aug. 10, 1970	23	975	192	4.3	1,450	7.6	16.5	86,000	1,200	Water clear; gray sediment on bottom; faint odor.
Sept. 22, 1970	11.5	1,045	185	3.3	1,850	7.5	17	4,100	960	Color very pale gray with faint odor.
Nov. 10, 1970	13.5	980	198	4.1	1,880	7.6	12.5	7,400	800	Strong septic odor; oily film on stream; sludge worms; pale grayish color.

1/ Flow measured using a pygmy meter.

2/ V-notch weir installed on this date.

3/ Flow measured 1 day after heavy rain.

4/ Total organic carbon 140 mg/l in this sample.

APPENDIX

Table 1A. Chemical, physical, and bacteriologic analyses of water from Parks Run, the control stream near the sanitary landfill at Texas, Md.

(Analyses by the U.S. Geological Survey)

Date of collection	Chemical and physical character						Bacteriologic character		Remarks
	Total hardness as $CaCO_3$ (mg/l)	Chloride (Cl) (mg/l)	Dissolved oxygen (O_2) (mg/l)	Specific conductance (micromhos @ 25°C)	pH	Temperature (C°)	Coliforms (colonies per 100 ml)	Fecal coliforms (colonies per 100 ml)	
Dec. 3, 1969	--	--	11.4	670	--	5.5	--	--	
Apr. 28, 1970	420	--	10.2	620	7.7	12	1,600	79	Water clear.
May 26, 1970	285	--	7.7	640	7.5	16.5	9,000	380	Flow estimated 1-2 cfs.
June 17, 1970	300	47	8.8	540	7.5	17	3,600	1,500	Water slightly murky after rain.
July 8, 1970	--	--	7.8	560	7.2	17.5	1,000	500	Water clear; gray-tan algae in stream bed.
Aug. 10, 1970 1/	295	55	8.0	580	7.8	17.5	39,000	380	Water clear; gray algae as above.
Sept. 22, 1970	240	40	8.0	500	7.9	18	5,000	2,100	Water clear; light brown filamentous algae in stream bed; sludge worms.
Nov. 10, 1970	315	55	9.0	610	7.8	12	2,800	94	Filamentous tan algae in stream bed as above.

1/ Total organic carbon 6 mg/l in this sample.

APPENDIX

Table 2. Chemical, physical, and bacteriologic analyses of water from the west branch of Boyer Run below the Reichs Ford Road sanitary landfill

(Analyses by the U.S. Geological Survey)

Date of collection	Measured discharge (gpm)	Chemical and physical character						Bacteriologic character		Remarks
		Total hardness as $CaCO_3$ (mg/l)	Chloride (Cl) (mg/l)	Dissolved oxygen (O_2) (mg/l)	Specific conductance (micromhos @ 25°C)	pH	Temperature (C°)	Coliforms (colonies per 100 ml)	Fecal Coliforms (colonies per 100 ml)	
May 12, 1970	--	310	30	4.1	440	7.0	21	1,000 2/	230 2/	Faint odor; pale gray color.
June 4, 1970	14 1/	200	--	4.0	430	6.9	19	300	5	Do.
July 15, 1970	29	230	30	5.5	360	6.9	19	--	--	Reddish algae in stream bed; black leachate issuing from toe of landfill.
July 29, 1970	12 3/	165	25	1.4	450	7.5	26	1,500	27	Odor of sulfur; red sludge worms on bottom.
Aug. 17, 1970	-- 4/	240	30	2.5	580	7.2	23.5	200	140	Brownish-red precipitate on rocks.
Sept. 30, 1970	12	190	17.5	3.3	420	7.2	14.5	2,000	4	Odor of sulfur; oil slick on water surface.
Nov. 10, 1970	13.5	235	50	6.3	520	7.3	11.5	4,400	29	Reddish sediment in ditch; slight septic odor.

1/ Weir installed on May 27, 1970; discharge measured 21 gpm.

2/ Bacteriologic sample collected on May 19, 1970.

3/ Weir moved approximately 210 feet downstream prior to measurement.

4/ Weir clogged with debris - unable to make measurement.

APPENDIX

Table 2A. Chemical, physical, and bacteriologic analyses of water from the spring-fed pond above the Reichs Ford Road sanitary landfill

(Analyses by the U.S. Geological Survey)

Date collection	Chemical and physical character						Bacteriologic character		Remarks
	Total hardness as $CaCO_3$ (mg/l)	Chloride (Cl) (mg/l)	Dissolved oxygen (O_2) (mg/l)	Specific conductance (micromhos @ 25°C)	pH	Temperature (C°)	Coliforms (colonies per 100 ml)	Fecal coliforms (colonies per 100 ml)	
June 4, 1970	65	--	6.2	170	6.9	22	1,000	30	
July 15, 1970	60	7.5	12.8	140	7.5	22	--	--	Water pale green.
July 29, 1970	65 1/	7.5 1/	12.6	130	8.1	29	150	120	Water clear.
Aug. 17, 1970	75	12	8.0	170	7.7	27	800	420	Water brown, turbid.
Sept. 30, 1970	60	5	8.0	140	7.4	13.5	4,600	130	Water brown, turbid; no odor.
Nov. 10, 1970	85	15	7.4	190	7.1	9.5	6,600	900	Water pale gray; slightly murky.

1/ Samples for chloride and hardness collected Aug. 4, 1970.

APPENDIX

Table 3. Chemical, physical, and bacteriologic analyses of water from Swan Creek near the Mullins sanitary landfill

(Analyses by the U.S. Geological Survey)

Date of collection	Measured discharge (cfs)	Chemical and physical character						Bacteriologic character		Remarks
		Total hardness as $CaCO_3$ (mg/l)	Chloride (Cl) (mg/l)	Dissolved oxygen (O_2) (mg/l)	Specific conductance (micromhos @ 25°C)	pH	Temperature (C°)	Coliforms (colonies per 100 ml)	Fecal coliforms (colonies per 100 ml)	
Mar. 19, 1970	--	25	--	13.2	230	8.4	4.0	3,200	120	
Apr. 28, 1970	--	80	--	--	130	8.9	15	800	40	Water clear.
May 26, 1970	6.6	45	--	7.8	140	7.0	19	2,200	400	Do.
June 17, 1970	16.0	45	15	8.2	140	6.7	19	8,200	3,200	Water murky brown due to heavy rain previous day.
July 8, 1970 1/	--	55	15	9.0	140	7.1	22	4,000	300	Water clear.
Aug. 10, 1970	4.7	45	12	9.4	110	7.6	21	4,200	1,200	Do.
Sept. 22, 1970	1.81	55	20	8.6	160	7.5	21	1,300	68	Water clear; weather clear, hot, and dry.

1/ Total organic carbon 2 mg/l in this sample.

APPENDIX

Table 3A. Chemical, physical, and bacteriologic analyses of water from Swan Creek at Maryland Route 22 above the Mullins sanitary landfill

(Analyses by the U.S. Geological Survey)

Date of collection	Measured discharge (cfs)	Chemical and physical character						Bacteriologic character		Remarks
		Total hardness as $CaCO_3$ (mg/l)	Chloride (Cl) (mg/l)	Dissolved oxygen (O_2) (mg/l)	Specific conductance (micromhos @ 25°C)	pH	Temperature (C°)	Coliforms (colonies per 100 ml)	Fecal coliforms (colonies per 100 ml)	
Mar. 19, 1970	--	25	--	12.7	230	8.3	4.5	3,900	84	
Apr. 28, 1970	--	80	--	--	130	8.9	16	2,600	190	Water clear.
May 26, 1970	--	45	--	8.0	140	7.3	19.5	3,800	620	Do.
June 17, 1970	--	--	17	8.7	160	6.9	20	7,800	3,400	Color murky brown due to rain.
July 8, 1970 1/	--	55	15	9.3	140	7.6	22.5	--	--	Water clear.
Aug. 10, 1970	--	45	15	9.9	100	8.2	23	7,800	2,500	Clear; no odor.
Sept. 22, 1970	--	45	17	9.4	182	8.0	22	3,600	150	Clear.

1/ Total organic carbon 2 mg/l in this sample.