GROUND-WATER FLOW AND THE POSSIBLE EFFECTS OF REMEDIAL ACTIONS AT J-FIELD, ABERDEEN PROVING GROUND, MARYLAND

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CONVERSION FACTORS, ABBREVIATIONS, AND VERTICAL DATUM

Multiply	By	To obtain
inch (in.)	2.54	centimeter
inch/year (in/yr)	2.54	centimeter/year
foot (ft)	0.3048	meter
foot per day (ft/d)	0.3048	meter per day
foot squared (ft^2)	0.0929	meter squared
foot squared per day (ft^2/d)	0.0929	meter squared per day
mile (mi)	1.609	kilometer
acre	0.4048	hectare
square mile (mi^2)	2.590	square kilometer
gallon per minute (gal/min)	3.785	liter per minute

Sea Level: In this report, "sea level" refers to the National Geodetic Vertical Datum of 1929--a geodetic datum derived from a general adjustment of the first-order level nets of the United States and Canada, formerly called Sea Level datum of 1929.

The standard unit for hydraulic conductivity is cubic foot per day per square foot $[(ft^3/d)/ft^2]$. This mathematical expression reduces to foot per day (ft/d).

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ABSTRACT

J-Field is located in the Edgewood Area of Aberdeen Proving Ground, Md., and has been used since World War II to dispose of munitions, explosives, chemical-warfare agents, and industrial chemicals. Ground-water, surface-water, and soil contamination has resulted from these past activities. The U. S. Geological Survey finite-difference model was used at J-Field to better understand ground-water flow at the site and to simulate the effects of remedial actions. Two layers were used in the model to simulate a surficial aquifer and a confined aquifer. A confining unit separates these two units and is represented in the model by the leakance between the two layers. The area modeled is 3.65 square miles; the model was constructed with a variably spaced 40 X 38 grid. The horizontal boundaries and the bottom boundary of the model are all no-flow boundaries. All the simulations were conducted under steady-state conditions. Tidal wetlands and estuaries were simulated as fixed heads and nontidal wetlands were simulated using either the river subroutine or the drain subroutine.

Ground water at all three of the solid-waste-management units under investigation flows from disposal pit areas toward discharge areas in the estuaries or in wetlands. Remedial actions were not simulated at the white-phosphorus disposal area, because rapid flow through the system has allowed much of the contamination at that site to discharge offshore. Simulations show that capping the riot-control-agent and toxic-materials disposal areas with an impermeable cover is effective at slowing advective ground-water flow by about 0.7 and 0.5 times, respectively. Barriers to horizontal ground-water flow were simulated and effectively prevented the movement of contaminated ground water toward discharge areas, such as wetlands. A horizontal hydraulic conductivity less than 0.005 feet per day was required for the simulated 10-foot-wide barriers to be effective.

Extraction wells were simulated as a way to contain ground-water contamination and as a way to extract contaminated ground water for treatment. Simulations indicated that two wells pumping 5 gal/min (gallons per minute) each at the toxic-materials disposal area and a single well pumping 2.5 gal/min at the riot-control-agent disposal area would be effective for containing contaminated ground-water at these sites. A combination of barriers to horizontal flow both north and south of the toxic-materials disposal area and a single extraction well pumping at 5 gal/min can be used at the toxic-materials disposal area to extract contaminated ground water and to prevent pumpage of wetland water. In the pumpage scenarios at the toxic-materials disposal area, ground-water discharge to the wetland is captured by the wells and could result in a reduction in the wetland area.

INTRODUCTION

J-Field is located at the southernmost tip of the Gunpowder Neck Peninsula on the western shore of the Chesapeake Bay in the Edgewood Area of Aberdeen Proving Ground, Md. (fig. 1). J-Field has been used by the U.S. Army to test chemical-warfare-agent filled munitions and to dispose of toxic chemicals, chemical-warfare agents, and explosives by open-pit burning. Testing began shortly after World War I and large-scale disposal operations began shortly after World War II and continued into the 1970's. Since the early 1980's, only emergency disposal operations are conducted at J-Field.

In 1986, J-Field was placed under the regulations described by the Resource Conservation and Recovery Act (RCRA) that govern operations at hazardous-waste-disposal sites. In 1987, the U.S. Army contracted the U.S. Geological Survey (USGS) to conduct a Hydrogeologic Assessment (HGA) of J-Field. The USGS began a study to determine the hydrogeologic framework and the extent of ground-water contamination at J-Field. In 1990, all of the Edgewood Area of Aberdeen Proving Ground, including J-Field, was added to the National Priority List (NPL) by the U.S. Environmental Protection Agency (USEPA) and hence came under the regulations described by the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), also known as Superfund. In order to complete all of the CERCLA requirements for a Remedial Investigation (RI) and Feasibility Study (FS), Argonne National Laboratory was contracted by the U.S. Army in 1991 to assist the USGS with the RI/FS. The USGS is responsible for a subset of the work required for the RI, primarily work that was originally planned for the HGA.

Several techniques were used to determine the hydrogeologic framework for the HGA. These included electromagnetic-resistivity surveys, continuous seismic profiling, drilling observation wells, subsurface mapping of hydrogeologic units, measuring water levels, conducting aquifer tests, and numerical modeling of ground-water flow. The USGS has also sampled soil, soil gas, surface water, and ground water to determine the types and extent of contamination.

Purpose and Scope

The purpose of this report is to describe ground-water flow and the possible effects of selected remedial actions on ground-water flow and contaminant movement at J-Field using the results of a ground-water-flow model.

Five remedial actions were simulated: installation of an impermeable cover, installation of barriers to horizontal ground-water flow, installation of extraction wells, and installation of a combination of barriers to horizontal flow and extraction wells. This report describes the possible effects of these remedial actions on ground-water flow and the likely movement of contaminants in ground water as a result of these remedial actions.

Description of Study Area

J-Field is located at the southernmost end of the Gunpowder Neck Peninsula (fig. 1). The topography is relatively flat. Tidal estuaries surround J-Field on three sides: the Gunpowder River on the west and the Chesapeake Bay to the south and east. North-south-trending uplands, 10 to 15 ft above sea level, are present along the west-central side of the study area and slope gently toward either the shores of the surrounding estuaries or toward wetlands. At some locations along the shore, wave erosion has produced 2- to 10-ft-high cliffs. J-Field is comprised of mowed fields, second-growth forest, forested wetlands, and open tidal and nontidal wetlands.

Approach

Hydrogeologic data obtained as part of the remedial investigation of J-Field were used to design a ground-water-flow model of J-Field and the surrounding area. Input data for the model included the distribution, thicknesses, and hydraulic conductivity of hydrogeologic units that were mapped as part of the HGA (Hughes 1993). Ground-water levels, tidal information, and surface-water levels in wetlands were also used as input

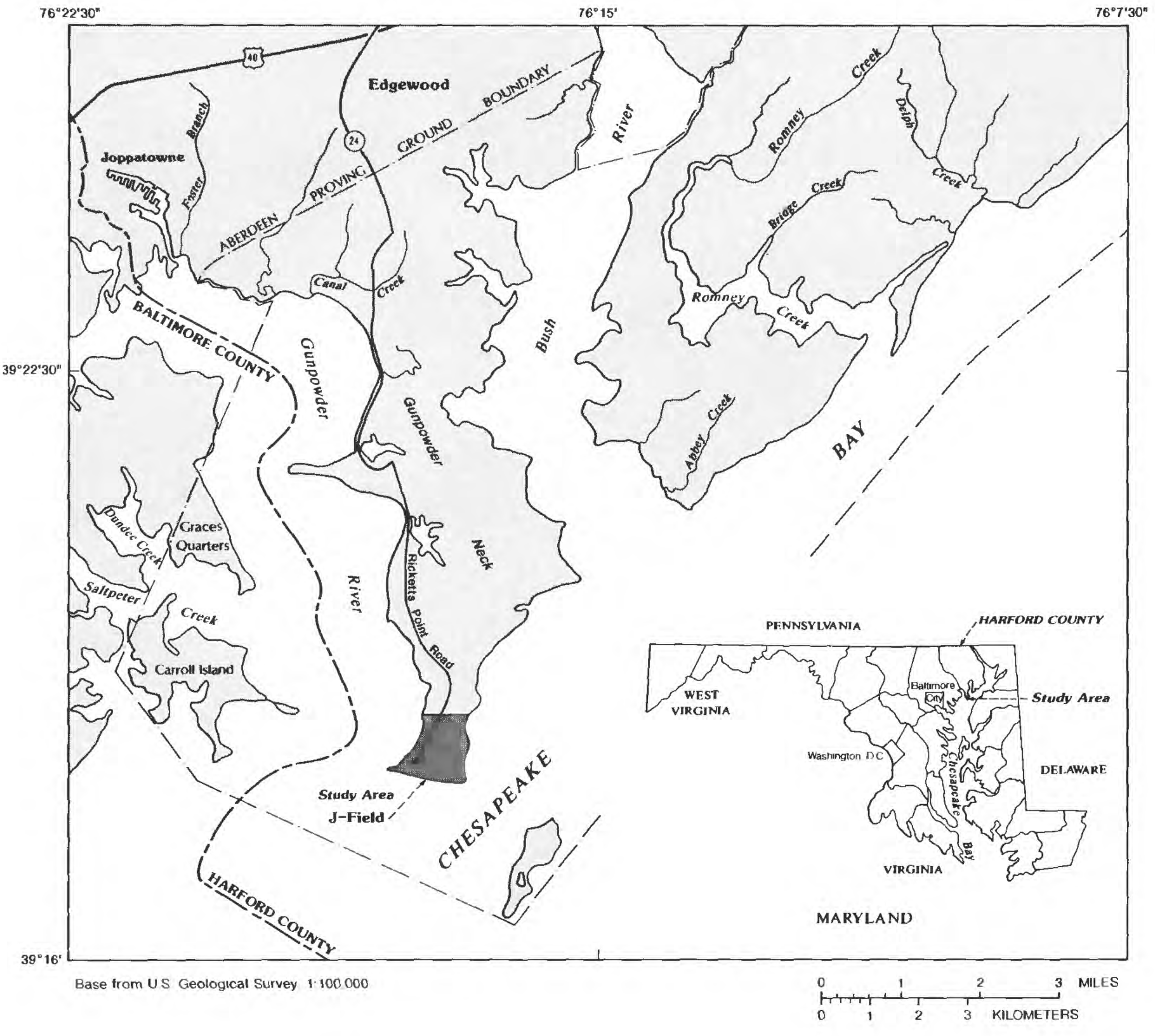


Figure 1. Location of J-Field study area.

data and to calibrate the model. Rainfall data collected by the U.S. Army was used in conjunction with hydrographs of ground-water levels to calculate recharge for the model. U.S. Army maps and field observations were used to distinguish between tidal and nontidal wetlands.

A numerical model of J-Field and the surrounding area was constructed using the USGS modular quasi three-dimensional finite-difference ground-water-flow model for an area of approximately 3.65 mi^2 (square miles) that encompasses extensive offshore areas. The model was calibrated to steady-state conditions by adjusting the hydraulic parameters so that the simulated heads and the heads measured in the field agreed within reasonable limits. Remedial actions were simulated with the steady-state model to determine the effects on ground-water flow. The ground-water-flow paths for remedial actions were plotted with the USGS particle-tracker subroutines.

History of Disposal Activities

J-Field has been used since World War I for testing high-explosive and chemical munitions. Detailed records of the location and nature of the tests are not available and a summary of the quantities of chemicals released on J-Field is not available. Nemeth (1989) suggests that there is little possibility of residual environmental contamination from testing because the chemical-agent tests were on such a small scale.

During 1940-70, open-pit burning at J-Field was used extensively to dispose of many types of chemical agents, high explosives, and chemical wastes. Although no records were kept of the quantities and types of chemicals and agents that were disposed of in this manner at J-Field, they probably included various nerve agents, vomiting agents, riot-control agents, and mustards. In addition, munitions containing these agents, white phosphorus, and high explosives were also disposed of at J-Field. Chemical wastes were primarily those generated from the industrial production of chemical-warfare agents at APG and probably consisted of organic solvents. Other materials disposed of at J-Field were napalm, liquid smoke materials, and agent-contaminated storage or manufacturing materials (Nemeth and others, 1983).

The typical procedure for open-pit burning was to place wood dunnage in a disposal pit, add the agents, munitions, and other chemicals, and then flood the pit with a flammable hydrocarbon fuel, such as fuel oil. The materials were ignited and containers were simultaneously opened by an explosive charge. After the burn was completed, the remaining materials were moved to the adjacent reburn pit, where the process was repeated. After completion of the second burn, some of the remaining debris was pushed into the adjacent wetlands and some metal debris was removed and sold as scrap. An unknown quantity of the liquid materials, such as fuels, organic solvents, and agents, probably infiltrated into the soil and could have contaminated soil and ground water (Nemeth, 1989).

Since the early 1980's, only laboratory chemicals from small-scale testing and unexploded ordnance (UXO) discovered during excavations at APG have been disposed of at J-Field. The disposals are conducted by detonating the UXO or laboratory containers with enough high explosives to destroy the chemicals in the resulting fireball.

Disposal of hazardous materials was primarily conducted in three solid-waste-management units at J-Field: the toxic-materials disposal area, riot-control-agent disposal area, and white-phosphorus disposal area (fig. 2). Both the toxic-materials and white-phosphorus disposal areas contain two parallel disposal pits approximately 15 ft apart. Each pit is 10 ft deep and approximately 200 ft long by 15 ft wide. At the toxic-materials disposal area, remnants of older pits extend approximately 100 ft into the wetlands southeast of the existing pits. The riot-control-agent disposal area contains a single pit, approximately 500 ft long by 15 ft wide. All of the pits were originally designed so that precipitation that collected in them would drain into the adjacent wetlands or river. Since the 1970's, all the pits at J-Field have been blocked by mounds of soil to prevent drainage from the pits.

The area to the east of the toxic-materials disposal area is where unburned materials and soil were pushed out of the disposal pits and into the wetlands (the "push-out" area). The area southeast of the toxic-materials disposal area contains numerous craters ranging from 5 to 20 ft in diameter and from 5 to 10 ft in depth. These small craters were probably used for burning or demolition

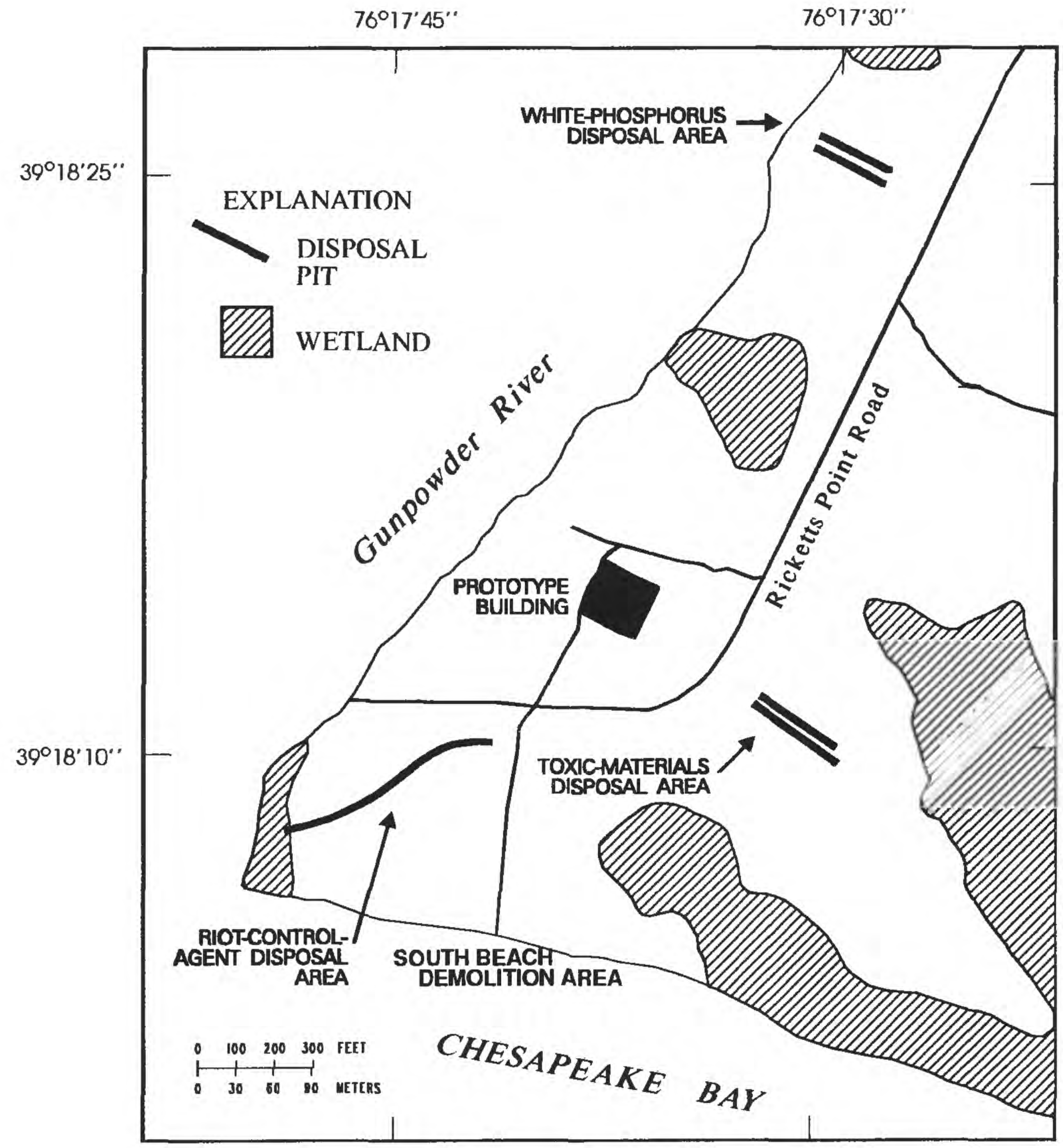


Figure 2. Location of solid-waste-management units.

(Gary Nemeth, U.S. Army Environmental Hygiene Agency, oral commun., 1988).

The prototype building (fig. 2) is a three-story, steel-reinforced, open concrete structure that was probably used to store chemicals. The prototype building was designed to simulate typical German construction practices used during World War II; it was used to test the effectiveness of various weapons on such structures. Although no records of such use are known to exist, the numerous circular stains on the concrete floor of the building indicate that rusty 55-gallon drums were stored there. A large open field surrounds the prototype building and a sidewalk extends from the building down to the Gunpowder River. There are no pits or other obvious signs of disposal activities in the immediate vicinity of the prototype building.

The South Beach demolition area was used primarily to detonate high-explosive munitions. Because of the high rates of shoreline erosion in this part of J-Field, the South Beach demolition area is now offshore in the Chesapeake Bay (fig. 2). Its presence is marked only by the abundant fragments of munitions that can be observed at low tide. More detailed descriptions of the solid-waste management unit (SWMU's) can be found in Nemeth and others (1983) and Nemeth (1989).

Previous Investigations

The first environmental survey of J-Field was conducted during 1977-78 by U.S. Army Toxic and Hazardous Materials Agency (USATHAMA) (Nemeth and others, 1983). The study involved searching records, collecting hydrogeologic data, and sampling soil, sediment, ground water, and surface water for chemical analyses. Wells installed for the study were screened approximately 15 ft below land surface. Nemeth and others (1983) concluded that deposits of interbedded sand and clay encountered during test-hole drilling belonged to the Cretaceous Potomac Group. Water levels measured in observation wells indicated that the horizontal component of ground-water flow was from the upland areas toward the adjacent rivers or wetlands, and that the water table generally followed the configuration of the land surface. Soil, bore-hole sediment, and surface-water samples collected during the study did not contain any contaminants. Ground-water samples contained low concentrations of volatile organic compounds. On the basis of low or undetectable concentrations, Nemeth and others (1983) concluded that the concentrations of contaminants at J-Field were not a threat to the environment and that future monitoring was not necessary.

A munitions-disposal study was conducted in 1983 by Princeton Aqua Science (1984) to evaluate the environmental effects of the disposal operations at J-Field. The study involved site inspections, interviews with appropriate site operations personnel, and field investigations. Nine observation wells were installed. During drilling, borehole samples were collected and analyzed for chemical constituents. Borehole sediment samples at the toxic-materials disposal area were found to contain concentrations of arsenic, cadmium, lead, and mercury that were higher than concentrations in adjacent areas. After the wells were completed, ground-water samples were collected and analyzed for chemical constituents. Water samples collected from wells at the toxic-materials disposal area exceeded the 1983 USEPA primary drinking-water regulations for nitrates, coliform bacteria, and gross-beta radiation. USEPA secondary drinking-water regulations for chloride, iron, manganese, and sulfate were also exceeded. At the white-phosphorus disposal area, the primary drinking-water regulation for coliform bacteria was exceeded as were the secondary regulations for iron and sulfate. Contrary to these findings, the study concluded that the burning operations did not adversely affect ground-water quality, and the disposal practices did not need to be substantially altered (Princeton Aqua Science, 1984).

The RCRA Facility Assessment (Nemeth, 1989) contains the most comprehensive information available on the disposal of chemicals in the study area. The report reviews and summarizes previous work at J-Field. Because several contaminants were identified in ground water collected at J-Field, Nemeth (1989) recommended continued investigation at the toxic-materials disposal area, white-phosphorus disposal area, riot-control-agent disposal area, prototype building, and South Beach area.

The hydrogeology and soil-gas contamination at J-Field was studied as part of the remedial investigation by the USGS (Hughes, 1993). This study showed that the geologic units at J-Field consisted of Cretaceous fluvial deposits overlain by Pleistocene paleochannel deposits. The thickness and distribution of the hydrogeologic units were mapped in onshore areas by well drilling, and offshore by continuous seismic profiling. Shallow ground water was shown to flow horizontally from upland recharge areas toward discharge areas in the wetlands, the Gunpowder River, and Chesapeake Bay. Some ground water flows vertically through a confining unit. Soil-gas analysis indicated that chlorinated solvents, aromatic hydrocarbons, alkanes, and phthalates are present in soil and/or ground water at the white-phosphorus disposal area, the toxic-materials disposal area, and the riot-control-agent disposal area.

GROUND-WATER FLOW

The regional flow system in the upper Chesapeake Bay region (fig. 3) consists of recharge on the eastern and western shores of the Chesapeake Bay with horizontal flow toward, and discharge into the bay (Otton and Mandle, 1984). On the western shore, recharge results from precipitation above the Cretaceous deposits and ground water generally moves downdip toward the east. On the eastern shore, recharge results from precipitation above Pleistocene units and ground water moves vertically downward through Tertiary deposits and Cretaceous deposits. Some of the water recharged on the eastern shore flows east toward the Atlantic Ocean; some flows west toward the Chesapeake Bay. On both sides of the bay, ground water following the shortest flowpaths moves from upland areas and discharges into adjacent wetlands, streams, or estuaries. Following intermediate flowpaths, ground water recharged on the eastern and western shores moves horizontally and discharges upward into the Chesapeake Bay. The longest flowpath is followed by ground water flowing from the western shore downdip beneath the Chesapeake Bay toward the Atlantic Ocean.

Hydrogeology of J-Field

Four major hydrogeologic units were identified beneath J-Field (fig. 4). From the surface downward these are (1) a surficial aquifer, (2) a confining unit, (3) a confined aquifer, and (4) confining units and confined aquifers in the Patapsco Formation. The first three units are subdivisions of the Talbot Formation, a fluvial and estuarine unit that in the J-Field area was deposited in a Pleistocene paleochannel (Hughes, 1991). The Patapsco Formation, a much older Cretaceous unit, was deposited in a fluvial environment.

The surficial aquifer is a heterogenous mixture of medium-grained to fine-grained sand with interbedded clay. The sand and clay beds are from 2 to 10 ft thick and are horizontally discontinuous. The sand is generally red to gray in color and the clay is dark to light gray. The total thickness of this unit is approximately 30 to 40 ft and horizontal hydraulic conductivity ranges from 0.29 to 1.0 ft/d as measured by slug tests (Hughes, 1993).

The confining unit is composed of olive-gray, silty, sandy clay. The sand is very fine grained and comprises less than 30 percent of the unit. Bivalve shells and shell fragments range from trace percentages in the upper part of the unit to as much as 70 percent of the sample in some sections near the base. Minor amounts of fine-grained organic particles are present in some zones, whereas leaves, stems, and large woody fragments comprise up to 75 percent in other zones. The confining unit ranges in thickness from 40 to 110 ft and the range of horizontal hydraulic conductivity as measured by slug tests was <0.01 to 0.20 ft/d (Hughes, 1993). Although the highest measured horizontal hydraulic conductivity in the confining unit is similar to the lowest measured in the surficial aquifer, the median values measured for the two units differed by an order of magnitude. A head gradient of as much as 6 ft between the surficial aquifer and the confined aquifer indicates that the confining unit significantly slows the movement of water between the two units.

The confined aquifer is composed primarily of gravelly sand mixed with abundant clay and clayey sand that ranges from 40 to 50 ft thick. The gravel is well-rounded and ranges in size from pebbles to small cobbles. The larger clasts are predominantely rock fragments of sandstone, granitic rock, or gneiss. The horizontal hydraulic conductivity of the confined aquifer as measured by slug tests ranged from 3 to 900 ft/d (Hughes, 1993).

The Patapsco Formation of the Potomac Group is a fluvial unit composed of fine-grained sand and dense red or light gray clay. The full thickness of the unit in this area is not known because drilling did not fully penetrate the

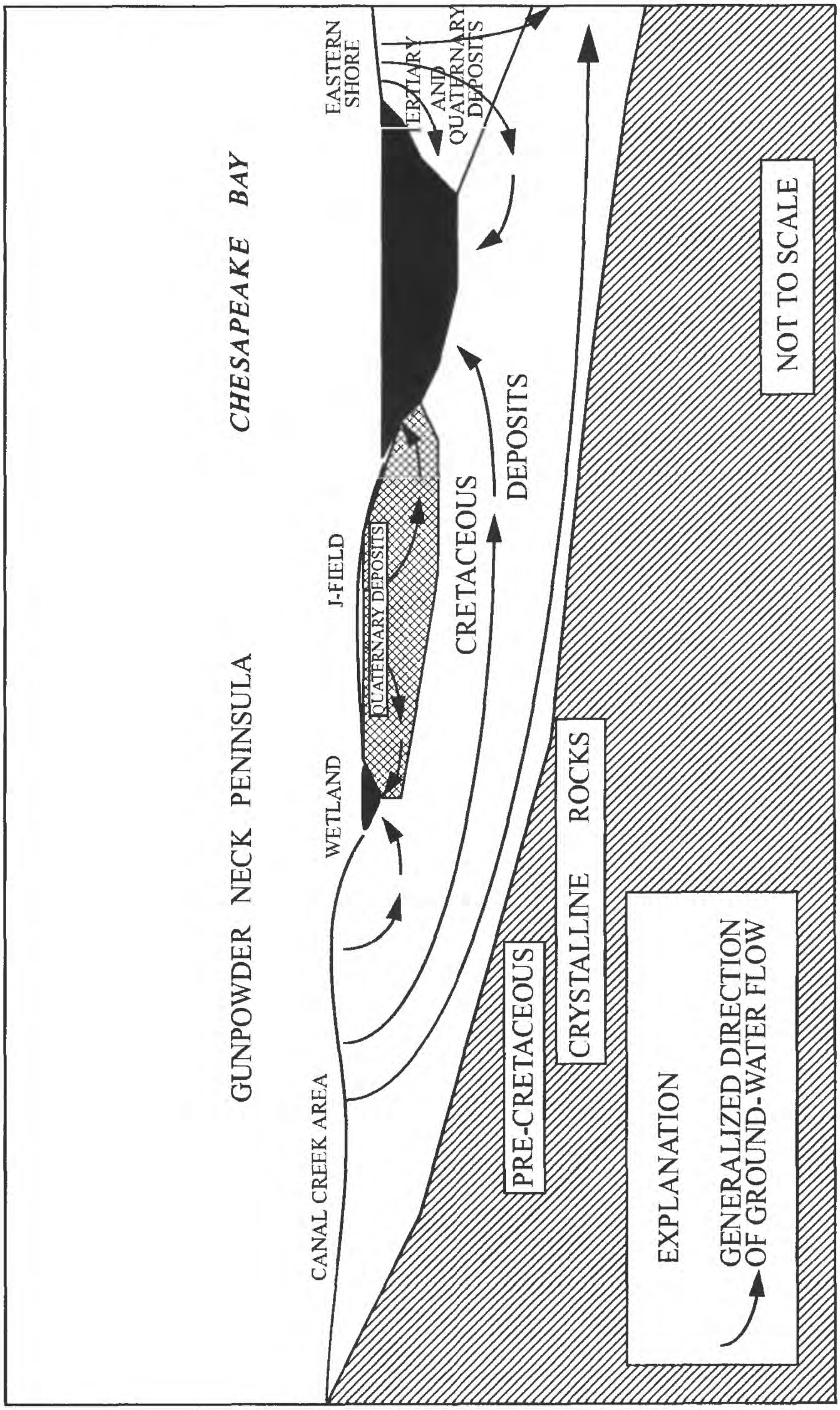


Figure 3. Geologic units and generalized ground-water flow directions in the upper Chesapeake Bay area.

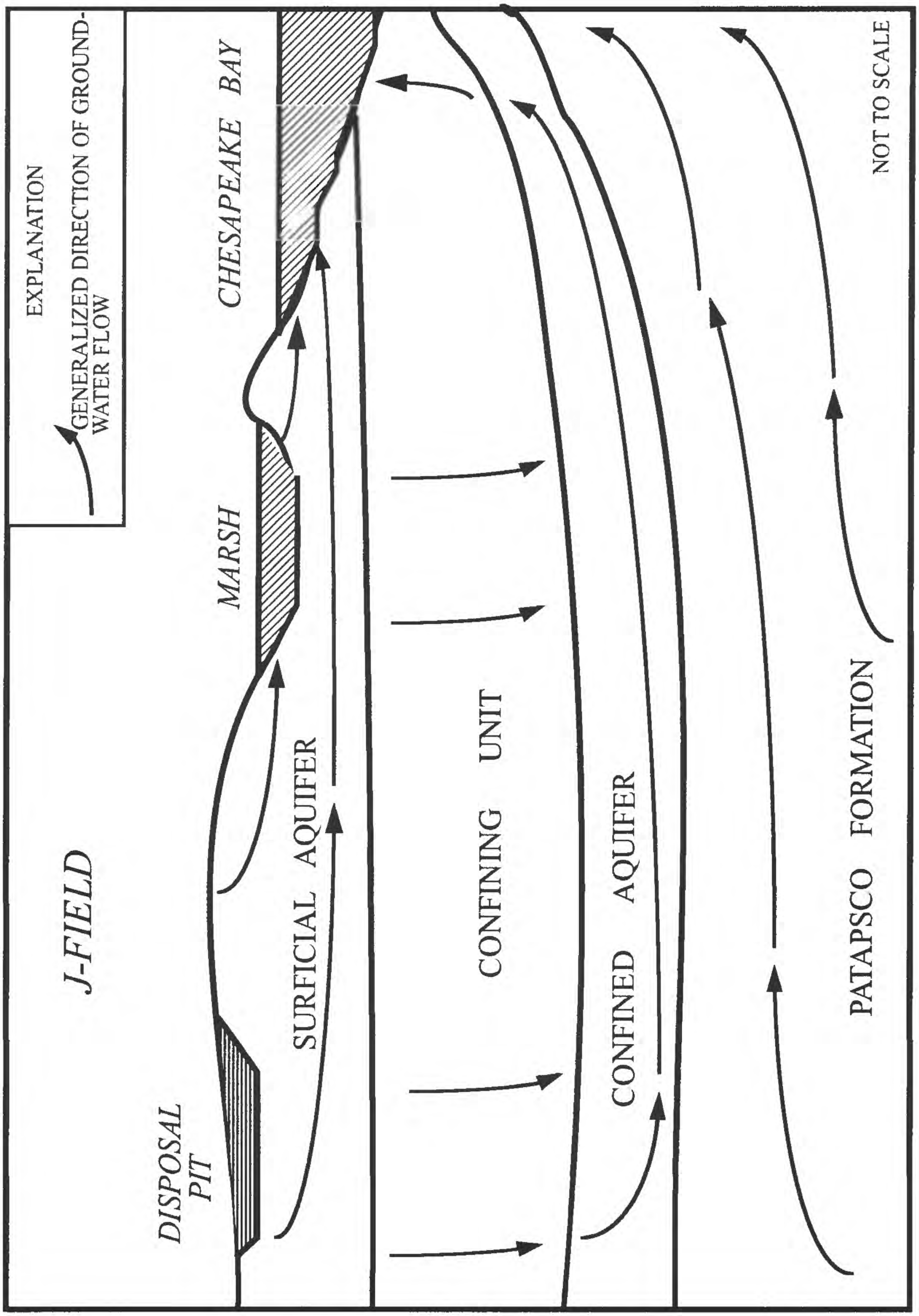


Figure 4. Hydrogeologic units and generalized ground-water-flow directions at J-Field.

formation. The deepest borings at J-Field penetrated as deep as 190 ft into the Patapsco Formation. All of the deep borings at J-Field encountered dense, red or gray clay 10 ft or less below the bottom of the confined aquifer, indicating that a confining unit underlies the confined aquifer (Hughes, 1993). The horizontal hydraulic conductivity of the first sand layer in the Patapsco Formation as measured by slug tests ranged from 0.06 to 0.61 ft/d. The hydraulic conductivities of the clay layers in the Potomac Group are unknown but are likely to be quite low.

Recharge and Discharge Relationships

Recharge to the shallow ground-water-flow system results from precipitation on the land surface at J-Field. Water percolates downward through the unsaturated zone to the water table in the surficial aquifer. Most of the water in the surficial aquifer flows horizontally toward discharge areas in the adjacent wetlands or estuaries. A small fraction of the water in the surficial aquifer flows slowly downward through the confining unit and eventually discharges into the confined aquifer. Horizontal ground-water flow in the confining unit and the confined aquifer is slow and toward the adjacent estuaries. Ground-water flow in the Patapsco Formation is dominated by the regional flow system. In this flow system, ground water recharged on the western shore of the Chesapeake Bay flows horizontally through the Cretaceous deposits and eventually discharges into the bay. Some of the ground-water recharged on the western shore passes beneath the bay and Delmarva Peninsula, eventually discharging to the Atlantic Ocean (Otton and Mandle, 1984).

Ground water at J-Field discharges to tidal estuaries on the east, west, and south, and to extensive wetlands located along the southern and eastern shores of J-Field. Sand accumulations on the southern and southeastern shores of J-Field have formed low ridges 3 to 5 ft above high tide. The sand ridges act as dams, preventing surface water in the wetlands from draining directly into the estuaries. Consequently, the water level in the wetlands, although seasonal, can be as high as 2 ft above the high-tide level in the estuaries. During storms and unusually high tides, estuary water can flood the wetlands, as is demonstrated by the abundant debris deposited in the wetlands.

Several ponds 50 to 600 ft in diameter are located in the wetlands. When the water level in the wetlands is high, the ponds are interconnected, separated only by raised mats of vegetation. Conversely, in late summer when water levels in the wetlands are low, the smaller ponds dry up and the larger ponds are separated by land. The largest body of open water in the wetlands is southeast of the toxic-materials disposal area. The water depth in this pond fluctuates seasonally from a maximum of approximately 3 ft in the spring to 1 ft in late summer. Two stream channels drain the eastern side of J-Field. The lower reaches of both streams are flooded by tides. Flow in the upper reaches, which are above the high tide mark occurs only during storms.

Description of Ground-Water Flow Model

Advective ground-water flow was simulated with the USGS finite-difference ground-water-flow model (McDonald and Harbaugh, 1984). Advection is the transport of dissolved substances in ground water at the same velocity as the ground water. Processes that tend to produce different results than advection, such as dispersion and adsorption, are not accounted for by the model. This model simulates flow in a quasi three-dimensional mode--flow is horizontal in aquifers and vertical in confining units. This differs from the conceptual model in that there is some vertical flow in the aquifers and some horizontal flow in the confining unit. The model assumes homogeneity of aquifer properties within a cell. Horizontal isotropy was also assumed for the aquifers and confining unit at J-Field. Heads were simulated in two aquifers, the surficial aquifer (layer 1) and the confined aquifer (layer 2) (fig. 5). The two aquifers are separated by a confining unit, which was simulated as the leakance between layers 1 and 2. No heads were calculated in the confining unit.

The model area is 3.65 mi^2 and is divided into 38 columns and 40 rows for a total of 1520 cells (fig. 6). A cell is an area where hydraulic properties are simulated to be uniform. The model calculates the head at the center of a cell. The cell sizes vary from 100 to 2,000 ft in length and from 100 to 1,300 ft in width. Each cell is no more than 1.5 times larger or smaller than any adjacent cell. Thicknesses of cells in the surficial aquifer are determined by the model by subtracting the

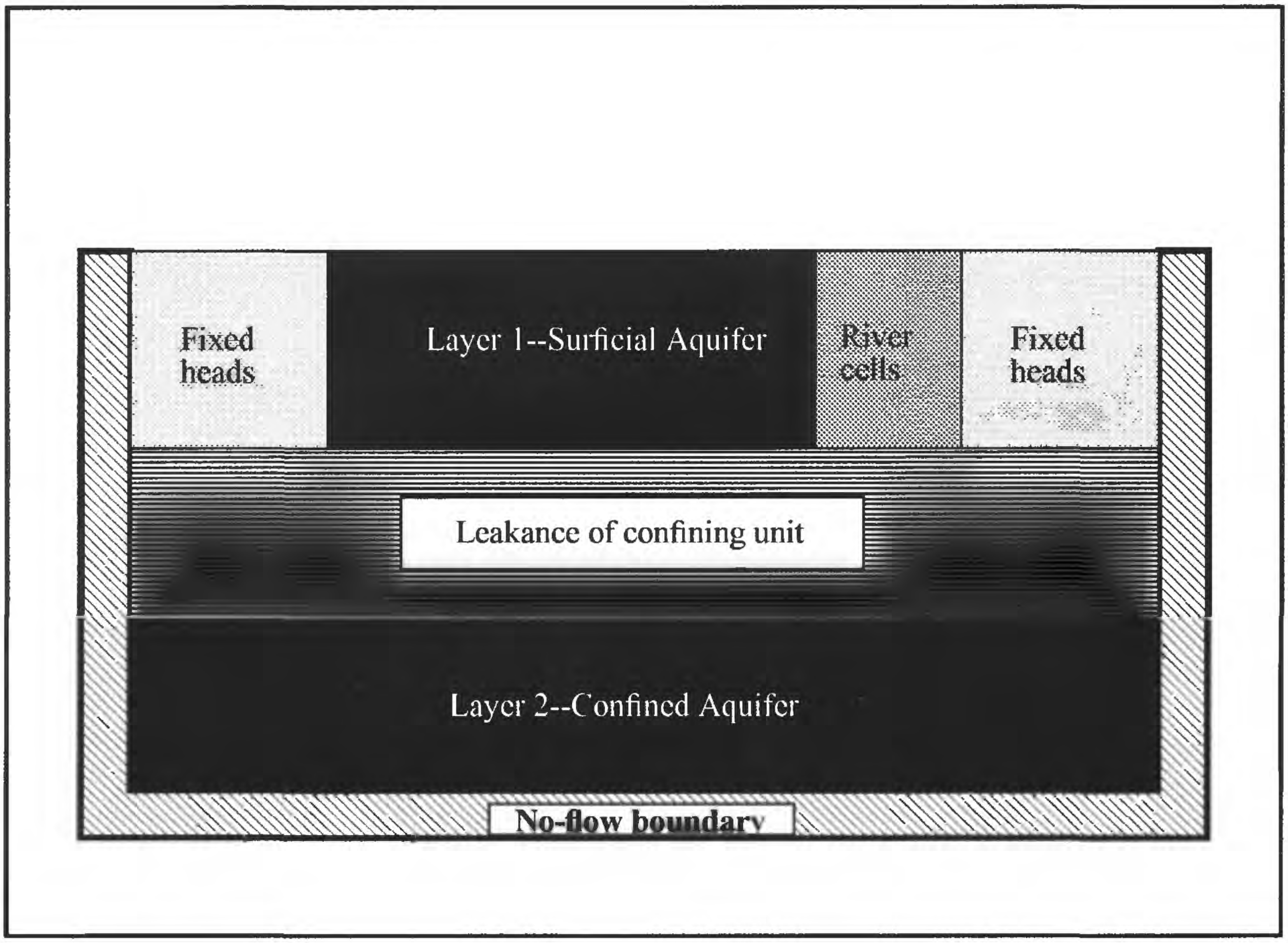


Figure 5. Model layers and boundaries.

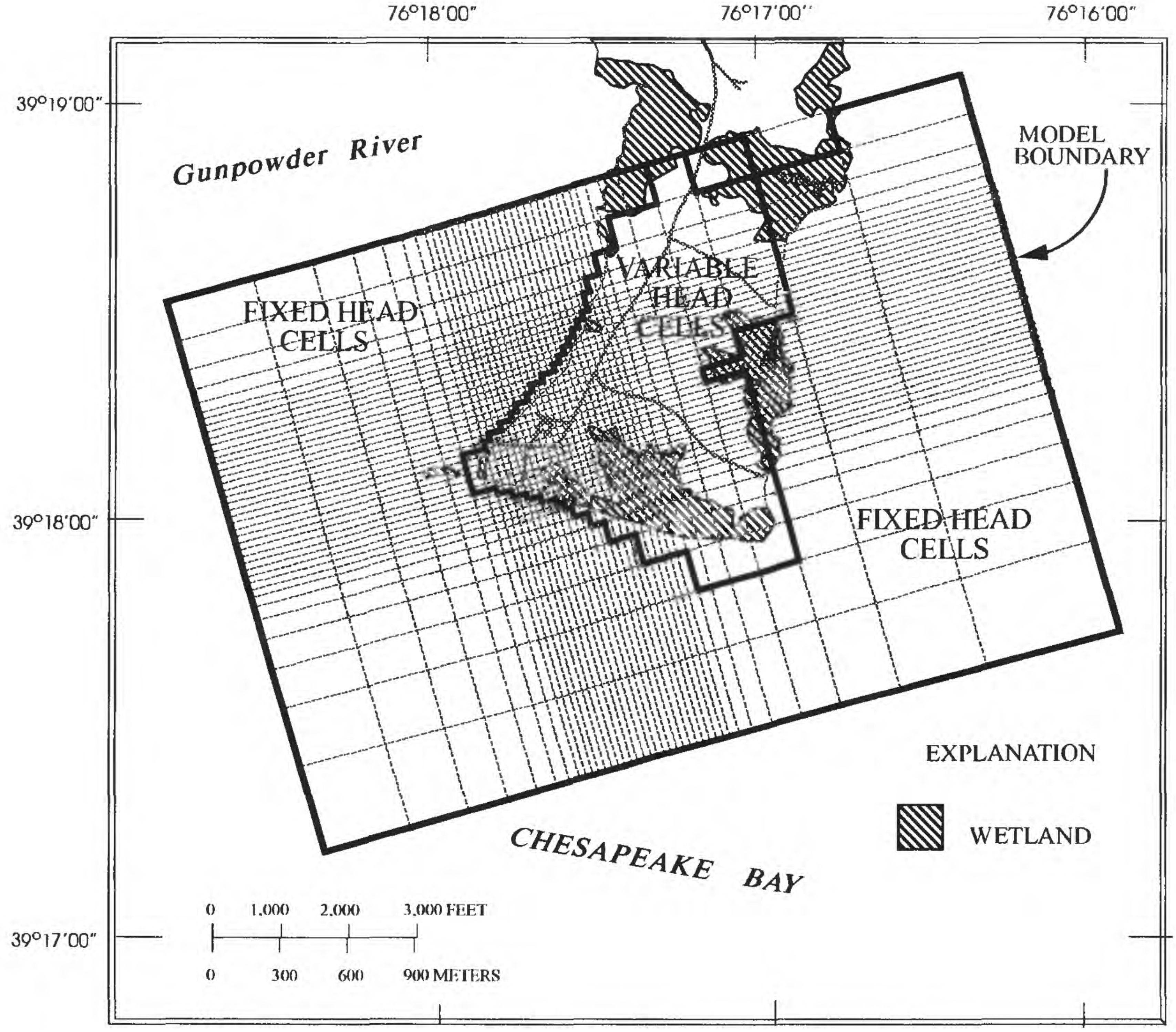


Figure 6. Finite-difference grid and lateral model boundaries.

elevation of the bottom of the surficial aquifer from the water-table elevation that was calculated during model operation. Thicknesses of the confining layer and the confined aquifer are not input directly to the model, but are a component of the leakance and transmissivity values, respectively, and are used to calculate the flow in these units.

Boundary Conditions

The model boundaries were chosen to closely approximate the hydrologic boundaries identified in the ground-water-flow system at J-Field. The upper boundary of the model is the water table, which is in layer 1 (the surficial aquifer) (fig. 5). Recharge enters the model in layer 1. The bottom of the model is a no-flow boundary and corresponds to the bottom of the confined aquifer. In the exploratory borings that were drilled early in the study, the confined aquifer was observed to overlie the clay of the Patapsco Formation (Hughes, 1993). The clay is extremely dense and actually appears dry in core samples. This information suggests that a no-flow boundary is a reasonable way to simulate the bottom of the confined aquifer.

The northern and southern boundaries of the model are no-flow boundaries and represent the margins of the Pleistocene paleochannel deposits (fig. 6). At these boundaries, the surficial aquifer, confining unit, and confined aquifer pinch out. These boundaries were mapped offshore with continuous seismic profiling (Hughes, 1991). The hydraulic characteristics of the Patapsco Formation deposits is not known at the paleochannel margins. A no-flow boundary was chosen to represent this boundary because the hydraulic conductivity of the Patapsco Formation in the Edgewood Area is generally lower than that of the shallow Pleistocene deposits, and because the shallow head gradients in offshore areas are likely to be predominantly vertical. The eastern and western boundaries also were simulated as no-flow boundaries, although no hydrologic boundaries were identified on the seismic profiles in these areas. These boundaries were placed so that they would not significantly affect the simulations in the area of interest near the center of the model.

In the surficial aquifer, cells 100 to 300 ft offshore in the Gunpowder River and Chesapeake Bay were simulated with constant heads. Tides in these estuaries generally range less than 3 ft and have an annual average of 0.9 ft above sea level (National Oceanic and Atmospheric Agency, 1993). Data collected at J-Field indicate that a tide range of 2.5 ft in the Chesapeake Bay caused tidal fluctuations of 0.6 ft in a surficial aquifer well adjacent to the western shore of J-Field, and a fluctuation of 0.1 ft in a well 1,200 ft from shore (Hughes, 1993). In the confined aquifer, tidal fluctuations of approximately 1 ft were observed throughout the study area. Because a steady-state model was used, no attempt was made to simulate these daily tidal cycles in the rivers or the aquifers. All cells in the Gunpowder River and Chesapeake Bay were set to a fixed head of 0.9 ft.

Hydraulic Parameters

The initial values of horizontal hydraulic conductivity input to the model were the median values calculated from the analysis of slug tests (Hughes, 1993). These values were then adjusted during calibration of the model. The median horizontal hydraulic conductivity for the surficial aquifer was used for the entire extent of the aquifer, because data were insufficient to define a more detailed distribution. The initial horizontal hydraulic conductivity of the surficial aquifer used in the model was 1.0 ft/d and was subsequently increased to 8.0 ft/d during calibration.

The model simulates vertical flow through a confining unit using the leakance, which is defined as the vertical hydraulic conductivity of a unit divided by its thickness. Thickness values of the confining unit in onshore areas were obtained from drilling exploratory borings and observation wells. Thickness values for offshore areas were approximated using the results of continuous seismic profiling (Hughes, 1991). The thicknesses were extrapolated from the shoreline, where the thickness was known, to the margins of the paleochannel, where the unit pinches out. Each model cell was then assigned a discrete thickness value. Leakance was then calculated by dividing an estimate of the median vertical hydraulic conductivity by the thickness assigned to each cell (fig. 7).

The initial estimate of vertical hydraulic conductivity used to calculate the leakance of the confining unit was 0.05 ft/d. This value was adjusted to 0.005 ft/d during calibration. This is considered a reasonable value based on the following reasoning: The median horizontal hydraulic conductivity

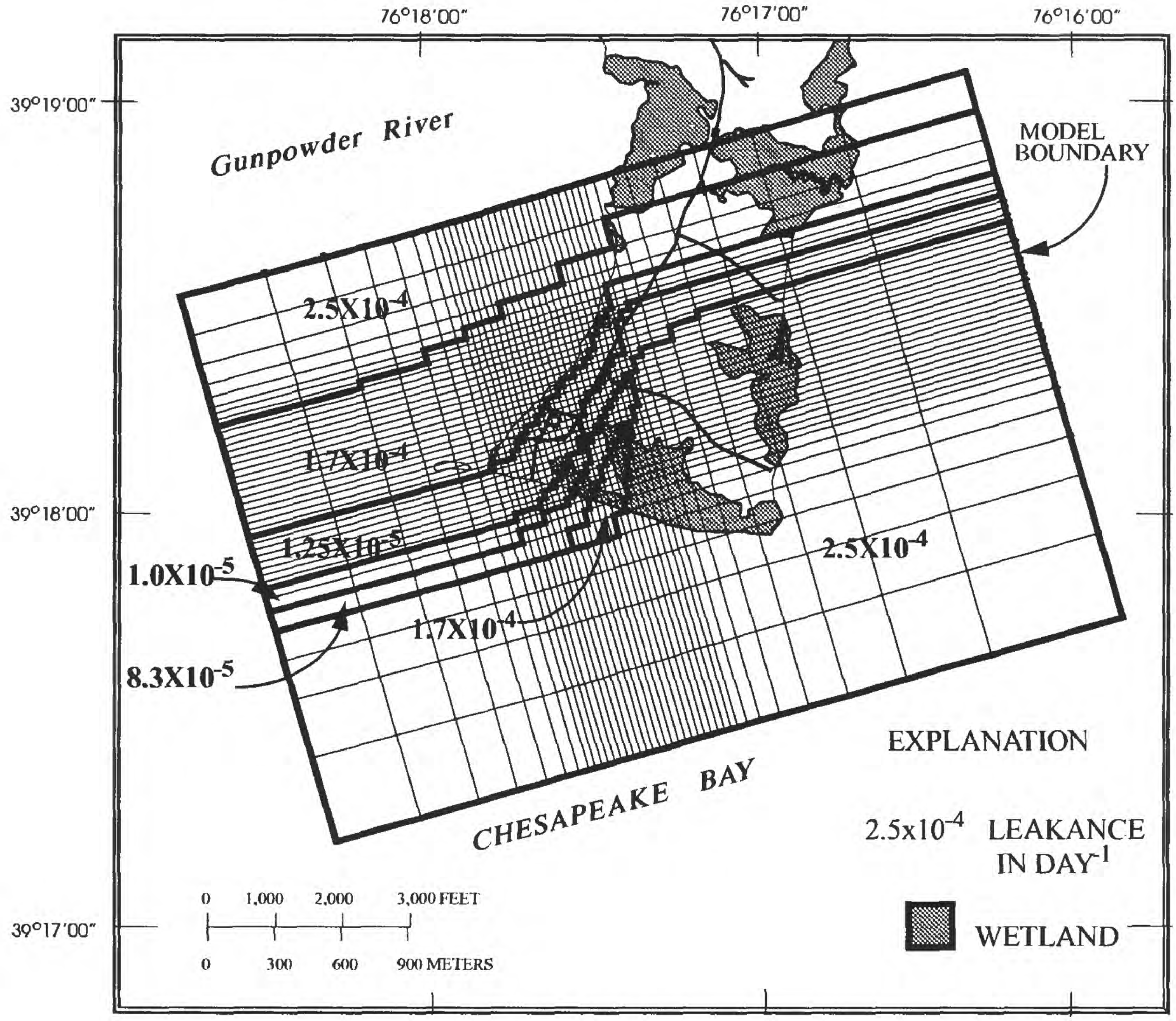


Figure 7. Distribution of leakance in the confining unit.

of the confining unit as measured by slug tests was 0.05 ft/d (Hughes, 1993). The model, however, requires the use of vertical hydraulic conductivity values to calculate the leakance values for confining units. From numerous measurements, the ratio of horizontal hydraulic conductivity to vertical hydraulic conductivity can range from 2:1 to 10:1 (Freeze and Cherry, 1979; Todd, 1980) and can range as high as 100:1 where clay is present (Todd, 1980). For this study, the ratio of median horizontal hydraulic conductivity to vertical hydraulic conductivity determined during model calibration is 10:1, which is within the range described by Freeze and Cherry (1979) and Todd (1980).

The measured horizontal hydraulic conductivity of the confined aquifer ranged from 3.16 to 932 ft/d. From the distribution of the slug-test data, the aquifer was originally divided into two areas with different horizontal hydraulic conductivities. The median horizontal hydraulic conductivity was 390 ft/d in the western half of the confined aquifer and 10.5 ft/d in the eastern half. During calibration, it was determined that the transmissivity of the confined aquifer was probably more uniform than the field data suggested. A critical factor in the measurement of horizontal hydraulic conductivity by slug tests is the efficiency of the well, which is primarily a function of the size of the screen opening. The openings in the USGS well screens in the confined aquifer are either 0.01 or 0.001 in. All of the wells where low horizontal hydraulic conductivites were measured are the wells with the smaller of the two screen openings. Although the 0.001-in. screens were used in sediments that were finer grained than sediments where the 0.01-in. screens were used, the smaller screen openings could account for a part of the difference between the measured horizonal hydraulic conductivity and the simulated horizontal hydraulic conductivity in the calibrated model. For the final calibration, the median value of horizontal hydraulic conductivity in the western part of the study area was multiplied by the thickness distribution to obtain the transmissivity distribution (fig. 8). The thickness distribution of the confined aquifer was determined in a manner similar to that described for the confining unit.

Recharge Calculations

The unconfined ground-water system at J-Field is recharged by precipitation and, accordingly, recharge rates depend on seasonal variations in precipitation and evapotranspiration. The total annual precipitation in 1990 was 46.81 in. and in 1991 was 40.68 in. (Wayne Kaiser, U.S. Army Test and Evaluation Command, written commun., 1992). These measurements were made approximately 1.5 mi north of J-Field. Estimates of recharge used in other models for the Coastal Plain of Maryland range from 31 to 52 percent of precipitation (Rasmussen and Andreasen, 1959; Vroblesky and others, 1989; Harsh and Laczniak, 1990; Achmad, 1991). Using these percentages, the annual recharge calculated from the H-Field precipitation data could range from 13.5 to 22.7 in.

A method described by Rasmussen and Andreasen (1959) was used to calculate the recharge for October 1990 to October 1991 to refine the estimate of recharge. This technique uses ground-water hydrographs, precipitation data, and a value of specific yield to estimate the amount of water that recharges the water-table aquifer. The specific yield of an aquifer is the ratio of the volume of water that drains from a saturated rock under the influence of gravity to the total volume of the rock (Fetter, 1980). Achmad (1991) compiled specific yields calculated for water-table aquifers in the Coastal Plain of Maryland and Delaware. These values range from 0.10 to 0.15 (Rasmussen and Andreasen, 1959; Johnston, 1976; Mack and Achmad; 1986; and Chapelle, 1985).

The annual recharge is estimated by recording the rise in water level resulting from individual storms on a hydrograph. If the rises are measured from an existing water level, the amount of recharge would be underestimated by the quantity of ground water that discharged to streams and rivers during the storms. The amount of ground-water discharge, therefore, is estimated by projecting the recession of the hydrograph to the date of the peak stage, after which, the distance from the peak stage to the antecedent hydrograph is measured. These measurements are converted to

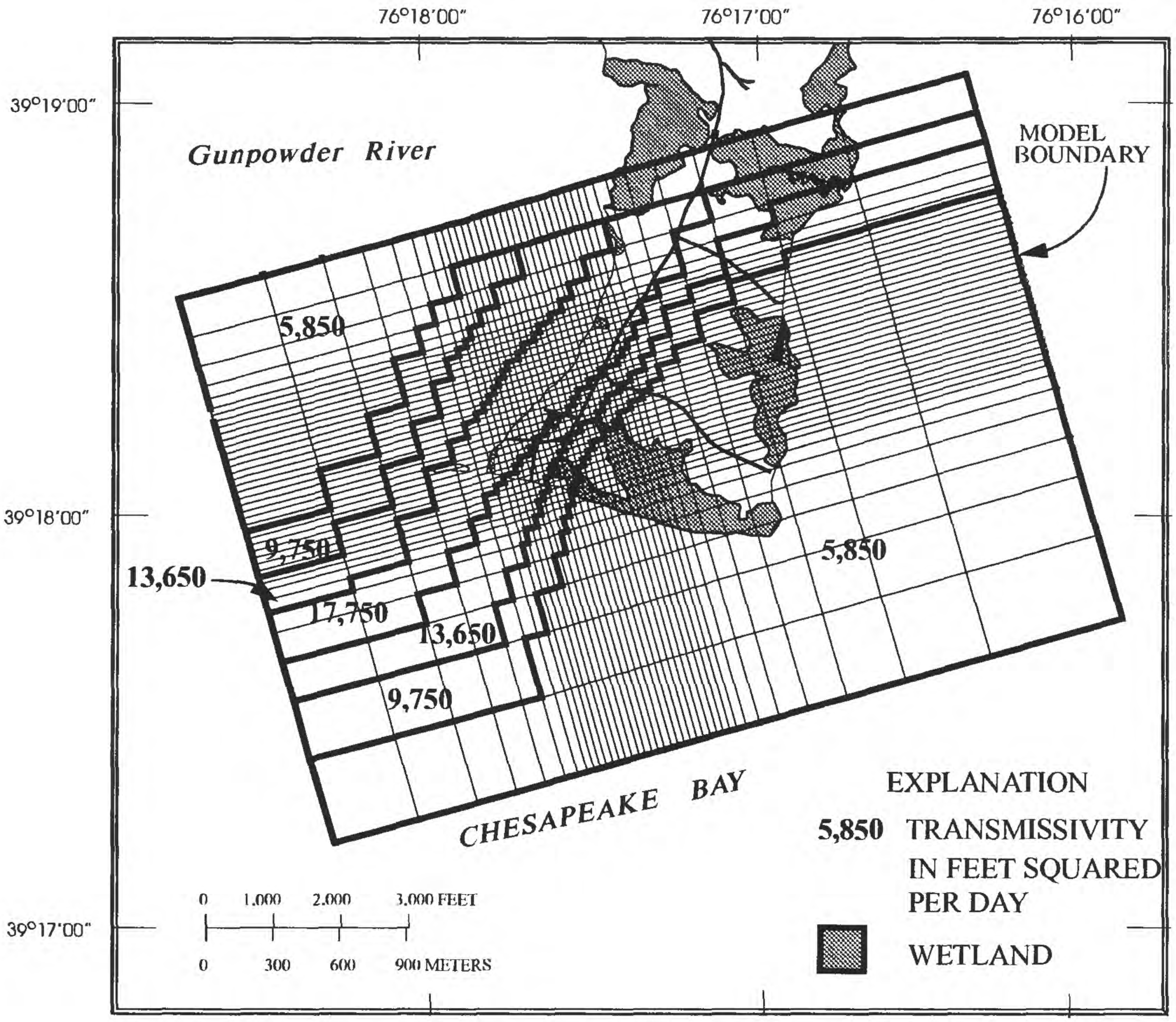


Figure 8. Distribution of transmissivity in the confined aquifer.

inches, multiplied by the specific yield, and summed to obtain the annual recharge. Using this technique, annual recharge for October 1990 to October 1991 is 17.5 in., or 37 percent of the annual precipitation. This value was used to calibrate the model. Data for this period were used because complete hydrograph data were available and because average annual rainfall and temperature conditions during the period were similar to those during the period of record (1903-92).

Simulation of Wetlands

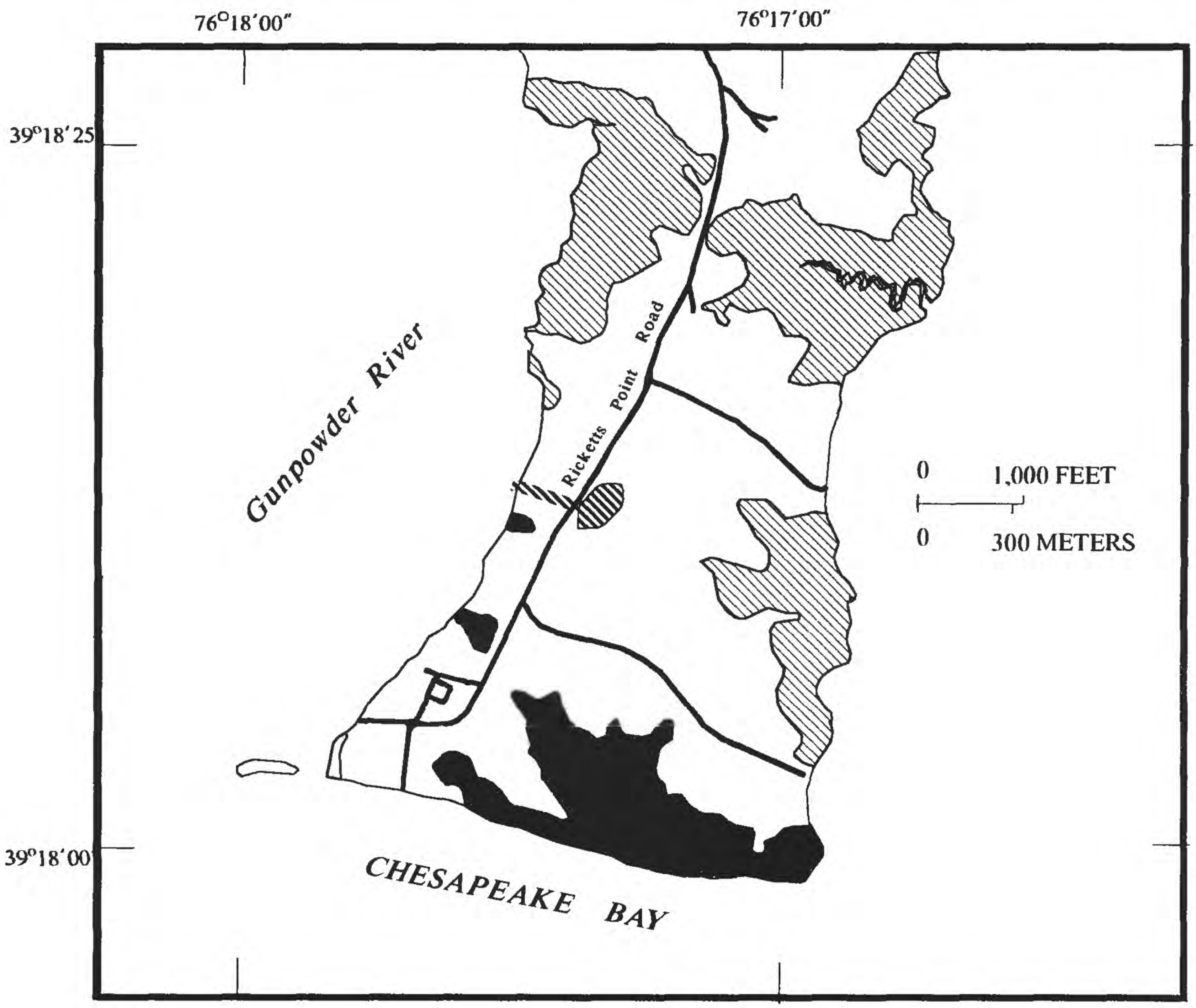
The wetlands at J-Field can be divided into two different types--tidal and nontidal (fig. 9). The tidal wetlands were simulated as fixed-head cells that are essentially identical to the cells that represent the Gunpowder River and Chesapeake Bay. The head in the tidal wetlands was set to the average tide level of 0.9 ft above sea level. The nontidal wetlands were simulated with the river subroutines in the USGS ground-water-flow model with the exception of a small area to the north of the white-phosphorus disposal area, which was simulated with the drain subroutines. The stage in the nontidal wetlands was set to 0.9 ft above sea level to represent an average annual water level. No water-level data for the nontidal wetlands have been recorded. On the basis of field observations, the water level in the wetlands ranges from several feet above the stage in the adjacent estuary in the spring, to almost dry conditions in late summer and fall. As an approximation, the average tide value in the estuaries was used to represent the average wetland water level in the model.

The river subroutine in the model requires that data for a fixed head and a river-bottom elevation be input. As long as the head in the river cell remains above the river bottom, the cell functions the same as any other fixed-head cell does, such as those that represent the Gunpowder River and Chesapeake Bay. Under these conditions, the nontidal wetlands act as discharge areas for ground water. During pumpage, however, the head in the wetland cells can drop below the bottom of the wetlands. When this happens, the flow of water from the wetlands is restricted and is governed by the conductance of the wetland-bottom sediments (McDonald and Harbaugh, 1984). The wetland-bottom sediments are composed of fine-grained silt and clay and contain abundant organic matter. Although no measurements were made, because of the fine-grained texture of the material, the conductance was presumed to be low and the vertical hydraulic conductivity used to calculate the conductance was initially set equal to 0.05 ft/d. Since no data were available, a uniform wetland-bottom sediment thickness of 2.5 ft was assumed. Using this thickness, a 100 x 100-ft cell has a conductance of 200 ft^2/d. The conductance distribution was adjusted during model calibration, but the initial values were found to work best in the calibrated model.

Sometime before 1970, a drainage ditch was dug completely across the peninsula to the north of the white-phosphorus disposal area. Although no historical records exist, the ditch was probably dug in a largely unsuccessful attempt to drain the wetlands in that area. During periods of high ground-water levels, the ditch fills with water and slowly drains into the Gunpowder River. This area was simulated with the drain subroutine in the model (fig. 9). The drain subroutine works similar to the river subroutine; however, a fixed head is not specified for the drain subroutine. This area was simulated differently from the nontidal wetlands, because those wetlands retain high water levels and have no surface-drainage pathway, whereas the drainage ditch only contains water during the wettest time of year and has a direct surface pathway to the Gunpowder River. Fixed heads enable nontidal wetlands to be a source of water for wells during pumpage. The drainage ditch is not a source of water for pumpage. The vertical hydraulic conductivity value used to calculate the drain-bottom conductance was initially set to 0.005 ft/d but was adjusted during calibration to 0.0032 ft/d. Since no data were available, a uniform drain-bottom sediment thickness of 1 ft was assumed. Using this thickness, the conductance value is 32 ft^2/d for a 100 x 100-ft cell.

Calibration

The steady-state ground-water-flow model was calibrated by comparing simulated heads to heads measured on May 20, 1992 (fig. 10). After final calibration, the absolute differences between measured and simulated heads in the surficial aquifer ranged from -1.03 to 1.28 ft (-52 to +32 percent difference) and in the confined aquifer ranged from -0.15 to 0.52 ft (-9 to +54 percent difference). Because the steady-state model was developed using data for average annual recharge, tide levels,



EXPLANATION

NONTIDAL WETLAND SIMULATED WITH DRAIN SUBROUTINE

NONTIDAL WETLAND SIMULATED WITH RIVER SUBROUTINE

TIDAL WETLAND SIMULATED WITH CONSTANT HEADS

Figure 9. Simulated wetlands.

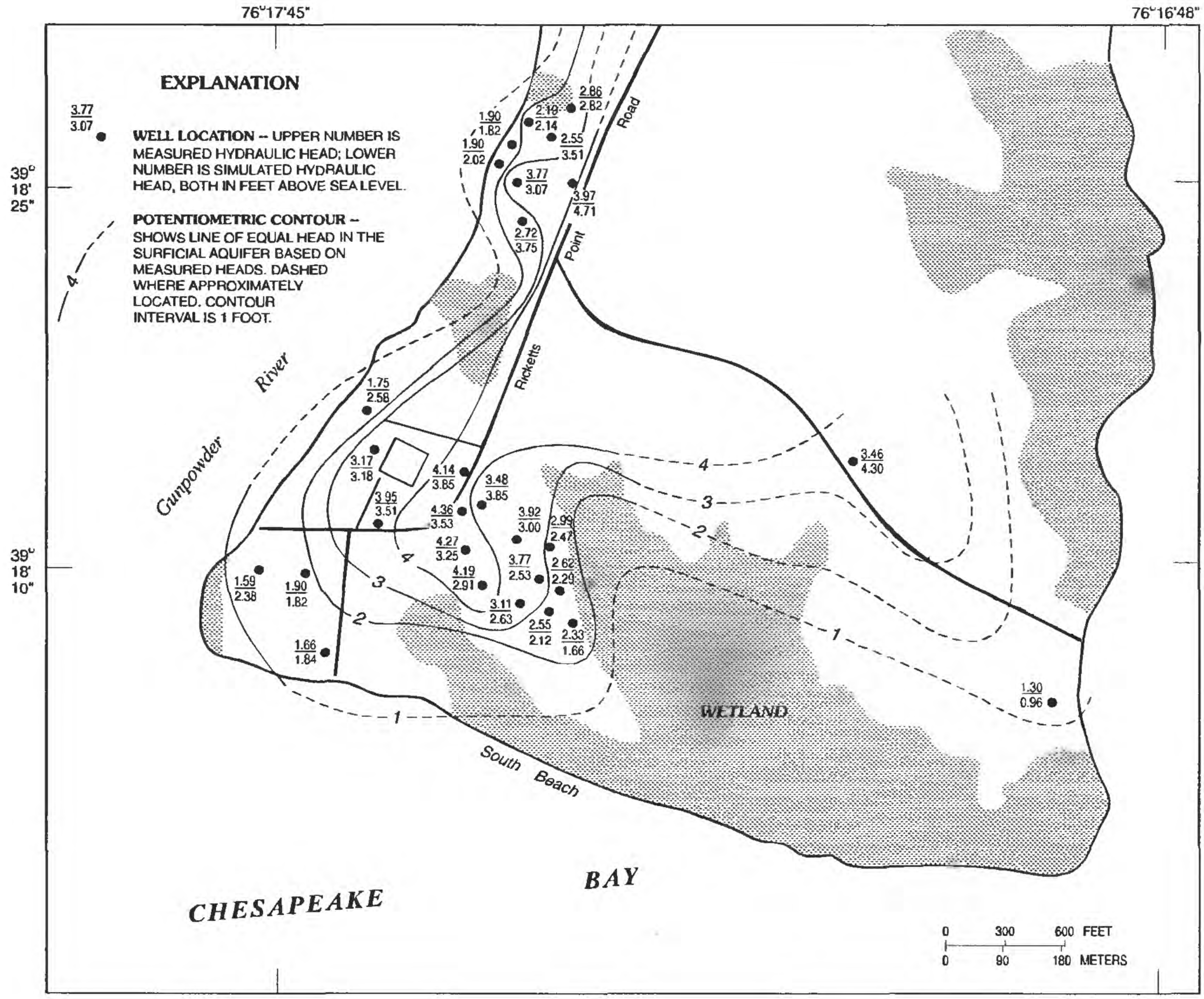


Figure 10. Water-level contours, heads measured on May 20, 1992, and heads calculated by the steady-state model for the surficial aquifer at J-Field.

and wetland water levels, a synoptic measurement for purposes of comparison was chosen that approximated average annual water levels for the study area. Water levels measured May 1992 were chosen, because when compared to hydrographs of ground-water levels measured at J-Field, the May 1992 water levels were found to lie near the mid-point of the annual water-level fluctuations.

The horizontal hydraulic conductivity of the surficial aquifer, the leakance of the confining layer, and the transmissivity of the confined aquifer were adjusted to obtain a reasonable agreement between the simulated and measured water levels. The model was not sensitive to changes in the transmissivity of the confined aquifer, so the major changes for calibration were in the horizontal hydraulic conductivity of the surficial aquifer and the leakance of the confining unit. Recharge was held constant for the calibration procedure. The horizontal hydraulic conductivity of the surficial aquifer and the vertical hydraulic conductivity used to calculate the leakance of the confining unit were systematically varied, and the squares of the differences between heads measured in 40 wells on May 20, 1992, and simulated heads were summed to obtain a single numerical value that represents the error for an entire simulation. Large errors indicated large differences between measured and simulated heads; small errors indicated small differences. These values were plotted and contoured to illustrate the range of horizontal hydraulic conductivity of the surficial aquifer and vertical hydraulic conductivity of the confining unit that yielded the smallest differences between simulated and measured heads (fig. 11).

A range of values of horizontal hydraulic conductivity of the surficial aquifer and vertical hydraulic conductivity of the confining unit yield similar values for error in the model and appears on figure 11 as a band of values that are less than 15. This figure illustrates how simulated heads respond to changes in calibration values. For example, as the hydraulic conductivity of both the surficial aquifer and confining unit are increased to values higher than the calibration values (upper right in figure 11), the error increases only gradually. With higher conductivities, water would move more easily through both the surficial aquifer and confining unit. The fixed heads at the boundary of the surficial aquifer would then become the dominant control on the simulated heads, placing a limit on how low the heads could drop. Error values would asymtopically approach the error value that would result if all the simulated heads were equal to the value of the fixed heads.

In contrast, the error increases abruptly when the conductivities of the surficial aquifer and confining unit are decreased below the calibration values (lower left in figure 11). Lower conductivities would restrict water movement through both the surficial aquifer and the confining unit, causing the simulated heads in the surficial aquifer to deviate from measured heads, with no upper limit on the consequently large error values. As the horizontal hydraulic conductivity of the surficial aquifer decreases and vertical hydraulic conductivity of the confining unit increases (lower right in figure 11), the hydraulic properties of the surficial aquifer and the confining unit become increasingly similar. Correspondence between measured and simulated heads would result, but these conditions do not fit the field data and observations.

A horizontal hydraulic conductivity of the surficial aquifer of 8 ft/d and a vertical hydraulic conductivity of the confining unit of 0.005 ft/d were chosen for use in the calibrated model. These values were chosen from the wide range of possible values for several reasons. Vertical hydraulic conductivity of the confining unit was chosen as the critical factor for calibration because it controls the movement of water to the confined aquifer. Simulations showed that for values greater than 0.02 ft/d, the majority of ground-water movement was vertical, with little horizontal flow and consequently little ground-water discharge to the wetlands. This is in contrast to the chemical data that were collected at J-Field, which indicates that only trace amounts of contaminants were detected in the confined aquifer, whereas high concentrations of contaminants were detected in the surficial aquifer and in some wetlands (Martha Cashel, U.S. Geological Survey, oral commun., 1994). A vertical hydraulic conductivity of 0.005 ft/d was chosen because it allowed some water to move through the confining unit, although most of the water in the surficial aquifer discharged to the wetlands or estuaries. For values of vertical hydraulic conductivity less than 0.004 ft/d, the model failed to converge to a solution. After determining a vertical hydraulic conductivity value for the confining unit, the horizontal hydraulic conductivity of the surficial aquifer that produced the smallest error was chosen.

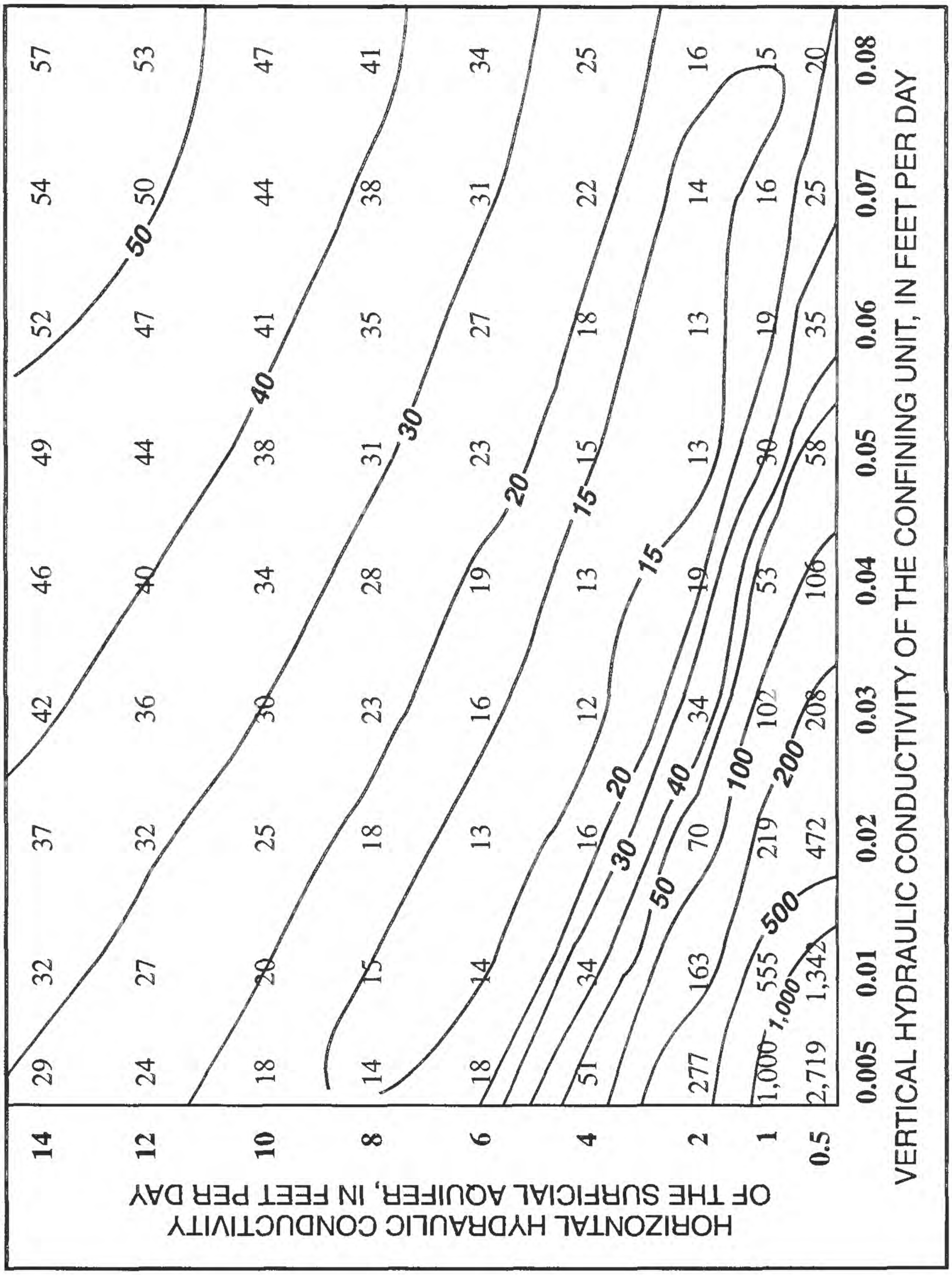


Figure 11. Error between measured and simulated water levels with respect to the horizontal hydraulic conductivity of the surficial aquifer and the vertical hydraulic conductivity of the confining unit.

Sensitivity Analysis

The model was tested to determine its sensitivity to changes in recharge, wetland-bottom conductance, transmissivity of the confined aquifer, and model boundaries. Sensitivity to changes in recharge was tested by running simulations with recharge values that varied between -50 to +100 percent of the calibration values (fig. 12). Median heads in the surficial and confined aquifers varied by -34 to +59 percent and -20 to +33 percent, respectively, in response to the variations in recharge. Head changes in the confined aquifer that resulted from changes in recharge were similar to those in the surficial aquifer, but less pronounced. The relation between recharge and median head change appears to be linear in the range that was examined (fig. 12). Based on this analysis, if estimates of recharge differ by no more than 20 percent, then heads can be expected to differ by 10 to 15 percent in the surficial and confined aquifer.

Sensitivity to changes in wetland-bottom conductance was tested by running simulations with conductance values that varied between -90 to +300 percent of the calibration values (fig. 13). The heads in the confined aquifer showed no or little change for the full range of conductance values tested. Median heads in the surficial aquifer ranged less than -10 to +10 percent within most of the range of conductance values tested, but at -90 and +300 percent of the calibration conductance, heads changed +23 and -13 percent, respectively. Even at two and three orders of magnitude times the wetland-bottom conductance (not shown in fig. 13), the heads in the surficial aquifer did not change more than -30 percent. Resistance to flow through the wetland bottom is insignificant at high conductance values, and the wetland cells respond similarly to fixed-head cells, allowing for little change in the simulated heads.

Sensitivity to changes in the transmissivity of the confined aquifer was tested by running simulations with transmissivity values that varied between -90 to +200 percent of the calibration values (fig. 14). Median heads in the surficial aquifer were virtually unaffected by changes in the transmissivity of the confined aquifer. The median head in the confined aquifer decreased less than 5 percent in response to a 200-percent increase in transmissivity and the model failed to converge to a solution when the calibrated transmissivity was increased 300 percent. Median head change in the confined aquifer increased 28 percent in response to a 90-percent decrease in transmissivity, but the increase was less than 10 percent for decreases in transmissivity less than 50 percent.

The boundaries of the ground-water-flow model were tested in two ways. First, the model boundaries were moved inward to make a much smaller grid. Second, a fixed-head boundary was substituted for the horizontal no-flow boundary of the confined aquifer. The smaller grid was established by truncating three cells on all four sides of the model. The new grid had 34 rows and 32 columns, but because the largest cells were truncated, the length and width of the grid was approximately halved and the area was reduced to approximately one-quarter of the original model area. Surficial aquifer heads were not sensitive to this change in the boundary conditions. Heads increased in the surficial aquifer from 0.04 to 0.21 ft (1 to 7 percent). Heads in the confined aquifer were more sensitive and increased from 0.34 to 0.40 ft (23 to 27 percent). The surficial aquifer heads were not sensitive to the placement of fixed-head cells at the boundary of the confined aquifer. Heads decreased for this test in the surficial aquifer and ranged from -0.01 to -0.11 ft (<-1 to -4 percent). Heads in the confined aquifer were also more sensitive to this change in boundary conditions and decreased by -0.46 to -0.55 ft (-31 to -37 percent). These tests show the boundary conditions have the largest impact on simulated heads in the confining unit. The focus of remediation efforts is on the surficial aquifer, however, which allows more error in the simulation of the confined aquifer heads.

Simulated Ground-Water Flow

The conceptual model of ground-water flow at J-Field (Hughes, 1993) was used to design the numerical ground-water-flow model. Simulated heads closely matched the measured heads, indicating that simulated ground-water flowpaths approximated the real flowpaths. Because the simulations were run under steady-state conditions, all of the water that entered the model came in the form of recharge and left as discharge. Simulated ground water flowed from the highest land-surface elevations toward topographically low discharge areas in the wetlands and estuaries surrounding J-Field. Approximately 74 percent of the simulated

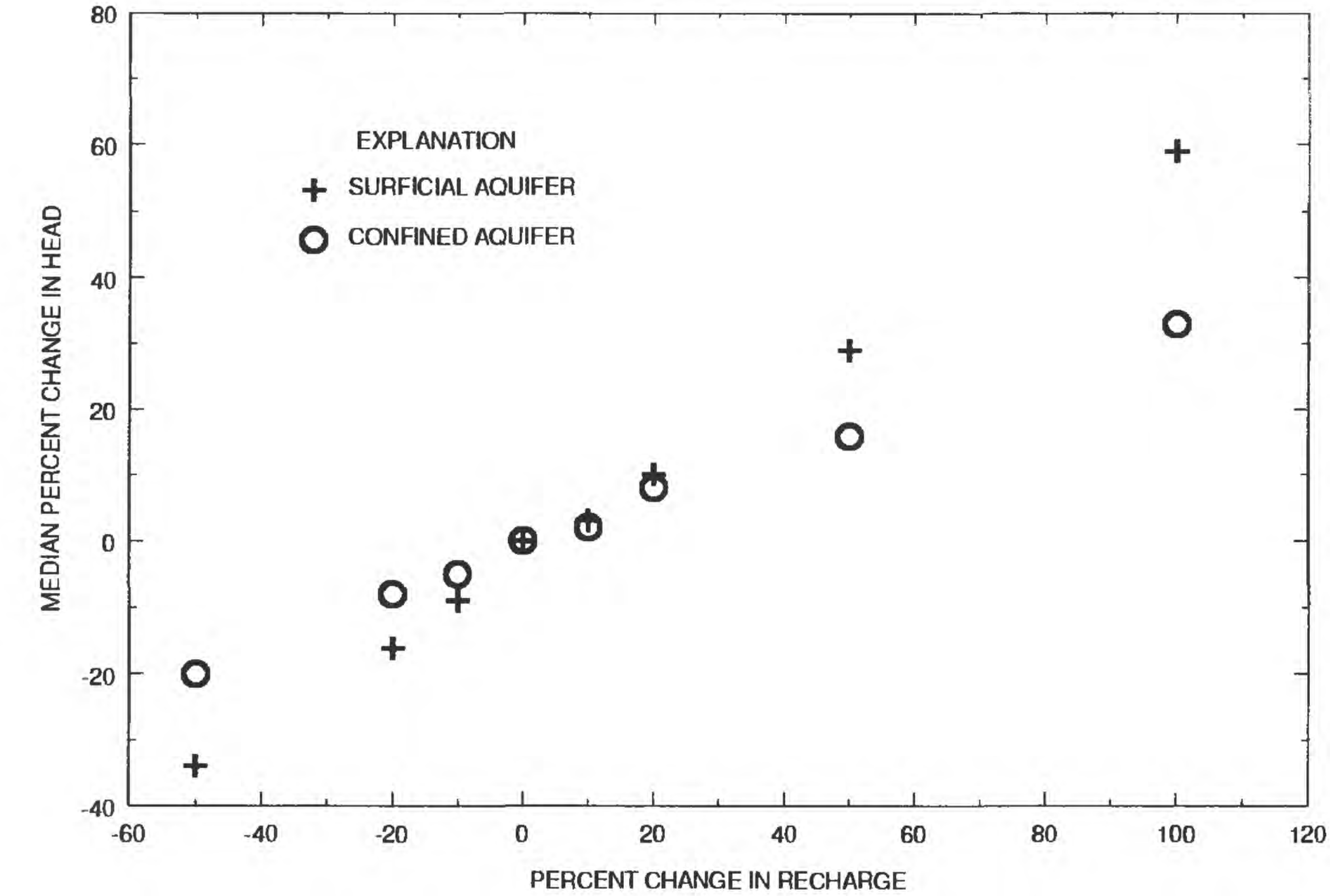


Figure 12. Median percentage of change in head in the surficial and confined aquifers with respect to percent change in recharge.

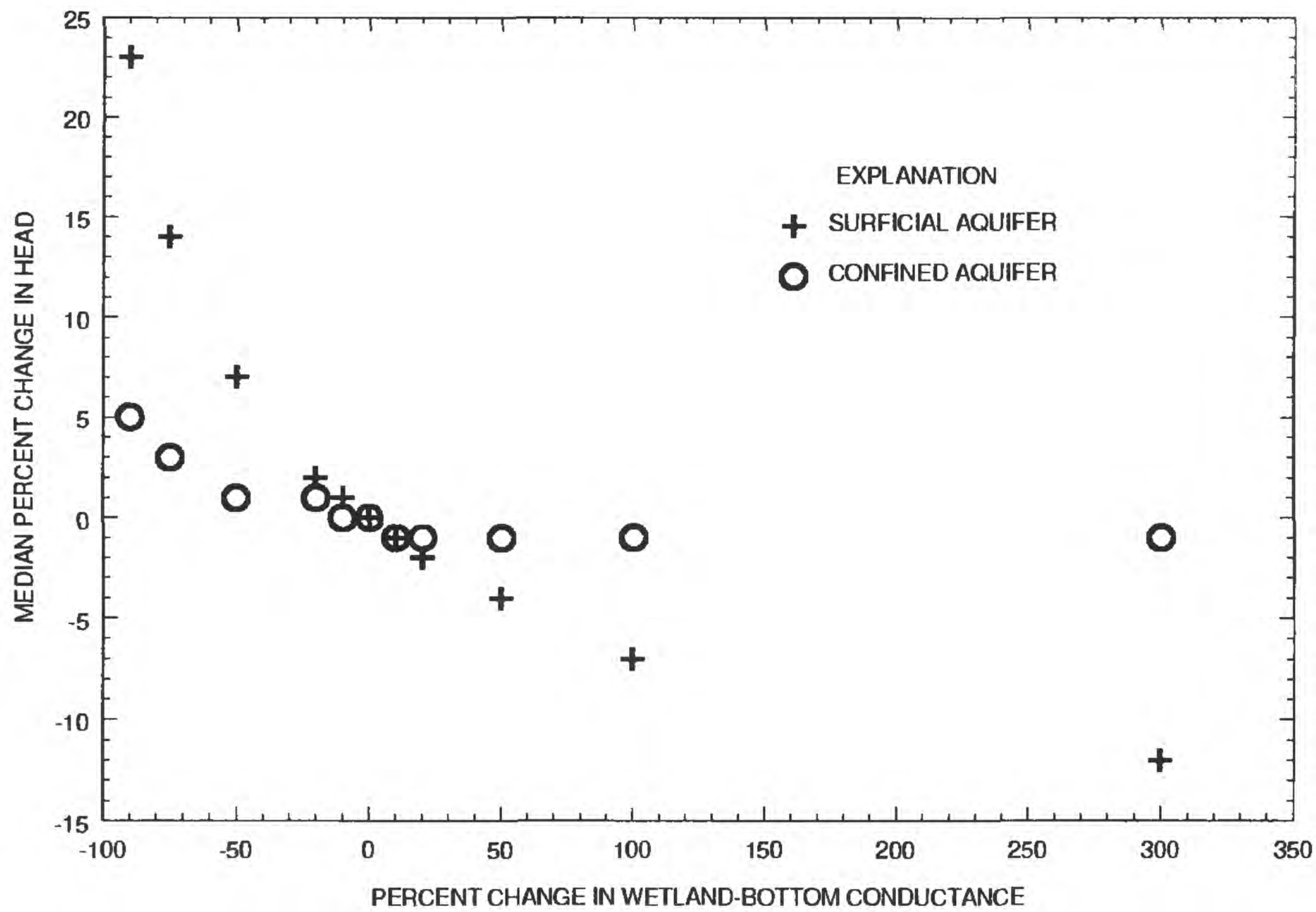


Figure 13. Median percentage of change in head in surficial and confined aquifers with respect to percent change in wetland-bottom conductance.

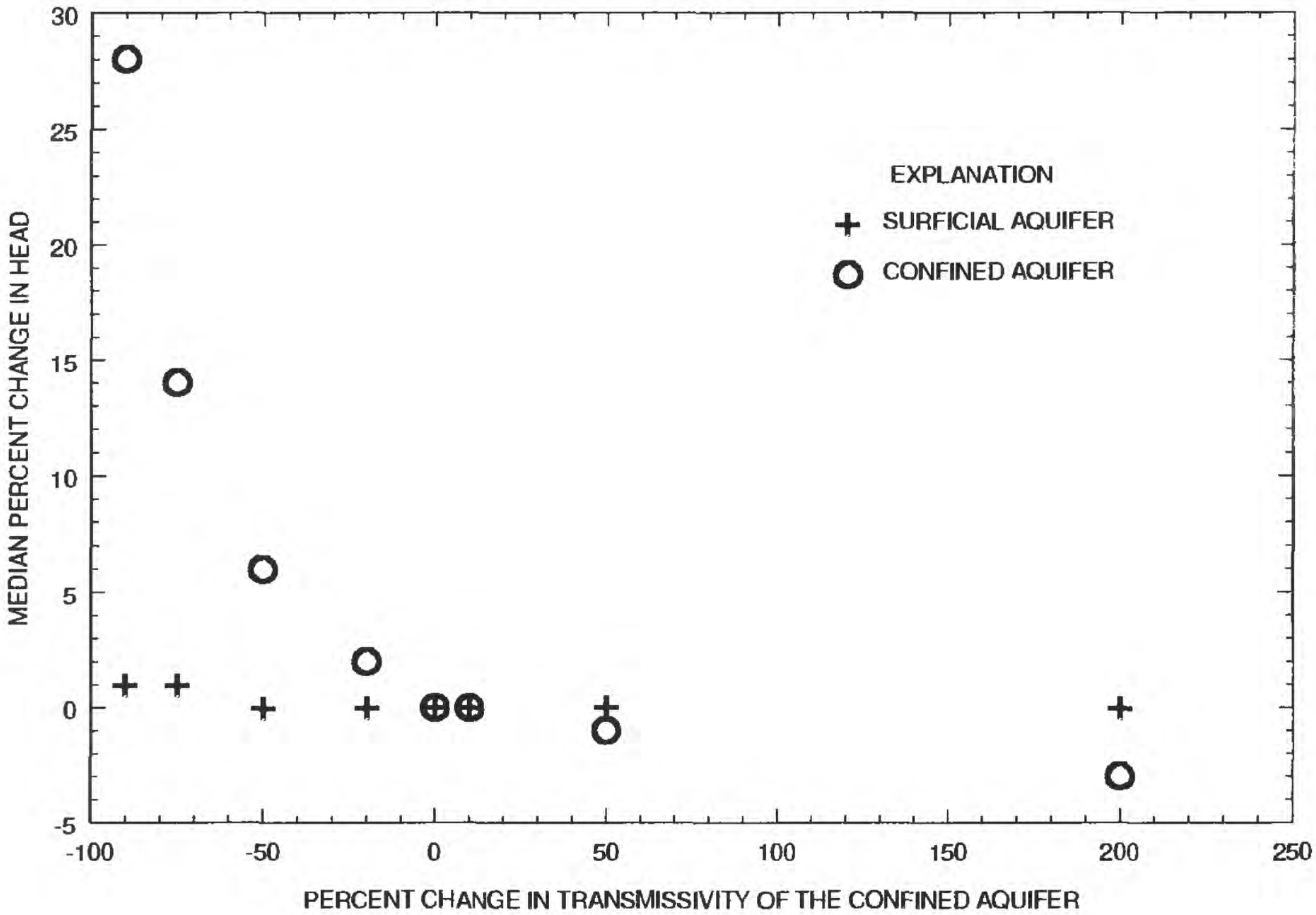


Figure 14. Median percentage of change in head in the surficial and confined aquifers with respect to percent change in transmissivity.

recharge eventually discharged to the estuaries near J-Field, most of it within 100 ft of the shore. Approximately 22 percent of the simulated flow discharges to the nontidal wetlands and the remaining 3 percent discharged to the drains north of the white-phosphorus disposal area.

Ground-water flow was simulated in detail for the white-phosphorus disposal area, riot-control-agent disposal area, and the toxic-materials disposal area with the particle-tracker subroutine for the ground-water-flow model (Pollock, 1989). The particle-tracker subroutine allowed the paths of individual particles of water to be tracked as they move through a ground-water-flow system. Particle tracking can be used as a simple means to evaluate the advective transport characteristics of ground-water systems, but cannot be used to calculate solute concentrations because it does not account for dispersion or adsorption. Travel time for the water particles can also be calculated with particle tracking. The particle-tracking algorithm assumes a uniform distribution of porosity and linear velocity components within a model cell. These conditions vary from field conditions, but the subroutine does provide approximations of flowpaths and travel times. The particle-tracker subroutine is limited in its predictive capabilities by its ability to deal with transient flow, discretization affects, and uncertainties in data and boundary conditions (Pollock, 1989). Discretization effects are most pronounced at weak sinks because it is impossible to determine whether particles should discharge or flow through cells containing weak sinks. For the simulations in this report, particles were assumed to discharge when they encountered weak sinks, such as cells located in wetlands.

The particle-tracker subroutine does not track the paths of chemical contaminants in ground water. These chemicals are subject to a variety of physical processes such as dispersion and adsorption, as well as chemical reactions that can alter their composition and transport characteristics. In general, these processes tend to slow down the movement of contaminants relative to ground water, and as a result, calculated travel times for ground water are likely to be shorter than the actual travel times of chemical contaminants. The particle-tracker subroutine can provide an indication of the general direction and relative rates of advective chemical transport within a ground-water-flow system and can be useful when selecting potential remediation techniques for a specific site.

The following simulations of flow at the solid-waste-management units were conducted by placing eight particles of water in each cell that is located beneath a disposal pit and tracking the particles forward through time to the point where they discharge. The elevations of the top and bottom of units that were input to the particle-tracker subroutine were the same as those used to calculate thicknesses for leakance and transmissivity for the ground-water-flow model and were derived from borehole data (Hughes, 1993). The porosities of the hydrologic units at J-Field were not measured; therefore, values input to the particle-tracker subroutine were based on average values of porosity for similar materials reported in Freeze and Cherry (1979). The values used were 30 percent for the surficial and confined aquifers and 40 percent for the confining unit.

Ground-Water Flow at the White-Phosphorus Disposal Area

Ground-water flow at the white-phosphorus disposal area is from the higher land surface near Ricketts Point Road to the west toward the Gunpowder River (fig. 15). The wetlands to the north and south of the disposal pits cause some deflection of flow toward these areas but do not capture water that flows through the pit area. Water that was recharged in the disposal pits and in the immediate vicinity of the pits flows directly toward the Gunpowder River, where it discharges. The model estimates travel times of 2 to 5 years for ground water from the white-phosphorus disposal pits to the discharge areas in the Gunpowder River. These travel times are faster than those for the other disposal sites, because the head gradients are considerably higher at the white-phosphorus disposal area than those at the toxic-materials and the riot-control-agent disposal areas.

Low levels of contaminants were present in the surficial aquifer at the white-phosphorus disposal area as compared to the toxic-materials and riot-control-agent disposal areas. These low levels are possibly a function of the time since disposal ended at the site and the rate of ground-water movement. Most of the disposal activities at the white-phosphorus disposal area ended in the mid-1970's, which allowed ground water for at least 18 years to move toward the Gunpowder River. Because 18 years is approximately three to nine times the travel time from the disposal pits to the river, the aquifer has essentially been flushed three to nine times. This large amount of flushing probably resulted in most contaminants moving offshore and discharging into the Gunpowder River. Much of the contaminated ground water that resulted from disposal during the 1940's and 1950's could have already discharged offshore, depending on the degree of dispersion and adsorption of contaminants. This is supported by the low levels of contamination present in the surficial aquifer at the white-phosphorus disposal area, as compared to the toxic-materials and riot-control-agent disposal areas. Under present conditions, the remaining contaminants in the aquifer will continue to discharge to the Gunpowder River.

Ground-Water Flow at the Riot-Control-Agent Disposal Area

Ground water flows from the riot-control-agent disposal area toward the Gunpowder River and Chesapeake Bay (fig. 15). Ground water from the northern part of the pit area flows toward the Gunpowder River and from the southern part flows toward the Chesapeake Bay. Concentrations of benzene were detected in samples from a well adjacent to the disposal pit and also in wells downgradient to the west. Water in wells lying along flowpaths to the south and southwest was uncontaminated indicating that there is no source of contamination in the southern part of the disposal pit. This is supported by the lack of munition fragments and debris in this area, which is associated with the other burning and disposal areas at J-Field.

Travel times for ground water at the riot-control-agent disposal area range from 6 to 14 years. Because most organic contaminants are nonconservative, they will travel at a slower rate than the ground water. Much of the disposal work at the riot-control-agent disposal area was conducted in the 1970's. Given that approximately 20 years has elapsed since disposal, and ground water takes

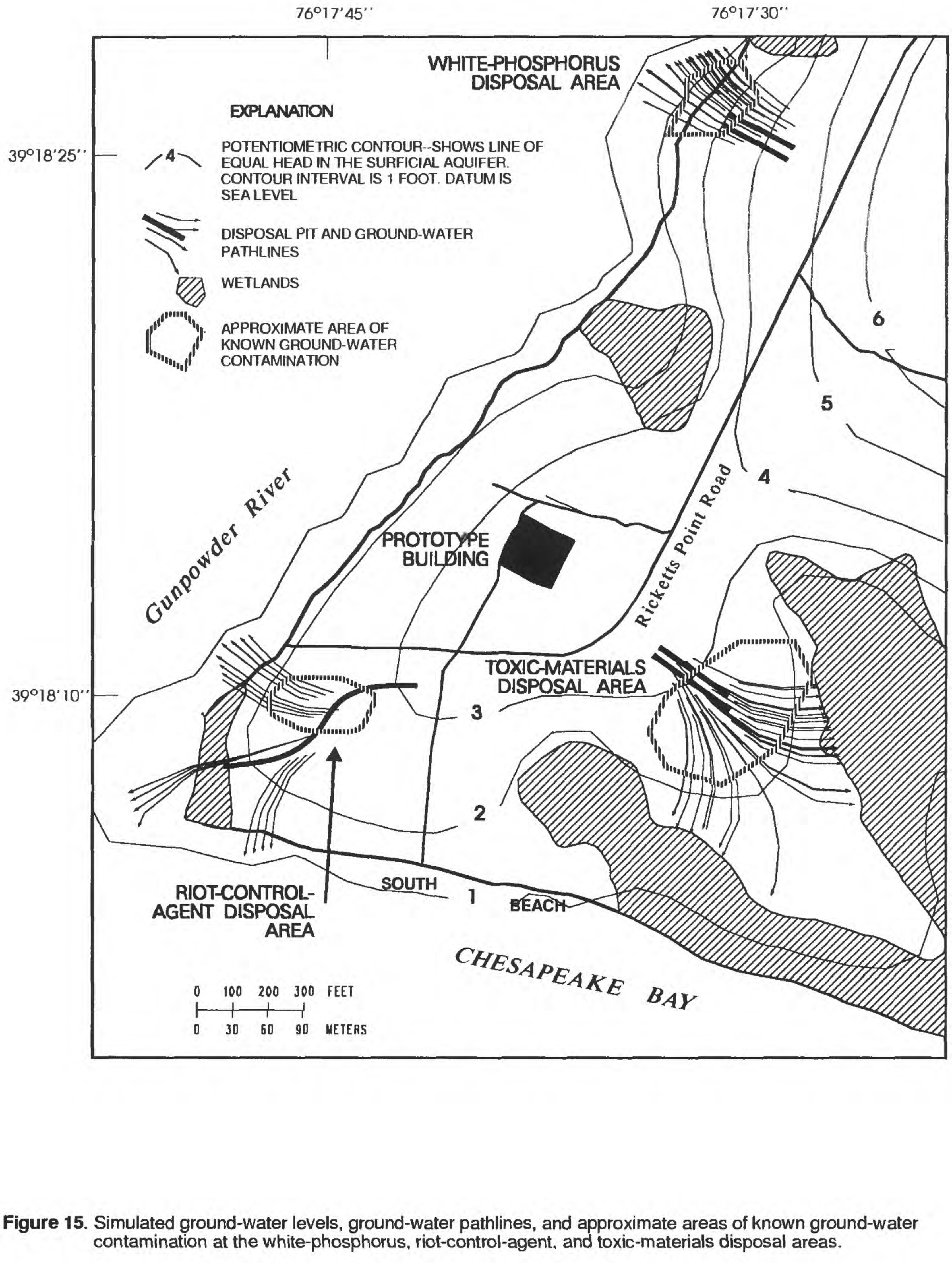


Figure 15. Simulated ground-water levels, ground-water pathlines, and approximate areas of known ground-water contamination at the white-phosphorus, riot-control-agent, and toxic-materials disposal areas.

6 years to travel from the disposal pit to a discharge point, contaminants that have a retardation factor of 3 or more (travel 1/3 or more the rate of ground water) should have reached the Gunpowder River. At this site, benzene is present in wells located approximately 100 ft from the river and is potentially discharging to the Gunpowder River. Under present flow conditions, contaminated ground water will continue to discharge to the Gunpowder River.

Ground-Water Flow at the Toxic-Materials Disposal Area

Ground water flows horizontally outward from the pits at the toxic-materials disposal area toward the wetlands located to the east and south (fig. 15). Chemical analyses of ground water at the toxic-materials disposal area indicate that ground water northwest of the disposal pits is uncontaminated and the water to the east and southeast contains high concentrations of chlorinated solvents (Martha Cashel, U.S. Geological Survey, oral commun., 1994). These areas of ground-water contamination lie on flowpaths simulated by the model. Surface-water samples collected in the wetlands in areas where the model indicates that ground water discharges also contain high concentrations of chlorinated solvents. Areas of the wetlands that were identified by soil-gas analysis as contaminated with chlorinated solvents (Hughes, 1993) correspond to ground-water discharge areas identified with the model. Chemical-contamination data along the simulated pathlines and discharge areas strongly support the results of the ground-water-flow model.

The model results indicate that ground water can take from 8 to 27 years to move from the disposal pits to discharge areas in the wetlands. Disposal operations at the toxic-materials disposal area began shortly after World War II and continued through the mid-1970's. The largest quantity of materials were disposed of during the 1970's. The travel times for contaminants are several times longer than the ground-water travel times because of retardation and dispersion. Consequently, much of the contamination should not have moved far from the disposal pits. Even assuming that the contaminants at the toxic-materials disposal area are conservative, based on the model-computed travel time, much of the 1970's contamination would still be in areas adjacent to the disposal pits or possibly beneath the wetlands.

The transport is complicated because of the presence of high concentrations of dense nonaqueous phase liquids (DNAPL's) in some wells at the toxic-materials disposal area (Martha Cashel, U.S. Geological Survey, oral commun., 1994). Because they are denser than water, DNAPL's, in the form of pure product, can sink in an aquifer until they reach a confining layer. Once there, the DNAPL can remain immobile or move slowly along the top of the confining layer due to gravity, all the while releasing contaminants into solution and acting as a continuous source of contamination. The particle-tracker subroutine calculates only advective flow or the approximate movement of the dissolved contamination. If pure product is present in large quantities, then it is likely that, under present conditions, some level of ground-water contamination will remain at the toxic-materials disposal area for an undetermined period of time. Under present flow conditions, the contaminants present in the ground water beneath the toxic-materials disposal area will continue to discharge into the wetlands.

Limitations of the Ground-Water-Flow Model

All ground-water-flow models are limited in their simulation capabilities because they are mathematical simplifications of infinitely more complex systems. The input data for the J-Field ground-water-flow model are geographically biased toward the disposal areas located in the southwest corner of the study area. Outside this area, the model is less finely discretized, data are more limited, and consequently, simulated heads are likely to be less accurate.

Because a steady-state model is being used for the simulations in this report, changes in water levels and aquifer storage were not considered. The simulations described in this report illustrate changes in heads and flows that could be expected as a result of the remedial actions with the flow system in equilibrium. The head distributions, travel times, and pathlines for the real flow system might not approach the simulated ones for several months to several years, depending on how long the system takes to approach equilibrium. Indeed, seasonal and annual variations in recharge, as well as tidal cycles, will always maintain a short-term disequilibrium in the system. The average conditions used to represent seasonal stresses should, however, result in simulations that approximate the real hydrologic system.

Calculations of travel times for ground water through the confining unit should be used with some caution. As currently calibrated, the model estimates travel times of several hundred years for ground water to pass through the confining unit. Some of the chemical data collected from wells contradict these findings. Analyses of water samples from the confined aquifer for tritium indicate that post-1950 water is mixed with water in the confined aquifer. Low concentrations of contaminants, which usually travel slower than ground water, also are present in the confined aquifer. As part of the calibration process, a simulation was conducted so that recharge from the surface reached the confined aquifer in 50 years, a scenario that would account for these chemical findings. Under these conditions, all the water in the surficial aquifer moved vertically downward, and the horizontal flow was inconsequential. These results, however, are incompatible with the horizontal spread of contamination that was evident at most of the disposal areas and indicate that this is an unrealistic simulation of the system.

There are several ways to explain these contradictory findings. The contamination in the confined aquifer could result from the movement of small quantities of DNAPL's through the confining unit along zones with higher than average vertical hydraulic conductivity. Since a single value of vertical hydraulic conductivity was used to calculate the leakance for the model, it does not account for limited zones of higher or lower hydraulic conductivity. Although there are no data to support this theory, it is also possible that the confining unit is not continuous beneath J-Field. Another explanation is that a small amount of water from the surficial aquifer could have been introduced into the confined aquifer during well drilling. In this case, the tritium and contaminants detected in the confined aquifer would have bypassed the confining unit altogether and would represent a minor source of contamination.

The calculations of contaminant transport are limited with the particle-tracker program. The model results can only be used to estimate transport of conservative chemical constituents. Many chemical compounds detected at J-Field are considered nonconservative and would be expected to move at a slower rate than ground water.

POSSIBLE EFFECTS OF REMEDIAL ACTIONS

Five remedial actions were simulated and analyzed to determine their effects on ground-water flow, movement of contaminants, and overall effectiveness. These include installing (1) an impermeable cap, (2) barriers to horizontal flow, (3) extraction wells, and (4) barriers to horizontal flow in combination with extraction wells. These remedial actions were simulated at the toxic-materials disposal area and the riot-control-material disposal area. Remediation was not simulated for the white-phosphorus disposal area because of the low concentrations and small distribution of contaminants there.

Installation of an Impermeable Cover

An impermeable cover could be installed at the disposal areas to prevent infiltration of precipitation, thus slowing the movement of water and dissolved contaminants. The primary result would be a lowering of heads in the surficial aquifer, smaller ground-water gradients, and a consequent reduction in ground-water velocities. The purpose of this type of remediation would be to prevent water from infiltrating into contaminant source areas, resulting in a slowing of contaminant transport. The impermeable cap is simulated by removing recharge to the area that would be capped.

Recharge was removed from approximately 8 acres of land surface to simulate an impermeable cap at the toxic-materials disposal area. The overall effect on the model was to reduce total model recharge by 2.7 percent. Ground-water discharge to the total model wetlands was consequently reduced by 5 percent and discharge to the estuaries was reduced by 1 percent. The heads in the toxic-materials disposal area were reduced by approximately 1 ft and simulated ground water moved half as fast from the disposal pits to the wetlands. At the riot-control-agent disposal area, recharge was removed from approximately 2 acres to simulate an impermeable cover. Recharge to the model decreased by 0.7 percent and simulated discharge to the bay was decreased by the same amount. The heads in the aquifer beneath the impermeable cover decreased by approximately 0.5 ft (16 percent) and simulated ground water moved from the disposal pit to the bay more slowly (travel times increased) by a factor of approximately 1.5.

Placing impermeable covers over the disposal sites would be effective for slowing down movement of contaminants but would be ineffective at containing or removing contaminants. Additional barriers to horizontal flow would be required to contain the contamination. Extraction wells or a drainage system would be required to remove contaminated ground water for treatment.

Installation of Barriers to Horizontal Flow

Barriers to the horizontal flow of ground water are used to contain contaminants or to prevent the migration of contaminants to a particular area. An example at J-Field would be a barrier to prevent the discharge of contaminated ground water into the Chesapeake Bay. Barriers can be constructed by injecting cement grout into the ground or by trenching and backfilling with a low permeability material (Canter and Knox, 1986). The barriers used for the following simulations were 10 ft wide, with a horizontal hydraulic conductivity of 0.00005 ft/d, and extended from land surface to the top of the confining unit. Because the model is only discretized to 100-ft^2 cells, the 10-ft-wide barriers were simulated by changing the conductivity of barrier cells to 0.0005 ft/d. This conductivity value is an average horizontal conductivity calculated from:

$$K_h avg = \sum_{m=1}^{n} (k_m d_m) / d \quad ;$$

where K_h avg is the average horizontal hydraulic conductivity, K_{hm} is the horizontal hydraulic conductivity, d_m is the thickness of each layer, and d is the total thickness of the unit (Fetter, 1980). In all the simulations with barriers, the recharge is removed from the barrier cells. The sensitivity ofthe model to the horizontal hydraulic conductivity of the horizontal barriers was tested. There was no leakage through the barriers with horizontal hydraulic conductivity values less than 0.005 ft/d. With horizontal hydraulic conductivity values larger than 0.005 ft/d, ground water began to leak through the barriers. The value of 0.00005 ft/d used for the simulated barriers is relative to the input values used to calibrate the model and is not necessarily representative of the actual conditions at the study site.

The first scenario tested was a barrier to the east of the disposal pits at the toxic-materials disposal area (fig. 16). The purpose of this barrier was to prevent contaminated ground water from discharging into the wetlands to the east of the toxic-materials disposal area. The barrier would result in a deflection of ground water toward the wetland on the south side of the disposal pits. Several different barrier placements were simulated to illustrate their hydrologic effects at the toxic-materials disposal area and the riot-control-agent disposal area (table 1). All of the simulated barriers are effective at preventing ground water from moving through the barrier; however, none of the barriers are viable remediation techniques in themselves. Ground water is deflected in another direction in all of the barrier simulations, resulting in contaminant transport in that direction. Some extraction method, such as a well or drain, is therefore needed to remove contaminated water from behind the barrier.

Complete encapsulation of the toxic-materials disposal area and the riot-control-agent disposal area was simulated. An impermeable cover and barriers to horizontal flow were placed around the pit areas as described above. The simulations indicated that water slowly circulates from the confined aquifer through the confining unit into the encapsulated zone and back through the confining unit and into the confined aquifer. The simulations also indicated that water inside the encapsulated area takes approximately 27,000 years to move from the toxic-materials disposal area to a discharge area in the Chesapeake Bay and approximately 25,000 years to move from the riot-control-agent disposal area to the Bay.

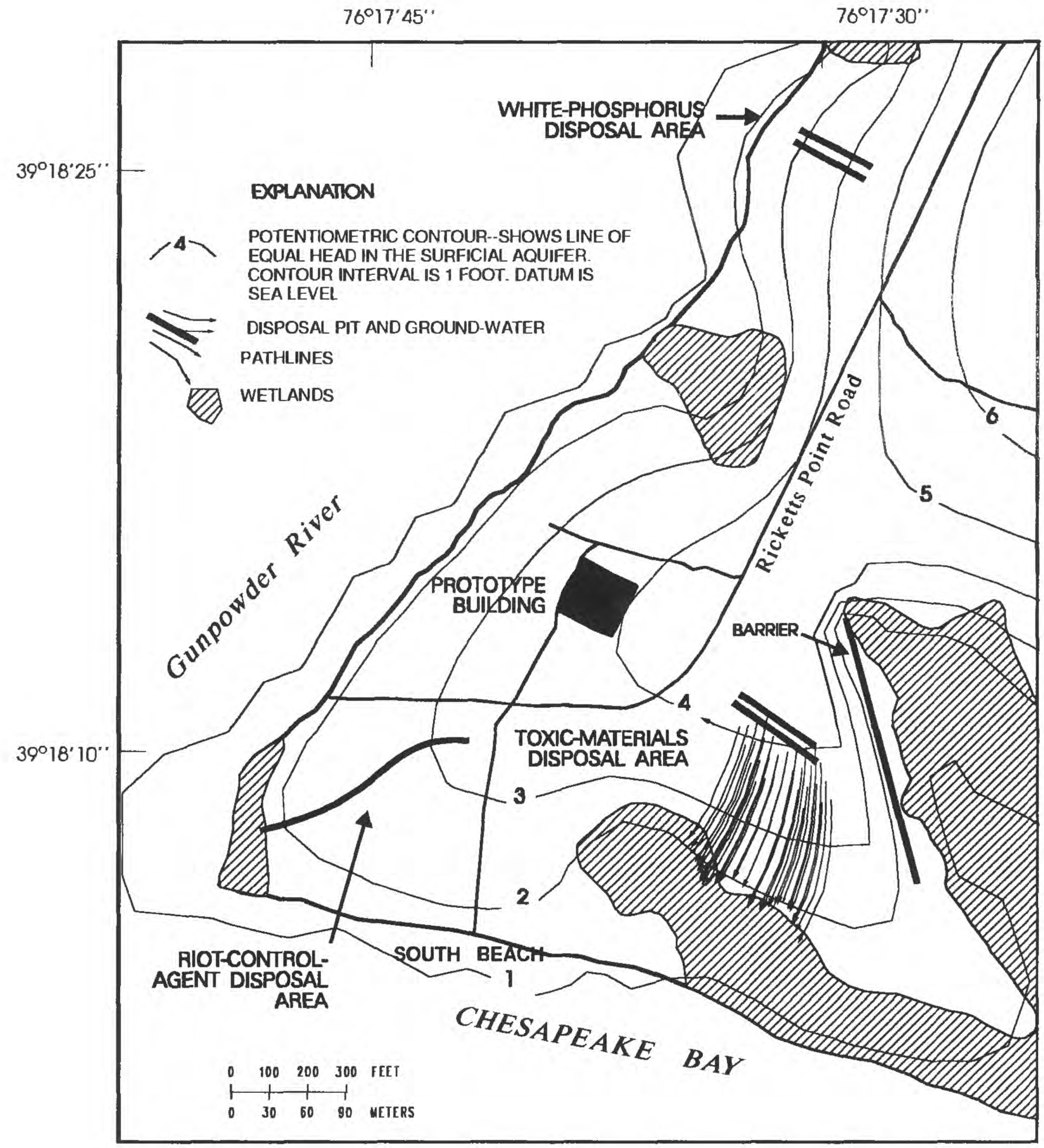


Figure 16. Simulated ground-water levels and ground-water pathlines for the toxic-materials disposal area with a barrier to horizontal flow east of the disposal pits.

Table 1. *Location and hydrologic effects of barriers to horizontal flow at the toxic-materials and riot-control-agent disposal areas*

Barrier location	Hydrologic effects
North side of toxic-materials disposal area	Stops movement of water into wetlands north of pits; deflects flow south of pits.
South side of toxic-materials disposal area	Stops movement of water into wetlands south of pits; deflects flow north of pits.
North and south side of toxic-materials disposal area	Stops movement of water into wetlands north and south of pits; gradient reversed with flow to west.
Surrounding toxic-materials disposal area	Heads build up to point where water would flow over the top of barriers.
West of riot-control-agent disposal area	Stops movement of water into Gunpowder River; flow deflected south into Cheasapeake Bay.
South and west of riot-control disposal area	Stops direct movement of water into Chesapeake Bay and Gunpowder River; flow goes around barrier to north and south.
Surrounding riot-control-agent disposal area	Water level builds up to point where flow is over the top of barriers.

Installation of Extraction Wells

Wells can be used to extract contaminated ground water which can then be treated onsite or removed to another location for treatment. Water treated onsite can often be released to nearby surface-water bodies or applied to the adjacent upland area to directly recharge the local system, thereby creating a semiclosed system. On the basis of the observed drawdowns in existing wells at the site during pumpage, sustained well yields on the order of 5 to 10 gal/min should be achievable. With this in mind, most of the pumpage simulations were conducted using well discharges of 2.5, 5, and 10 gal/min. In some cases, it was obvious from the 5-gal/min simulation that the 10-gal/min simulation would draw excessive amounts of water from the wetlands or bay; therefore, a 10-gal/min simulation was not conducted. Pumping water from the wetlands is undesirable because it could result in the loss of wildlife habitat if parts of the wetlands dried out, and because it would be costly and inefficient to pump and treat water that is relatively uncontaminated. If heads at the toxic-materials disposal area are lowered sufficiently by pumpage, the ground-water gradient can be reversed, allowing surface water in the wetlands to recharge the surficial aquifer. Since the object of the pumpage is to remove and treat contaminated ground water, extracting surface water would be costly and inefficient. A major element to be considered when choosing a remediation scheme is to determine the amount of wetland water that is drawn into the aquifer. Because all leakage is into the wetlands under existing conditions, the amount of water pumped from the wetlands was estimated by the amount of water leaking into the surficial aquifer from the wetlands during the pumpage scenarios. Not only is the amount of water removed directly from the wetlands a major consideration, but so is the quantity of water that is captured by the wells that originally discharged to the wetlands. This was examined in the pumpage scenarios by comparing the leakage to the wetlands under pumpage to the leakage without pumpage.

The application of the results of the pumpage scenarios with respect to the wetlands is limited because of the lack of data on sediment thickness and vertical hydraulic conductivity of wetland-bottom sediments. The results are further limited by the simplistic simulation of the wetlands using fixed heads when the actual heads vary seasonally; the steady-state model does not take these seasonal variations into consideration. The pumpage rates used in the scenarios, the quantity of water removed from the wetland, and the quantity of discharge captured by the wells are only estimates and do not represent absolute values that can be used for a remediation design. Further testing at the field site will be necessary to determine optimal pumpage rates, number of wells, and well placement.

Well placement for the pumpage scenarios at the toxic-materials disposal area included a single well on the northeast side of the disposal pits (fig. 17) and another simulation with two wells, one located to the northeast and one to the southwest of the pits (fig. 18). Simulated wells were placed in areas with the highest levels of ground-water contamination. The simulation with a single well pumped at 2.5 gal/min captured water from the northwest side of the pits (not shown). Ground-water flow on the southeast side of the pits was only slightly affected at this pumpage rate. The drawdown in the cell containing the well was approximately 1.0 ft and no water was removed from the wetlands under these conditions, but about 3 percent of the discharge to the wetland was captured by the well. The same well pumped at 5 gal/min captures water from a much broader area (fig. 17). Drawdowns in the vicinity of the well are approximately 2.1 ft and none of the water pumped by the well comes from the wetlands. About 7 percent of the wetland discharge is captured at this pumpage rate. At 10 gal/min, this well produces a drawdown of almost 4.5 ft near the well and captures water from much of the toxic-materials disposal area (not shown). The southwest side of the area, however, is still largely unaffected by the increased pumpage. About 3.5 percent of the water pumped at 10 gal/min comes from the wetlands, as indicated by leakage through the wetland bottom. A significant portion of the wetland discharge, about 13 percent, is captured at this pumpage rate.

The two-well pumpage scenario was tested to try to capture water from both the north and south sides of the toxic-materials disposal area. Pumpage was simulated with a total pumpage of 5 gal/min (not shown), where two wells pumped 2.5 gal/min each, and a total pumpage of 10 gal/min (fig. 18), where two wells pumped 5 gal/min each. With a total pumpage of 5 gal/min, most, but not all, of the area of identified contamination is contained within the capture area for the wells and no water is removed from the wetlands. Drawdowns at this pumpage rate are about 1.2 ft in both cells that contain the wells. No water is removed from the wetlands, but about 7 percent of the wetland discharge is captured by the wells. With a total pumpage of 10 gal/min, the capture area for the wells is more extensive and contains all of the contaminated area at the site (fig. 18). At this pumpage rate, less than 1 percent of the water pumped is from the wetlands and about 14 percent of the wetland discharge is captured by the wells. Drawdowns at this pumpage rate are about 2.4 ft in both wells.

To better understand the effectiveness of the model at simulating the amount of water removed and discharge captured from the wetlands, the sensitivity of the model to changes in the wetland-bottom conductance was tested under pumpage conditions. The vertical hydraulic conductivity value used to calculate wetland-bottom conductance in the calibrated model was 0.1 ft/d. Values of 0.05 ft/d and 0.2 ft/d were tested with the 5 and 10 gal/min, two-well pumpage simulations described above. The analysis showed that the amount of water removed from the wetlands and the amount of wetland discharge captured by the wells is not sensitive to the range of wetland-bottom conductances tested. The variation in these two values was negligible over the range tested.

The simulations at the toxic-materials disposal area indicate that a single well would not be sufficient to contain and/or remove contaminated ground water at pumpage rates less than 10 gal/min. Sustained pumpage rates greater than 10 gal/min are probably not achievable at this site. A minimum of two wells pumping at 5 gal/min would be needed to contain the contaminants at the site. More than two wells pumping at lower rates

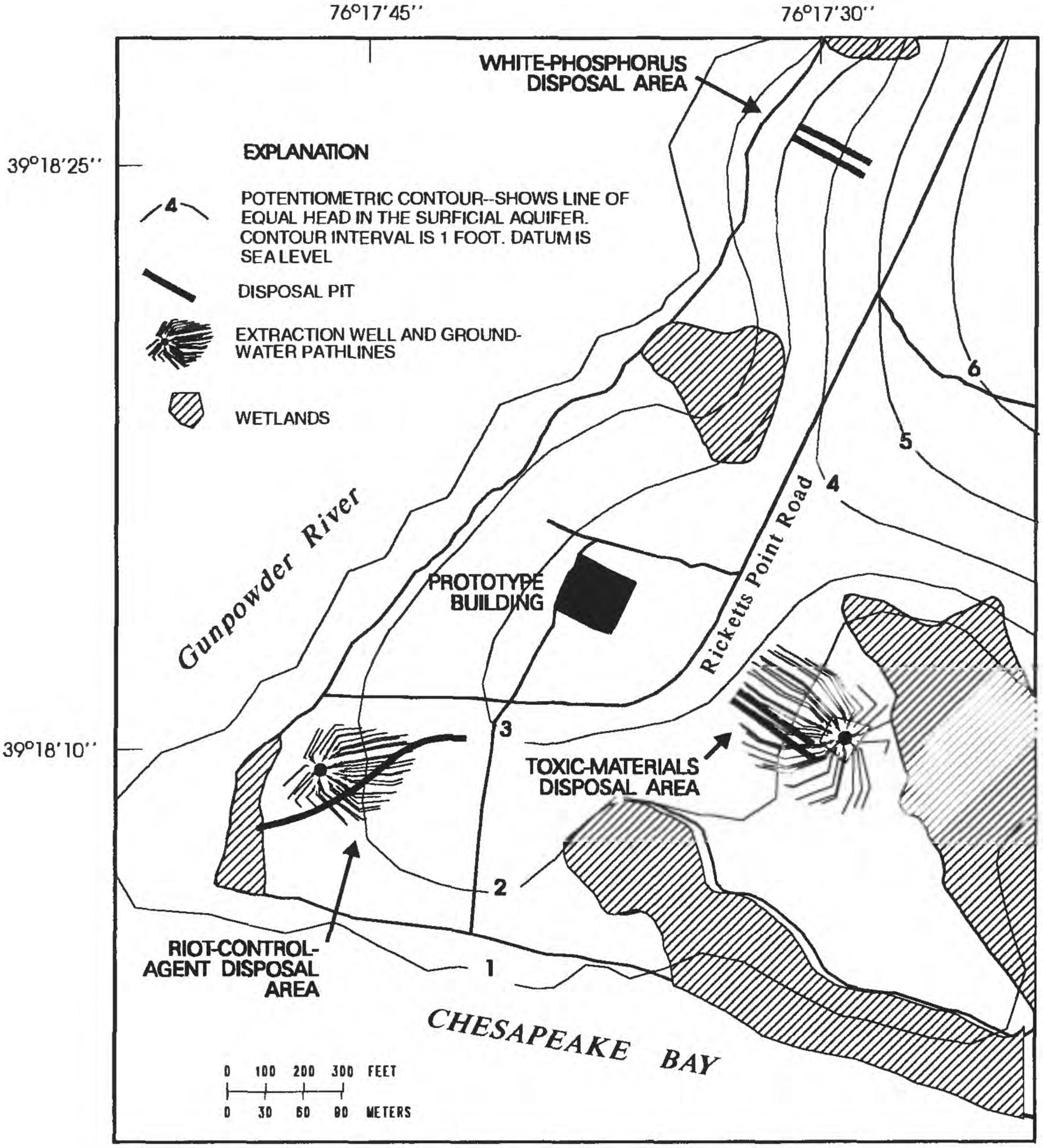


Figure 17. Simulated ground-water levels and ground-water pathlines for the toxic-materials disposal area with a single well pumping 5 gallons per minute and the riot-control-agent disposal area with a single well pumping 2.5 gallons per minute.

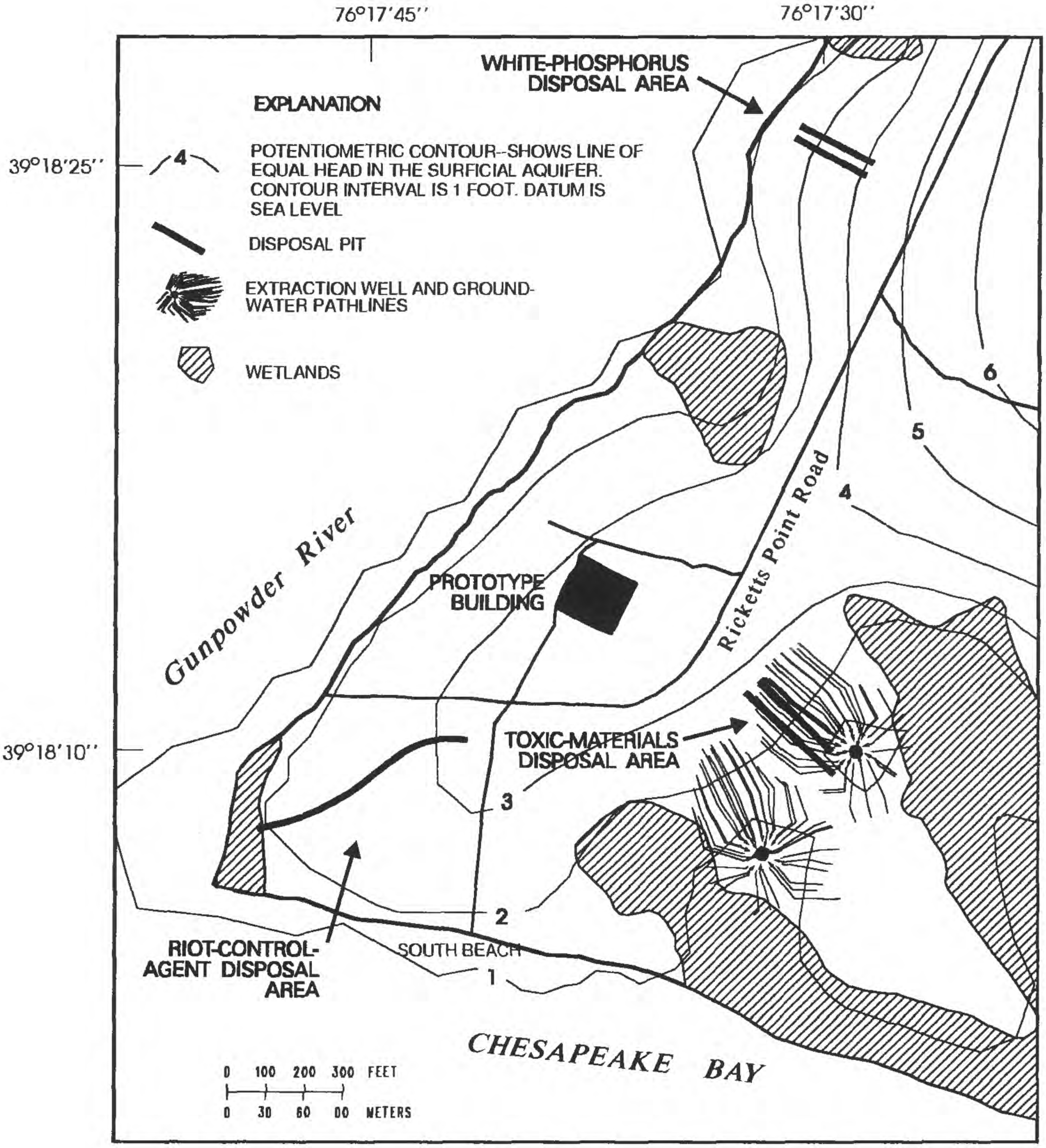


Figure 18. Simulated ground-water levels and ground-water pathlines for the toxic-materials disposal area with two wells pumping 5 gallons per minute each.

were not simulated but could also be effective for containing contaminated ground water. At higher pumpage rates, significant quantities of wetland discharge could be captured by the wells and affect the aquatic ecosystems there. This problem could be easily solved if contaminated ground water can be treated onsite and discharged to the wetland.

Pumpage at the riot-control-material disposal area was simulated for a single well downgradient of the disposal pit at 2.5 (fig. 17) and 5 gal/min. With a pumpage rate of 2.5 gal/min, water is captured from the area of known contamination at the riot-control-agent disposal area. None of the water that is pumped is removed from the Gunpowder River or Chesapeake Bay. At this pumpage rate, the existing flow is merely deflected toward the well and no flow reversal from river to well occurs. Drawdown in the cell containing the well was approximately 1.1 ft. At 5 gal/min, the area of capture for the well is slightly larger than at 2.5 gal/min, and there is some reversal of flow from the Gunpowder River toward the well; however, no river water is actually pumped by the well. Drawdown in the cell where the well is located is approximately 2.3 ft. At the riot-control-agent disposal area, a single well with a low pumpage rate will probably suffice to contain the contaminated ground water.

Installation of Extraction Wells and Barriers to Horizontal Flow

A combination of barriers to horizontal flow and pumping wells has several useful features. The barriers prevent the migration of contaminated ground water offsite and can also prevent unwanted pumpage of water from the wetlands or bay by extraction wells. The barriers will essentially enable the wells to be used most efficiently to extract water from the area desired. Because the simulations in the previous section indicated that no water is removed from the bay at the riot-control-agent disposal area under reasonable pumpage scenarios, no simulation of barriers in combination with pumpage was made at that site.

At the toxic-materials disposal area, barriers to horizontal flow to the north and south of the disposal pits and a single pumping well downgradient were simulated at 5 gal/min (fig. 19) and 10 gal/min. With 5 gal/min pumpage, the main area of ground-water contamination is captured by the well. Drawdown in the cell containing the well is -0.05 ft, indicating that the head is slightly higher in the well for this simulation than it is under ambient conditions. Because the barriers do not allow lateral flow, heads are higher in this simulation than they are under ambient conditions. Although the heads upgradient of the pits are 1 to 2 ft higher than they are under ambient conditions, the pumping well downgradient captures all the water from within the barrier and maintains heads at such a level that no water flows over the top of the barrier. The 10-gal/min simulation has a much larger capture area for the well, extending upgradient across Ricketts Point Road. Drawdowns at this pumpage rate are much greater, approximately 6.8 ft in the cell containing the well. Some water is pulled through the confining layer from the confined aquifer because of the increased gradient between the surficial and confined aquifers. For both the 5- and 10-gal/min simulations, none of the well discharge comes from the wetlands or bay; however, about 22 percent of the wetland discharge was captured by the well/barrier combination. The 10-gal/min pumpage would probably not be necessary to contain the contaminated ground water or to remove it for treatment.

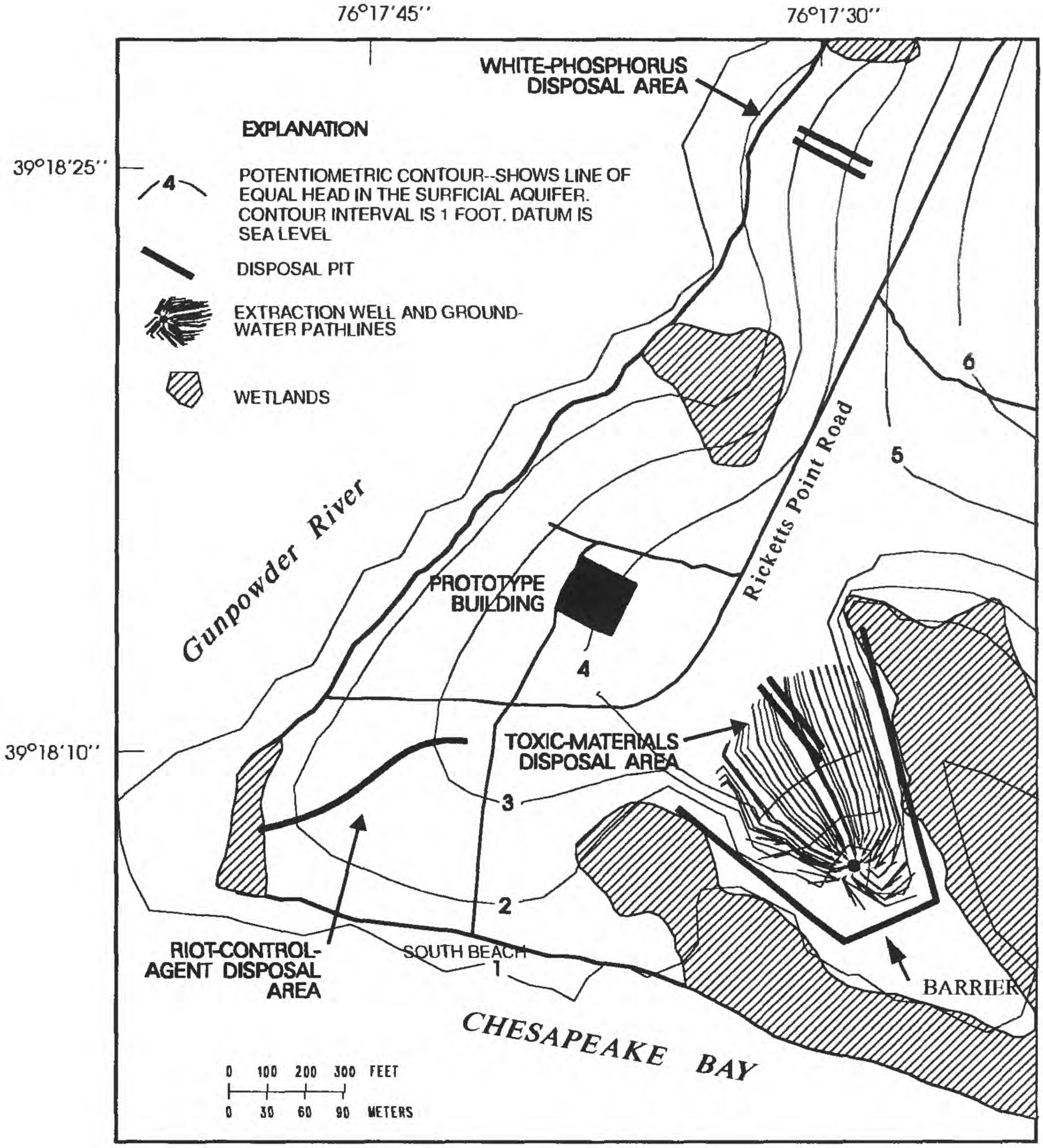


Figure 19. Simulated ground-water levels and ground-water pathlines for the toxic-materials disposal area with a single well pumping 5 gallons per minute and a barrier to horizontal flow.

SUMMARY AND CONCLUSIONS

J-Field is part of the Edgewood Area of Aberdeen Proving Ground, Md., and was used by the U.S. Army to test and dispose of explosives, chemical-warfare agents, and industrial chemicals. Ground-water, surface-water, and soil contamination has resulted from approximately 30 years of disposal operations. In order to understand the ground-water-flow system of the study area and to predict the possible hydrologic effects of remedial actions, a ground-water-flow model of the site was developed by the U.S. Geological Survey.

The major geologic units at the site are Cretaceous sand and clay, which is overlain by Quaternary paleochannel deposits consisting of sand, silt, and clay. Two aquifers are located in the Quaternary deposits--a surficial aquifer composed of interbedded sand and clay and a confined aquifer of gravel, sand, and clay. These aquifers are separated by a confining unit composed of silty and sandy clay. Most of the flow in the system is through the surficial aquifer, which is under water-table conditions. The surficial aquifer receives recharge at the land surface from precipitation and ground water flows horizontally toward the surrounding wetlands and estuaries, where it discharges. A small quantity of ground water moves vertically downward through the confining unit and into the confined aquifer. Water in the confined aquifer moves horizontally toward the estuaries, where it discharges upward through the confining unit and eventually into the Chesapeake Bay or Gunpowder River.

The U.S. Geological Survey finite-difference ground-water-flow model was used to simulate the flow and remedial actions at J-Field. The area modeled is 3.65 mi^2, using a variably spaced grid with 40 rows and 38 columns. The upper boundary of the model is the simulated water table. The lower boundary of the model is a no-flow boundary and corresponds to a clay layer in the Cretaceous deposits that underlies the confined aquifer. The horizontal boundaries are all no-flow boundaries. The north and south boundaries correspond to the contact between the Cretaceous deposits and the Quaternary deposits at the edge of a Pleistocene paleochannel. No physical boundaries were conveniently located to the east and west, so these boundaries were located far enough away from the area of interest to insure that they would not affect the simulations. All simulations were conducted under steady-state conditions.

Because of limited data, median values of hydraulic conductivity were used for all three hydrologic units. For the surficial aquifer, a horizontal hydraulic conductivity of 8.0 ft/d was used. A value of 0.005 ft/d was used for the vertical hydraulic conductivity of the confining unit and a value of 390 ft/d for the horizontal hydraulic conductivity of the confined aquifer. These values were multiplied by the thickness distributions of the respective units to obtain the distribution of leakance and transmissivity. Recharge for the model was determined by calculating a net water-level increase in the surficial aquifer for the water year 1990 and multiplying this by the specific yield of the aquifer. A value of 17.5 in/yr was obtained with this technique and corresponds well to the recharge values used for other models in the region. The estuaries and tidal wetlands were simulated with fixed heads that used a value of 0.9 ft above sea level, the average tide value for the Chesapeake Bay near J-Field. The nontidal wetlands were simulated with the river subroutine, and a vertical hydraulic conductivity value of 0.05 ft/d was used to calculate the wetland-bottom conductance. A small wetland northeast of the white-phosphorus disposal area was simulated with the drain subroutine. A vertical hydraulic conductivity value of 0.0032 ft/d was used to calculate the drain-bottom conductance.

The model was calibrated by comparing the simulated heads to heads measured on May 20, 1992. The horizontal hydraulic conductivity of the surficial aquifer and the vertical hydraulic conductivity used to calculate the leakance of the confining unit were varied to match the May 1992 heads. The absolute differences between measured and simulated heads in the surficial aquifer ranged from -1.03 to 1.28 ft (-52 to +32 percent) and in the confined aquifer ranged from -0.15 to 0.52 ft (-9 to +54 percent) for the calibrated model. The model calibration was sensitive to changes in recharge and was not sensitive to changes in the wetland-bottom conductance and transmissivity of the confined aquifer. Moving the horizontal boundaries of the ground-water-flow model inward by approximately 30 percent had a negligible effect on simulated heads in the surficial aquifer but changed heads in the confined aquifer by as much as +37 percent.

Simulation of ground-water flow at the white-phosphorus disposal area indicates that water from the disposal pits flows northeast toward the Gunpowder River. The travel times for ground water range from 2 to 5 years, indicating that the aquifer was flushed out from three to nine times since major disposal operations ended in the mid-1970's. Flushing the system has enabled much of the contamination to move offshore and probably discharge into the Gunpowder River. At the riot-control-agent disposal area, ground water flows horizontally from the disposal pit toward the Chesapeake Bay and Gunpowder River. Travel times vary from 6 to 14 years, much longer than at the white-phosphorus disposal area. The longer travel times probably contribute to the presence of elevated levels of contaminants adjacent to the disposal pit and immediately downgradient. At the toxic-materials disposal area, ground water diverges from the disposal pit area and flows toward wetlands located both east and south of the pits. Travel times for ground water range from 8 to 27 years. Because the travel times are similar in length to the time period since disposal operations ended, much of the contamination has not moved far from the pit area.

Remedial actions were simulated at J-Field to better understand their effects on the hydrologic system and their potential effectiveness. The remedial actions are limited in their application, because of assumptions about aquifer and confining-unit properties and simplifications that were used to develop the model. No remedial actions were simulated at the white-phosphorus disposal area because of the lack of any major contamination at that site. Simulations indicated that the installation of an impermeable cover at the toxic-materials and riot-control-agent disposal areas would effectively slow down the movement of ground water and would probably result in a slowing of contaminant transport. The time required for water to reach a discharge point was doubled at the toxic-materials disposal area and increased 1.5 times at the riot-control-disposal area.

Barriers to horizontal flow could be used to contain the contamination onsite but would require a method to reduce heads within the containment area, such as extraction wells to prevent water from flowing over the barriers. Simulations indicate that a 10-ft-wide slurry wall or other containment structure with a horizontal hydraulic conductivity as high as 0.005 ft/d could be used to prevent the horizontal flow of ground water. Extraction wells could be used to contain the contamination onsite by creating a cone of depression and could also be used to remove and treat contaminated water. Simulations indicated that a single well at the toxic-materials disposal area is insufficient to capture water from the entire area of contamination. Two wells located downgradient of the toxic-materials disposal area, pumping at 5 gal/min each, can effectively capture water from the contaminated area. In this scenario, no water is pumped from the adjacent wetlands, but about 14 percent of the wetland discharge is captured by the wells.

At the riot-control-agent disposal area, a single well pumping at 2.5 gal/min is sufficient to capture water from the area of known contamination. If extraction wells alone fail to prevent the spread of contamination, a combination of barriers and extraction wells can be used. A barrier to horizontal flow and a single extraction well was simulated for the toxic-materials disposal area. With the barrier in place, a single well pumping at 5 gal/min would capture water from the contaminated area and not remove any water from the adjacent wetlands; however, about 22 percent of the wetland discharge would be captured by the well/barrier combination. The captured discharge could be returned to the wetland if contaminated ground water can be treated onsite and discharged to the wetland.

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