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Effects of Underground Coal Mining on Ground Water in the Eastern United States

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EPA-600/7-80-120
June 1980

EFFECTS OF UNDERGROUND COAL MINING ON GROUND WATER IN THE EASTERN UNITED STATES

by

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FOREWORD

When energy and material resources are extracted, processed, converted, and used, the related pollutional impacts on our environment and even on our health often require that new and increasingly more efficient pollution control methods be used. The Industrial Environmental Research Laboratory - Cincinnati (IERL-Ci) assists in developing and demonstrating new and improved methodologies that will meet these needs both efficiently and economically.

This report summarizes documented effects of underground coal mining on ground water in the eastern United States and evaluates these effects on a regional basis. The findings provide a basic assessment of the mechanisms by which mining activities affect ground water and the extent of such changes. An understanding of these mechanisms can help to lead to changes in mining techniques or mine planning that will help to minimize adverse effects on ground water. For further information, contact the Energy Pollution Control Division, Industrial Environmental Research Laboratory - Cincinnati.

David G. Stephan
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ABSTRACT

This report addresses the past effects and the possible future effects of underground coal mining activities on ground-water resources in the region east of the 100th meridian. Such effects are highly dependent on the location of the mine with respect to natural flow systems. Freely draining up-dip drift mines, as well as actively pumped slope, shaft, and down-dip drift mines, act as sinks that reduce ground-water storage. This is especially true where mine roofs cave in and subsequent subsidence occurs over shallow mines. In these cases, secondary fractures extend up through overlying strata and may increase rock permeabilities by several orders of magnitude. Lowered ground-water levels around active mines commonly do not recover to pre-mining conditions after closure.

Studies indicate that contamination of ground water exists in many places in the immediate vicinity of coal mines. The nature and extent of the contamination is governed by the geochemistry of the individual seam being mined, the nature of flow around the mines, the presence or absence of calcareous material in associated strata, and the time of contact with various minerals.

Most refuse piles and impoundments in the Appalachian States are located near surface waters, which increases the opportunity for refuse leachate to enter streams by means of surface seeps or seepage into shallow ground-water systems. No underground mines and few refuse areas were found to have monitoring wells at locations and depths where water-quality problems or water-level changes might reasonably be expected. On a regional basis, there is little evidence from the scanty data available of gross ground-water contamination in heavily mined areas.

From the viewpoint of the value of ground-water resources, it is most likely that future underground mining in the Eastern Interior Basin and the southern Appalachians will result in adverse ground-water effects in only very limited areas. The central Appalachians, and in particular parts of western Pennsylvania and southern West Virginia, have a greater potential for such impacts. Pre-mine planning based on knowledge of local hydrogeology and geochemistry can lead to changes in mining techniques or mine planning that will help to minimize adverse effects on ground water.

This report was submitted in fulfillment of Contract 68-03-2467 by Geraghty & Miller, Inc., under the sponsorship of the U.S. Environmental Protection Agency. The report covers the period from September 27, 1976, to September 31, 1979, and work was completed as of September 31, 1979.

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LIST OF ABBREVIATIONS

bgd	- billion gallons per day
cm	- centimeter
cm/s	- centimeter per second
ft	- foot
gpd	- gallons per day
gpd/acre	- gallons per day per acre
gpd/ft^2	- gallons per day per square foot
gpd/mi^2	- gallons per day per square mile
gpm	- gallons per minute
g	- gram
g/m^3	- grams per cubic meter
ha	- hectare
in	- inch
kg/ha/d	- kilograms per hectare per day
km	- kilometer
lb	- pound
lb/acre/d	- pounds per acre per day
liter/min	- liter per minute
l/s (L/S)	- liter per second
m	- meter
m^3	- cubic meter
m^3/d	- cubic meters per day
$m^3/d/ha$	- cubic meters per day per hectare
$m^3/d/km^2$	- cubic meters per day per square kilometer
meq/gm	- milliequivalents per gram
mg/l	- milligrams per liter
mgd	- million gallons per day
mi	- mile
mm	- millimeter
ppb	- parts per billion
µg/l	- micrograms per liter
µmhos	- micromhos
µmhos/cm	- micromhos per centimeter
tons/yr	- tons per year
yd^3	- cubic yard

LIST OF CONVERSIONS

To Convert	Into	Multiply By
centimeters	inches	0.39
centimeters	feet	0.033
centimeters per second	gallons per day per square foot	21204
centimeters per second	feet per day	2835
cubic meters	gallons	263.18
cubic meters	cubic feet	10.76
cubic meters	cubic yards	1.30
cubic meters per day	gallons per day	264.2
cubic meters per day per hectare	gallons per day per acre	106.91
cubic meters per day per square kilometer	gallons per day per acre	1.07
cubic meters per day per square kilometer	gallons per day per square mile	688.13
grams	pounds	0.002
hectare	acre	2.47
kilograms	pounds	2.2
kilograms per hectare per day	pounds per acre per day	0.89
kilometers	miles	0.62
liters	gallons	0.26
liters per minute	gallons per minute	0.26
liters per second	gallons per minute	15.85
liters per second	gallons per day	22824
millimeters	inches	0.039
meters	feet	3.28
metric tons per year	short tons per year	1.10
thousand cubic meters per day	million gallons per day	0.264

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SECTION 1

INTRODUCTION

Coal mined by underground methods has been a major source of energy in the United States for over two hundred years. Historically, most coal production has come from underground mines east of the Mississippi River, and it is likely that much of the future mining of coal reserves in this region will also be by underground techniques.

In recent years, concern has arisen over the adverse effects of this mining on the environment, with most of the concern being centered around degradation of surface waters by coal mine drainage. Little study has been made of effects on underground water resources. The objective of the present investigation is to attempt to fill this void by summarizing documented effects on ground water of past underground mining and by assessing the potential for such effects from future mining. The study region comprises Alabama, Illinois, Indiana, Kentucky, Maryland, Ohio, Pennsylvania, Virginia, and West Virginia (Figure 1).

Owing to the broad scope of the subject and the small amount of relevant ground-water data available in useful form, the investigation relied heavily on personal communications with state, federal, and university personnel, and on information contained in published and unpublished reports that was primarily of a site-specific nature. Some additional information was obtained through visits to coal waste-disposal sites and from examination of ground-water data compiled on a regional and county basis. In addition, a review was made of legal case histories involving coal and water problems in all of the states except Indiana. Alternatives for minimizing or controlling ground-water quality or quantity problems related to subsurface mining activities are evaluated for their effectiveness.

Because most work in this subject area has been done on only a limited site-specific basis, the first task was to assess the relative magnitude of effects from different mining activities on a regional basis. In addition, an attempt has been made to describe the extent and distribution of such effects as far as is possible using existing data. The condition of ground water near mining activities is assessed by consideration of well water quality and levels, baseflow, springs, and other unspecified discharges where these data were available in useable form. Point-source discharges to streams from mines or from coal-waste pile seeps are addressed only insofar as they are related to, or are an indication of, changes in subsurface conditions. Several regulatory

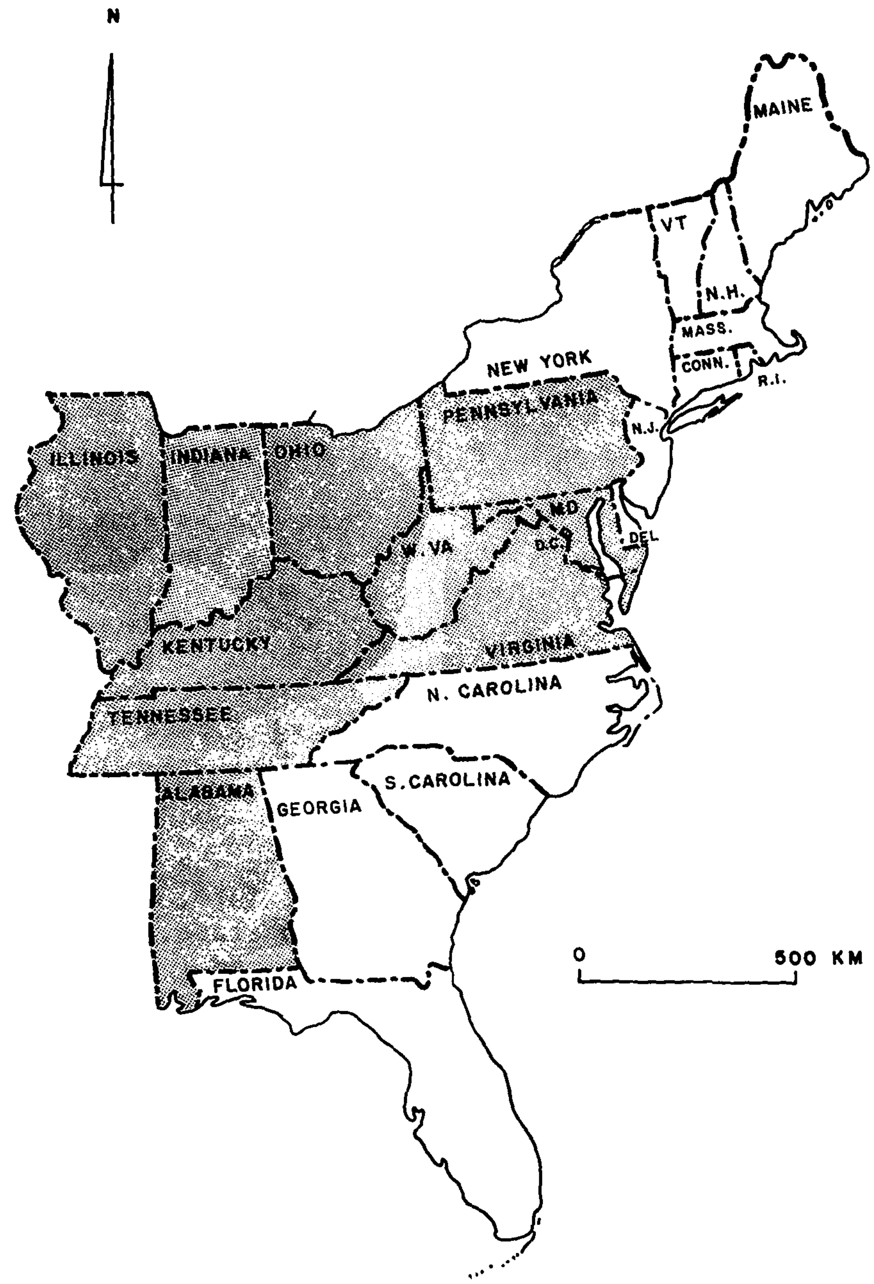

Figure 1. Coal-bearing states included in this investigation.

controls already exist for these point sources, including issuance of discharge permits under EPA's National Pollution Discharge Elimination System. Numerous river basin studies have been made to investigate and abate these kinds of discharges.

Background information presented in this report includes maps of the location and distribution of underground coal mines and reserves in the eastern United States, the availability of ground water on a statewide basis, and ground-water use on a county basis. For purposes of this report, the value of the ground-water resource is determined from a combination of use and availability. Use is taken to be a measure of the present worth of the ground water, and availability is an indication of its future worth. Using this value system, a series of maps have been generated which indicate areas with potential for significant ground-water problems from future underground mining.

Complex patterns of ground-water occurrence and flow are analyzed by use of conceptual models and review of case histories. Impacts of mining on subsurface flow systems can cause changes in water levels, ground-water storage, and streamflow. Changes in ground-water quality as a result of mining are assessed largely on the basis of results of previous studies and case histories. A review of basic data on ground-water quality was performed in a region in Pennsylvania that has been heavily mined by underground methods for many years.

In addition to the effects of the actual underground mining operations, coal wastes from the underground mines that are placed on the land surface also have a potential for causing degradation of the quality of shallow ground-water resources and streams. In order to help assess the potential for ground-water contamination from coal waste disposal, 23 sites were inspected in Illinois, Pennsylvania, Ohio, and West Virginia. Refuse piles and slurry impoundments at these sites were inspected for physical characteristics, geologic and geographic setting, recharge/discharge relationships, and water-quality conditions.

SECTION 2

FINDINGS

1. The effects of underground mining activities on ground-water flow and/or ground-water quality are highly dependent on the location of the mine with respect to natural flow systems. For example, mining or placement of coal wastes in a local ground-water recharge area is likely to affect shallow ground water and baseflow to small streams. Similar activities in a regional recharge area on a broad upland may affect deeper aquifers over long periods of time. In contrast, mining operations in ground-water discharge areas may have only a small effect on shallow ground water but a significant impact on nearby surface-water bodies.

2. It is estimated that millions of cubic meters of ground water are diverted from natural flow systems every day via drainage from abandoned mines and pumpage from active mines. Declines of ground-water levels resulting from the diversion can result in the drying up of shallow wells. When mines are sealed or are naturally filled up after active mining, water levels may recover, but, because of the fracturing of overlying rocks, usually not to original levels.

3. Removal of water from mines typically reduces the rate of recharge to underlying aquifers and may cause a shift in the position of ground-water divides as a result of changes in recharge/discharge relationships. In addition, baseflow to nearby streams may be significantly changed as a result of diversions of ground water into or around mines in heavily mined watersheds. Losses and gains in streamflow have been observed for various watersheds within a given mining district.

4. Freely draining up-dip drift mines, as well as actively pumped slope, shaft, and down-dip drift mines, act as sinks that lower ground-water levels and reduce the amount of ground water in storage above the mine. This is especially true where mine roofs cave in and subsequent subsidence occurs over shallow mines. Where roof fractures reach the land surface and subsidence occurs, there is an increase in recharge to the subsurface and a decrease in overland runoff. In such areas, the rate of travel of ground water from the surface to points of discharge is typically greatly increased.

5. Interconnection between underground mines and surface mines, as well as interaquifer flow via abandoned oil and gas wells, can result in the alteration of directions and rates of natural subsurface flow.

6. Underclays directly below coal seams retard vertical leakage of poor quality water to underlying aquifers. Underclays in the Appalachian Basin tend to be more brittle than those in the Midwest and, therefore, are probably less effective in retarding downward seepage of contaminants. It is uncertain whether this seepage could lead eventually to large-scale degradation of deeper aquifers.

7. Ground water contaminated by underground mining activities and refuse disposal may ultimately enter nearby streams in the form of baseflow seepage, discharge from springs, and in some places, discharge from flowing oil and gas wells. Such discharges undoubtedly contribute to the existing poor quality of some streams in mining areas. Degraded ground water from single refuse piles will typically move in discrete and defineable plumes toward nearby points of discharge. Significant quantities of dissolved solids will continue to be generated from refuse piles for many years after initial disposal.

8. The chemical quality of ground water around mines is governed, in part, by the geochemistry of the individual seam being mined. Where calcareous sediments are associated with the coal, the acidity in ground water near mines is generally low. In places, the reactivity of pyrite associated with the coal may be related to the type of paleoenvironment under which the coal was deposited.

9. Naturally mineralized shallow ground water occurs in parts of every state studied at depths of less than 91 m (300 ft), especially in Illinois and western Kentucky. Further contamination of this poor quality ground water by mining activities may not be of serious concern.

10. There is some evidence that inundation of shaft, slope, and down-dip mines following periods of active mining will result in reduced levels of contamination. It is likely that ground-water quality above and around abandoned underground mines will be least affected where exposure of coal to air is eliminated and where ground-water flow conditions are most like pre-mining conditions. However, where the rate and volume of ground-water circulation from the land surface have been significantly increased as a result of fracturing, exposed pyrite will be subject to increased oxidation, which in turn could lead to contamination. This situation is most likely above relatively shallow underground mines in areas of moderate or rugged relief.

11. Almost all refuse piles and impoundments in the Appalachian states are close to surface-water drainage features, posing threats primarily to streams and very shallow ground waters. Coal refuse piles in the midwestern states are not necessarily associated with drainage courses but may be located on deposits of glacial till where leachate may slowly infiltrate into underlying sediments. In general, coal refuse on till deposits represents a threat mainly to the quality of ground water in the immediate vicinity of the waste.

12. Coarse refuse generally absorbs precipitation at higher rates than either the bedrock or till deposits in coal regions. Assuming a recharge rate of 38 cm (15 in) per year, a typical 40-ha (100-acre) pile would absorb approximately 379 m^3/d (100,000 gpd). Depending on the hydrogeologic setting, part of this degraded water will enter the shallow ground-water system.

13. On both county-wide and regional bases, there is little evidence of widespread ground-water contamination in heavily mined areas. However, the data base is inadequate, and where detailed studies of ground-water quality have been conducted, the findings almost always indicate that contamination is present in the immediate vicinity of the mines.

14. Existing ground-water quality monitoring systems are inadequate to assess the extent of ground-water contamination in the coal regions of the study area. In most states, the monitoring systems make use of water-supply wells that are remote from sources of contamination and, therefore, are of little value in detecting contamination. No underground mines and few refuse disposal areas were found to have special monitoring wells placed at locations and depths where water-quality problems might reasonably be expected to occur.

15. From the viewpoint of the value of ground-water resources, it is most likely that future underground mining in the Eastern Interior Basin and the southern Appalachians will result in adverse ground-water effects in only very limited areas. The central Appalachians, and in particular parts of western Pennsylvania and southern West Virginia, have a greater potential for such impacts.

16. Engineering and hydrologic controls, such as mine sealing implemented during and after the mining process, are seldom fully effective in controlling adverse effects on ground water. From a ground-water quality viewpoint, mine seals probably have little benefit except in cases where post-mining flow patterns are restored to pre-mining conditions. Such restoration is most likely to occur only in deep mines with shaft or slope openings, or at drift mines with sufficient pillars to support roof rocks indefinitely. Ground-water diversion around mines by means of dewatering or connector wells holds promise for minimizing ground-water impacts in some hydrogeologic settings.

17. Studies of the hydrogeology and ground-water geochemistry during the pre-mining site-selection process can provide information that will alert mining companies and environmental regulatory agencies to potential ground-water problems. Consideration of this potential may indicate that a particular site is environmentally or economically unsuitable for mining or disposal, or that special mining techniques and disposal procedures will be needed.

SECTION 3

UNDERGROUND COAL MINING IN THE EASTERN UNITED STATES

The type of mine openings, depth of mining, and intensity of underground mining operations influence mine drainage and the hydrology at and near underground coal mines. Entryways to mines are of three general types: drift, slope, and shaft. In drift mining the coal is first removed from a hillside outcrop and the coal is removed progressively along the seam into the hill. Slope mining is used where the coal is too deep to remove economically by strip mining. In this technique, a sloped tunnel extends from the surface to the coal seam. Shaft mining is generally used to reach coal at still greater depths below the land surface. Vertical shafts are sunk and mining takes place in a horizontal direction from the shafts. Portals can serve as points of ground-water discharge or recharge both during mining and after abandonment of the mine. Mining may result in fracturing of overlying strata and eventually land subsidence from mine roof collapse. Disturbance of the stratification and permeability of these rocks can result in changes in recharge/discharge relationships, water levels, and ground-water flow patterns.

The intensity of underground coal mining in the states east of the 100th meridian is indicated by Figures 2 through 7, which show underground production and reserves by county. The relative order of production by states in 1975 (from highest to lowest) is as follows: Kentucky, West Virginia, Pennsylvania, Illinois, Virginia, Ohio, Alabama, Tennessee, Indiana, and Maryland (Table 1). Comparison between production and reserves in individual states gives an indication of the potential problem areas associated with future underground mining. The production in most of the states coincides with the areas of greatest reserves. However, in parts of Illinois, Indiana, and northern West Virginia, production at present is low, in spite of the availability of significant underground reserves. For purposes of this report, reserves are defined as coal seams .71 m (meters) (28 in.) or more thick that occur at depths to 305 m (1,000 ft.). Resources include any coal units which have been mapped, indicated, or inferred at depths to 915 m (3,000 ft.).

Figures 8 through 17 show the coal resources or reserves in eight states and the number and distribution of active underground mines producing more than 180,000 metric tons/yr (200,000 tons/yr). The number of these active mines as of 1975 ranges from 0 in Indiana (not shown) to 126 in West Virginia. These numbers include mines not reporting production but that are in the >180,000 metric ton class. They may, therefore, differ somewhat from the U. S. Bureau of Mines (1975) estimates. In some states, maps were available showing mined-out seams. This term does not

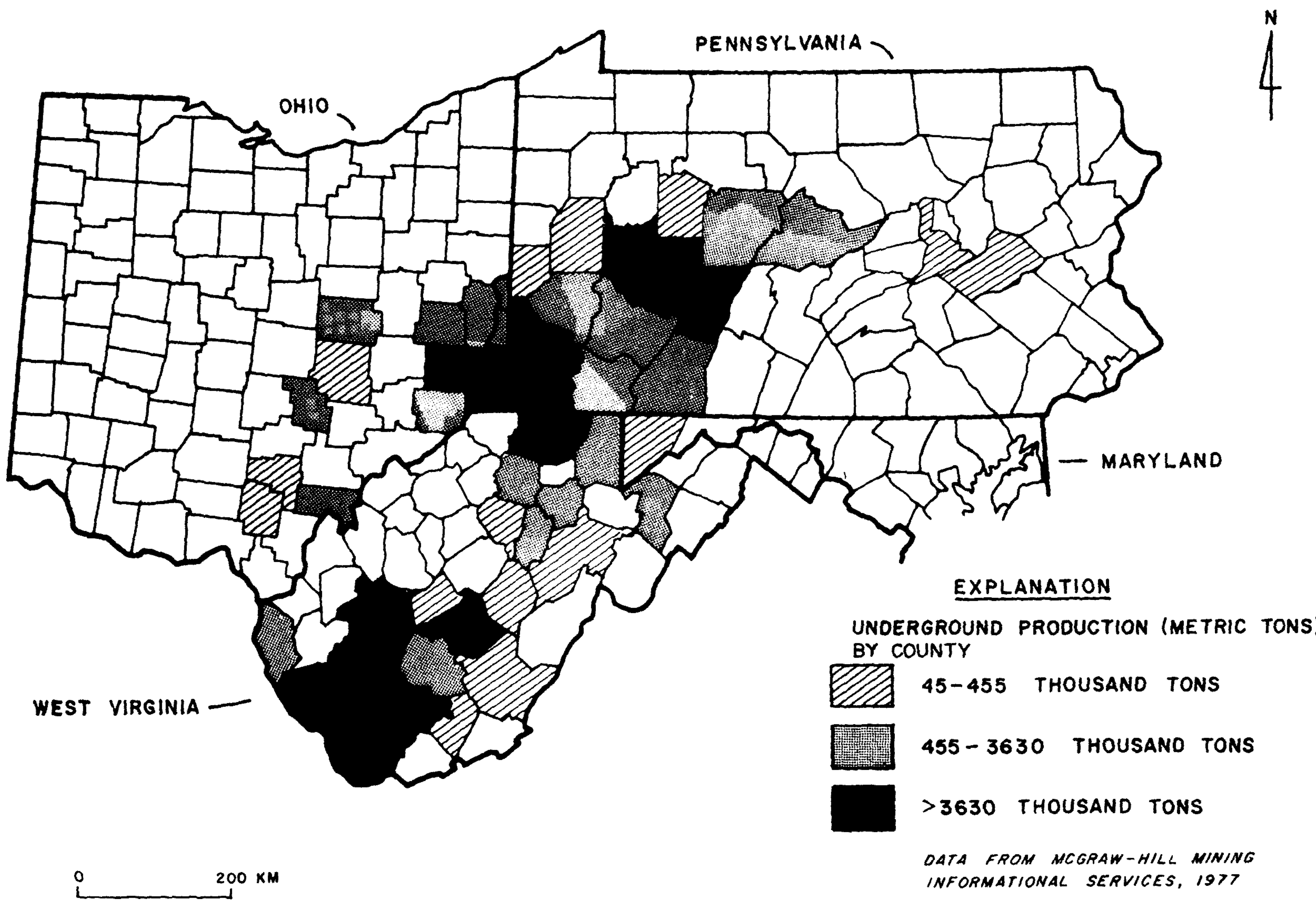

Figure 2. Production of coal from underground mines in the central Appalachians in 1975.

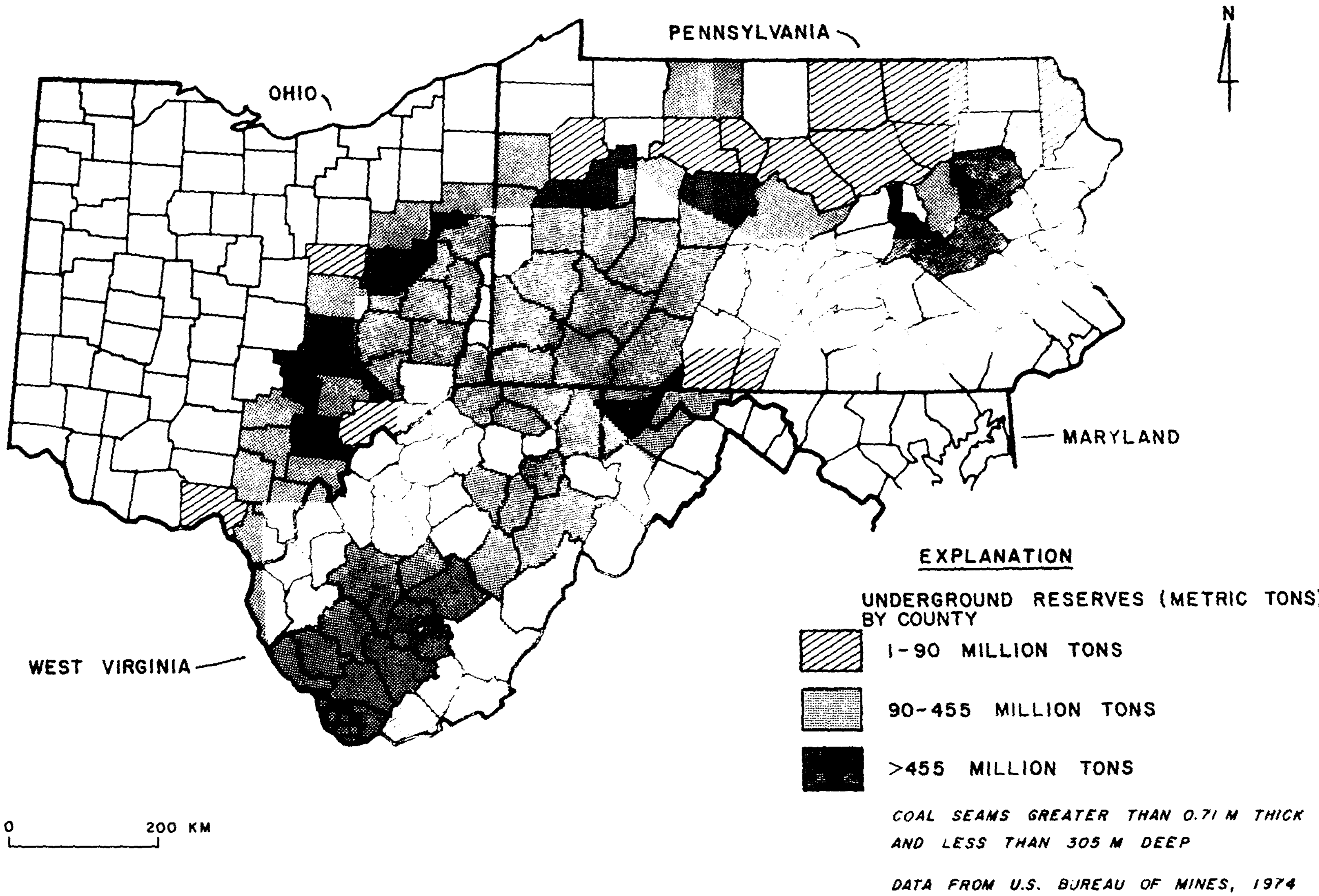

Figure 3. Coal reserves for underground mining in the central Appalachians.

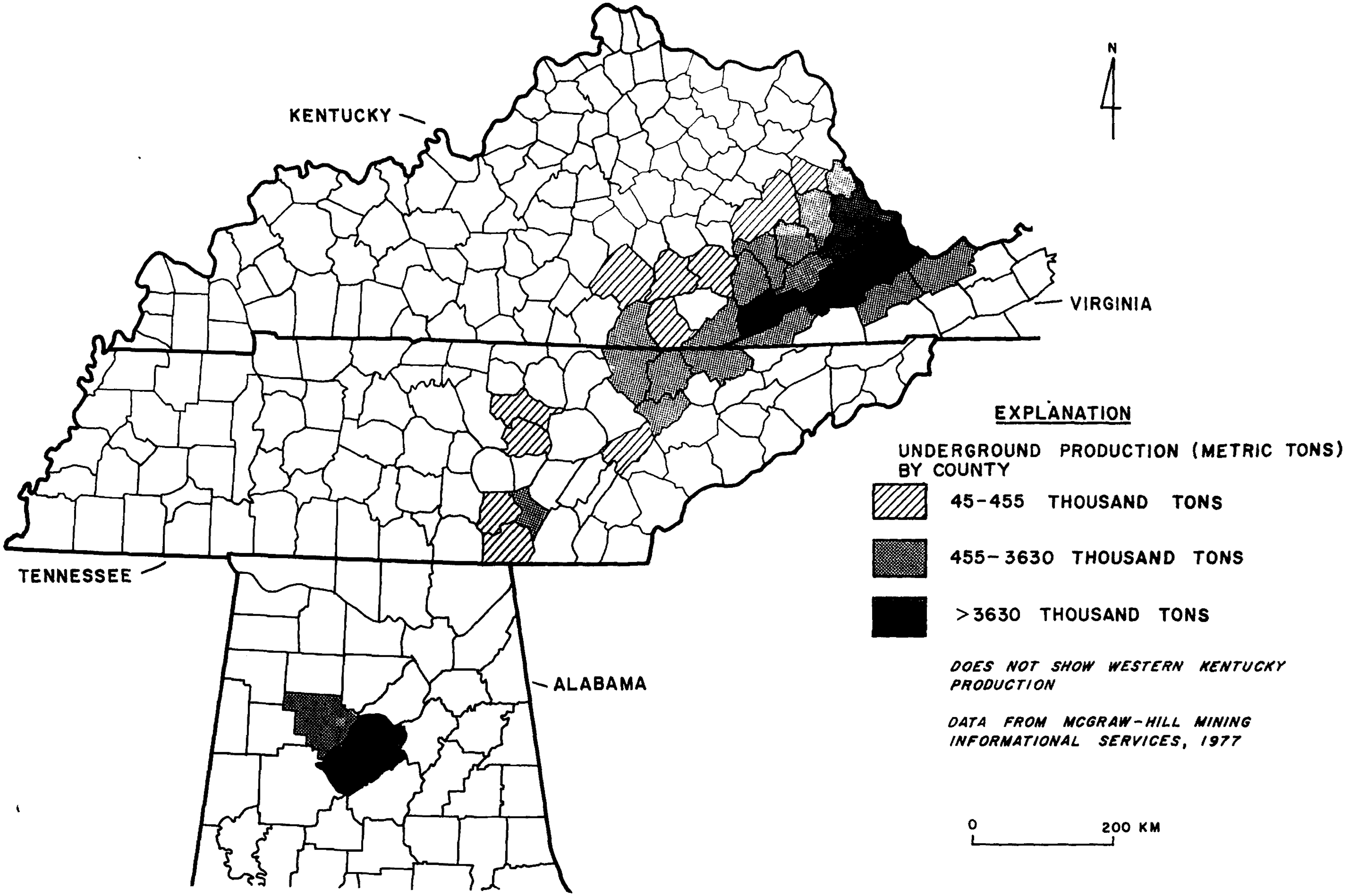

Figure 4. Production of coal from underground mines in the southern Appalachians in 1975.

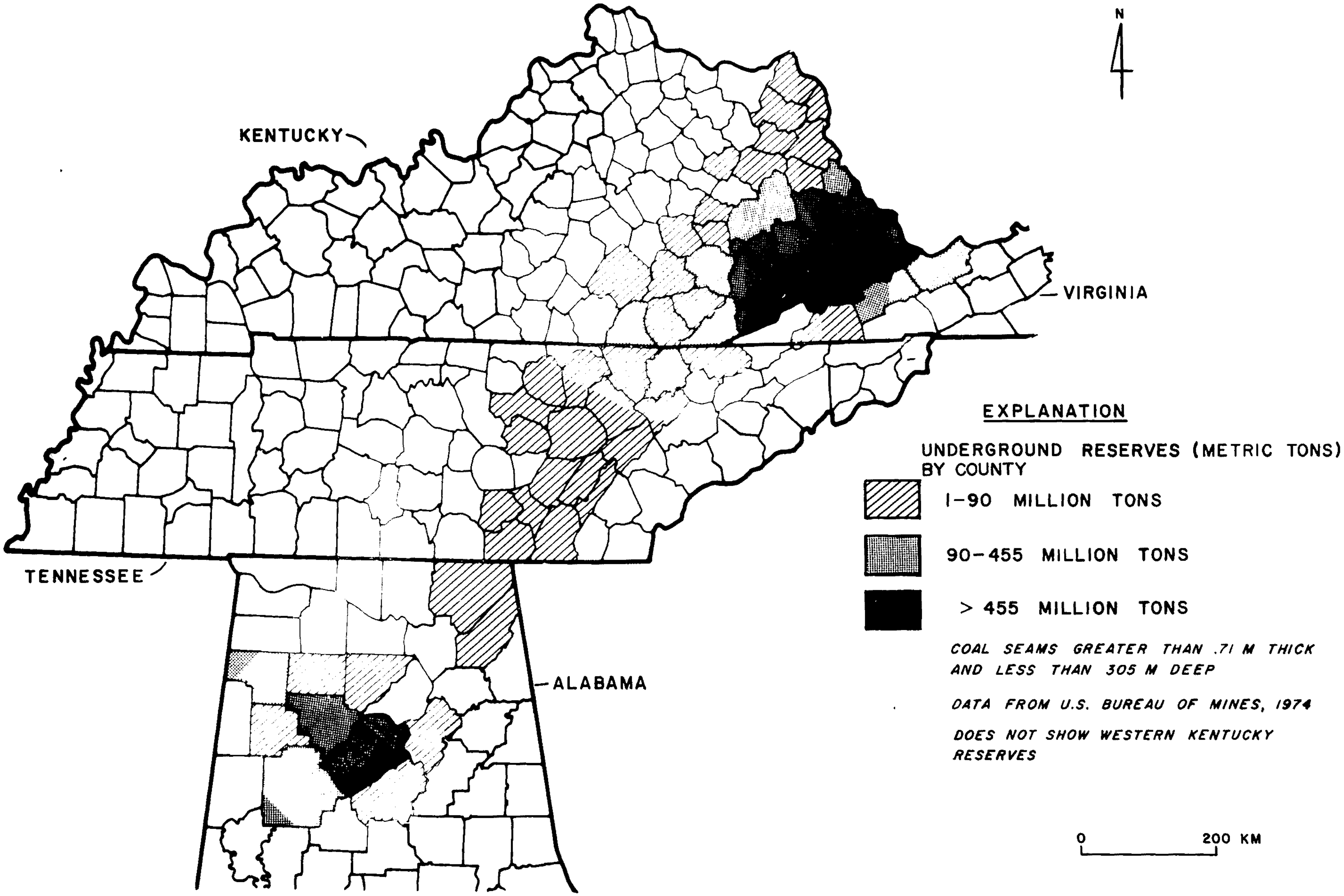

Figure 5. Coal reserves for underground mining in the southern Appalachians.

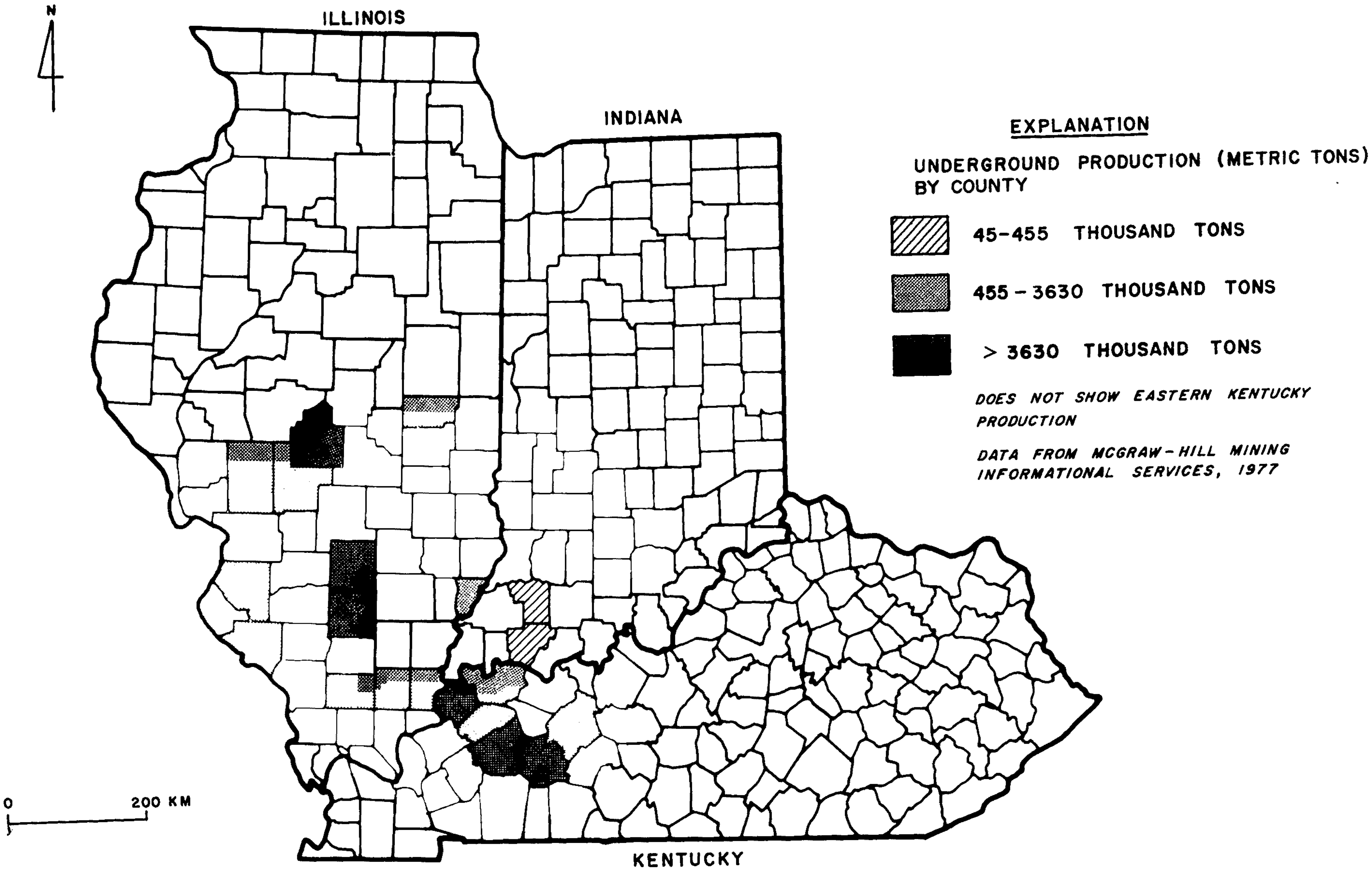

Figure 6. Production of coal from underground mines in the Eastern Interior Basin in 1975.

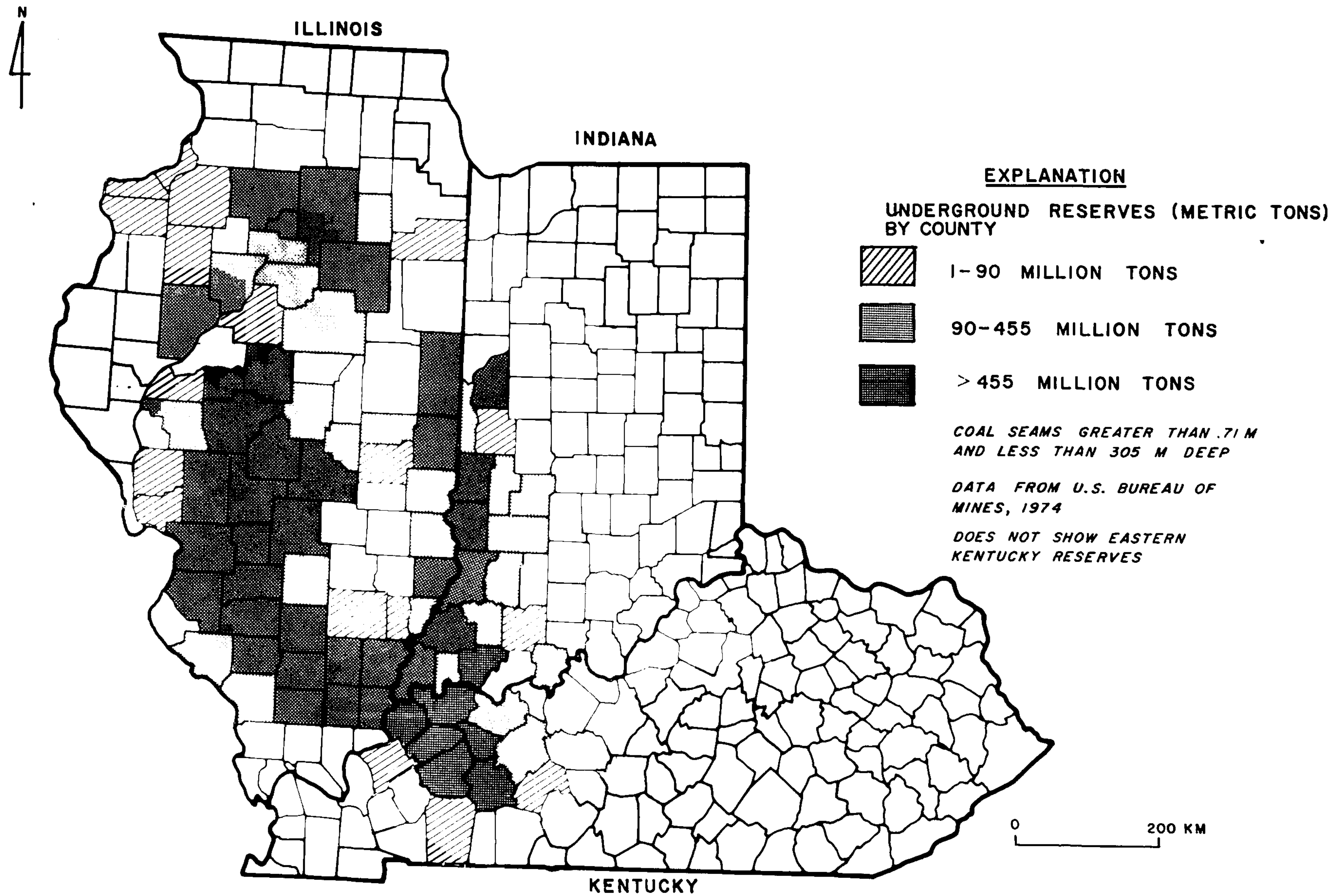

Figure 7. Coal reserves for underground mining in the Eastern Interior Basin.

TABLE 1. STATE PRODUCTION OF COAL MINED BY UNDERGROUND METHODS IN 1975 (Source: U. S. Bureau of Mines, 1976)

State	Production (thousand metric tons)
West Virginia	80,210
Kentucky	59,581
Pennsylvania	40,516
Illinois	28,936
Virginia	21,043
Ohio	14,030
Alabama	6,912
Tennessee	3,455
Indiana	171
Maryland	94

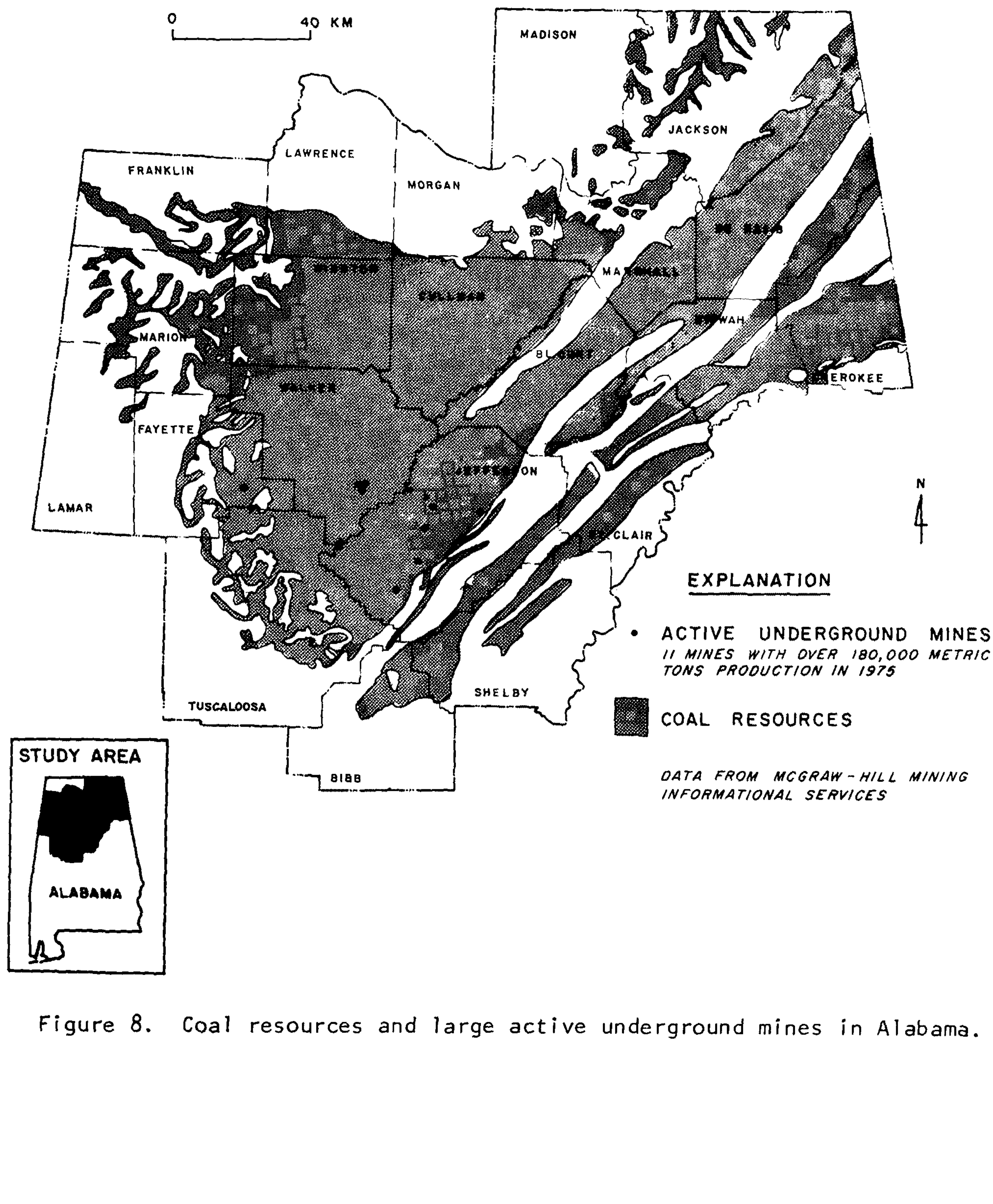

Figure 8. Coal resources and large active underground mines in Alabama.

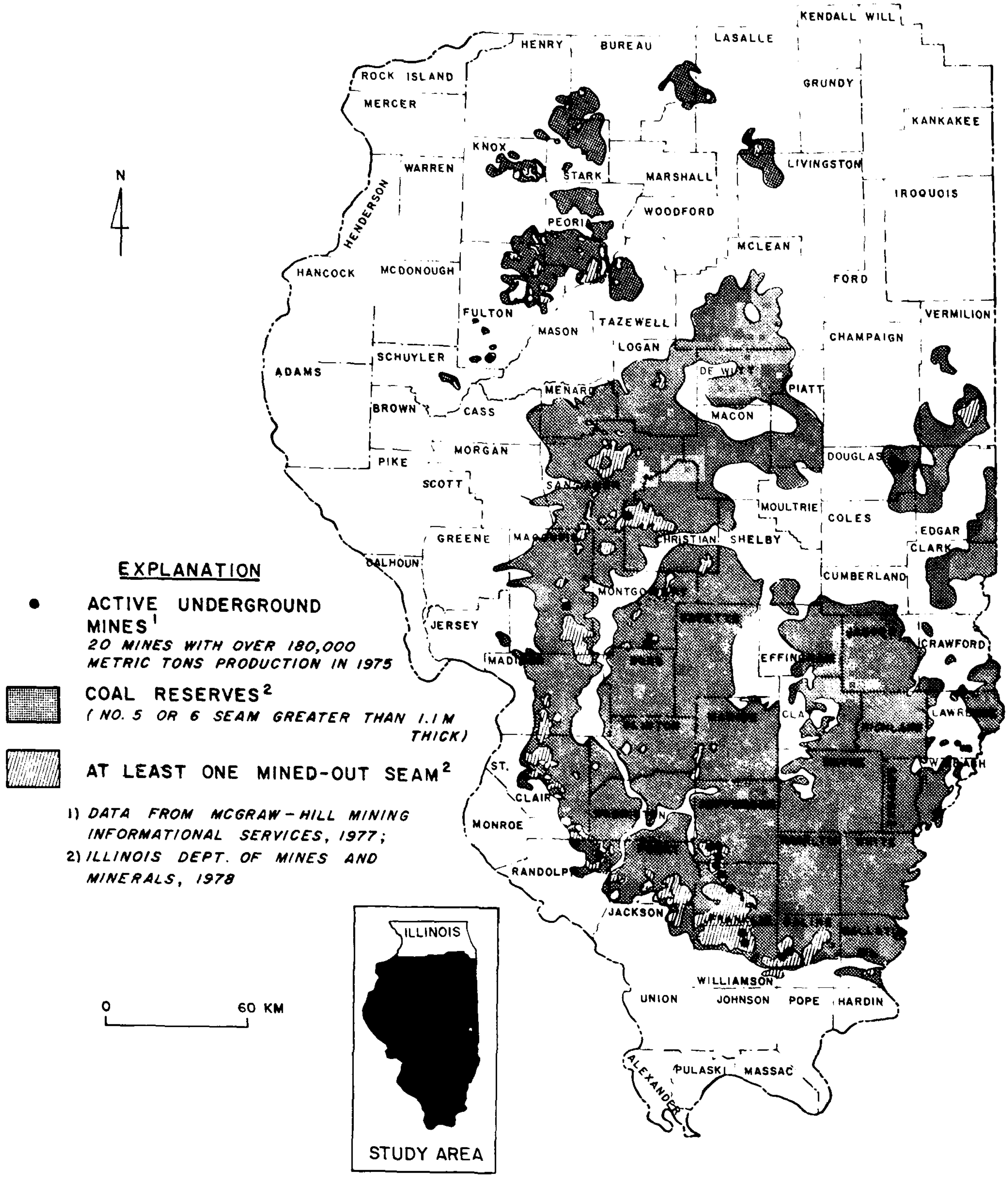

Figure 9. Coal reserves, large active underground mines, and mined-out areas in Illinois.

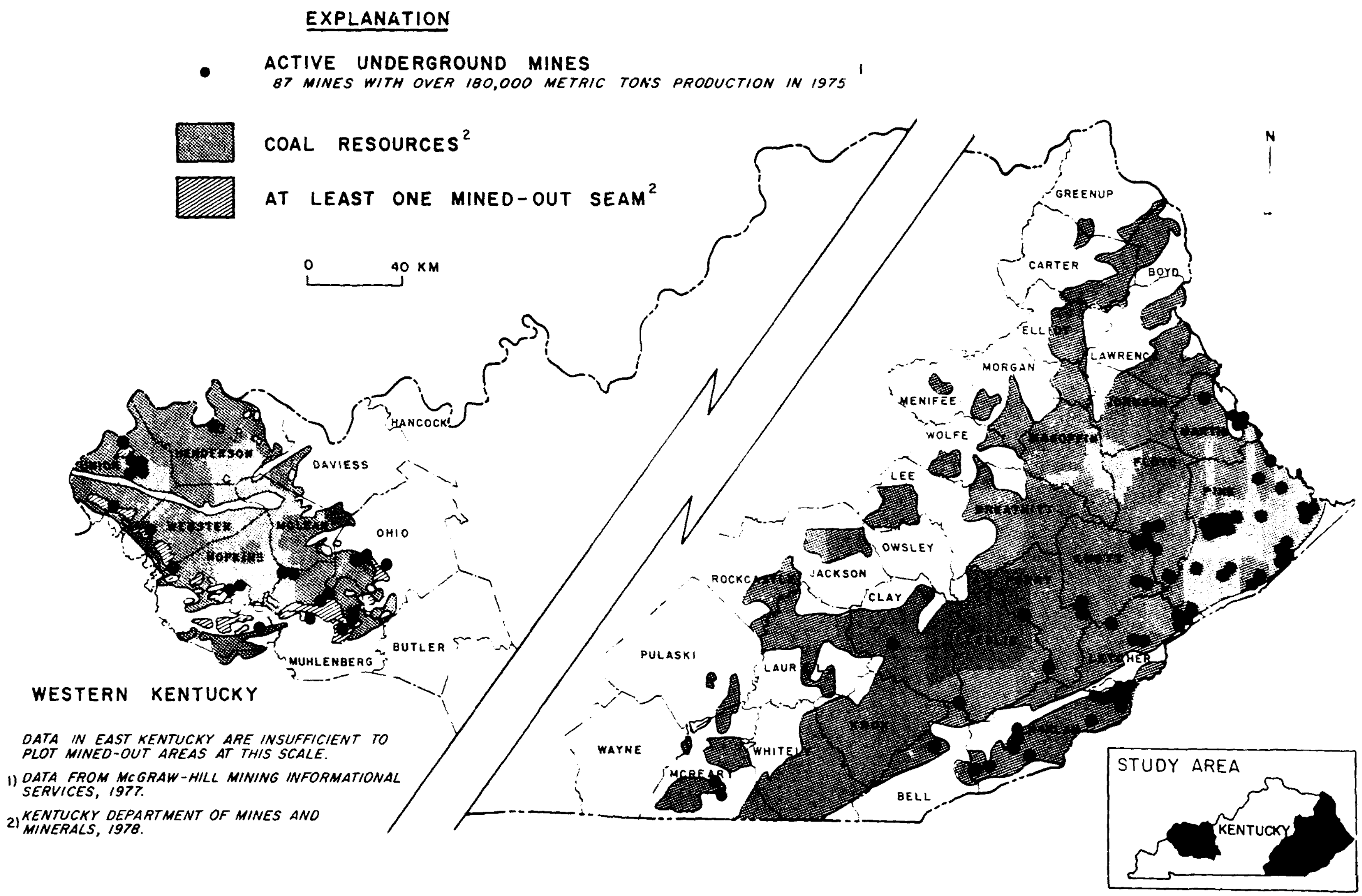

Figure 10. Coal resources, large active underground mines, and mined-out areas in Kentucky.

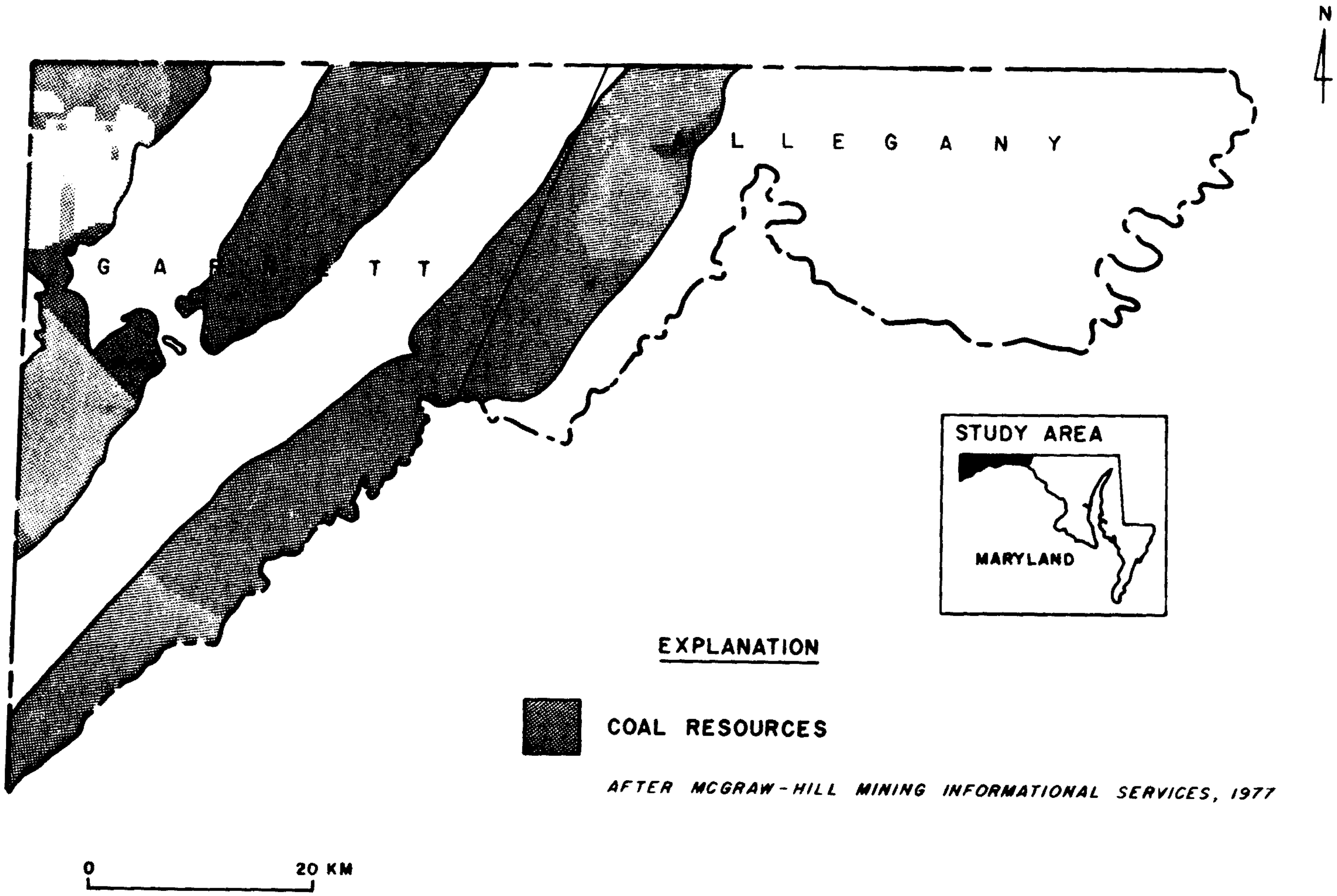

Figure 11. Coal resources in Maryland.

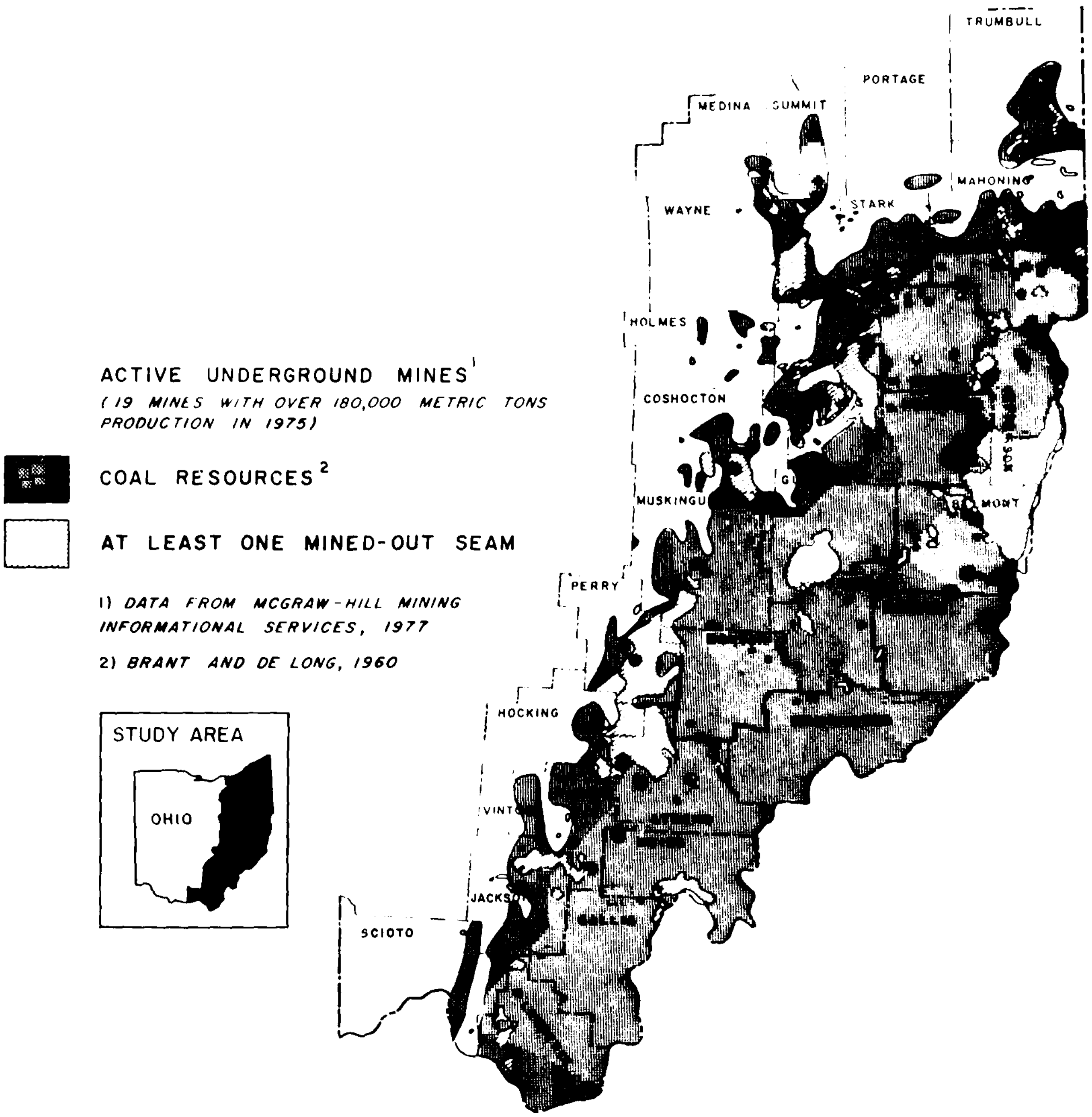

Figure 12. Coal resources, large active underground mines, and mined-out areas in Ohio.

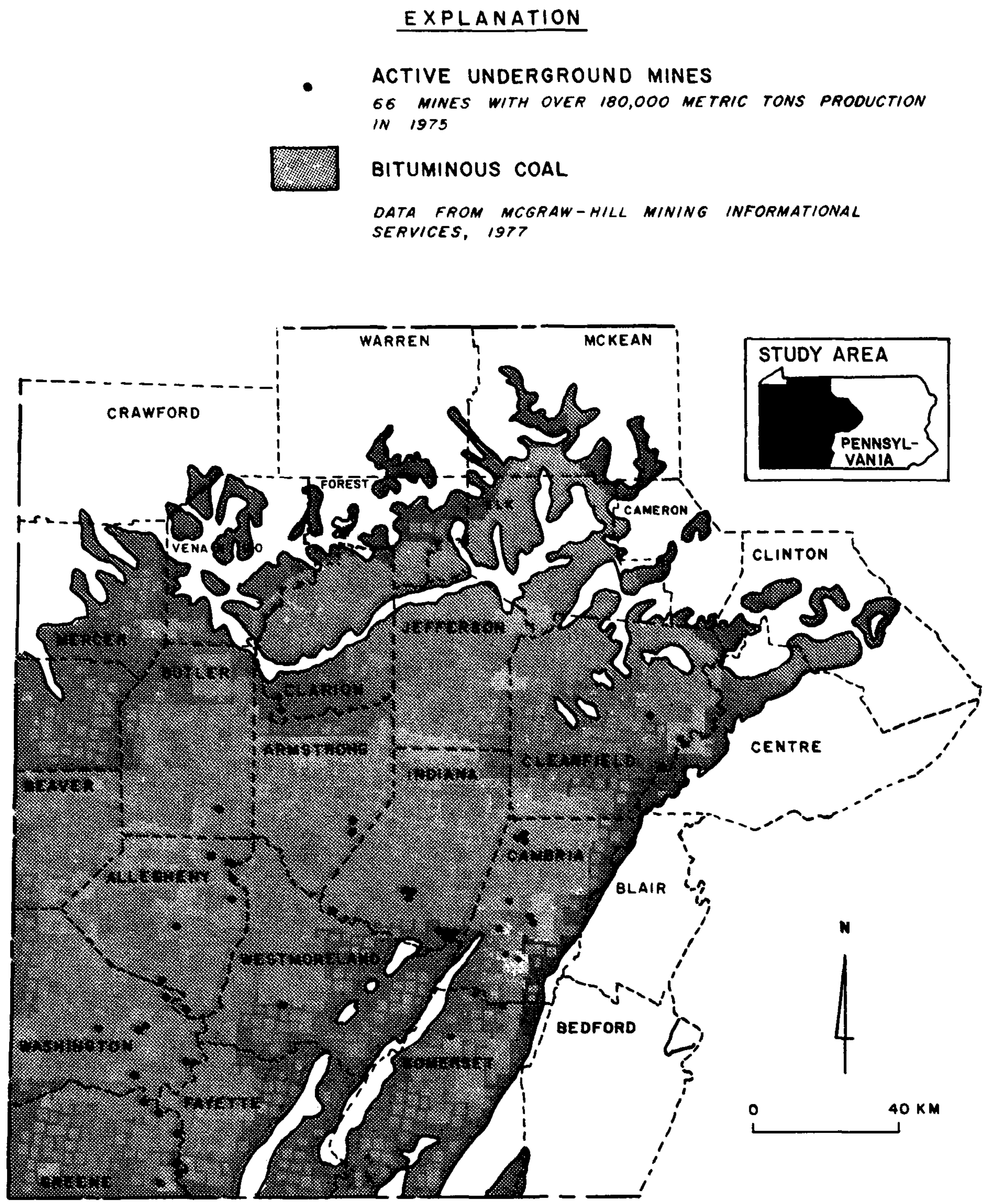

Figure 13. Bituminous coal resources and large active underground mines in western Pennsylvania.

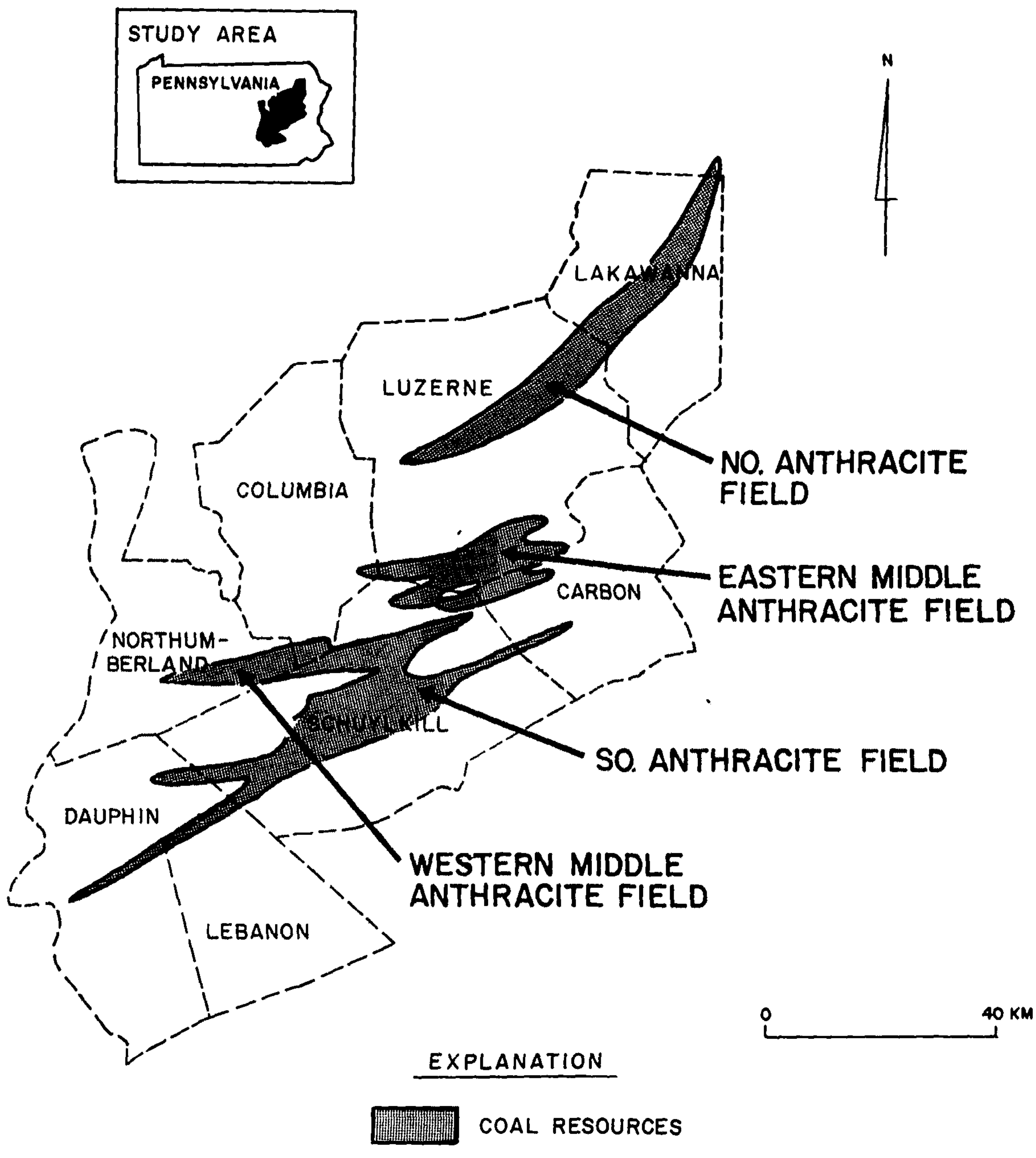

Figure 14. Anthracite coal resources in eastern Pennsylvania.

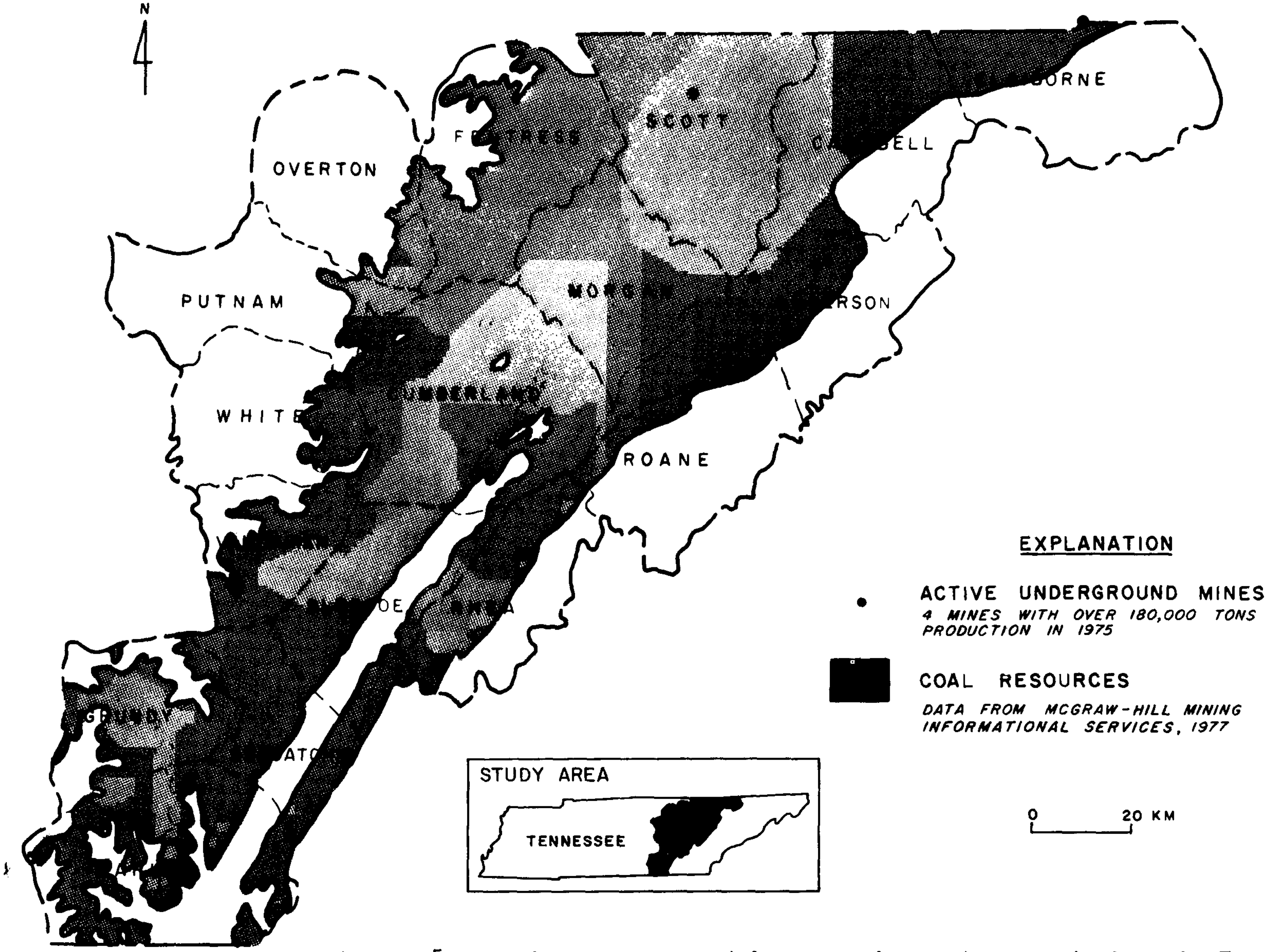

Figure 15. Coal resources and large active underground mines in Tennessee.

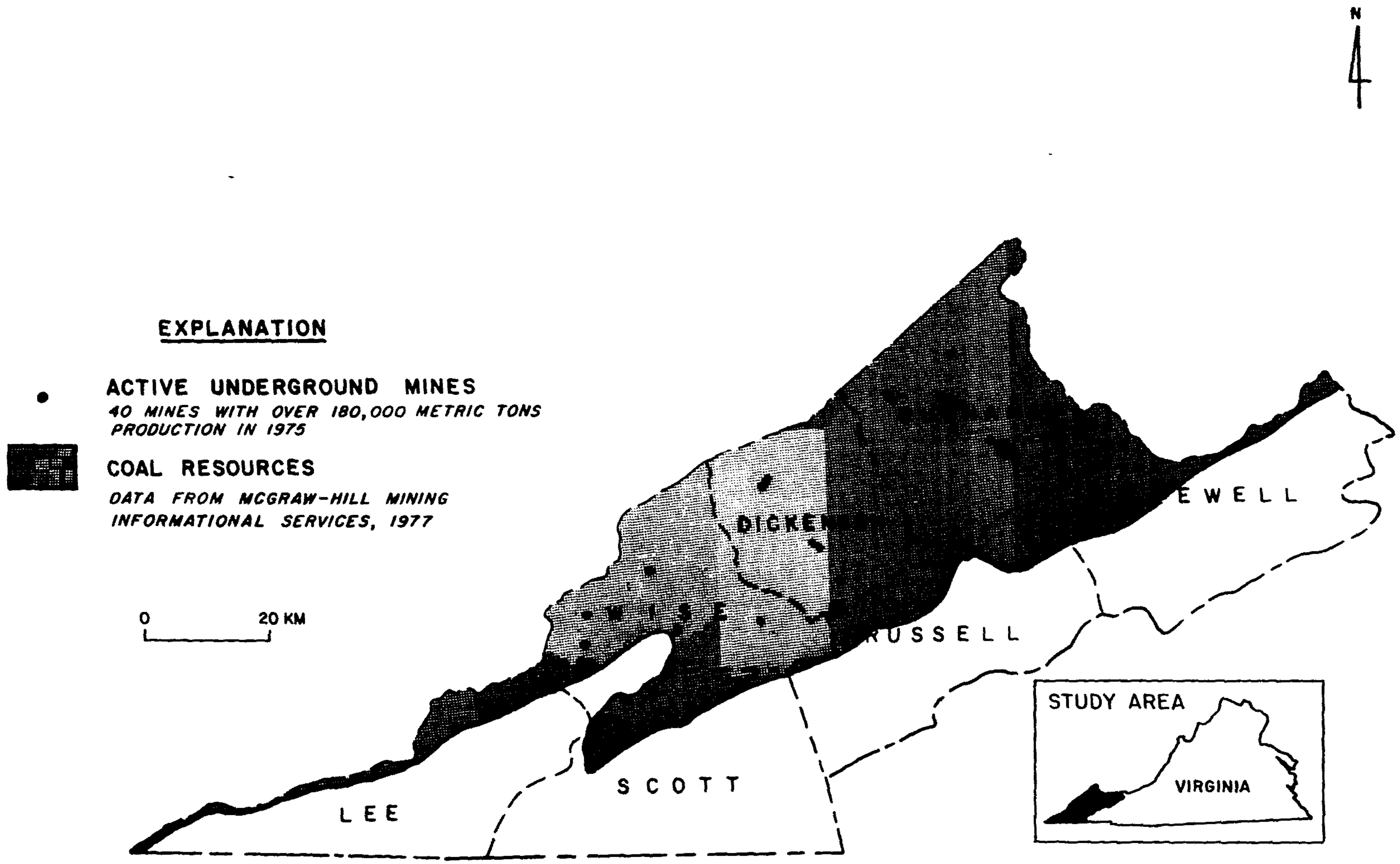

Figure 16. Coal resources and large active underground mines in Virginia.

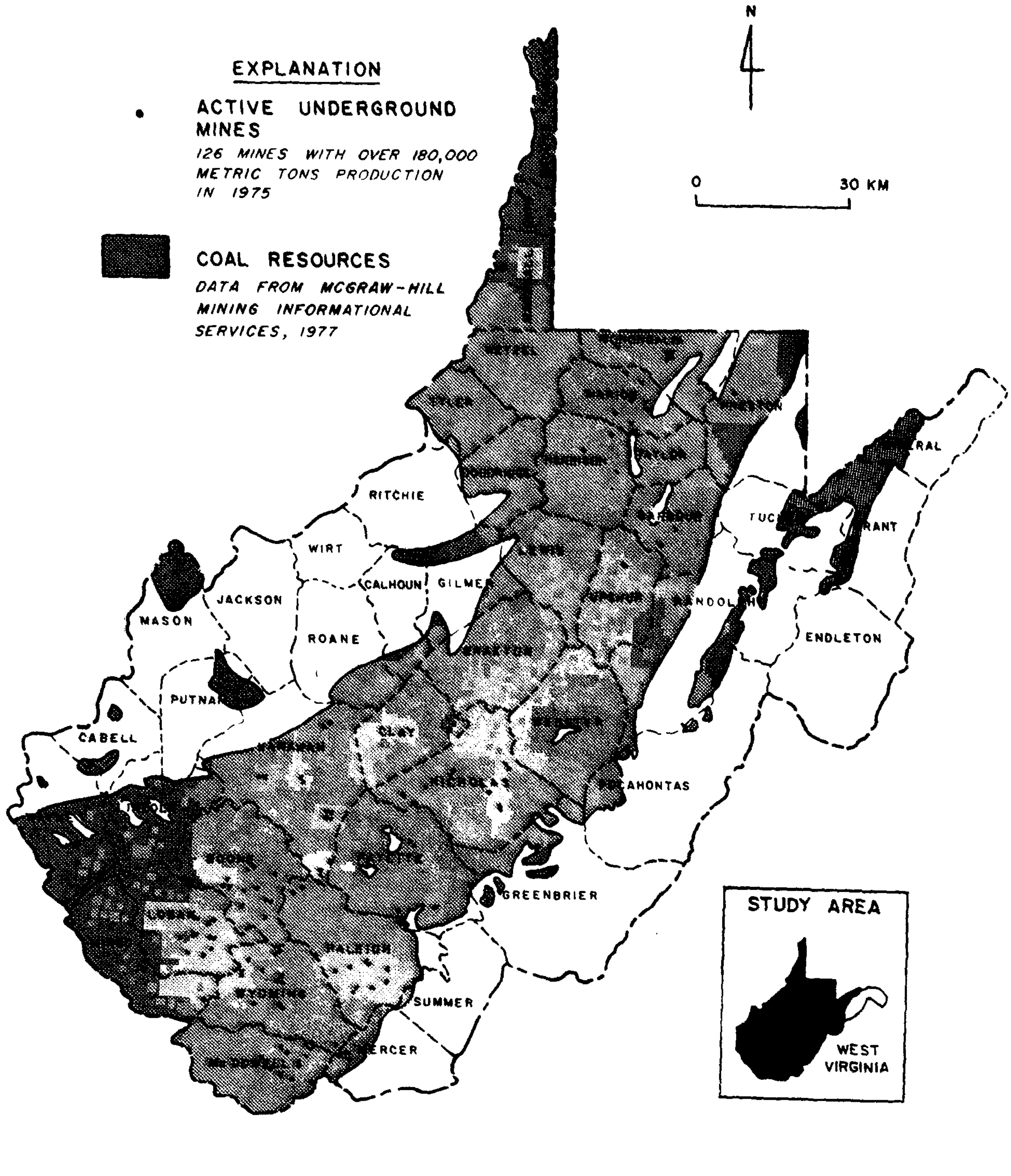

Figure 17. Coal resources and large active underground mines in West Virginia.

apply extraction of all available coal from a seam or particular area; only that some mining has occurred.

Reserves available by underground mining are much greater than reserves available for surface mining in most eastern states. Because of the high costs of mining and compliance with safety and environmental regulations, the number of active small underground mines has decreased in recent years. During the last 30 years, surface coal-mining production has increased relative to underground coal mining in many areas because of the introduction of new high-capacity mining equipment. Surface coal-mining production exceeded underground-mining production in 1975 in Indiana, Ohio, Alabama, Tennessee, and Maryland.

The type of mine entryway employed is largely dependent on the topography of the area and the depth of the coal seam being mined. This can be inferred from the data in Table 2. For example, in states and areas with rugged topography, such as eastern Kentucky, Pennsylvania, West Virginia, and Virginia, drift mines are most abundant. In states and areas with relatively low relief, such as western Kentucky, Alabama, and Illinois, large drift mines (production greater than 180,000 metric tons/yr or 200,000 tons/yr) are not found.

Coal mine depths differ according to the local topography and the structure of the coal seam. Because of these differences, individual mine depths may range from relatively shallow depths below valleys to many hundreds of feet below uplands. The depths of active coal mines in the study area have a wide range. In Illinois, for example, typical depths of active mines are as much as 60 m (196 ft) in St. Clair County, 180 m (590 ft) in southern Illinois, and 300 m (984 ft) in Christian County. In general, active mines are being extended to greater depths because most of the easily reached shallow coal deposits have been extracted. Depths of 200 to 250 m (656 to 820 ft) are common in active mines with slope or shaft entryways. Typical maximum depths of underground coal mines are: 600 m (1,968 ft) in Alabama, 400 m (1,312 ft) in Virginia and eastern Kentucky, and 120 m (394 ft) in Ohio.

The relative density of mining in states where data were available is shown in Figures 8 through 24. Greater densities suggest places where ground-water effects associated with past mining are most likely to occur. Maps showing areas underlain by mined-out seams were used where available. However, this type of data was usually not available on a regional basis. Consequently, figures showing the number of abandoned underground mines in each 7.5-minute U. S. Geological Survey topographic quadrangle have been used. These numbers have not been weighted according to the size of the mine or the mined-out acreage since data of this sort are generally not available on a regional or area-wide basis.

In order to determine whether the number of abandoned mines per quadrangle is a reasonably good indicator of the actual extent of the mined-out areas, two methods of estimating the extent of mined-out areas were applied to data for Pennsylvania's bituminous region. Figure 18

TABLE 2. TYPES OF MINE ENTRYWAYS USED IN LARGE UNDERGROUND COAL MINES, 1975*
(Source: Data from McGraw-Hill Mining Informational Services, 1977)

State	Type of mine entryway				Number of mines
	Shaft	Slope	Shaft and slope†	Drift	
Alabama	1	6	2	0	9
Illinois	7	8	3	0	18
Kentucky	2	5	0	57	64
Ohio	1	10	2	2	15
Pennsylvania	12	13	7	28	60
Tennessee	1	0	0	2	3
Virginia	5	2	0	33	40
West Virginia	16	29	5	111	161
				TOTAL:	370

* Mines producing more than 180,000 metric tons/yr (200,000 tons/yr).

† Both shaft and slope entryways used in a single mine.

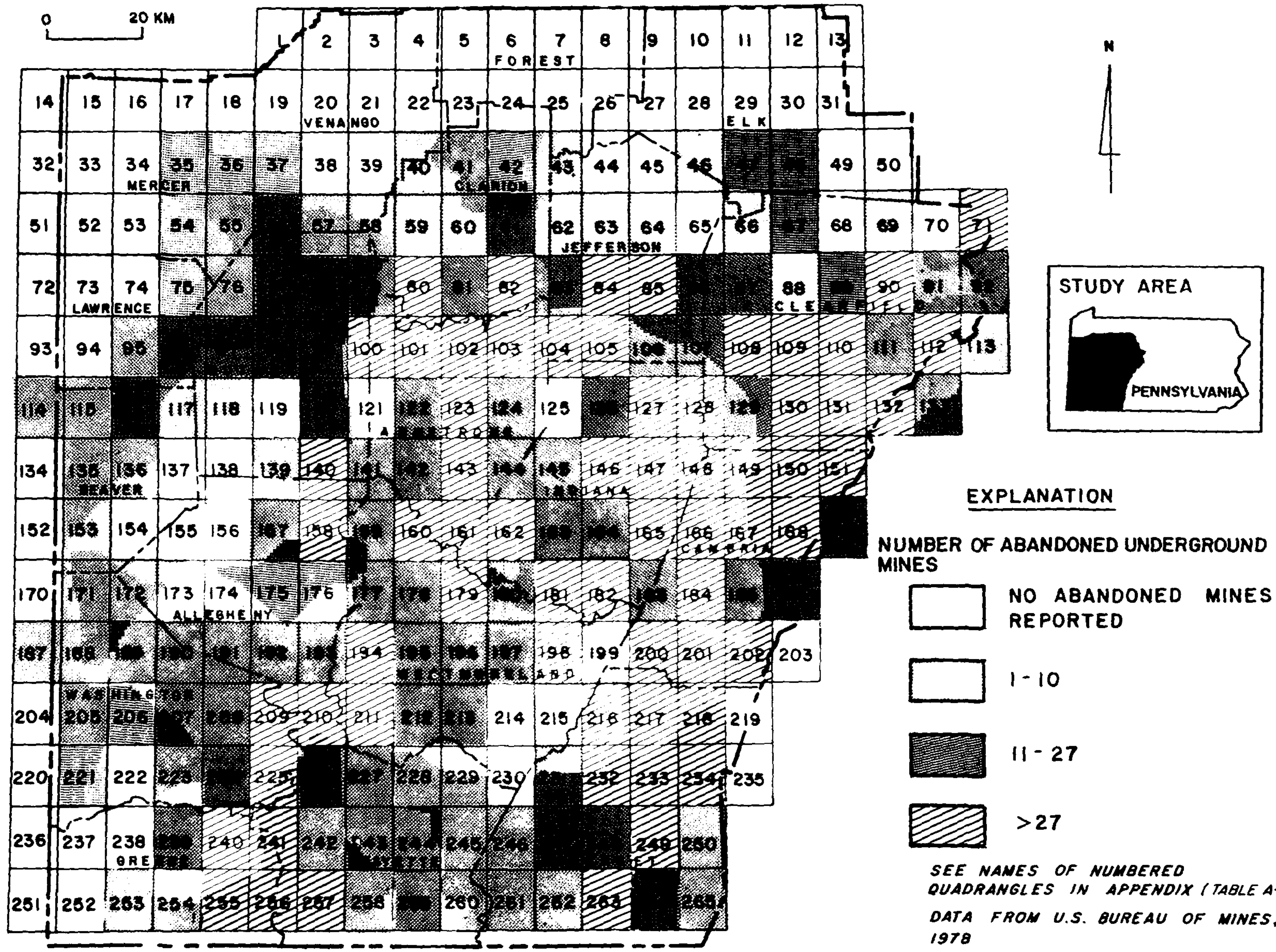

Figure 18. Density of abandoned underground coal mines in western Pennsylvania.

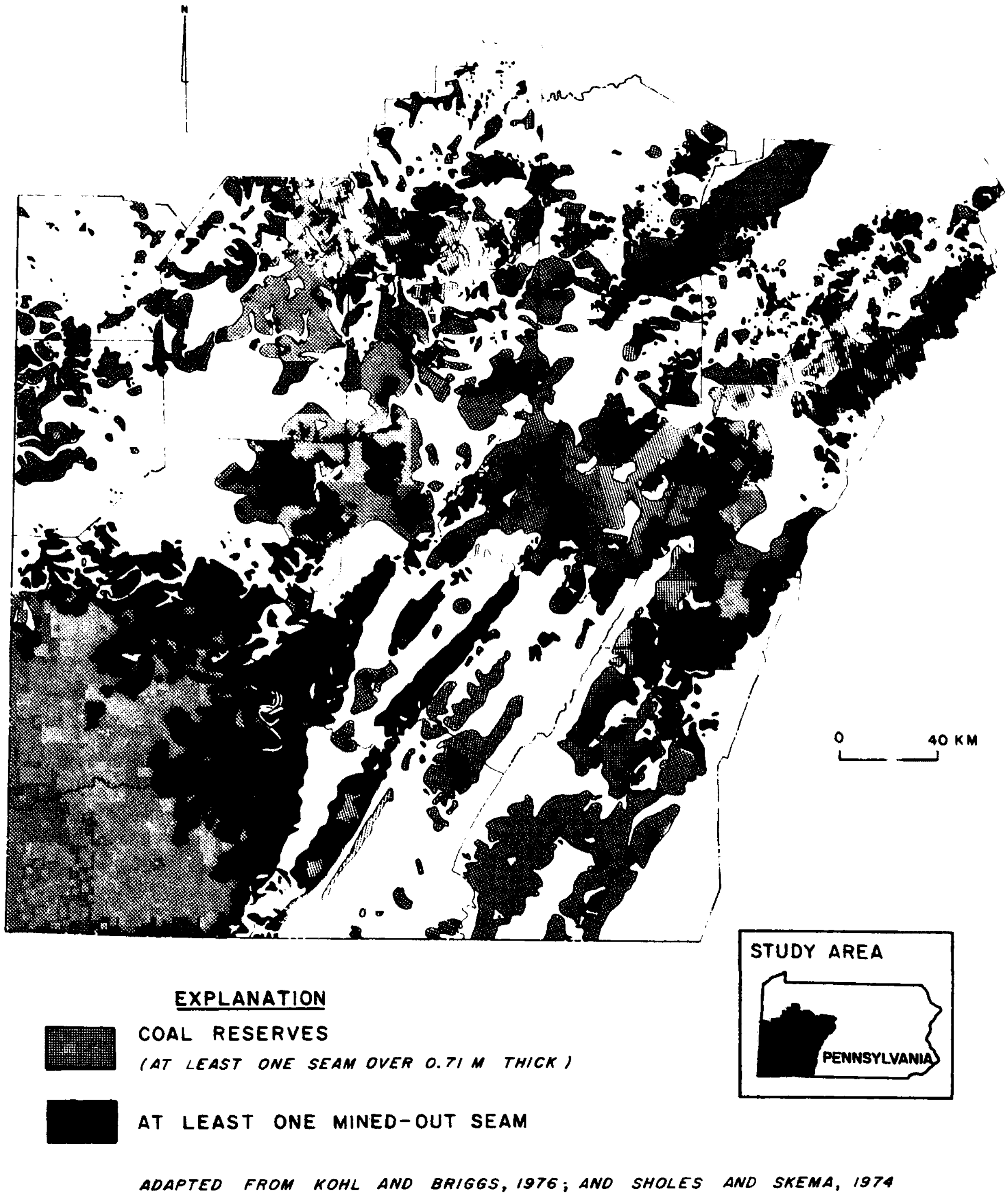

Figure 19. Coal reserves and mined-out areas in western Pennsylvania.

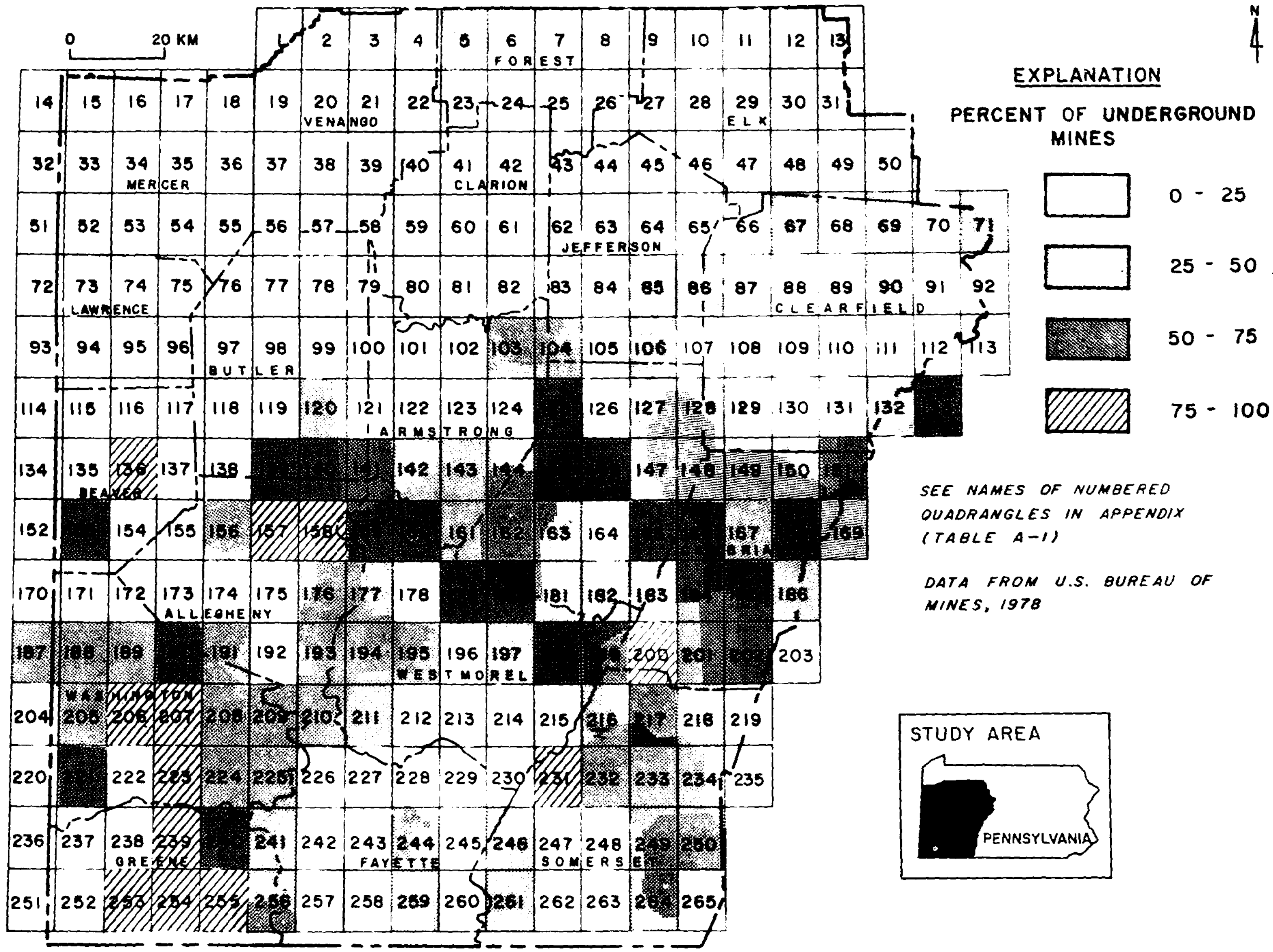

Figure 20. Number of underground coal mines as a percentage of total mines in western Pennsylvania.

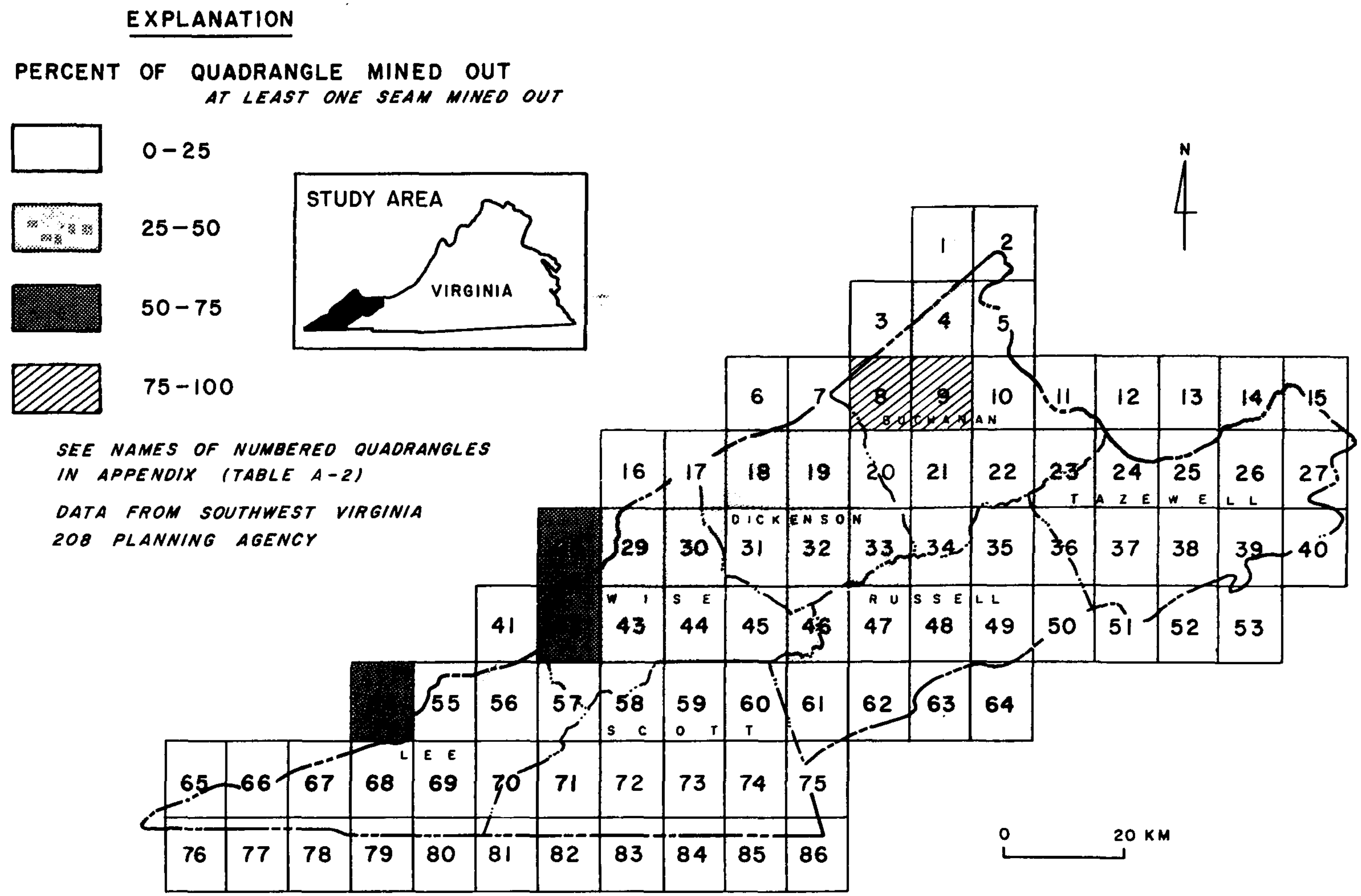

Figure 21. Percent of land surface underlain by mined-out seams in Virginia.

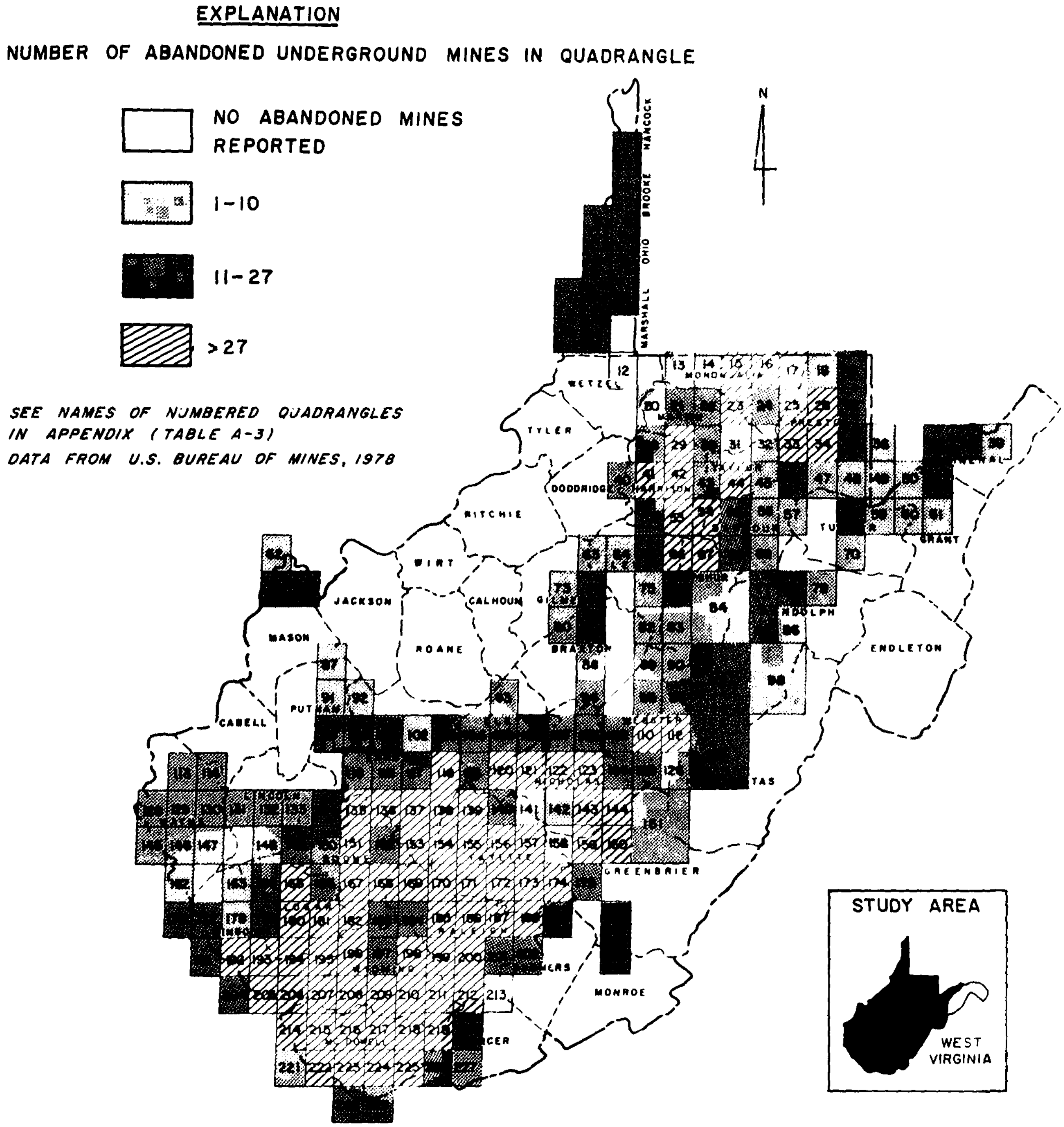

Figure 22. Density of abandoned underground coal mines in West Virginia.

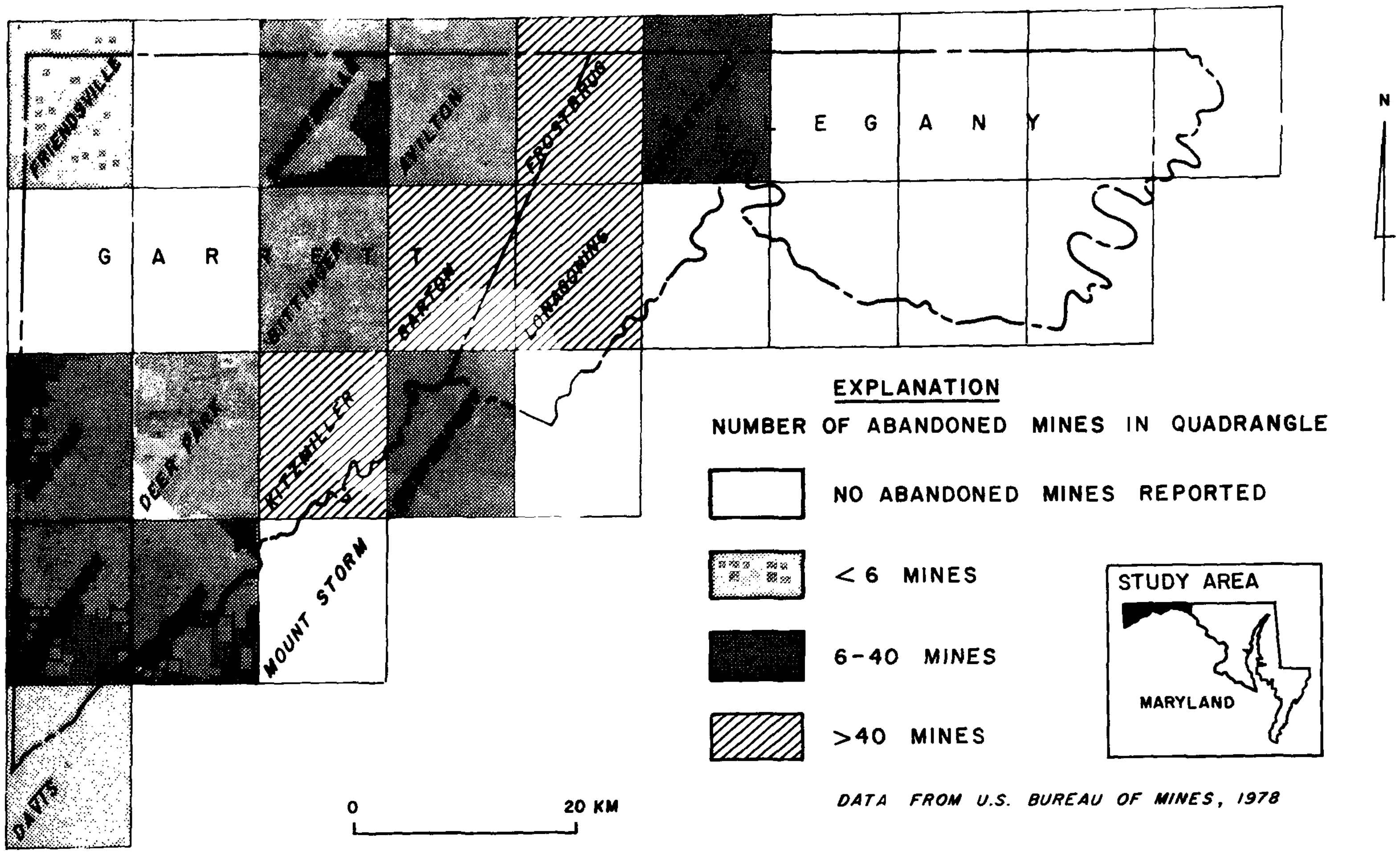

Figure 23. Density of abandoned underground coal mines in Maryland.

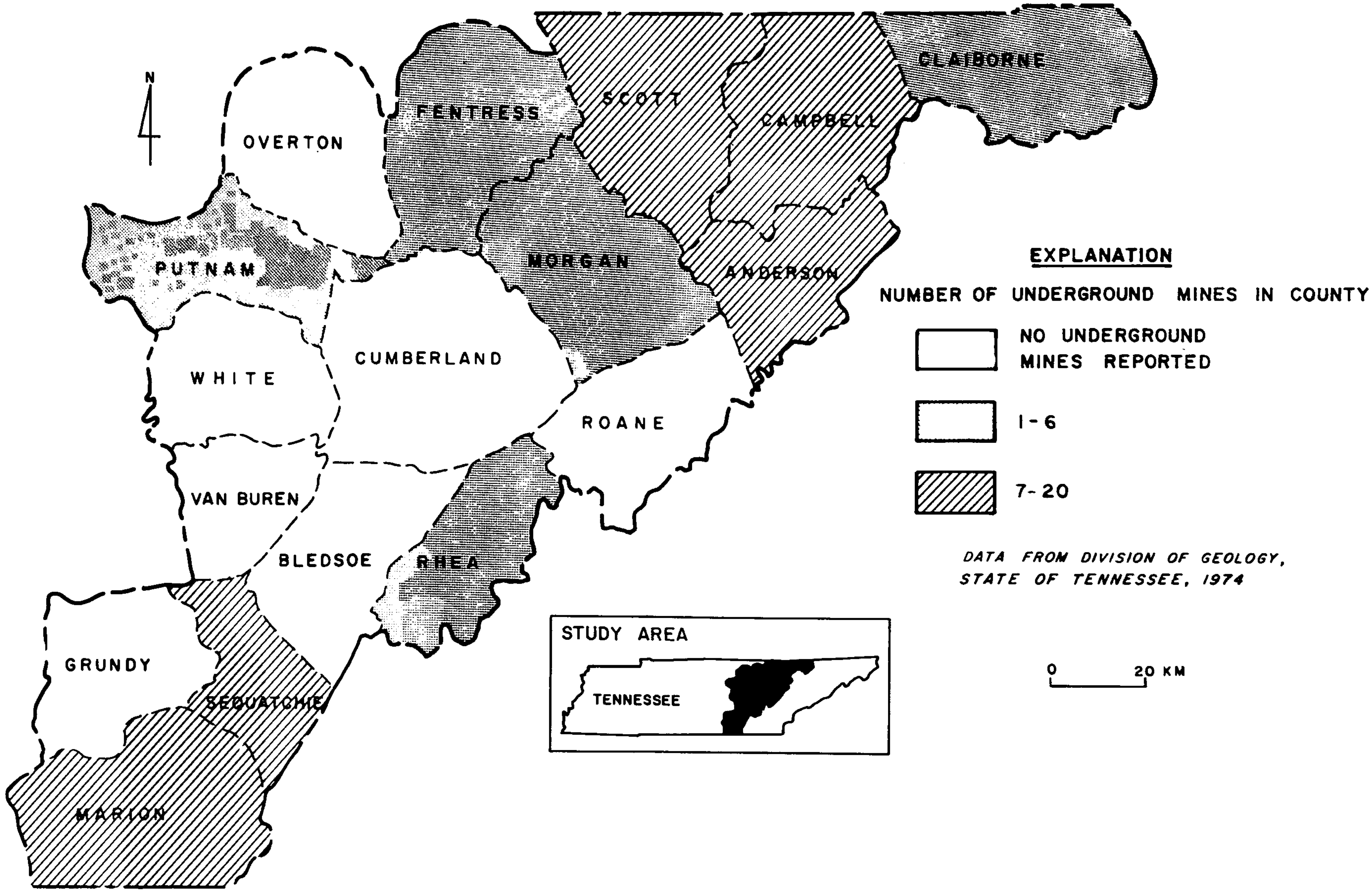

Figure 24. Density of underground mines in Tennessee.

shows the rather wide distribution of areas with large numbers of abandoned underground mines, whereas Figure 19 shows a somewhat more restricted distribution of mined-out areas (primarily of the Pittsburgh Coal) in Allegheny, Westmoreland, Fayette, and eastern Washington Counties. Approximately 30 to 35 percent of the land area underlain by coal reserves in western Pennsylvania has at least one mined-out seam. The percentage of underground coal mines to the total number of coal mines in Pennsylvania is shown by patterns in Figure 20. Counties with quadrangles where underground mines represent over 75 percent of the total number of reported mines include Washington, Westmoreland, Fayette, Beaver, Allegheny, Somerset, and Cambria.

The amount of data differs from state to state. The most complete data on abandoned mines or mined-out areas were obtained for Ohio, Illinois, Virginia, and Pennsylvania's bituminous coal region. The information for eastern Kentucky, Alabama, and Pennsylvania's anthracite region was most incomplete. Pennsylvania's anthracite region was heavily mined by underground methods in the past, but underground mining is now much less intensive, and production is predominantly from surface mines or from refuse bank recovery. For this area, the data were on too small a scale to compile regional maps of the mined-out areas.

Figure 21 is a map of Virginia showing the percent of surface area that is underlain by at least one mined-out coal seam. The number of underground mines in two quadrangles in Buchanan County represents over 75 percent of the total number of reported mines. Mapping of mined-out areas in West Virginia is currently in progress. However, the data were not considered to be sufficiently tabulated for use in this report. Therefore, estimates of the number of abandoned mines by quadrangle were used for the West Virginia assessment (Figure 22).

Underground coal mining is of small extent in Maryland and Tennessee. The number and distribution of abandoned underground mines, by county and/or quadrangle, were obtained from state publications and from lists of abandoned mines maintained by the U. S. Bureau of Mines (Figures 23 and 24). Underground mining in Indiana is of small extent and is not included as a map in this section.

SECTION 4

GROUND-WATER AVAILABILITY

GENERAL CONCEPTS

The ultimate value of a ground-water resource is a combination of both its availability and the need for the water. Mining and associated waste production in areas where aquifers are productive and well yields are high are likely to affect a greater quantity of useable ground water than in an area with low ground-water availability. Physical factors, such as aquifer permeability and thickness, presence of joints and fractures, and topography affect ground-water availability as well as the movement of contaminants in the system.

For purposes of this report, generalized ground-water availability maps have been prepared for those portions of each state where rocks of Pennsylvanian age are at the land surface. In addition, a very brief description of background information for each state is included. The maps characterize availability of ground water at or near the surface. Any inferences regarding effects on ground water from mining relate primarily to shallow underground mining or to coal wastes placed on the land surface. Effects on ground water from deeper mining must be treated on a smaller scale and in three dimensions, as described in the section entitled Hydrologic Effects of Mining. The maps and well-yield figures in this section are highly generalized and should be used only to evaluate the general potential for contamination of the aquifers in coal-bearing areas and not as a guide for ground-water development or site evaluation.

In many areas, unconsolidated aquifers are more important than bedrock aquifers as sources of ground water. This results from the fact that where unconsolidated aquifers are sand and gravel they have much higher storage capacities and permeabilities than bedrock aquifers. Such deposits are typical of thick, well sorted alluvial sediments, and glacial outwash. Where unconsolidated deposits have poor water-bearing characteristics because they are thin or have low permeability, bedrock aquifers are the principal sources of ground water.

Reported well yields from bedrock aquifers of the Pennsylvanian period are generally low to moderate. Yields are usually adequate for domestic purposes and, in some cases, for small public and industrial supplies. Ground-water availability from these rocks is particularly poor in Illinois, Kentucky, Tennessee, and much of Ohio, Virginia, and

Alabama. Bedrock which will potentially yield moderate amounts of water (>1.6 l/s or 25 gpm) underlies a large portion of Pennsylvania and West Virginia. Moderate to high yields are found in certain alluvial deposits in Illinois, western Kentucky, Ohio, and West Virginia. Yields are also high in outwash deposits of northwestern Pennsylvania.

LOCAL HYDROGEOLOGIC CONTROLS

The maps in this section show potential regional ground-water availability differentiated mainly on the basis of rock type. Although this is an important factor, several others are also important on a local basis. These include topographic position, fracture patterns, structure, and the effects of glaciation. Well yields in the same aquifer can differ widely according to topographic position. Studies have shown that aquifers beneath valleys commonly yield much more water than those beneath uplands. Among the reasons for this difference is the fact that rocks below and adjacent to valleys commonly have a greater number of joints and fractures than elsewhere. Moreover, the depth to the water table beneath valleys is generally much less than that beneath uplands.

The thickness and porosity of an aquifer determine its storage capacity. Thick, coarse-grained alluvial deposits, such as those along the Ohio River, generally have large storage capacities and can supply large amounts of water to wells. In contrast, thin or fine-grained alluvial deposits, such as those along most small to moderate size streams in the Appalachian Plateau, have relatively low water-yielding capacities. For instance, in Harrison County, West Virginia, alluvial deposits are minor sources of ground water owing to their thinness, and yields of only 0.13 to 0.19 l/s (2 to 3 gpm) can be obtained (Nace and Bieber, 1958). In contrast, yields of up to 50 l/s (800 gpm) are obtained from wells tapping thick alluvial deposits in Ohio County, West Virginia (Robison, 1964).

Local concentrations of fractures in a bedrock aquifer greatly increase its permeability and, therefore, the potential yields of wells. These fracture zones also facilitate vertical leakage and probably are the primary paths for movement of contaminated water to underlying aquifers (Parizek, 1978).

The effects of glaciation on ground-water availability are most evident in the northern part of the Appalachian Basin. In parts of Illinois and northern Pennsylvania, wells in permeable glacial outwash deposits have high yields. Over most of southern Illinois, however, the land surface is covered by low-permeability glacial till, and the underlying bedrock is commonly a relatively poor producer of water.

AVAILABILITY IN SELECTED STATES

Alabama

Ground-water data for the coal-mining area in Alabama were obtained from river-basin reports, county reports, and one Alabama Geological Survey report. Average yields of aquifers were used in the preparation of Figure 25. In the northern part of the state, where the bituminous reserves are located, ground-water availability is generally rather low compared to that in other states such as Pennsylvania and West Virginia. For convenience in showing the relative yields of the aquifers in the coal area, the yield categories in Figure 25 were set at intervals of less than 2 l/s, 2 to 4.4 l/s, and more than 4.4 l/s (less than 30, 30 to 70, and more than 70 gpm).

Illinois

Figure 26 is based on data obtained from state water-resources reports. Ground-water availability in Illinois is closely related to the nature of glacial or alluvial deposits in a given area. The highest yields are from wells tapping sand and gravel deposits and the lowest yields are from bedrock aquifers that are overlain by low permeability till deposits.

According to Smith and Stall (1975), ground-water availability is highest in the sand and gravel deposits of the Ohio, Wabash, Mississippi, Illinois, and buried Mahomet River valleys. Potential yields of wells tapping these deposits are as much as 31 l/s (500 gpm). Sand and gravel deposits in the Embarras and Kaskaskia River valleys have moderate ground-water availability; potential well yields range from 6.3 to 31 l/s (100 to 500 gpm).

The remainder of the coal region is underlain by bedrock aquifers, where yields of wells are generally less than 1.6 l/s (25 gpm), but occasionally may be as much as 6.3 l/s (100 gpm) (Illinois State Water Survey Division, 1967).

Indiana

Ground water is available from both consolidated rocks and unconsolidated deposits in Indiana (Figure 27). The unconsolidated sands and gravels, present as both Recent alluvium and Pleistocene glacial outwash, are far more productive than the consolidated Mississippian and Pennsylvanian rocks. Typical yields of sand and gravel deposits in the northern coal counties range from 0.06 to 76 l/s (1 to 1,200 gpm) and average about 22 l/s (350 gpm). Well depths range from 4.6 to 70 m (15 to 230 ft) below land surface and are commonly 20 to 27 m (65 to 90 ft). Yields up to 170 l/s (2,700 gpm) are obtainable from sand and gravel along the Wabash River, yields of 6.3 l/s (100 gpm) to over 31.5 l/s (500 gpm) are possible along the White River, and in the south, deposits along the Ohio River yield from 3.2 l/s (50 gpm) to over 63 l/s

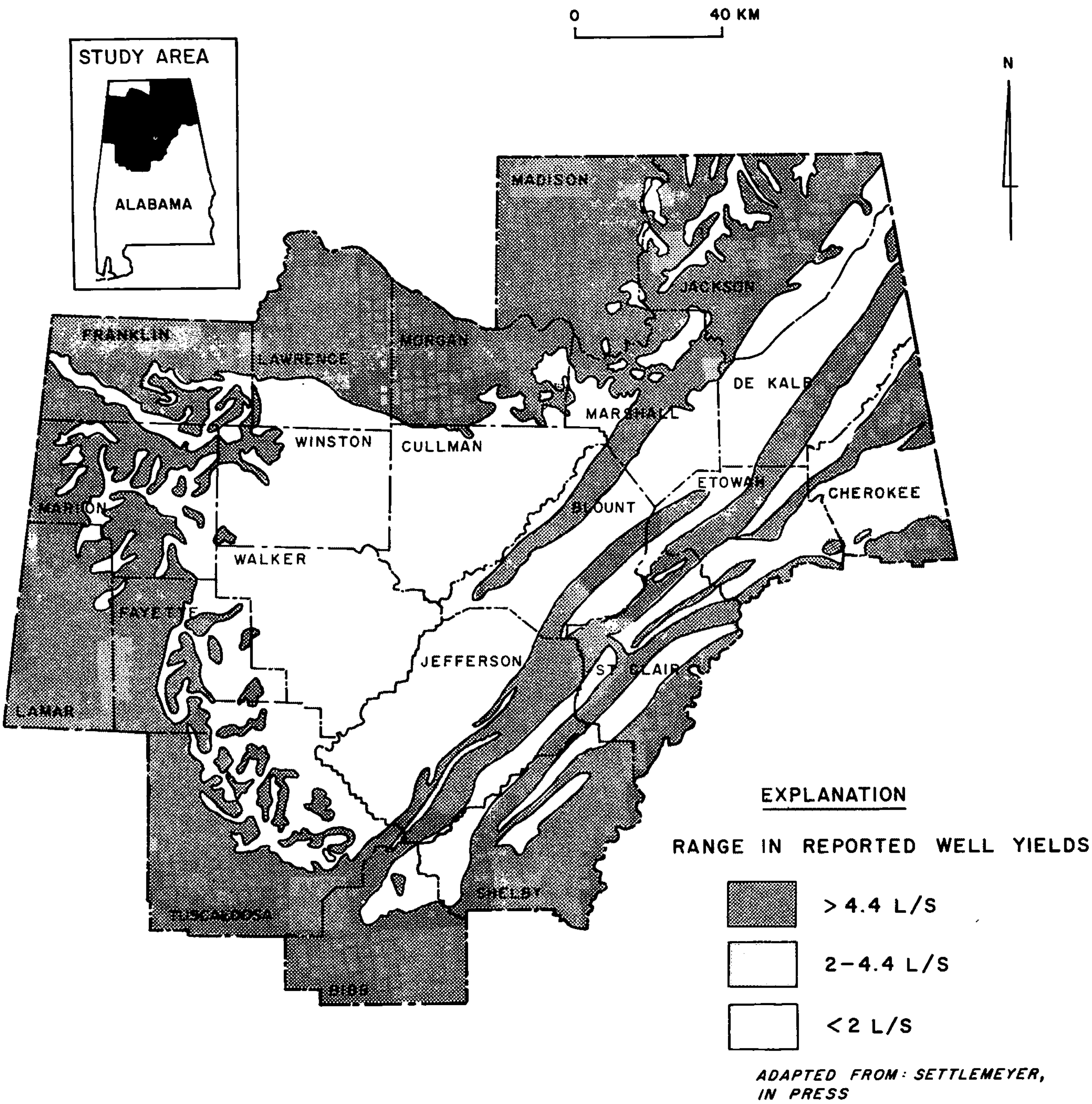

Figure 25. Potential ground-water availability in the coal-bearing region of Alabama.

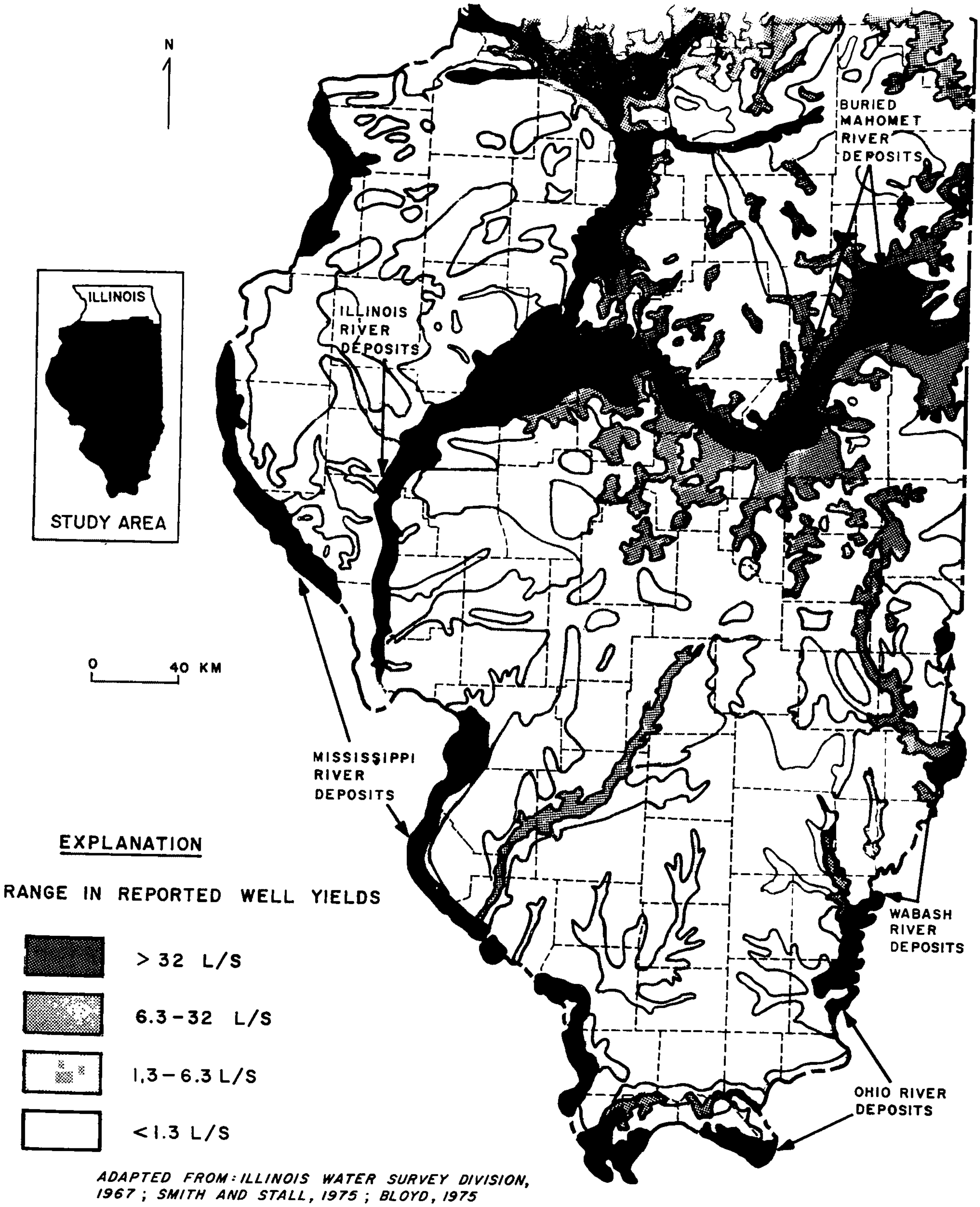

Figure 26. Potential ground-water availability in the coal-bearing region of Illinois.

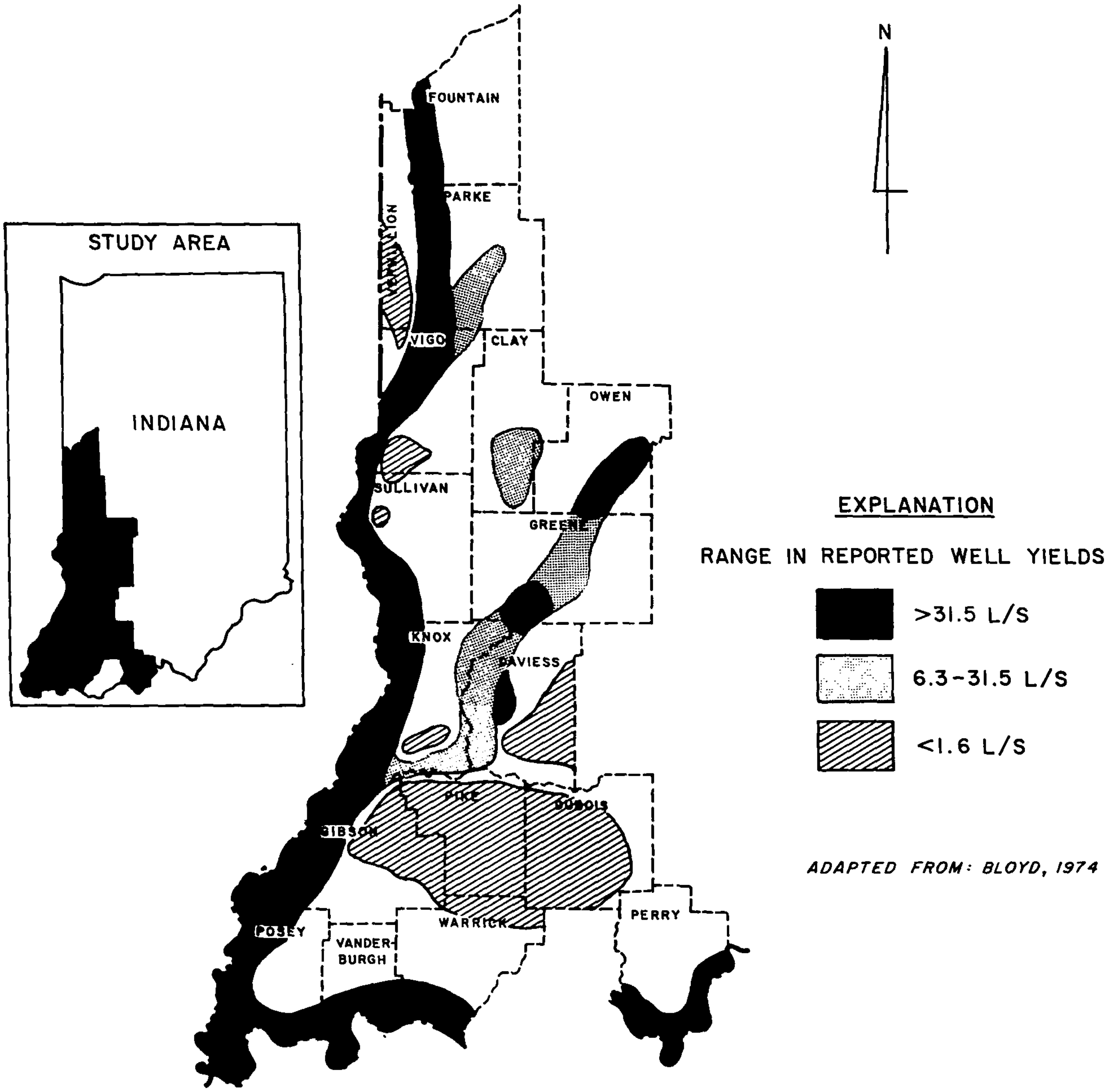

Figure 27. Potential ground-water availability in the coal-bearing region of Indiana.

(1,000 gpm). Generally, small industrial and municipal supplies are possible from these deposits while large supplies are available in Sullivan and Greene Counties.

Pennsylvanian rocks are the second major source of ground water in Indiana coal counties, but yields are suitable only for domestic and small industrial or municipal supplies. These rocks are composed of sandstone and shale with minor beds of coal, limestone, and fire clay occurring in cycles as cyclothems. The basal sandstone in each cyclothem is the principal water bearer. Mississipian rocks are shallow enough to be tapped for water only in the northern coal counties. They are composed of sandstone, shale, siltstone, and limestone. All the rock types are water-bearing to some degree, but thick-bedded limestone and sandstone are the main sources for domestic and stock supplies.

Kentucky

Information on ground water was available for both the eastern and western coal fields, mainly from county reports. Figure 28 was prepared using average anticipated well yields. Ground-water availability differed significantly from east to west; consequently, the two coal regions are described separately.

Ground-water availability in the eastern coal field region is low to moderate. Yields of wells in bedrock aquifers are generally less than 0.31 l/s (5 gpm) and are barely suitable for domestic use. The greatest potential well yields, greater than 1.6 l/s (25 gpm), occur in alluvial aquifers, particularly along the Big Sandy River and Levisa Fork.

Ground-water availability in the western coal field region is generally higher than that in the eastern region. Yields of wells in the alluvial aquifers in the Ohio River valley are usually greater than 6.3 l/s (100 gpm) and, in some places, may be as high as 63 l/s (1,000 gpm). Yields of wells in alluvial aquifers in other river valleys such as the Green and Tradewater may be as much as 6.3 l/s (100 gpm). In contrast, maximum potential yields of wells in bedrock aquifers are generally less than 0.31 l/s (5 gpm) and, therefore, these aquifers are unsuitable for most supply purposes.

Maryland

Well yields in the coal counties of northwestern Maryland vary to a certain degree among the different coal fields. In general, the Pottsville Formation is a fairly good aquifer; yields of wells range from about 1.3 l/s (20 gpm) to 12.6 l/s (200 gpm). Wells tapping the Allegheny and Conemaugh Formation usually yield less than 1.3 l/s (20 gpm), and are adequate mainly for domestic use. Since the Allegheny and Pottsville Formations are not distinguished in geologic maps of this region, no figure of ground-water availability is included for Maryland.

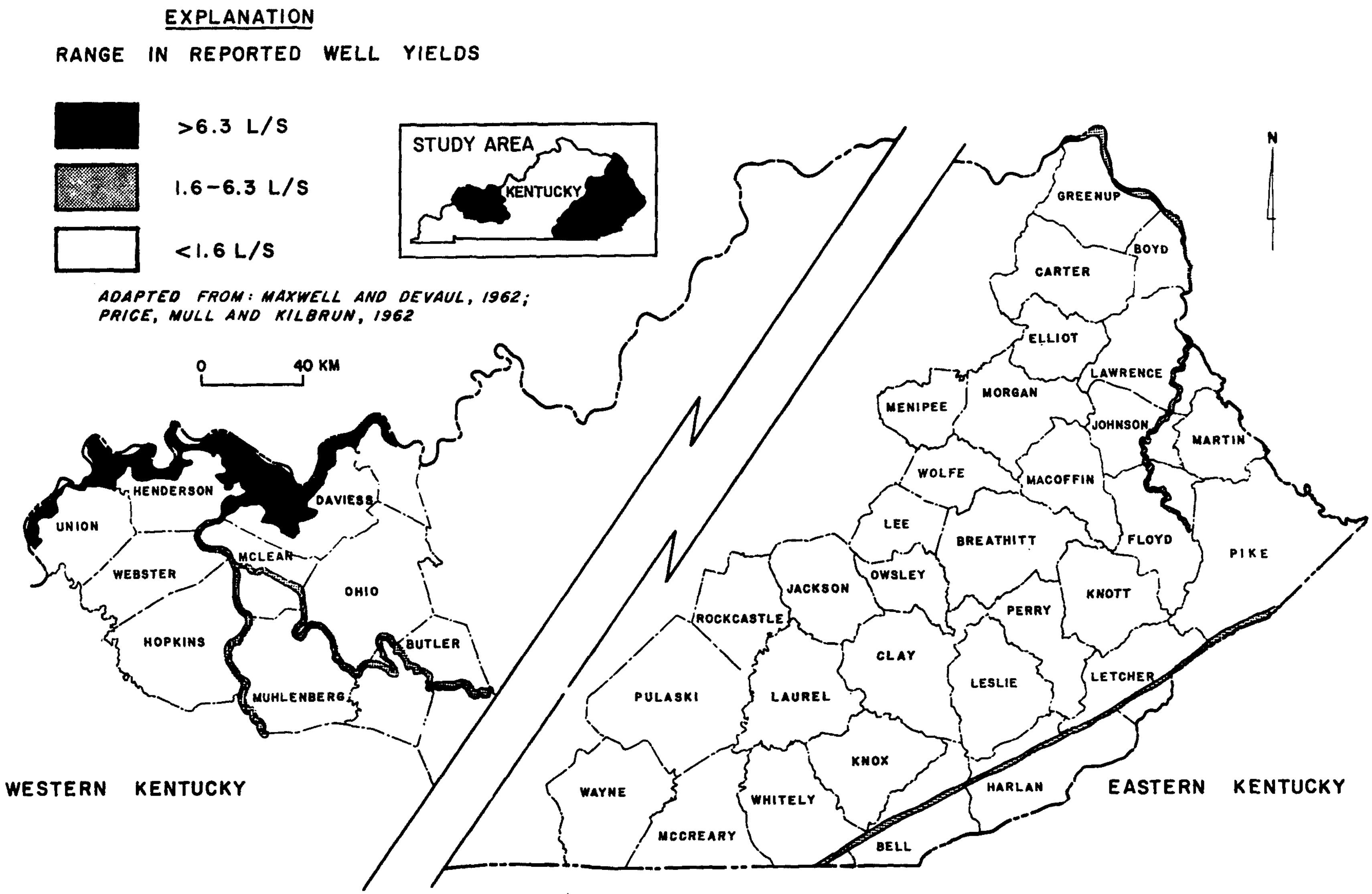

Figure 28. Potential ground-water availability in the coal-bearing region of Kentucky.

Ohio

The Pennsylvanian age rocks of Ohio generally yield only small amounts of ground water to wells (Figure 29). These yields are, in some cases, barely suitable for domestic supplies (less than 0.31 l/s). Slightly higher average yields are obtained from older rocks to the northwest. Moderate to large yields suitable for public supplies are obtainable from certain alluvial deposits, especially those of the Ohio and Muskingum Rivers.

Pennsylvania - Bituminous Region

Ground-water data on the bituminous coal region of Pennsylvania are abundant. The types of data available range from county studies to studies of multi-county regions. The yields used to define the categories of ground-water availability shown in Figure 30 are based on reported average yields given in published reports. The areas of highest yield shown on Figure 30 represent areas underlain by permeable outwash in the northwest and alluvial deposits elsewhere. The boundary between low and moderate yields of these aquifers is 1.6 l/s (25 gpm). Yields above this level are suitable for development of small industrial or municipal supply wells. The poorest yields from bedrock and alluvial deposits are associated with rocks of the Monongahela, Washington, and Greene Formations in the southwest corner of the state.

Pennsylvania - Anthracite Region

Reported well yields from rocks in and around the four anthracite fields were evaluated in preparing Figure 31. The complex structure of the rocks affects the yields of wells tapping different aquifers in the northern and southern anthracite fields. Yields are highest in glacial outwash deposits. In defining yield categories, the average yields are divided into intervals of less than 3.2, 3.2 to 16, and more than 16 l/s (less than 50, 50 to 250, and more than 250 gpm, respectively).

Tennessee

The availability of ground water in the coal-bearing region of central Tennessee is generally very low (Figure 32). The yields of most of the wells and springs are suitable only for domestic supplies. A few wells with somewhat higher yields most likely tap fracture zones in topographically low areas.

Virginia

Ground-water data on the coal-bearing region of southwestern Virginia are sparse. Consequently, it was not feasible to construct a map showing the distribution of potential ground-water availability. Potential well yields in Virginia's coal area range from low to moderate (0.3 to 3.1 l/s, or 5 to 50 gpm) (Denning, 1977). These yields are generally suitable for domestic or small public-supply or industrial use. Most public water systems in the area use surface-water sources.

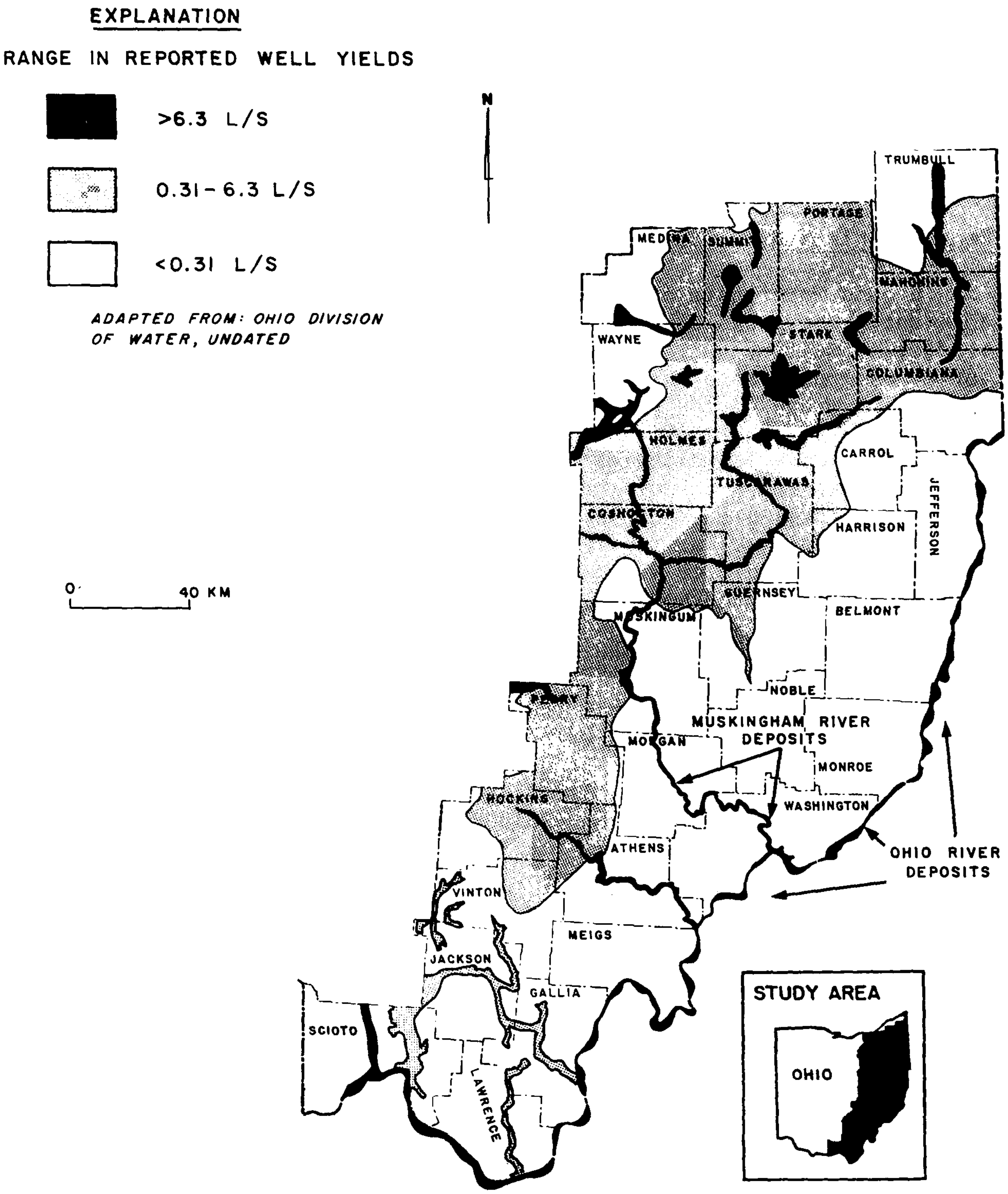

Figure 29. Potential ground-water availability in the coal-bearing region of Ohio.

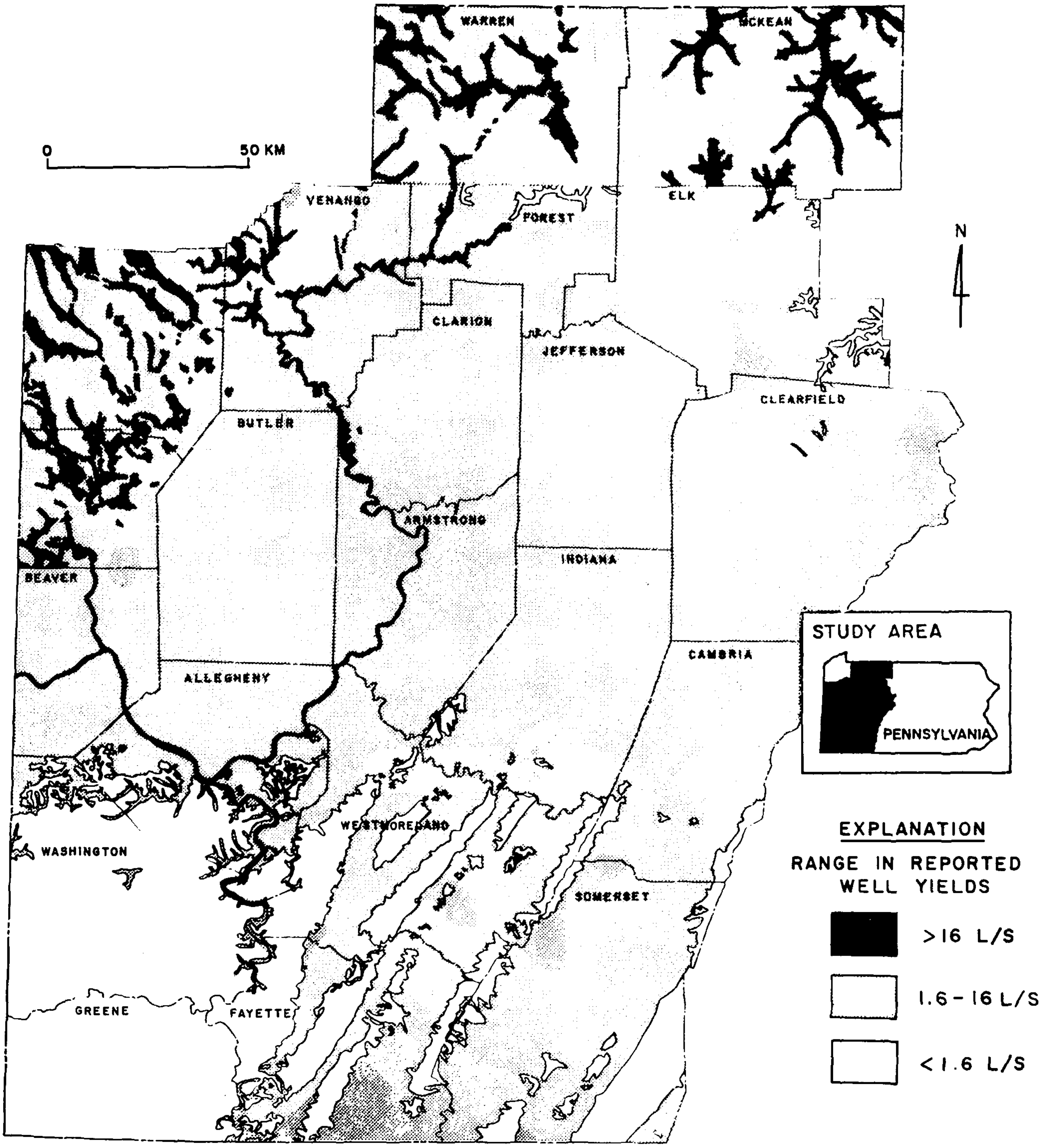

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Figure 30. Potential ground-water availability from near-surface deposits in western Pennsylvania.

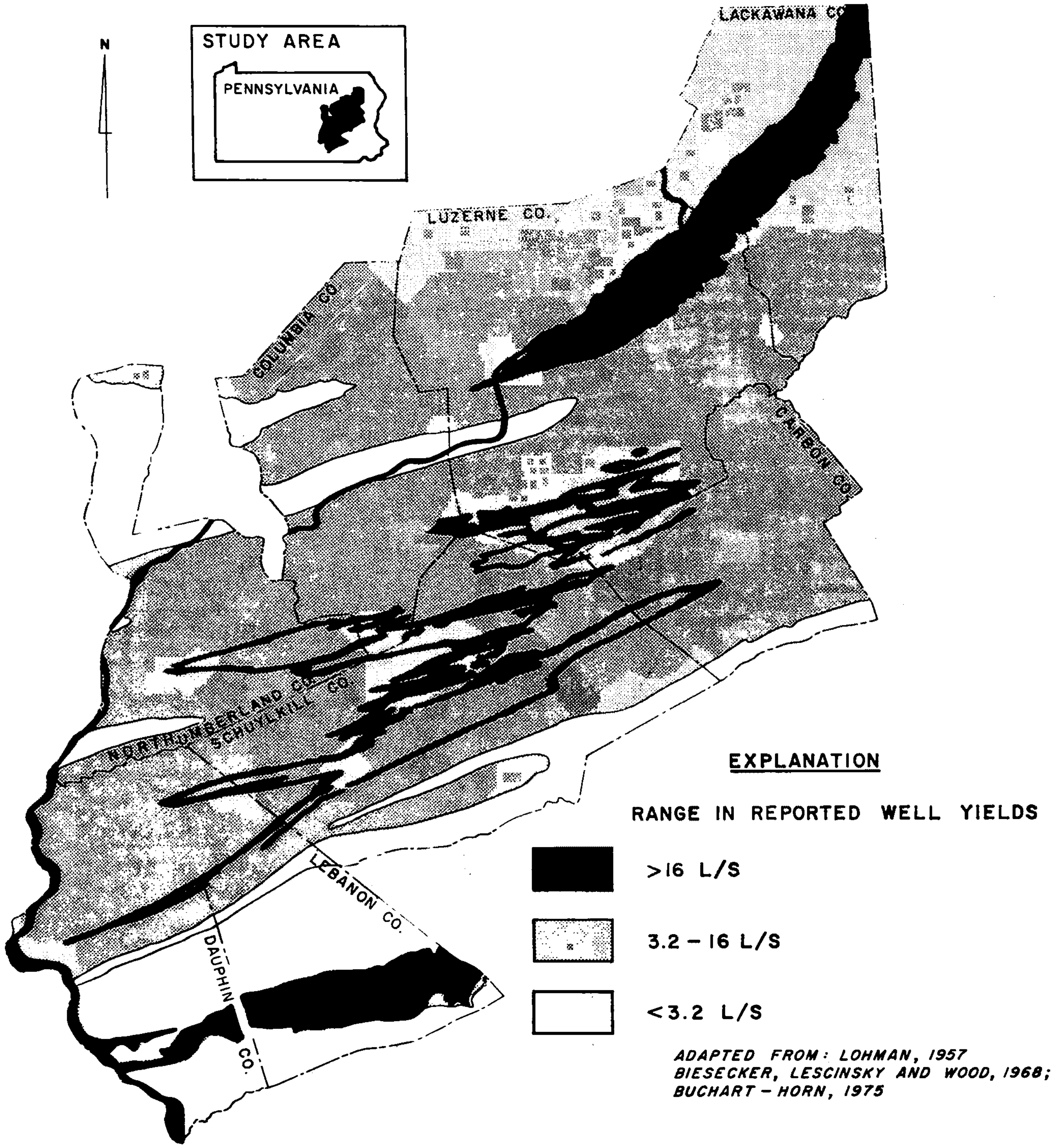


Figure 31. Potential ground-water availability in the anthracite fields of eastern Pennsylvania.

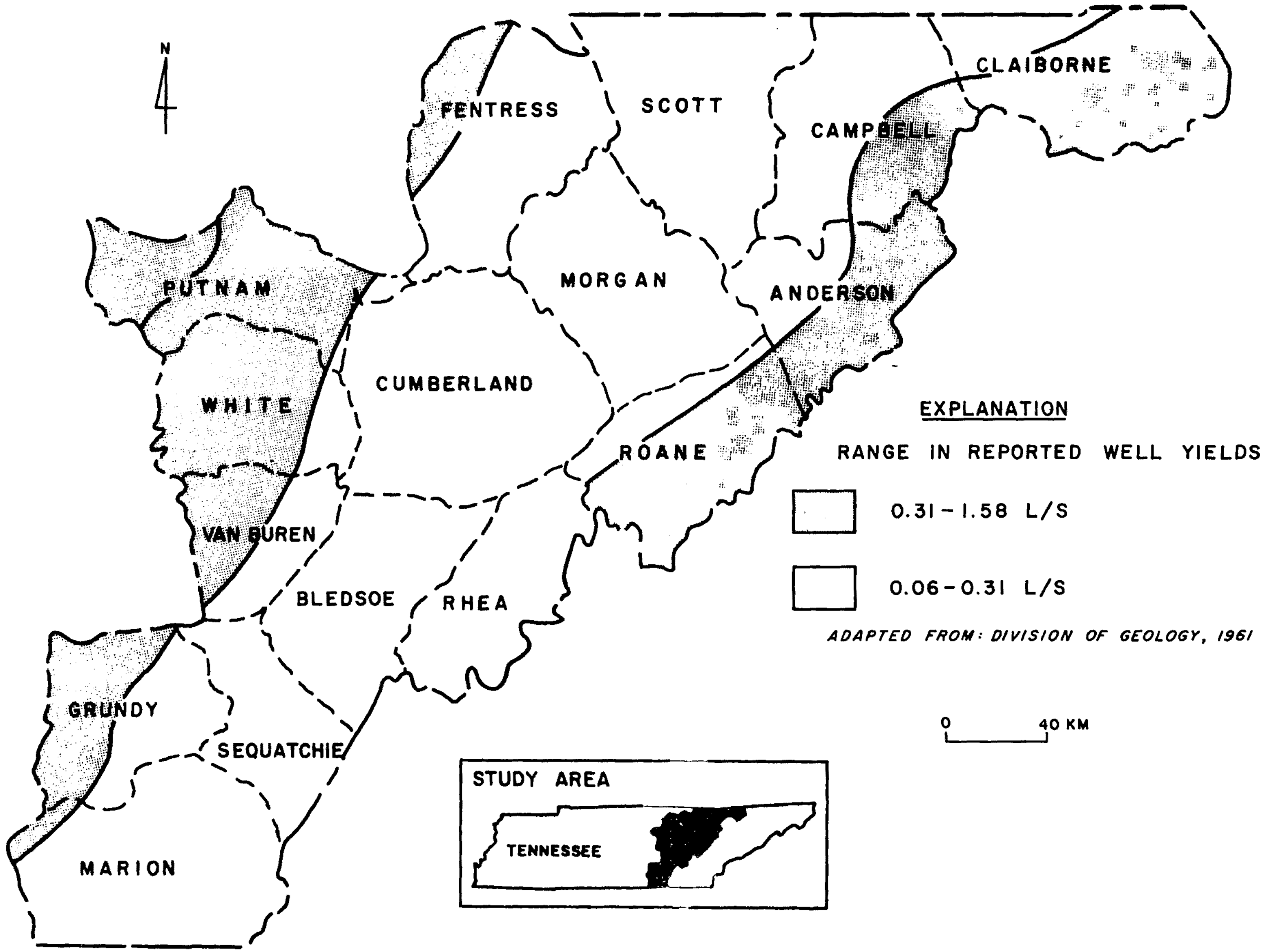


Figure 32. Potential ground-water availability in the coal-bearing region of Tennessee.

West Virginia

Figure 33 shows the distribution and location of the various potential well yields in the state. Poor water-producing sediments are associated with rocks of the Conemaugh, Monongahela, and Dunkard Groups. Fine-grained alluvial deposits are common in West Virginia and do not yield large amounts of water. Wells along the Ohio River, however, are very productive and may yield as much as 63 l/s (1,000 gpm). The lowest yield category is adequate mainly for domestic supplies only.

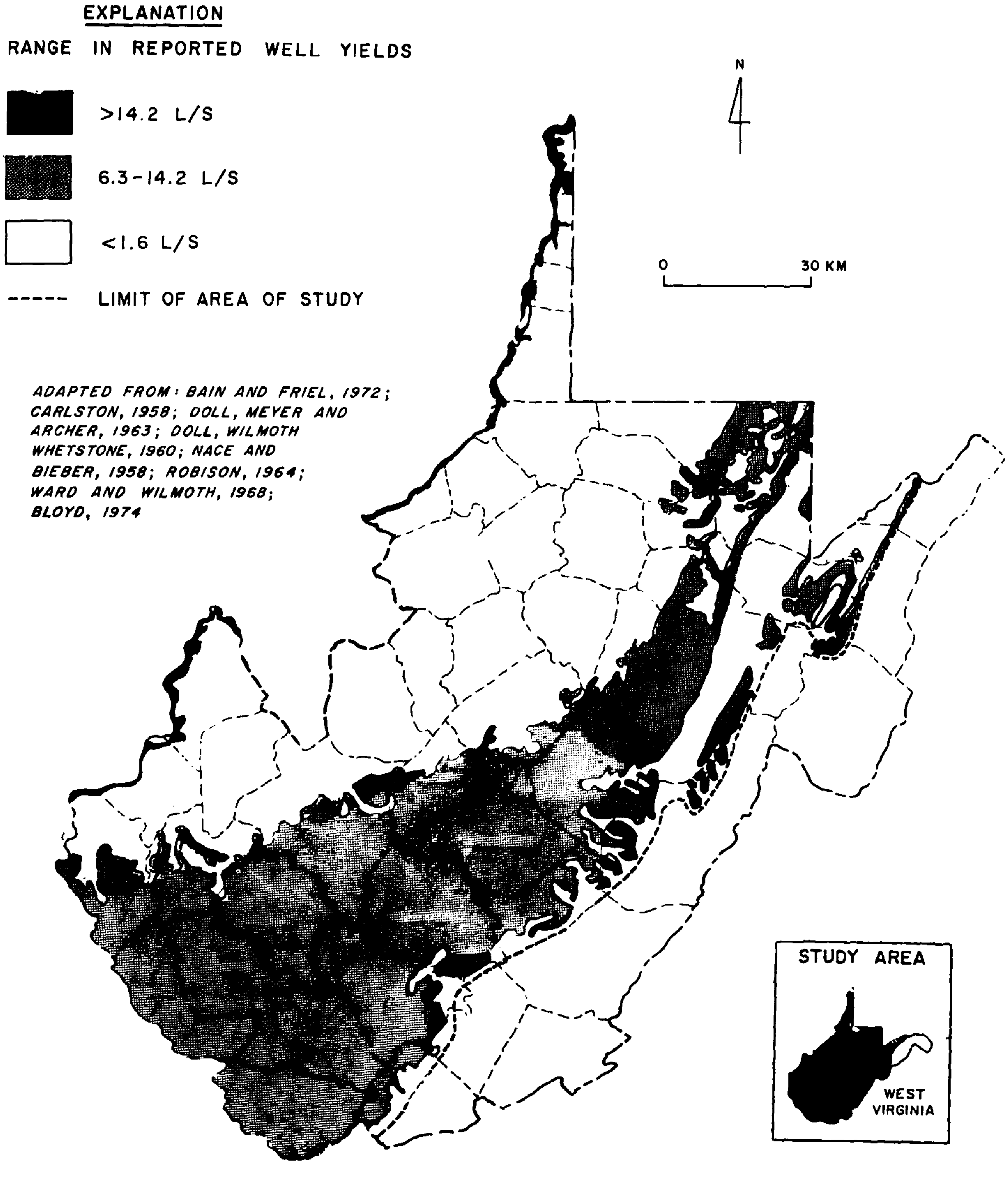


Figure 33. Potential ground-water availability in the coal-bearing region of West Virginia.

SECTION 5

WATER USE

SOURCES AND SIGNIFICANCE OF THE DATA

Water-use data for the coal-bearing counties in the states investigated range from fair to good but are generally incomplete owing mainly to a lack of uniformity in the collection and tabulation of the data among the states. Public supply and industrial use estimates are generally available from the U. S. Geological Survey and state agencies; however, domestic use is estimated from 1970 housing census data or from state tabulations. Domestic supplies are considered to be entirely from ground-water sources by some agencies because surface-water use is insignificant in rural areas of these states. Irrigation and stock-watering use are not included in the overall estimate because of the scantiness of such information. Also, withdrawals for hydroelectric power generation is not included because the water is generally not used consumptively.

The total water-use figures referred to in this section are for coal-bearing counties only and are not for the states as a whole (Figures 34 through 42). Coal counties are defined as those counties having coal reserves for underground mining of more than 0.9 million metric tons (one million U. S. short tons) as determined from U. S. Bureau of Mines statistics (1974). The states in which the coal-bearing counties had the largest ground-water withdrawals in the early 1970's were Ohio and Illinois (Table 3).

In the coal-bearing counties of most states, ground-water withdrawals were generally much less than 380 x 10^3 m^3/d (100 mgd), but 5 to 68 percent of all water used in these counties came from ground-water sources. Ground-water withdrawals in the coal-bearing counties of all eastern states, except Indiana, in the early 1970's were more than 3.4 million m^3/d (0.9 bgd), or 7 percent of the total water withdrawals (Table 3). Ground water generally is a more important source of supply for the dispersed population in rural counties than in the more urbanized counties.

SUMMARY OF USE BY STATES

Alabama

Ground-water withdrawals in coal-bearing counties in Alabama were 163 x 10^3 m^3/d (43 mgd) in 1970, which represented 13 percent of the

TABLE 3. WATER USE IN COAL-BEARING COUNTIES OF THE EASTERN STATES

State	Total Water Used ($10^3 m^3/d$)	Ground Water Used ($10^3 m^3/d$)	Percent of Total Water Use Met By Ground Water
Alabama	1,238	163	13
Illinois	18,433	886	5
Kentucky	861	254	29
Maryland	340	23	7
Ohio	13,074	997	8
Pennsylvania*	13,330	814	6
Tennessee	1,623	83	5
Virginia	126	87	68
West Virginia	682	155	23
Total	49,707	3,462	7

* Data for Indiana and Pennsylvania were available by river basin rather than by county. Much of the area in Indiana included for water-use tabulations was not actually in the study area and therefore is not included in this tabulation.

total water use in coal-bearing counties (Figure 34). The largest use categories were rural-domestic, 72 x 10^3 m^3/d (19 mgd); public supply, 49 x 10^3 m^3/d (13 mgd); and industrial, 42 x 10^3 m^3/d (11 mgd). The largest amount of ground water, 53 x 10^3 m^3/d (14 mgd), was pumped in Jefferson County. Ground-water use in individual coal-bearing counties ranged from 6 percent to 100 percent.

Illinois

Ground-water withdrawals in coal-bearing counties in Illinois were 886 x 10^3 m^3/d (234 mgd) in 1970, which represented 5 percent of the total water use in coal counties (Figure 35). The largest use categories were industrial, 431 x 10^3 m^3/d (114 mgd); public supply, 322 x 10^3 m^3/d (85 mgd); and rural-domestic, 132 x 10^3 m^3/d (35 mgd). With the exception of St. Claire County, all counties south of Marion generally use less than 3.8 x 10^3 m^3/d (1 mgd) of ground water. Counties north of Marion pumped from 2.7 to 155 x 10^3 m^3/d (0.69 to 40.91 mgd). In heavily mined Christian, Franklin, and Jefferson Counties, ground-water withdrawals were 14.8, 2, and 2.8 x 10^3 m^3/d (3.91, 0.53, and 0.71 mgd), respectively. Ground-water use in individual coal counties ranged from less than 1 percent to 100 percent.

Indiana

Ground-water withdrawals for public supply and industrial use could not be computed by county for Indiana because the U. S. Geological Survey compiled existing data by river basin and no state agency collects similar data. It therefore does not give an accurate impression of water use in the coal counties because so many counties outside the study area are included. Rural-domestic use was tabulated by county from housing census information (Bureau of Census, 1972) to help determine which coal-bearing counties are most dependent on ground water as a water supply. Only the counties of Vigo and Vanderbrugh use more than 4.0 x 10^3 m^3/d (1.1 mgd). The rural-domestic water use figures show that the rural population in all the counties is heavily dependent on ground water, deriving 67 to 98 percent of its supply from this source. Ground-water withdrawals for this use in coal-bearing counties in Indiana were 41 x 10^3 m^3/d (11 mgd) in 1971, which represented about 89 percent of the total rural-domestic withdrawals.

Kentucky

Ground-water withdrawals in coal-bearing counties in Kentucky were about 254 x 10^3 m^3/d (67 mgd) in 1968, which represented about 29 percent of the total withdrawals (Figure 36). The largest ground-water use categories were rural-domestic, 110 x 10^3 m^3/d (29 mgd); industrial, 98 x 10^3 m^3/d (26 mgd); and public supply, 45 x 10^3 m^3/d (12 mgd). In western Kentucky, the ground-water withdrawals in the heavily mined counties of Hopkin, Muhlenberg, and Union were less than 3.8 x 10^3 m^3/d (1 mgd). In eastern Kentucky, withdrawals in Pike, Harlan, Letcher, and Knott Counties were 2.9 to 12.8 x 10^3 m^3/d (0.77 to 3.4 mgd). Ground-

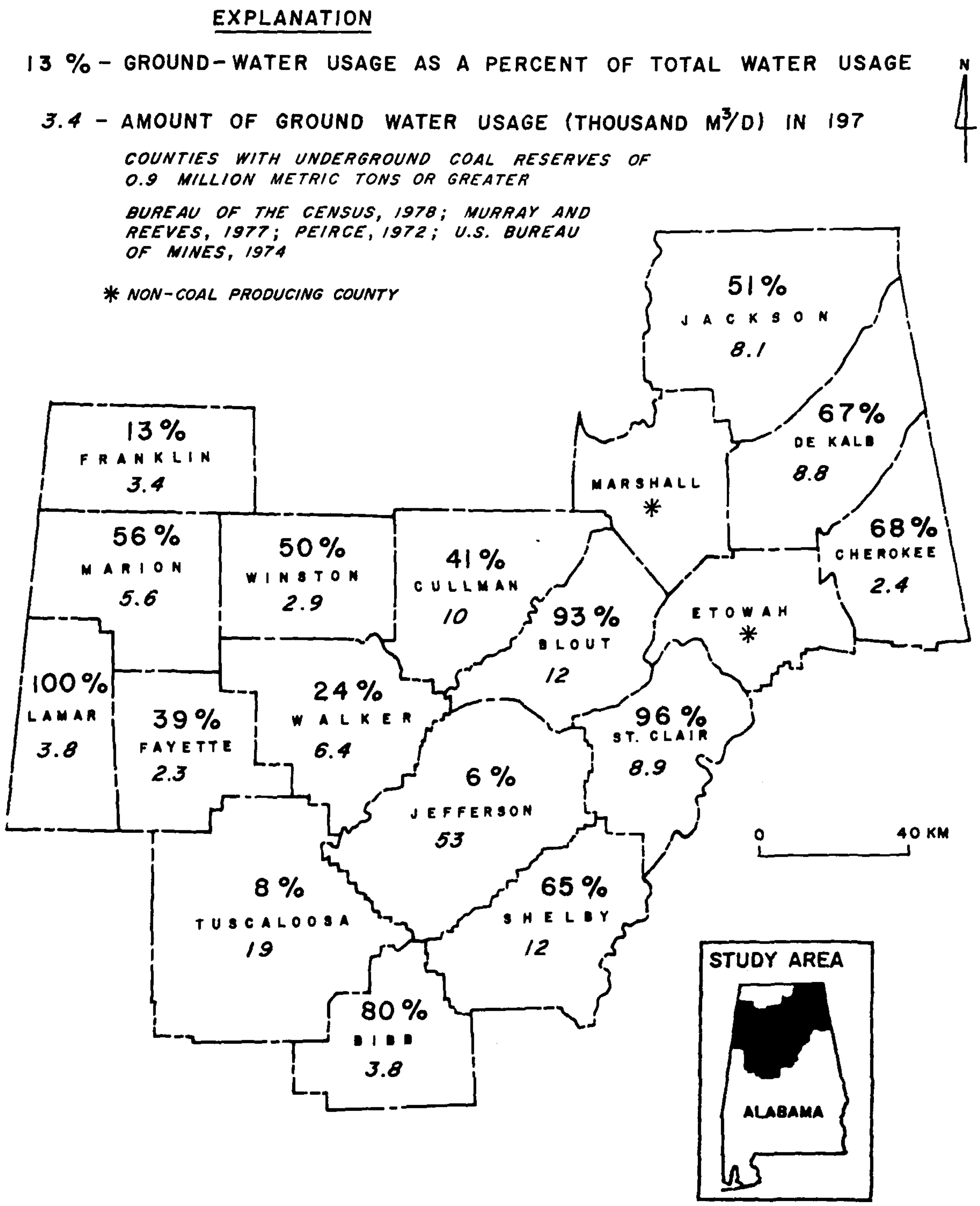


Figure 34. Ground-water use in coal counties in Alabama, 1970.

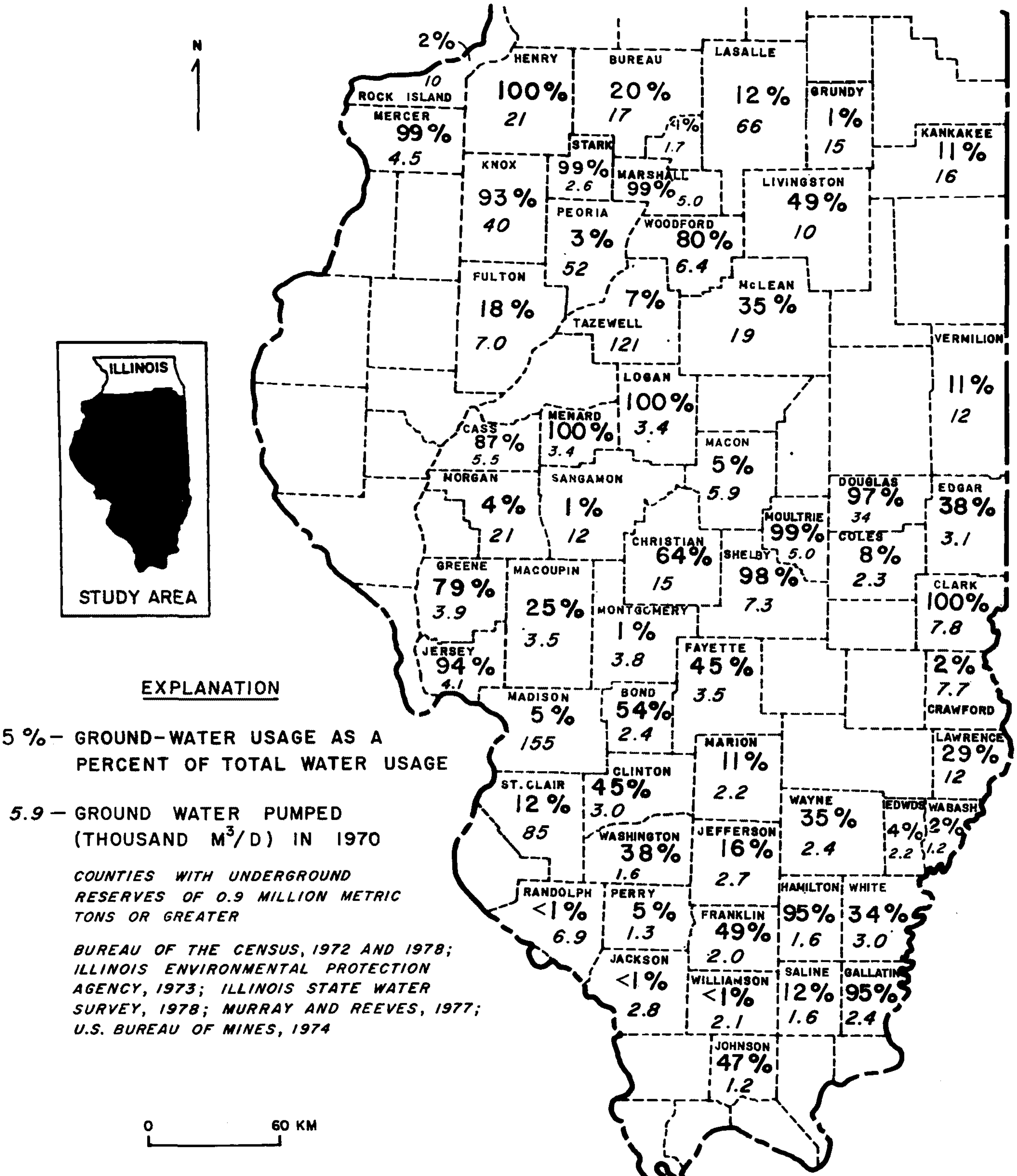


Figure 35. Ground-water use in coal counties in Illinois, 1970.

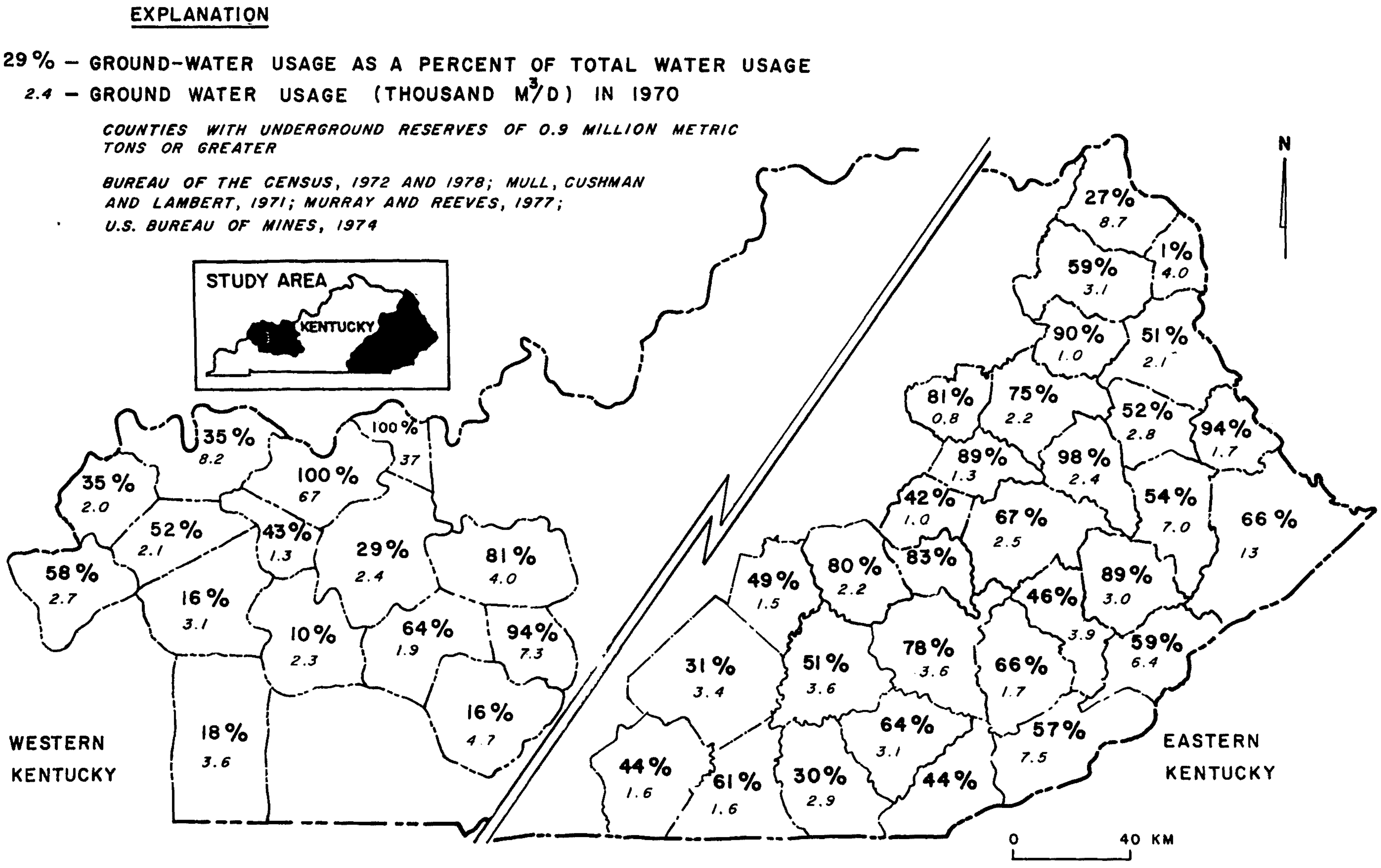


Figure 36. Ground-water use in coal counties of Kentucky, 1970.

water use in individual coal-bearing counties ranged from 1 percent to 100 percent.

Maryland

Maryland has only two counties, Allegany and Garrett, that have significant coal reserves (Figure 37). The Maryland Water Resources Administration (1978) reports that nearly all of Allegany's water supplies are from surface-water sources, and only 2 percent is from ground water. In contrast, Garrett County's water supplies are mostly from ground-water sources, and about 17 percent is from surface water. In 1978, an estimated 23 x 10^3 m^3/d (6 mgd) was pumped from ground-water sources, which represented 7 percent of the total water used in the two coal-bearing counties. Ground-water use in individual counties was 2 percent (Allegany) and 83 percent (Garrett) of the total water use.

Ohio

Ground-water withdrawals in coal-bearing counties in Ohio were 997 x 10^3 m^3/d (263 mgd) in 1975, which represented 8 percent of the total water withdrawals (Figure 38). Incomplete pumpage figures show that industry used 488 x 10^3 m^3/d (129 mgd) of ground water; public supply used 348 x 10^3 m^3/d (92 mgd); and rural-domestic users withdrew 163 x 10^3 m^3/d (43 mgd). Heavily mined Belmont County pumped 41 x 10^3 m^3/d (11 mgd) of ground water. Ground-water use in individual counties ranged from less than one percent to 100 percent of the total water use.

Pennsylvania

Water use in western Pennsylvania had to be compiled by drainage subbasin because the data were available in that format. Only subbasins with significant underground coal reserves are shown in Figure 39.

Ground-water withdrawals in subbasins containing coal-bearing counties were 814 x 10^3 m^3/d (215 mgd) in 1970, which represented 6 percent of the area's total water withdrawals. Ground-water use was 390 x 10^3 m^3/d (103 mgd) for industry; 242 x 10^3 m^3/d (64 mgd) for public supply; and 170 x 10^3 m^3/d (45 mgd) for rural-domestic use. Ground-water use in individual subbasins ranged from 1 percent to 65 percent of the total water use.

Tennessee

Water-use estimates for 1975 were still in preparation by the Tennessee Department of Conservation at the time of preparation of this report. Therefore, data from an earlier tabulation were used as an approximation (Wilson and Johnson, 1970). Ground-water withdrawals in coal-bearing counties were 83 x 10^3 m^3/d (22 mgd) in 1964 (Figure 40), which represented 5 percent of the total withdrawal. Ground-water use was divided as follows: rural-domestic, 61 x 10^3 m^3/d (16 mgd); public supply, 10 x 10^3 m^3/d (2.7 mgd); and industrial, 9.5 x 10^3 m^3/d (2.5

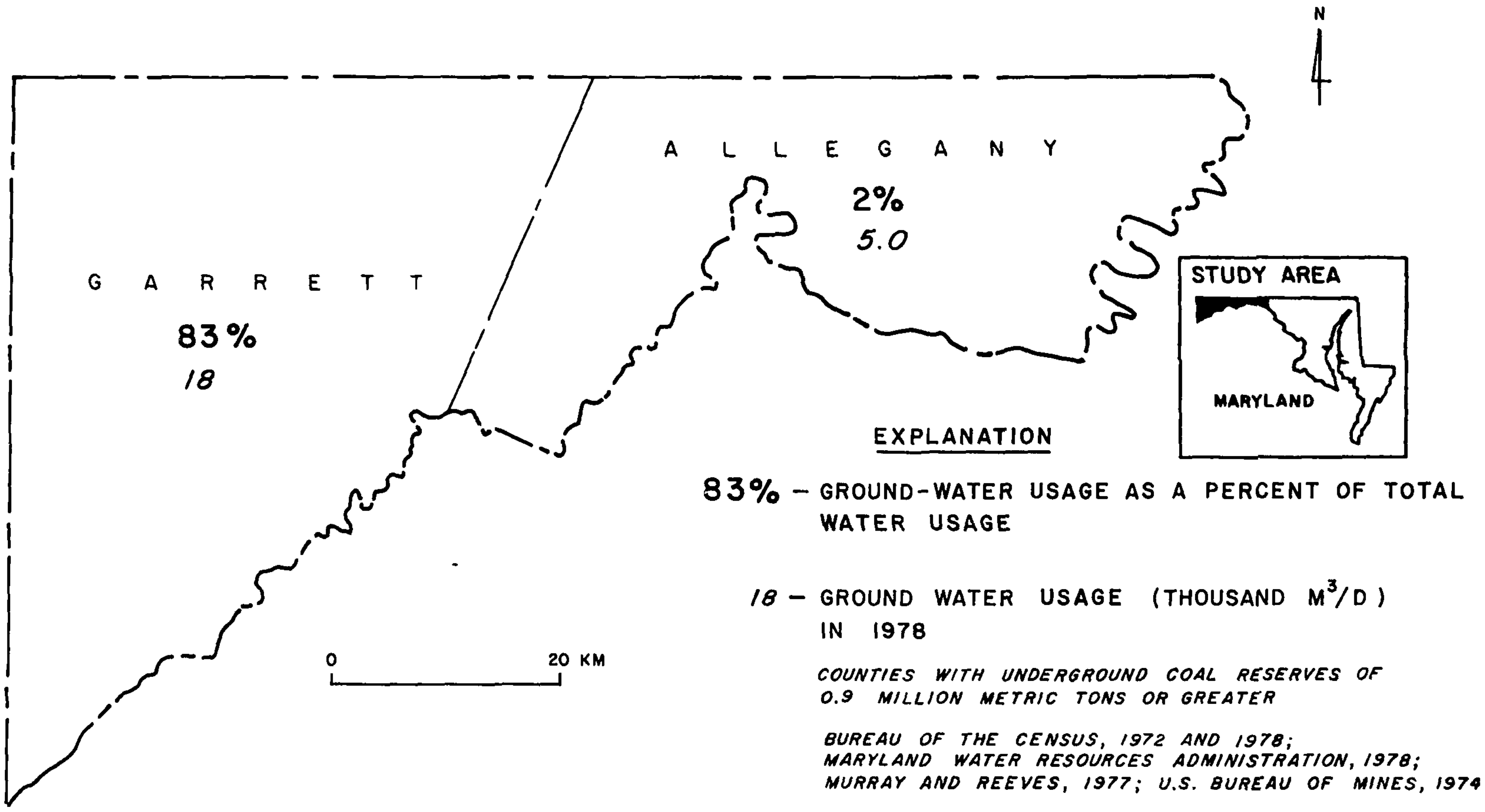


Figure 37. Ground-water use in coal counties of Maryland, 1978.

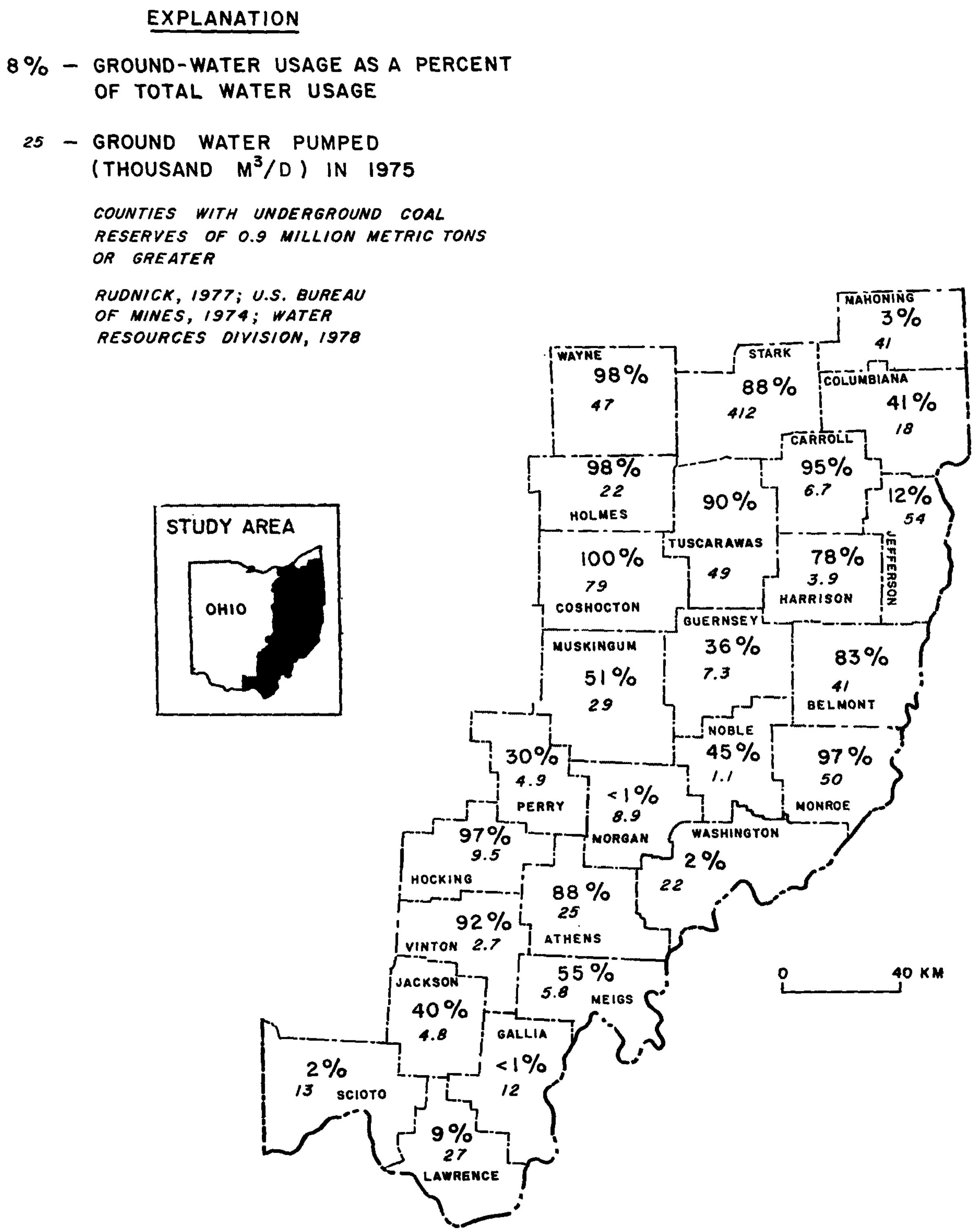


Figure 38. Ground-water use in coal counties of Ohio, 1975.

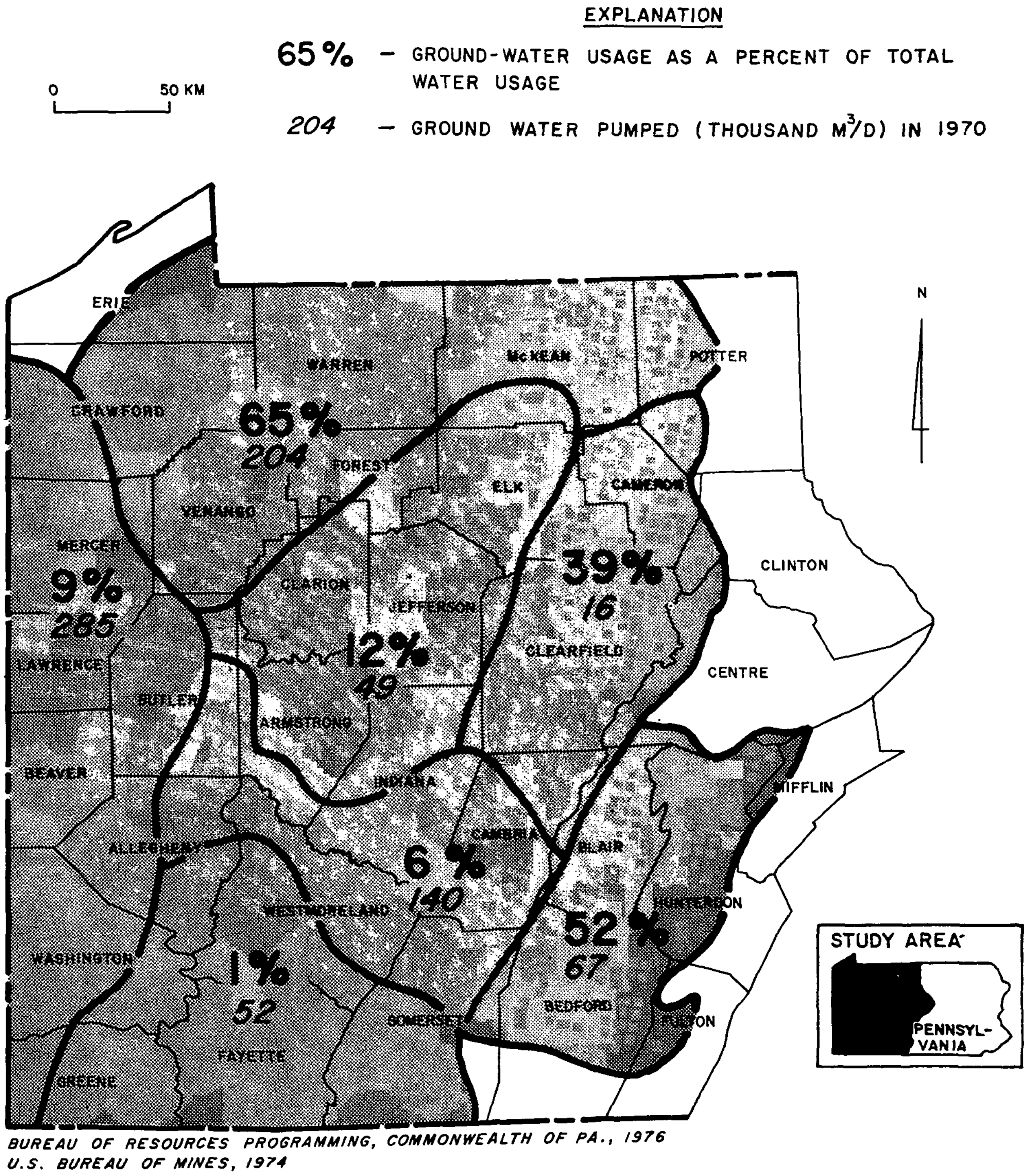


Figure 39. Ground-water use in coal subbasins of western Pennsylvania, 1970.

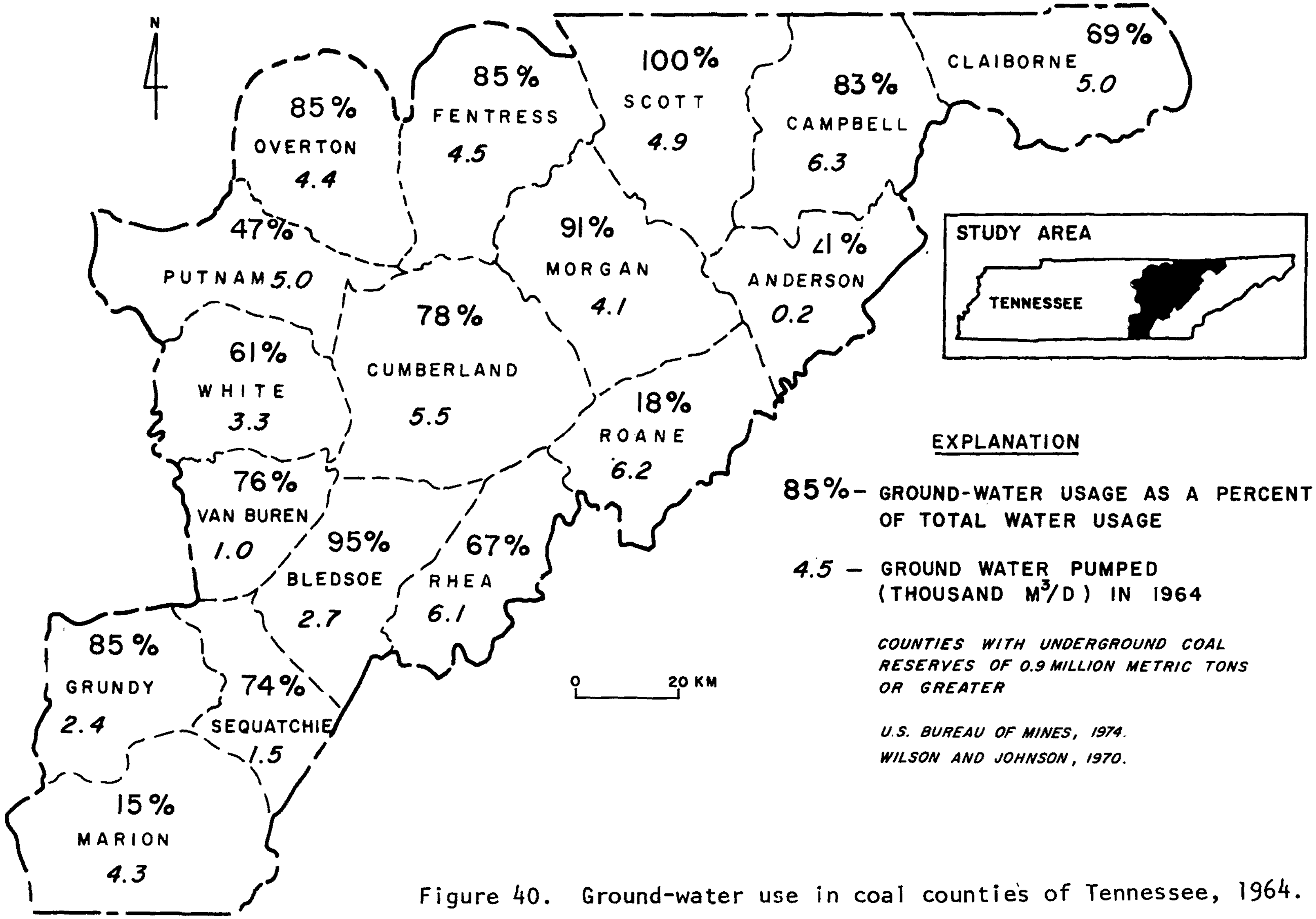


Figure 40. Ground-water use in coal counties of Tennessee, 1964.

mgd). Ground-water use in individual counties ranged from zero percent to 100 percent of the total water use.

Virginia

Ground-water withdrawals in coal-bearing counties were 87 x 10^3 m^3/d (23 mgd) in 1975, which represented 68 percent of the total withdrawals (Figure 41). Ground-water use was divided as follows: rural-domestic, 60 x 10^3 m^3/d (16 mgd); public supply, 22.7 x 10^3 m^3/d (6 mgd); and industrial, 0.95 x 10^3 m^3/d (0.25 mgd). In heavily mined Buchanan, Dickenson, and Wise Counties, ground-water withdrawals were 26, 8.3, and 11 x 10^3 m^3/d (6.8, 2.2, and 2.9 mgd). Ground-water use in individual counties ranged from 48 percent to 98 percent.

West Virginia

No industrial water-use data were available for this state. Known ground-water withdrawals in the coal-bearing counties were 155 x 10^3 m^3/d (41 mgd) in 1966, which represented 23 percent of the total withdrawals (Figure 42). Ground-water use was divided as follows: rural-domestic, 91 x 10^3 m^3/d (24 mgd), and public supply, 64 x 10^3 m^3/d (17 mgd). McDowell, Monongalia, Wyoming, Boone, and Logan Counties, which are heavily mined, used from 2.9 to 15 x 10^3 m^3/d (0.77 and 3.96 mgd). Ground-water use in individual counties ranged from 12 percent to 98 percent of the total water use.

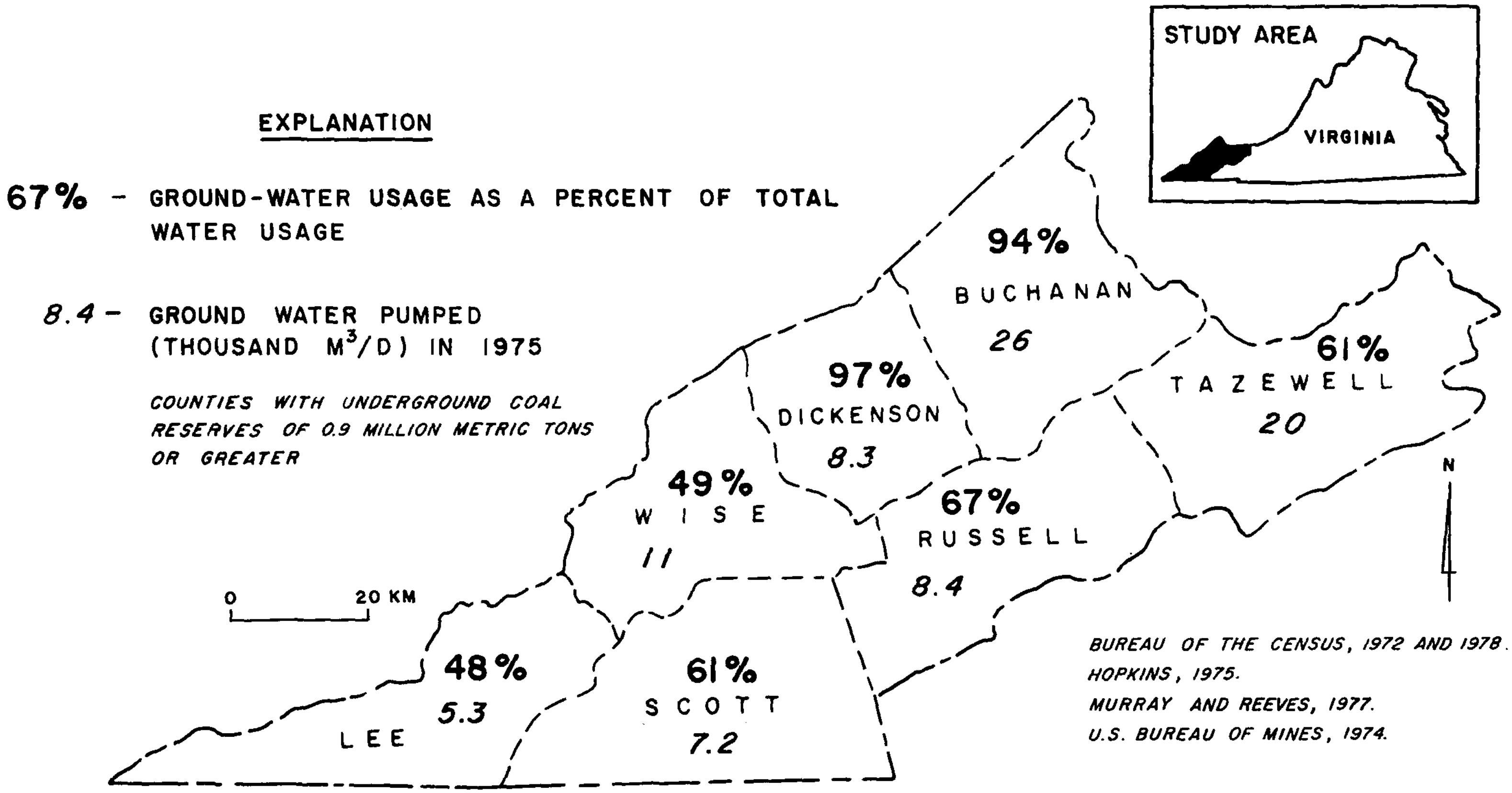


Figure 41. Ground-water use in coal counties of Virginia, 1975.

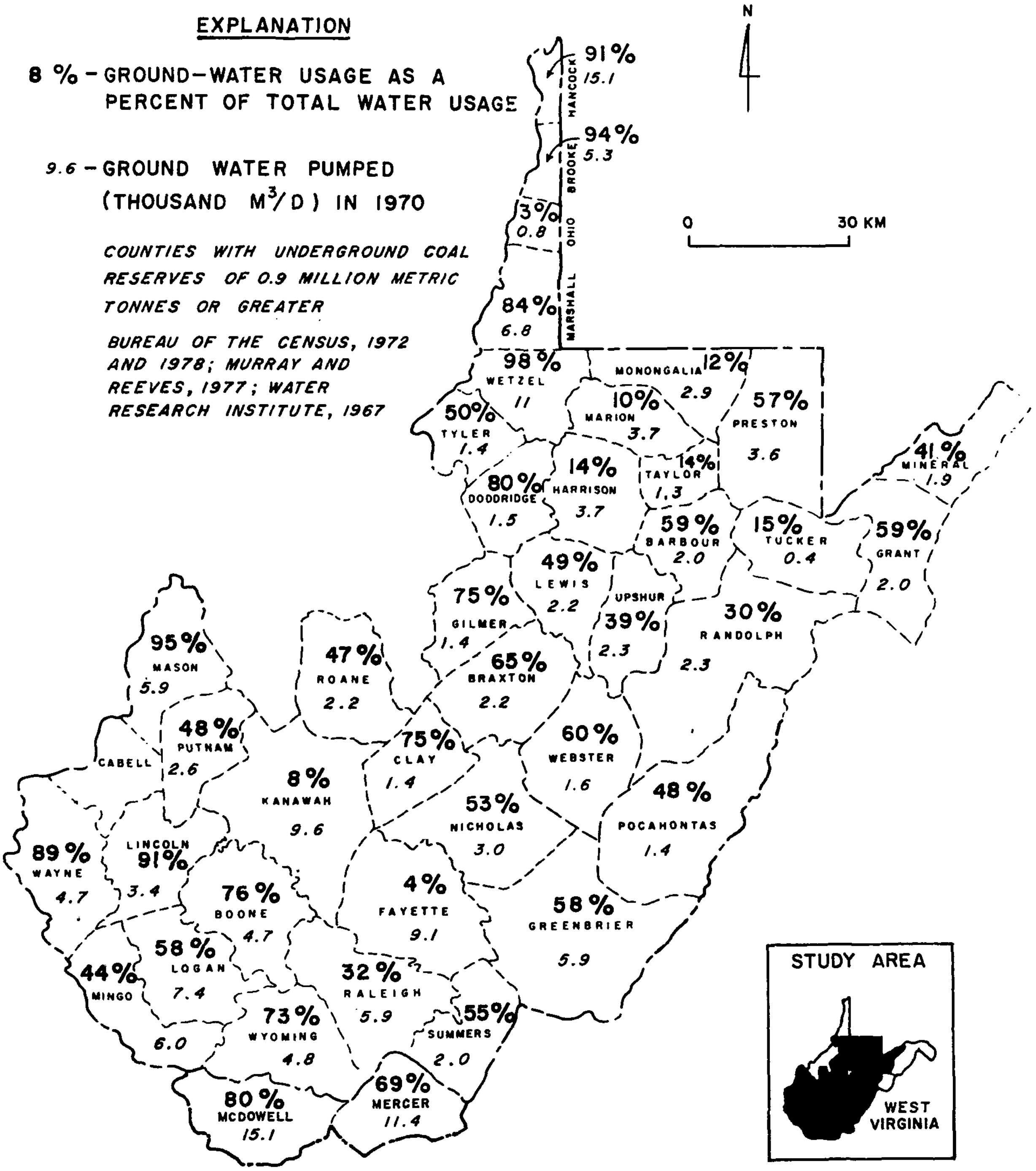


Figure 42. Ground-water use in coal counties of West Virginia, 1966-1970.

SECTION 6

NATURAL GROUND-WATER FLOW PATTERNS

GENERAL CHARACTERISTICS AND CONTROLS

An understanding of the rate, direction, and overall pattern of ground-water movement in the several types of aquifer systems in the coal regions is essential in developing plans for predicting, preventing, and coping with ground-water quality and quantity problems associated with underground mining. Among the key hydrogeologic controls on ground-water movement are aquifer lithology, structure, permeability, water levels, topographic setting, and pumping patterns.

The nature of ground-water flow in the Appalachian Plateau differs considerably from that in the flat lying Eastern Interior Basin. Previous studies of ground-water flow in the midwestern states have concentrated primarily on alluvial and glacial outwash aquifers. These aquifers, which tend to reflect recharge and discharge impacts relatively quickly, are generally shallow and narrow, and occur mainly in present-day or buried bedrock valleys that are filled with sand and gravel. In contrast, most of southern and central Illinois is covered with deposits of low permeability till. These deposits retard the infiltration of precipitation to the underlying bedrock aquifers, and thereby prevent dilution and flushing of natural brines in the bedrock. Trend analysis of the water chemistry of aquifers at depths of less than 76 m (250 ft) in Illinois shows no evidence of dilution and flushing beneath river valleys (Davis, 1973). Apparently topography plays a relatively minor role in the ground-water flow systems of this part of Illinois.

Graf et al (1966) and Bredehoeft et al (1963) suggest that the major circulation pattern in the Illinois Basin is from basin margins, down-dip toward basin center, and then upward across shale units toward the surface at basin center. Cartwright (1970) investigated the location and amount of ground-water discharge in the Illinois Basin by analysis of temperature anomalies. Warmer temperatures near the center of the Basin indicate this is a general discharge area. However, several structural features such as the Wabash Valley Fault Zone and the Du Quoin-Louden anticlinal belt also seem to serve as principal discharge features. Shallow brackish water is known to coincide with the main structural features of the basin. Estimates of the volume of discharge from the deep flow system range from about 136×10^3 m^3/d (36 mgd) to approximately 204×10^3 m^3/d (54 mgd) (Cartwright, 1970). Possibly as much as 95 percent of this discharge may be moving upward through vertical fractures, assuming shales in the Basin have a permeability of about 0.0000000046 cm/s (0.000013 feet per day).

Considerable study has been made of flow systems in the Appalachian Basin, especially in Pennsylvania, and to some extent in West Virginia and Maryland. Studies in these states have provided a fairly detailed understanding of the hydrologic framework of basins and subbasins. Many of the findings of these investigations apply to conditions elsewhere in the Appalachians also.

An example of a simplistic model of theoretical flow patterns in the Appalachian Plateau is illustrated in Figure 43. In this model, only topographic factors have been taken into account; the aquifer is assumed to be homogeneous and isotropic. The diagram shows that there are sets of flow lines in flow systems at progressively greater depths, which control the points of entrance (recharge) and departure (discharge) of ground water. For general discussion, these flow systems may be designated as shallow, intermediate, and deep subsystems. In general, the flow is from areas of high head to areas of lower head with scattered minor flow systems superimposed on the regional pattern.

Regional recharge occurs on upland areas where the flow has a downward component, as indicated by water levels in wells in the uplands. The water levels or heads in wells penetrating successively deeper aquifers are lower in elevation than in shallow wells. The reverse is true in discharge areas, such as stream valleys, where the flow is upward and water-level elevations increase with increasing well depth. Additional evidence of these water-level relationships is given by the common occurrence of flowing artesian wells in many stream valleys and only rare occurrence of such wells in the uplands. Because of the nature of the flow patterns described above, underground mining activities at or near the land surface in recharge zones are likely to affect both shallow and deep ground water, whereas mining activities at or near the land surface in discharge areas are likely to have little effect on deep ground water but could impact nearby surface water.

Figure 44 shows an idealized flow pattern in a series of interbedded permeable aquifers (mostly sandstone) and confining units (mostly shale). Contrasts in the permeability of successive strata can result in marked deflection of flow lines and in greater horizontal flow components than in homogeneous aquifers. High contrasts in permeability result in the development of perched and semi-perched water tables beneath the uplands. In these cases, horizontal flow in highly permeable sandstones may be sufficiently rapid to allow the water table to drop below overlying low permeability shales. This condition can vary seasonally, so that saturated artesian conditions occur when recharge rates are high and perched conditions when recharge rates are low.

Under both perched and semi-perched conditions, ground water can discharge as springs in the valley walls and floors. Springs typically occur at the contact zone where a poorly permeable unit is below a highly permeable zone of fractured rocks. Spring activity is also

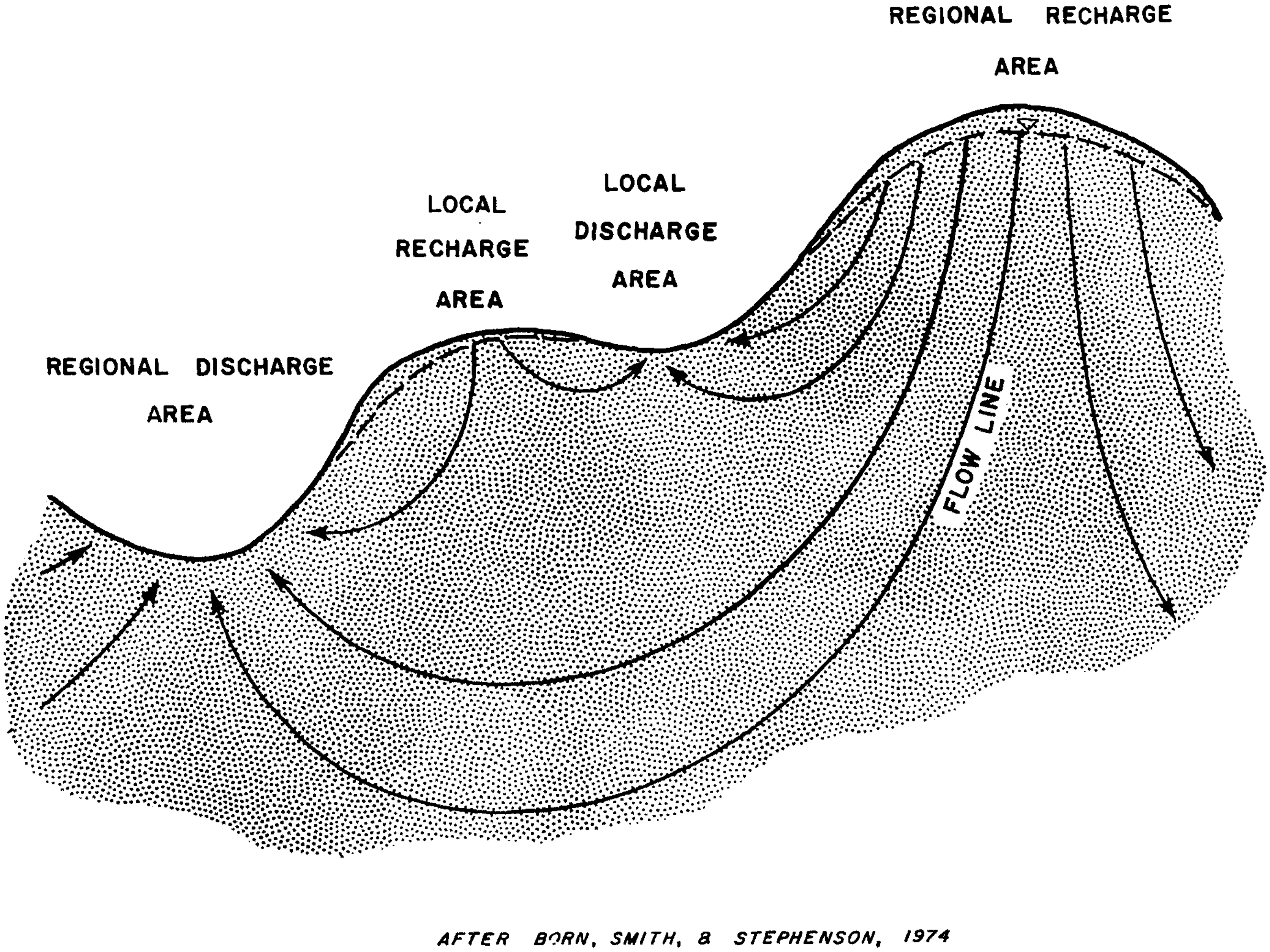


Figure 43. Section showing ground-water flow pattern in a homogeneous, isotropic aquifer with moderate relief.

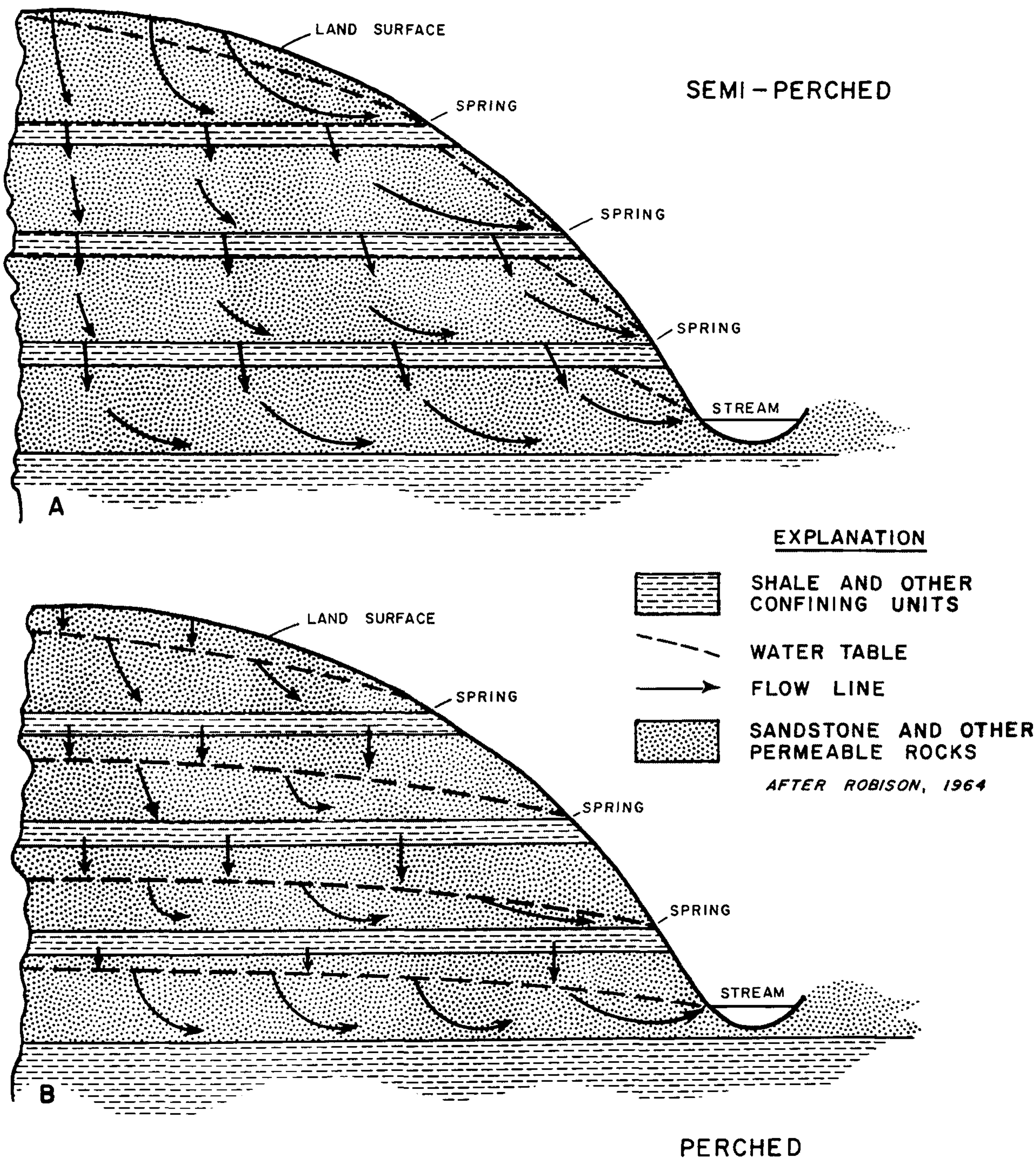


Figure 44. Idealized ground-water flow patterns under semi-perched and perched water-table conditions in stratified rocks of contrasting permeability.

pronounced where deposits of poorly permeable fragipans, colluvium, alluvium, or glacial drift cover the bottom and lower slopes of a valley (Parizek, 1978).

TYPES OF FLOW SYSTEMS

Shallow flow systems typically exist near the land surface and in the vicinity of surface-drainage features. The depth of shallow ground-water circulation below the land surface is typically a few to several tens of meters (Parizek, 1978). The time of travel in shallow flow systems is variable, but in general, water in such systems probably discharges a few weeks to a few years after entry. The ground-water drainage divides of shallow flow systems usually coincide with surface-water divides, and can be approximated from topographic maps.

Intermediate flow systems originate below minor uplands and are generally some distance away from surface-drainage features. Ground-water movement from points of recharge to points of discharge may range from several tens to hundreds of meters (Parizek, 1978). Travel times may be measured in terms of months to tens of years, and variations in flow quantity and quality are less dependent on daily and seasonal climatological fluctuations than in shallow systems.

Deep flow systems originate as precipitation infiltrating over broad recharge areas on major uplands and terminate as discharge to major stream systems. Ground water in these systems may move hundreds of meters to kilometers from points of recharge to points of discharge, and transfer between surface basins is the rule rather than the exception (Parizek, 1978). Typical residence times of ground water in the aquifers are tens to hundreds of years. Water quality and flow volume are not significantly affected by daily or seasonal variations in recharge and discharge.

The depth of deep ground-water circulation may extend many hundreds of meters below the land surface. In some places, this circulation pattern will result in mixing of fresh water with highly mineralized ground water or natural brines. In these cases, discharge from the deep flow system serves as a source of recharge of natural poor quality water to major streams.

As an example, the Burgoon Member is a sandstone of the Pocono Formation (Mississippian age). It is a deep fresh-water aquifer known as the Elliot Park-Burgoon Aquifer, and underlies extensive areas subject to strip and underground coal mining in west-central Pennsylvania. The aquifer is 30 to 183 m (100 to 600 ft) thick, and outcrops in the valleys of some of the major stream systems, including the Clarion, Susquehanna, and Allegheny Rivers. Figure 45 shows a schematic flow system for the Elliot Park-Burgoon and adjacent aquifers. The system consists of a series of minor aquifers and confining beds composed of shale, coal, and clay that overlie the principal aquifer. A part of the water in the minor aquifers eventually recharges the Elliot Park-Burgoon Aquifer by

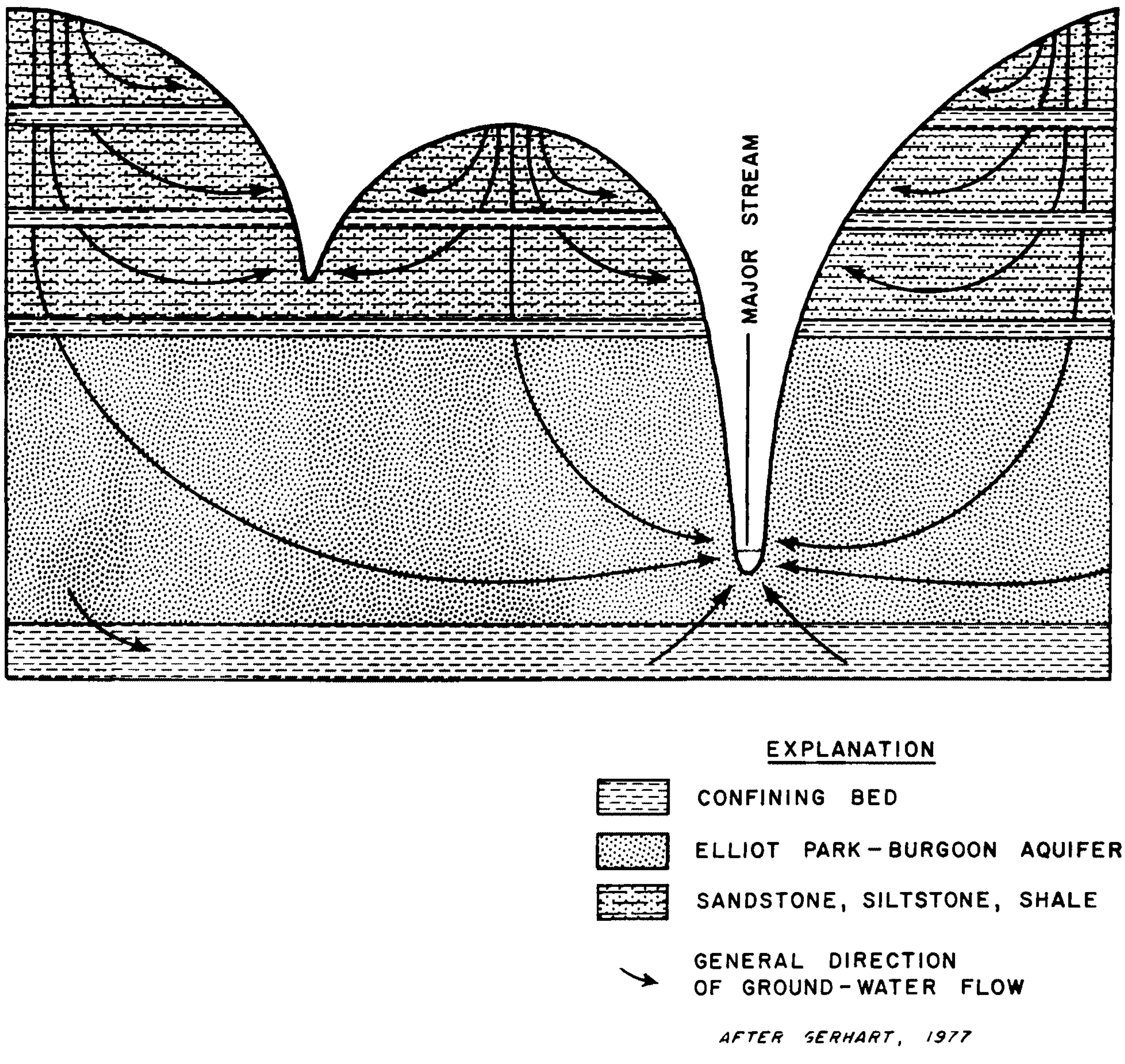


Figure 45. Section showing generalized ground-water flow system in unconfined and confined aquifers in west-central Pennsylvania.

slow leakage through the confining beds. Based on estimated aquifer characteristics and using computer simulations of pumping patterns, the yield of the Elliot Park-Burgoon Aquifer has been estimated to be as much as 150 to 190 x 10^3 m^3/d (40 to 50 mgd) in west-central Pennsylvania (Gerhart, 1977).

ROLE OF PERMEABILITY

The permeability of rocks is a measure of their ability to transmit water. Permeability may be classified as primary or secondary. Primary permeability depends on the size and shape of inter-granular openings or pore spaces and their interconnection. Secondary permeability results from interconnection of openings along joints, fractures, bedding planes, solution channels, and faults. In the coal regions, primary permeability of rocks such as shale, siltstone, limestone, clay, coal, and some some sandstones is very low. Studies of well yields in relation to topographic position throughout the Allegheny Plateau indicate that in many places secondary permeability of rocks is relatively high in valley floors and walls. There is no evidence that topography plays a major role in controlling secondary permeability of the rocks in the Eastern Interior Basin. Increased secondary permeability in this region is more related to structural features, such as faults and ancient anticlines and synclines.

Substitution of permeability, porosity, and hydraulic gradient values in Darcy's Law can provide a rough calculation of ground-water velocity through a given cross sectional area of an aquifer. Only a limited number of permeability values for coal-associated rocks are found in the literature. Brown and Parizek (1971) conducted laboratory tests on oriented cores using an air-type permeameter to determine vertical and horizontal primary permeability of various rocks associated with coal deposits in Clearfield County, Pennsylvania. Average permeability coefficients reported in this study were very low. The reported horizontal and vertical permeabilities of shales, claystones, and siltstones were 0.00000002 cm/s (0.0004 gpd/ft^2) and 0.000000005 cm/s (0.0001 gpd/ft^2), respectively. Cores of rocks near Elkins in northern West Virginia had somewhat higher permeability coefficients, ranging from 0.00000002 cm/s to 0.000047 cm/s (0.0004 to 1.0 gpd/ft^2), as reported by the Resource Extraction and Handling Division of the U. S. EPA (1977).

The most effective method of determining permeability is by means of controlled pumping tests. The results of only a few such tests are documented for the coal regions. Most tests were run in boreholes open to several rock types; therefore, the results represented an average permeability of all the rocks penetrated. Pumping tests conducted by Brown and Parizek (1971) in two areas near Kylertown, Pennsylvania, yielded coefficients of permeability ranging from 0.000057 cm/s to 0.03 cm/s (0.12 gpd/ft^2 to 680 gpd/ft^2) and averaging about 0.0029 cm/s (61 gpd/ft^2). Coefficients determined by Schubert (1978) in Pennsylvania were lower and ranged from 0.0000011 cm/s to 0.0001 cm/s (0.023 to 2.20 gpd/ft^2).

Underclays are typically thought of as impermeable units that prevent mine water from seeping to lower strata. Parizek (1978) reports, however, that underclays commonly contain hairline joints and fractures, and highly polished slickenside joint surfaces, and are usually thin. Consequently, although local leakage on a unit basis through these clays is small, the clays may extend over large areas and regional leakage could be large.

The magnitude of ground-water recharge in an area is an indicator of the relative permeability of surficial rocks and unconsolidated sediments. Estimates of recharge in subbasins of the Eastern Interior vary from 3 to 11 percent of the total precipitation. This estimate is based on 60-percent streamflow duration data (Bloyd, 1975). Assuming an average annual rainfall of about 103 cm (40 in), recharge ranges from 3 cm (1.2 in) to 11 cm (4.4 in). Recharge estimates in subbasins of the central Appalachian Basin range from 3 to 18 percent of the total precipitation (Bloyd, 1974). The greatest recharge (10 to 18 percent) is found in the Allegheny, Monongahela, Upper Ohio River, and Kanawha River Valleys. Assuming an average annual rainfall of about 115 cm (45 in), recharge ranges from 3 cm (1.2 in) to 21 cm (8.3 in).

The importance of permeability in the form of fracture traces and lineaments in determining the occurrence of ground water was mentioned in the section entitled Ground-Water Availability. The presence of these features can be responsible for a 10- to 100-fold increase in permeability of various sedimentary rocks (Parizek, 1976). Mundi (1974) reported significantly higher yields from wells penetrating fracture traces in shale, sandstone, and limestone in central Pennsylvania. These fractures undoubtedly provide a mechanism for vertical flow through rocks of otherwise very low permeability. Parizek (1978) reports that more than 50 percent of springs that issue from bedrock in Pennsylvania, Maryland, West Virginia, and similar mining regions are probably controlled by zones of fracture concentration which intersect valley walls and are only partly covered by deposits of low permeability. It is also likely that fracture traces and lineaments in underground mines are responsible for increased amounts of water encountered during active mining. A study of this relationship is in progress by Roebuck (1978). Regional maps of linear structural features are not common for coal-mining regions. One such map has been prepared for Pennsylvania by Kowalik and Gold (1975).

SECTION 7

HYDROLOGIC EFFECTS OF MINING

ALTERATION OF GROUND-WATER FLOW PATTERNS

Effects of underground coal mines on natural ground-water flow systems occur during both active mining and after abandonment. Active mining creates the most immediate and most noticeable changes in water levels and ground-water flow as a result of removal of water from the mine. This removal occurs by gravity drainage in the case of up-dip drift mining or by pumping in down-dip drift, slope, or shaft mining (Figure 46). Freely draining abandoned mines continue to discharge by gravity after mining ceases. In other situations, abandoned mines become flooded and water levels in the mine and vicinity recover to a new equilibrium position. Water discharged from a mine is a measure of the ground water lost from storage. Direct flow of mine drainage from active mines to streams is fairly well documented as a result of state and EPA coal-mining effluent guidelines and standards.

Active mines throughout the study area have a wide range in discharge. In the past, attention has been focused only on visible drainage and its effect on surface-water quality. In this section, emphasis will be given to changes in ground-water flow and availability, and to the hydrologic impacts of such changes.

In drift entry coal mines, the main and lateral headings act as line sinks which intercept water that normally would flow above or through the coal seam. As headings are advanced, the coal is removed from floors and panels over a very large area. The mine then acts as a broad sink or underdrain, which receives ground water that percolates downward from overlying strata. This water is derived from storage until a new equilibrium between inflow and outflow is established. Reduction in storage is indicated by declining water levels and reductions in flow at discharge points. During active mining, recharge to the deep part of ground-water flow systems virtually ceases in the vicinity of the mine (Hollyday and McKenzie, 1973).

Shaft and slope entry mining create a similar situation although the dewatering configuration may be considerably different. During active mining, the area influenced by drainage into the mine extends beyond the limits of the mine. In all types of underground mining it may take years before new equilibrium conditions are reached and, in many places, the mine may be abandoned before this occurs.

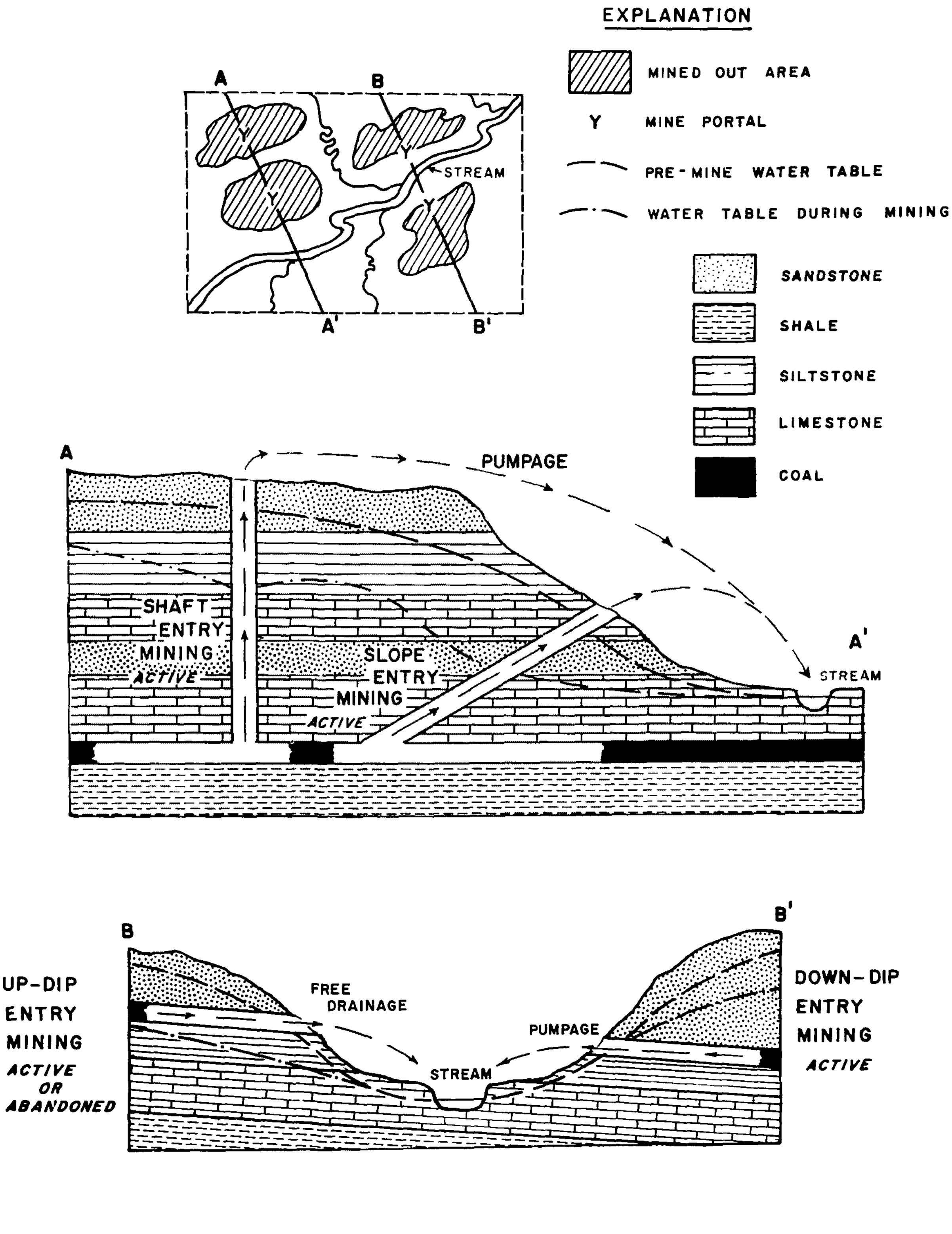


Figure 46. Types of underground mine entryways and their effects on ground-water levels.

Water can enter a deep mine by means of seepage from overlying strata, through faults and fractures in coal seams and adjacent strata, and by entering abandoned sections of surface or underground mine workings. A study by Lovell and Gunnett (1974) found a strong correlation between fracture traces visible at the land surface, joint density, and roof seepage in an underground mine in Clearfield County, Pennsylvania (Figure 47). In the uncaved portion of the B Seam mine, all of the fracture traces coincided with seepage or increased joint density in the shale roof rock. The other portion of the B Seam mine is an older section where the overlying strata have largely caved in. This area produces most of the water pumped from the mine as a result of increased vertical leakage. Pumpage from the mine in June 1972 was 1.5×10^3 m^3/d (400,000 gpd).

The authors found that, in the altered flow system, the water level drops below the E Seam in dry periods and rises to the mine workings in times of heavy rainfall. Calculations based on existing data indicate that the average vertical permeability of the strata between the B Seam and the E Seam is approximately 0.0000024 cm/s (0.05 gpd/ft^2). This value is more than 400 times the number estimated by Brown and Parizek (1971) for similar undisturbed beds of shale, claystone, and siltstone near Kylertown, Pennsylvania. The higher value probably reflects the increased permeability in the caved area (Lovell and Gunnett, 1974).

Documented hydrogeologic studies around mines in the Eastern Interior Basin are scarce. Cartwright and Hunt (1978) reported preliminary results of two studies in Illinois. Core holes and piezometers were drilled and constructed around an abandoned shallow underground mine in western Illinois. Initial data suggests the shallow ground-water flow system is unaffected by the mine 20 m (66 ft) below the water table, despite the mine collapse which has occurred. Another investigation, this time at an active mine, was initiated to study water-related problems in the mine. Although the mine is relatively dry (less than 50 cm/d) infiltration, local structural features were found to control the limited infiltration of water into the workings. Structural features mapped include joints, faults, clay dikes, and related features.

Another detailed hydrogeologic investigation was conducted at a site with both deep underground and strip mines near Kylertown in Clearfield County, Pennsylvania, by Brown and Parizek (1971) (Figure 48). The Clarion Coal Seam, or "A" Coal, is generally 0.6 to 1.2 m (2 to 4 ft) thick and has been mined primarily by stripping along the outcrop. The Lower Kittanning, or "B" Coal Seam, is 1.2 to 2.1 m (4 to 7 ft) thick and was extensively mined by underground methods in the 1950's. Since that time much of the Lower Kittanning Coal outcrop has been stripped. Three vertically separated flow systems were recognized at the site (Brown and Parizek, 1971). The upper flow system extends downward from the water table beneath the topographic highs to the roof of the deep mines in the Lower Kittanning "B" Coal Seam. The mine roofs are broken and fractured. Much of the water from this flow system discharges directly to the land surface as springs along the lower slopes of hillsides and in valley bottoms.

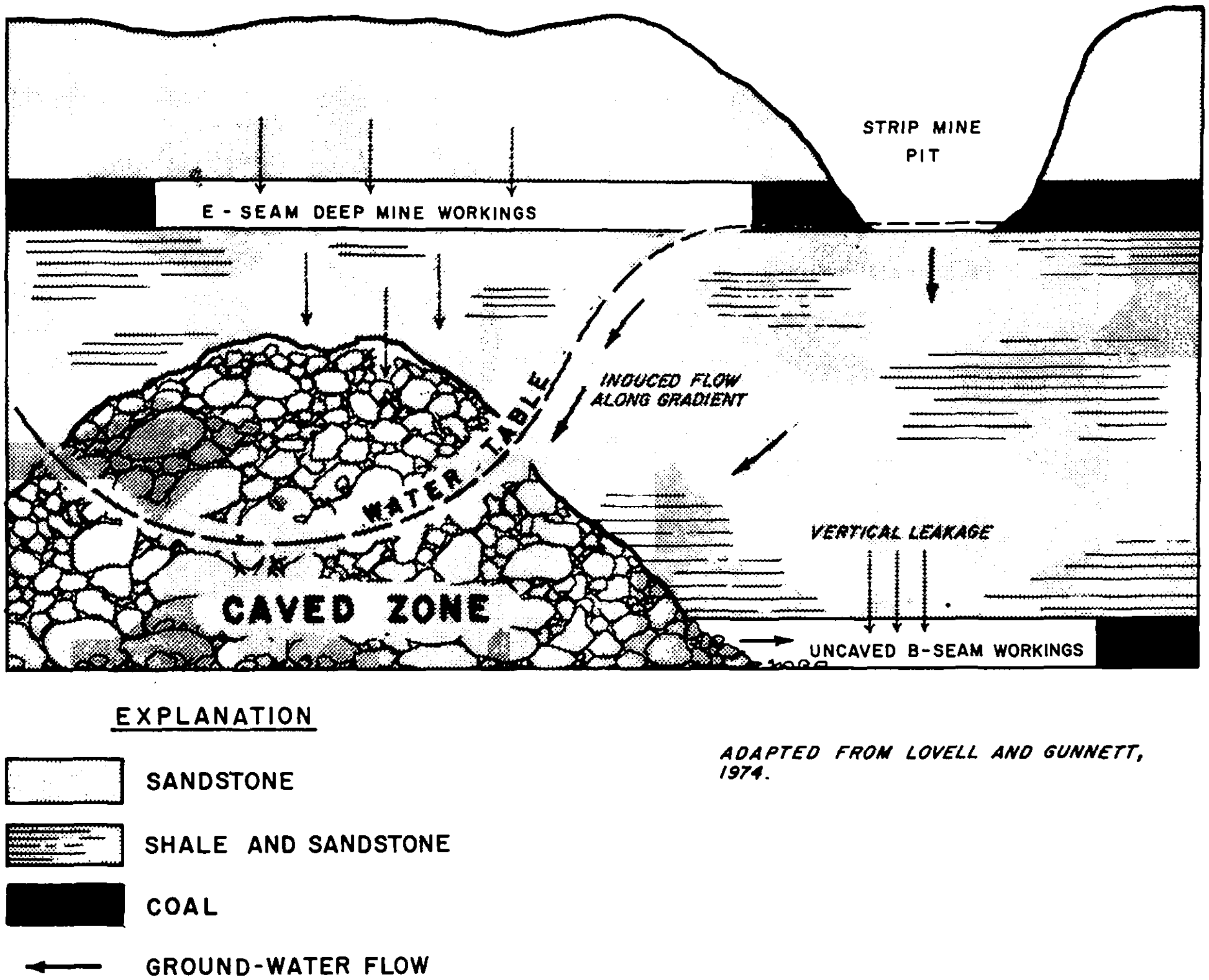


Figure 47. Idealized section of ground-water flow pattern into a mine in Clearfield County, Pennsylvania.

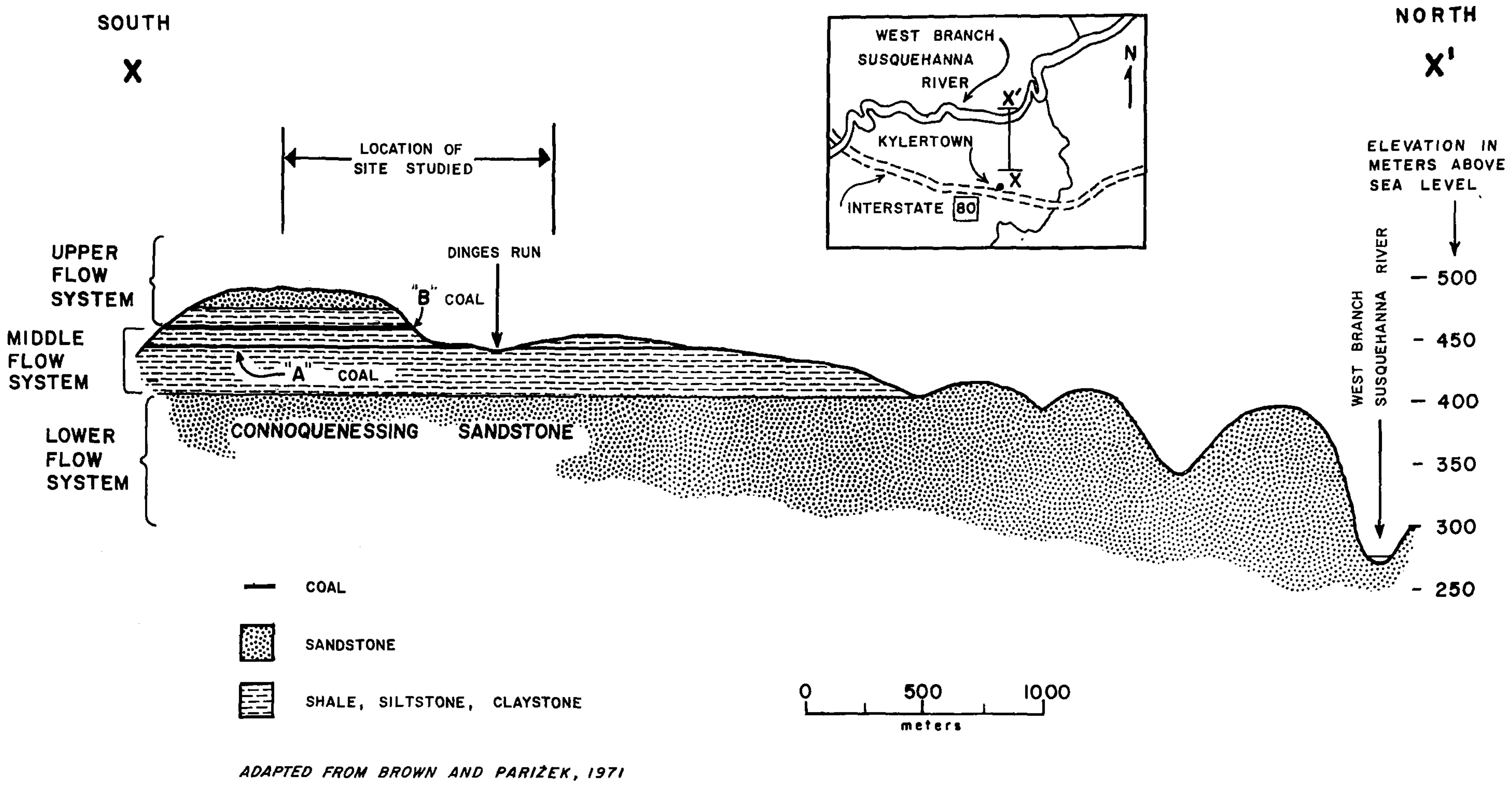


Figure 48. Generalized cross section of a mine near Kylertown, Pennsylvania.

The middle flow system identified by Brown and Parizek (1971) is bounded at the top by the B Coal Seam and at the bottom by the top of the Connoquenessing Sandstone which is below the A Coal Seam. Ground-water recharge occurs primarily from the floor of the B Seam deep mines and from B Seam strip mines. Discharge from the middle system occurs in the minor valley bottoms and lower walls as springs, swamps, seepage areas, and streamflow. Part of the discharge also takes place as re-charge to the Connoquenessing Formation by slow vertical filtration through overlying shale and claystone down to the water table in the sandstone. Water from this lower regional flow system moves into dis-charge zones along incised streams such as the West Branch of the Susquehanna River.

A flow system which has been affected by underground mining in Painters Run and McLaughlin Run basins, Allegheny County, Pennsylvania, is shown on Figure 49 (Subitzky, 1976). The author reports that inter-connected joints and mine subsidence fissures exert strong control on ground-water circulation. The flow lines in this system generally follow partly angular paths. Some flow lines reflect ground-water movement directly to the underlying mined-out part of the Pittsburgh coal seam, and others follow shorter paths to the streams.

Controlled pumping tests conducted above an underground mine near Carrolltown, Pennsylvania, underscored the importance of fracture sys-tems in controlling flow patterns. The drawdown pattern measured in observation wells near a well being pumped at 1.9 l/s (30 gpm) was assymetrical and indicated preferred paths of subsurface flow (W. A. Wahler and Assoc., 1978). In addition, despite the depth of the mine (160 m or 525 ft), there was a rapid response to heavy rainfall both as a rise in water levels in wells and as increased inflows to the mine, indicating good hydraulic connection of fracture zones with the land surface.

Local or subregional structural features, such as broad anticlines and synclines, provide exceptions to typical flow systems found in horizontal rocks. The effect of the dip of the rocks on the flow pat-terns is accentuated during mining. Figure 50 illustrates how a flow system may function where coal beds are inclined. In this situation, the location of the ground-water divide does not correspond exactly to that of the surface-water divide. Mining up dip in this case can result in a large shift in the ground-water divide in an up-dip direction. Mining down dip will cause some shifting of the ground-water divide in the down-dip direction. Spring flow and ground-water underflow that previously discharged to streams on either side of the upland will be reduced correspondingly.

The effect of underground mining on the flow system below the mining zone is largely dependent on the nature and continuity of strata below a coal seam. Specifically, the confining nature of the underclay plays an important role in retarding downward flow of mine water. As previously discussed, the primary permeability of most underclays is very low, although the clay may have small secondary fissures. In

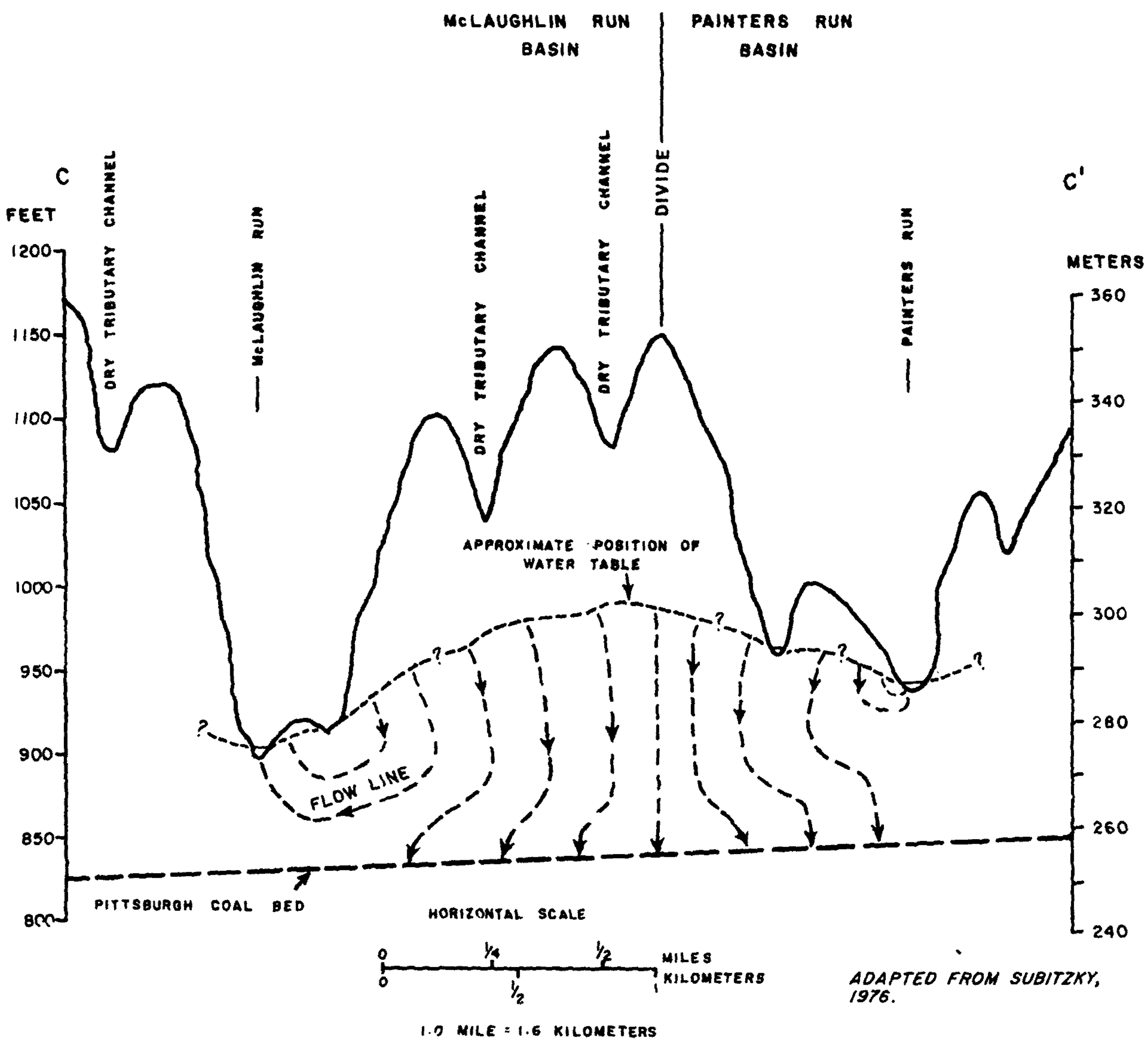


Figure 49. Idealized ground-water flow pattern showing effects of underground mining in Allegheny County, Pennsylvania.

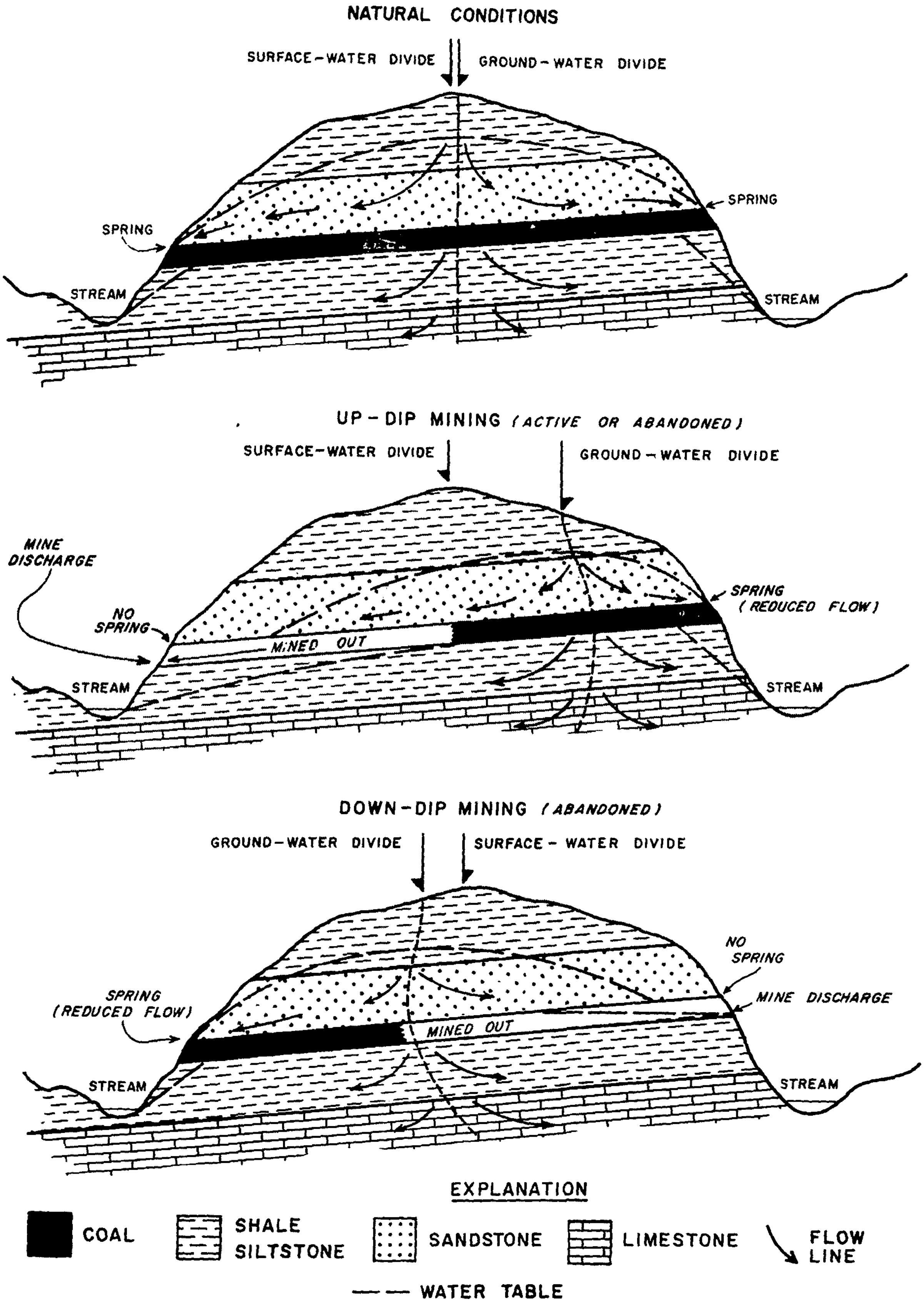


Figure 50. Relative locations of ground-water divides in inclined strata under mining and non-mining conditions.

addition, such deposits are very extensive. In most places, the underclay is as extensive as the coal itself, although channel sandstone deposits may intersect both coal and underclay in many seams. In the Upper Freeport and Pittsburgh seams in southwestern Pennsylvania, for instance, the coal is sometimes replaced by sandstone or shale deposited as channel filling material. Large washouts such as these are generally avoided during mining (Bushnell, 1975). The vertical leakage rates through plastic underclays of the Illinois Basin may be less than those through the brittle and jointed underclays of the Appalachian Basin (Parizek, 1978). However, no data were available to document the range of leakage rates.

Ground-water flow in Pennsylvania's anthracite region is complicated by the complex folding and faulting as well as by the intricate network of mined-out coal seams, boreholes, and subsidence features. Figure 51 is a schematic cross section showing the ground-water flow pattern in an idealized synclinal anthracite coal basin. Recharge is increased in areas of man-made openings such as surface depressions, subsidence areas, and strip-mine pits that are connected directly to the underground mine workings. In addition, surface water can enter the underground mine workings from streams that traverse broken and stripped ground (Growitz, 1978). Discharge is also increased by mechanisms similar to those found in the bituminous region except that intermine connections are probably more extensive.

SUBSIDENCE

As a result of pillar failure years after abandonment or intentional removal of pillars during mine retreat, the roofs of many underground mines eventually collapse. Secondary faults and fractures of mine roof failure extend up through the overlying strata and, depending on the depth of the mine and competence of the overburden rock, subsidence features may appear at the land surface. Besides the hazardous effects of depressions at the surface, subsidence can alter the hydrologic system in the following ways: (1) increased secondary permeability at the surface will result in increased infiltration of precipitation and decreased overland runoff; (2) travel time of ground-water flow from the surface to points of discharge will be greatly decreased; (3) water levels will fluctuate over a larger range, and average levels typically will be lower than under pre-mining conditions; (4) ground-water base flow to nearby streams may be increased as a result of the relatively free subsurface circulation created by fractures over mined-out seams; and (5) surface-water flow may be decreased where fracturing extends under a stream bed.

In room and pillar mining, it is generally not possible to predict the development of subsidence over abandoned mines without knowledge of the mine dimensions and size and height of pillars. Pillars may fail after years have elapsed and the amount of movement will depend on the room space available into which they can collapse. In some cases, pillars are forced into a soft floor, such as underclay. Longwall mining results in the greatest vertical displacement of overlying rocks

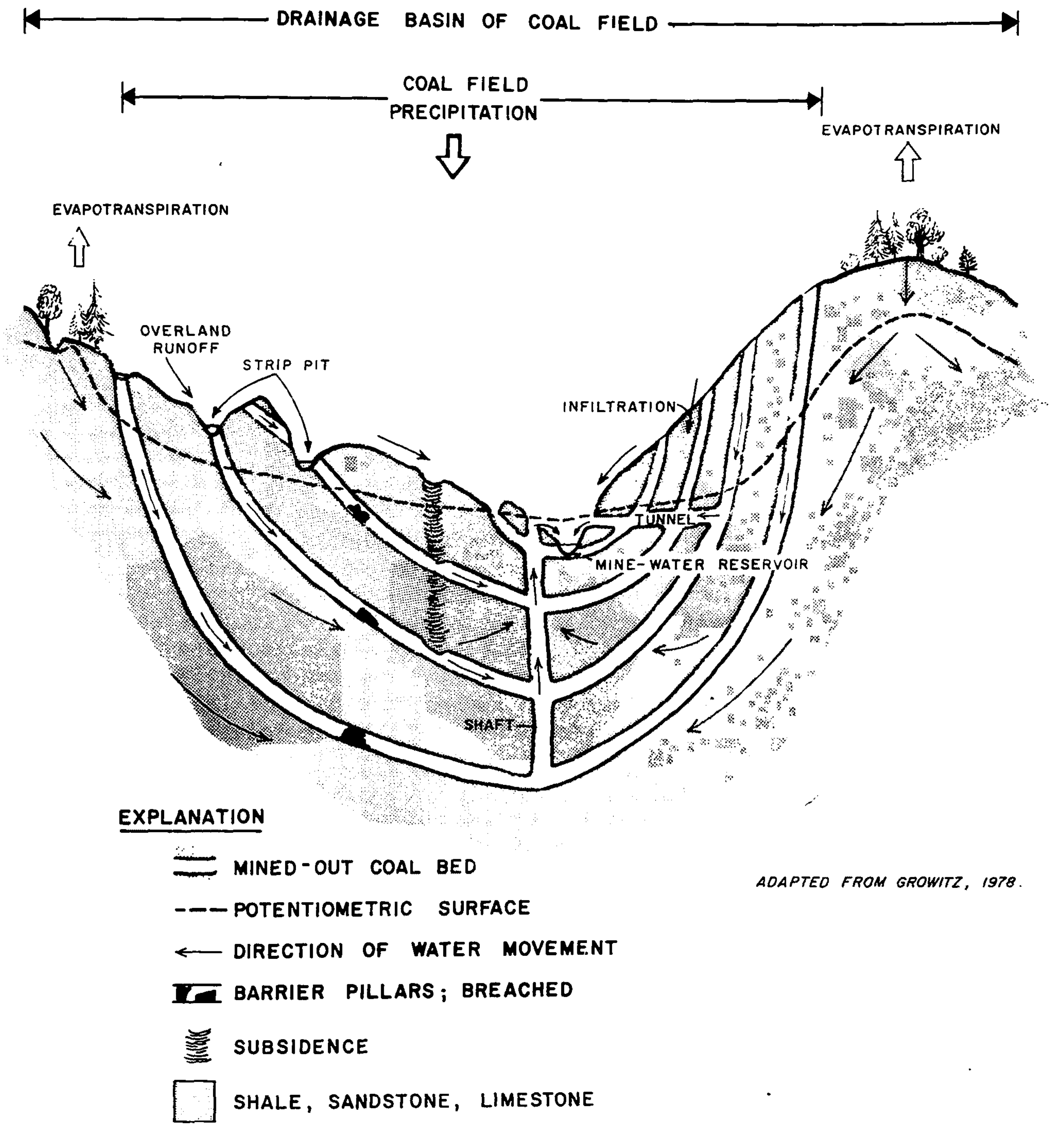


Figure 51. Schematic cross section showing the hydrologic cycle and flow patterns in an idealized anthracite coal basin.

since this method allows for controlled roof collapse during mining. Davies (1968) reports that the amount of surface subsidence in the Appalachian Basin is approximately 50 percent of the thickness of material removed from the mine when roof failure occurs at depths of 46 m (150 ft) or less. Surface subsidence is 25 to 30 percent of the thickness when roof failure is at depths of about 91 m (300 ft).

Any assessment of the extent of subsidence in the eastern coal fields is dependent on the reliability of reported data, which varies widely among the states. Generally, problems in the more populated urbanized portions of the study region are recognized to a much greater degree than in rural areas. The U. S. Bureau of Mines (1969) has estimated that, of the 8 million acres of land underlain by mines in the United States, more than 2 million acres have subsided. Some 1.9 million acres of land is over bituminous mines and most of this is in the study area. Anthracite mines alone have caused subsidence in an area of about 90,000 acres. Widespread subsidence has occurred in Pennsylvania and Illinois, relatively serious subsidence in localized areas of West Virginia, and isolated incidents are reported in Maryland (Singer, 1977).

One of the regions most impacted by subsidence is the Scranton/Wilkes-Barre area in the northern part of the Pennsylvania anthracite fields. A survey of subsidence in this area identified 726 incidents from 1947 to 1973 (Martin, 1974). Subsidence has also occurred over a wide area in the bituminous coal regions of Pennsylvania. Studies in Washington, Allegheny, and Westmoreland Counties (Bushnell, 1975) have shown that subsidence correlates with extensive mining of the Pittsburgh and Upper Freeport Coals. Subsidence began occurring locally more than 30 years after mine abandonment in these counties. Apparently 46 to 61 m (150 to 200 ft) of overburden is a critical depth in these counties, because very few cases of subsidence damage have been reported in areas with more than 61 m (200 ft) of overburden (Bushnell, 1975). In deeper mines, fracturing of strata overlying mined-out areas may affect groundwater flow, but may not be indicated by subsidence of the land surface.

In Illinois, mine subsidence was not considered a serious problem until recently, but the deterioration of mine supports and the spread of urbanization into areas underlain by mines is currently causing serious subsidence problems (Singer, 1977). Problems have arisen in St. Clair and Madison Counties, which contain the eastern suburbs of St. Louis. Although mining terminated there in 1970, coal had been mined by the room and pillar method for more than 100 years. As deterioration of mine supports continues, problems are also likely to develop in agricultural areas throughout central Illinois (Glover, 1977).

Mine subsidence damage in West Virginia has been limited largely to the northern part of the state. Reported subsidence incidents exist primarily in Harrison and Madison Counties (Wilson, 1976, and Gilley, 1977).

Major subsidence problems in Alabama are found in karst limestone areas rather than in coal regions. Several minor subsidence incidents have been reported in Jefferson County (Zabro, 1978). Reported cases of subsidence in Ohio have also been relatively minor, although some problems have occurred in the Canton-East Liverpool area (Delong, 1978). Subsidence in Virginia is reportedly a minor problem (Linkous, 1978).

Subsidence-related mechanisms can significantly alter the groundwater flow in the vicinity of underground mines. For example, subsidence and/or mine roof collapse (Figure 52) can convert formerly impermeable overlying strata into permeable zones that permit entry of water into mines. As discussed in the following sub-section entitled Mine-Water Discharge, another common mechanism by which water may enter mines is by interconnection with other underground mines or surface mines.

MINE-WATER DISCHARGE

A brief review of discharges from active and abandoned mines leads to two general conclusions: (1) flows from different mines in the same region can display a wide range in rates from virtually zero to hundreds of thousands of cubic meters per day, and (2) in many places large volumes of ground water are being diverted from natural flow systems above and adjacent to mines. Some of the important variables controlling water in mines are geologic structure, including fracture and joint concentration; mine roof rock type; type of openings; dip of coal seam; proximity to other mines; and type of mining. An examination of more than 250 eastern coal mines (Crichton, 1928) showed that the average rate of infiltration for all mines was approximately 10 $m^3/d/ha$ (1,100 gpd/acre). Parizek (1970) estimated the quantity of water leaking from the roofs of underground mines to be 6.3 $m^3/d/ha$ (670 gpd/acre).

Flow from 18 abandoned underground mines in Maryland has been related to the area of mine workings (Hollyday and McKenzie, 1973). Figure 53 indicates that in this region there may be a linear relationship between area of mine opening and mine discharge. The average infiltration rate for these mines is approximately 215 $m^3/d/km^2$ (148,000 gpd/mi^2) or 2.2 $m^3/d/ha$ (231 gpd/acre).

A survey of 59 active underground mines in the Monongahela River Basin in West Virginia showed that 48 had mine discharges (U. S. EPA, 1973). The total flow to surface-water bodies within the basin was about 329 x 10^3 m^3/d (87 mgd) from the mines studied, exclusive of refuse pile drainage. Drainage from mines in the Pottsville and Allegheny Groups, the principal coal-bearing rocks in the southern part of West Virginia, averaged about 19 $m^3/d/ha$ (2,000 gpd/acre) of coal mined in 1953 (Doll and others, 1963). Gravity drainage from several thousand abandoned mine openings in West Virginia in the late 1930's was estimated to be 2,119 x 10^3 m^3/d (560 mgd). It is likely that drainage has increased since this estimate was made because additional large areas have been mined, and only limited sealing has been accomplished. Some of the higher flow rates from active bituminous mines in Pennsylvania have been summarized in Table 4. Somerset, Cambria, Washington, and

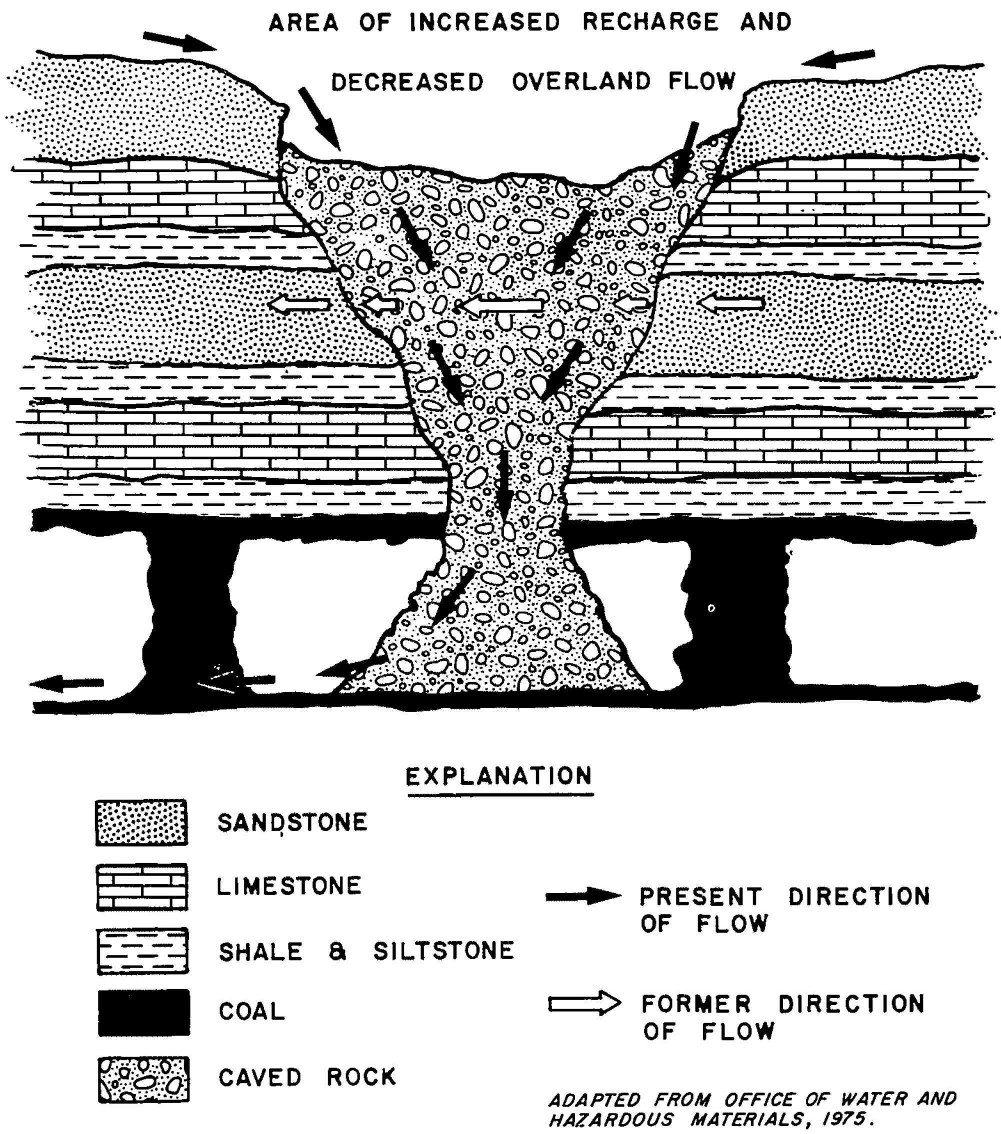


Figure 52. Idealized section showing increased infiltration of water and changes in ground-water flow directions in subsided area.

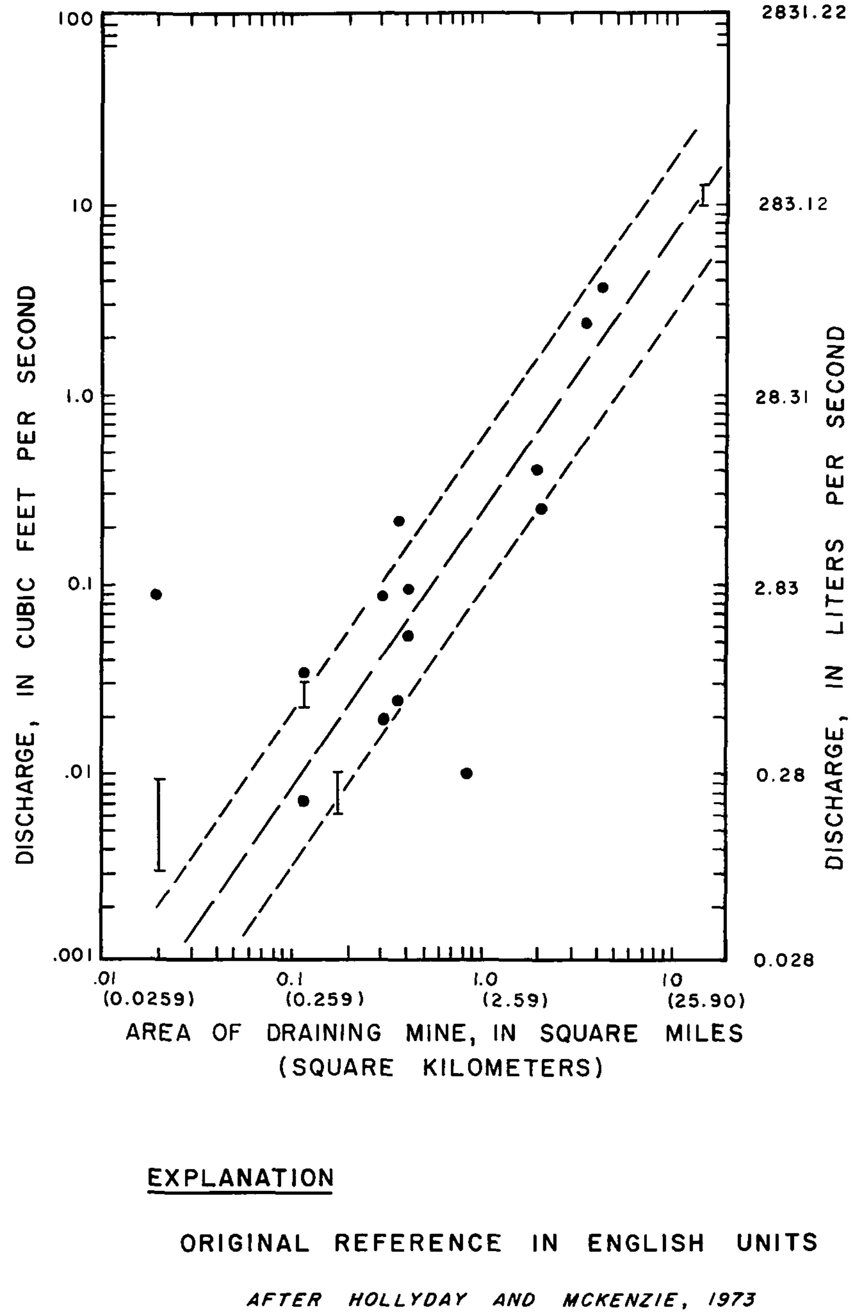


Figure 53. Relation of measured discharge of mines to area of mine workings in 18 underground mines in Maryland.

TABLE 4. SELECTED DISCHARGES FROM ACTIVE UNDERGROUND COAL MINES IN WESTERN PENNSYLVANIA*
(Source: Data from files of Pennsylvania Dept. of Environmental Resources.)

County	Range of discharge flow (m^3/d)	Number of discharges ($>38 m^3/d$)
Somerset	38 - 26,495 (10,000 - 7,000,000 gpd)	9
Cambria	53 - 24,600 (14,000 - 6,500,000 gpd)	7
Centre	15,140 (4,000,000 gpd)	1
Allegheny	38 - 1,515 (10,000 - 400,000 gpd)	7
Armstrong	190 - 6,435 (50,000 - 1,700,000 gpd)	5
Beaver	3,030 (800,000 gpd)	1
Fayette	11 - 680 (3,000 - 180,000 gpd)	1
Greene	5.3 - 3,405 (1,400 - 900,000 gpd)	13
Indiana	23 - 4,540 (6,000 - 1,200,000 gpd)	13
Washington	110 - 2,840 (29,000 - 750,000 gpd)	14
Westmoreland	1,630 - 22,710 (430,000 - 6,000,000 gpd)	3
Clearfield	16 - 530 (4,300 - 140,000 gpd)	5

* Discharges are recent instantaneous readings, not average values.

Westmoreland Counties each have several mine discharges exceeding 3.8×10^3 m^3/d (1 mgd).

Coal mines in Illinois and Kentucky tend to have relatively low inflows of ground water. Many of these mines are characterized by soft shale roof rocks rather than sandstone. The majority of mines in Illinois report pumpages of less than 100 m^3/d (26,400 gpd) (Cartwright and Hunt, 1978). Where water does enter mines, the mines are generally either quite shallow or are near buried channel deposits consisting of sandstone or unconsolidated deposits (Cartwright, 1977). Water entering deep mines at depths greater than 244 m (800 ft) is frequently brackish or salty (Gluskoter, 1977).

Ground water can be diverted from underground coal mines by a number of mechanisms which involve interaction with other underground mines or surface mines. A problem with surface mining up dip from a coal outcrop in the vicinity of underground mines is the possibility of leakage from the underground mine around or through coal barriers. Leakage of this type may result in the release of large quantities of poor quality, mine-pool water to other mines or to the land surface. Such a situation occurred near Russelton, Pennsylvania, where mine-pool water having a high iron content seeped through coal barriers from abandoned mines into the active Harmer Coal Mine. The operators of the active mine were required to treat the poor quality ground water that was released from storage (Thompson, 1977).

Surface mining progressing down dip from a coal outcrop can cause water to flow into an underground mine that is lower than the surface mine elevation. Under recharge conditions, water accumulating in strip pits or through surface mine refuse can enter deep mine workings. In some places, surface mine floors have been blasted in order to facilitate drainage, or have been perforated by open drill holes that later unintentionally serve as recharge wells. In other cases, the underclay has been stripped during the mining operation and more permeable strata are exposed (Parizek, 1978). Conditions in the Big Laurel Run Mine in Maryland provide an example of an underground mine that is receiving drainage from a large strip mine. As a result, drainage from this mine is several orders of magnitude greater than the average discharge of other mines in the vicinity having the same drainage area (Hollyday and McKenzie, 1973).

Interconnections between underground mines are responsible for numerous instances of mine water transfer from abandoned mine workings to active mines. Such cases result in temporary or permanent loss of mine-pool and ground-water storage. For example, the Banning No. 4 Mine in Westmoreland County, Pennsylvania, is located in the lower part of the Irwin Syncline and receives mine water from surrounding abandoned underground mines. The mine, which is in the Pittsburgh Coal, is the last active one in an area that has been mined since the mid-eighteen hundreds. Prior to abandonment of the adjoining Hutchinson Mine, the volume of the mine pool was estimated to be 1.3×10^7 m^3 (350 billion gallons). Pumpage from the Hutchinson Mine was more than 11,000 m^3/d

(3 mgd) in 1969. About 6,400 m^3/d (1.7 mgd) of this flow was estimated to be from abandoned mine workings to the north (Thompson and Emrich, 1969). Detailed studies indicate that many underground mine barriers have been breached and flow will increase in the future.

Abandoned mines in the Pittsburgh and Sewickley coals between Frostburg and Midland, Maryland, are drained by an underground tunnelling system. A 3,245-meter (10,646-foot) tunnel (Hoffman Drainage Tunnel) was drilled from a tributary of Wills Creek to the lowest workings of the Pittsburgh Coal. Mine workings were then connected to the tunnel. Where the tunnel is overlain by Georges Creek, seepage from this creek may constitute a significant part of the flow of the Hoffman Drainage Tunnel (Hollyday and McKenzie, 1973). Effects on ground-water levels have not been specifically studied but probably are significant.

Another factor involved in mine-water problems is the type of underground mining used. Entrance of water into mines is increased when the roof rock is dropped in longwall and retreat mining, and subsequent fracturing develops (Edmunds, 1977). Although most of the water entering underground mines enters through roof rocks, water can flow upward through the underclay floor in some mines, especially deep ones. If sufficiently high hydraulic heads are developed after abandonment and the mine is flooded, water will flow downward through the underclay. Conventional mining does not normally disturb the underclay; in some low seams, however, the underclay or portions of the roof rock may be removed in order to provide more working space (Boyer, 1976). In addition, in some places, hydrostatic and rock pressure in deep mines may cause the floor to buckle and heave up to the roof. This has occurred in the Pittsburgh seam in southwest Pennsylvania (Kebblish, 1977).

WATER LEVELS, WELL YIELDS, AND STREAMFLOW

Alteration of natural ground-water patterns by underground mining is commonly indicated indirectly by changes in water levels in aquifers, by the amount of ground-water seepage to streams, and by ground-water quality. The scanty existing information shows that, in many places, water levels in supply wells have been affected by mining, but the problem is not universal. The lack of documentation on these types of impact is most likely due to the low population density, the small number of wells in rural areas, and the slow rate of change in water levels. In addition, small gradual declines in water levels in domestic or public supply wells are rarely noticed until yields markedly decrease. The majority of such problems have been reported in states in the northern part of the Appalachian Basin.

Active mines as well as abandoned mines with natural drainage serve as hydraulic sinks or discharge zones that function like large horizontal pumping wells. Consequently, water levels in aquifers above and adjacent to the mines decline in response to the loss of water. The potentiometric levels below the mines also will decline, but to a lesser degree, in response to reduced natural recharge or to upward leakage into the mine. Where mines are sealed or become filled naturally after abandonment, water levels tend to recover substantially but not necessarily to original levels. Piper (1933) reported that the Pittsburgh Sandstone and its equivalents have been essentially dewatered wherever the underlying Pittsburgh coal has been mined and mine roofs have collapsed. Similar dewatering occurs above the Upper Freeport coal.

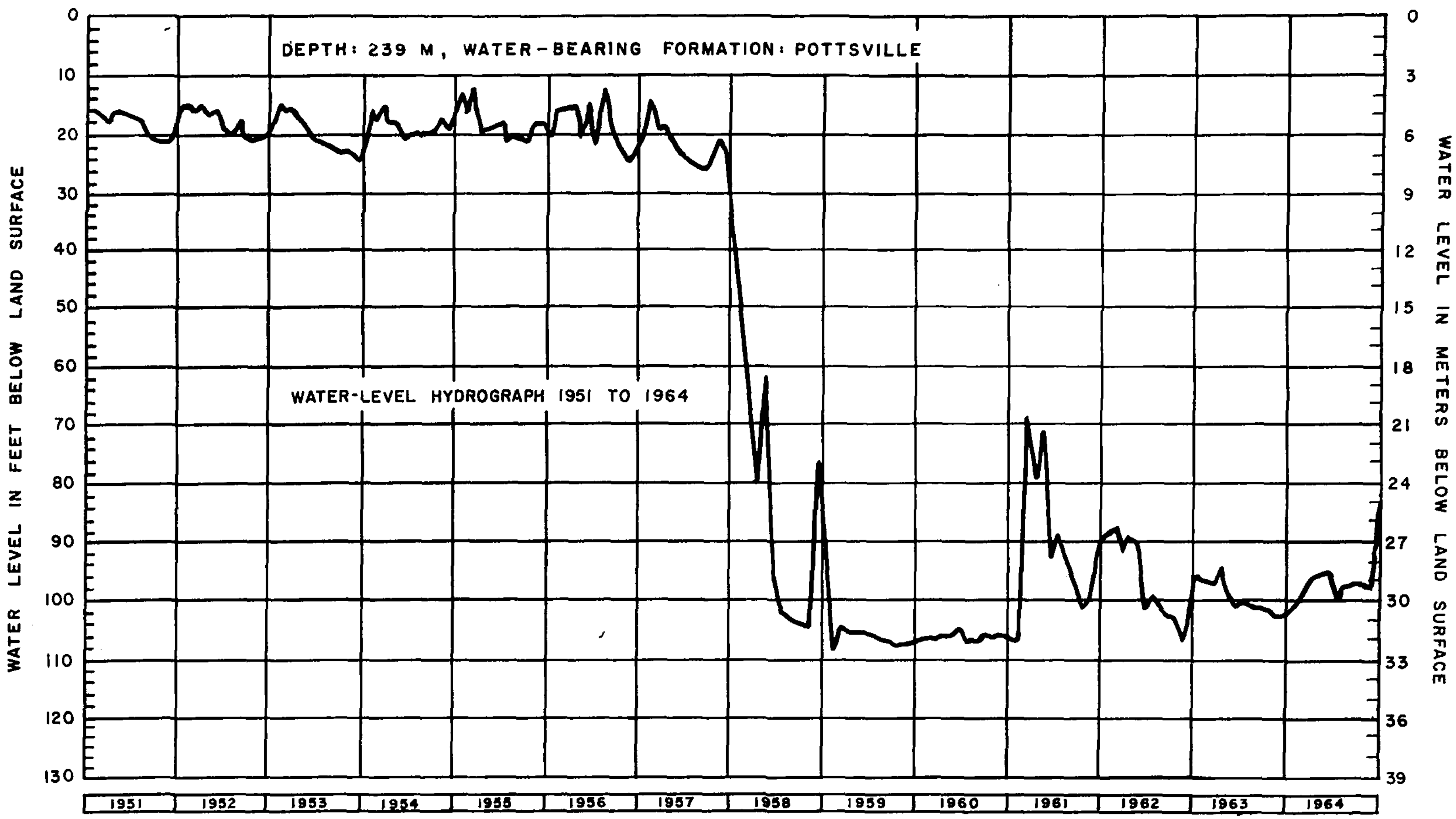
In West Deep Township, Allegheny County, Pennsylvania, drying up of wells was reported as early as 1933 in the vicinity of Russelton and again in 1970-71, when static water levels in approximately 20 wells in the northwestern part of the township dropped below the bottom of the wells (Subitzky, 1976).

At Flemington and Fairview in Marion County, West Virginia, partial dewatering of aquifers by mining activities has reportedly caused some wells to go dry (Ward and Wilmoth, 1968). Some well owners reported that the water levels in their wells recovered to the approximate premining levels about 12 to 15 years after mine pumping stopped. Water-level fluctuations due to mining near Masontown in Preston County, West Virginia, are indicated by the hydrograph in Figure 54. The sharp decline of water levels in 1958 is attributed to mining. Mining activity decreased shortly thereafter and, from 1961 to 1964, the water levels show a slight recovery.

An inventory of water levels in domestic wells in Marion County, West Virginia, showed that variations in long-term dewatering are related to mine depth (Rauch, 1978). The results are summarized as follows:

Mine depth		Effect on wells completed above mining zone
(m)	(ft)	
<61	<200	All wells permanently dewatered
61-76	200-250	Most wells permanetly dewatered
76-91	250-300	Some wells occasionally dewatered
>91	>300	No wells dewatered

Rauch (1978) also cites several other mechanisms by which water levels may decline locally as an indirect result of underground mining. For example, blasting in a strip mine in Marion County apparently caused subsurface fracturing and subsequent dewatering of wells that tapped aquifers above an existing deep mine. Also, vertical air shafts sunk for underground mines may receive as much as 19 to 25 l/s (300 to 400 gpm) of ground water which is commonly diverted to streams. Water rings (used in the construction of air shafts) which are not grouted result in the greatest water-level effects, with some declines being noted in wells as much as 1.61 km (1 mi) away.



AFTER WARD AND WILMOTH, 1968

Figure 54. Water-level fluctuations in an observation well near mining operations in Preston County, West Virginia.

A review of legal cases in eight states indicates that the effects of declining water levels in wells and reduced spring flow attributable to underground mining lead to occasional problems (Table 5). The majority of cases involve very abrupt and obvious declines. Doubtless there are other cases where gradual declines occur, but in these cases, it is difficult to correlate mining activities with the water-level declines.

Changes in ground-water flow can also have an impact on streamflow rates. Assessing changes in stream hydrology as a result of mining is complicated by the presence of surface mined areas, active and abandoned underground mine discharges, seepage from refuse piles and surface impoundments, as well as ground-water seepage to streams. Although a number of river-basin studies have addressed some of these variables, few published reports have dealt with changes in the baseflow of streams as a result of underground mining. In addition to impacts on flow rates, changes in ground-water flow can also affect stream quality as discussed in a later section of this report.

The U. S. Geological Survey in West Virginia is presently investigating the effects of deep mining and mine collapse on hydrology. Preliminary results of a detailed study of the hydrology of the Buffalo Creek and Indian Creek subbasins in northern West Virginia indicate that baseflow from mined-out areas is greater than that from unmined areas (Hobba, 1978). Typical yields from rocks in undisturbed basins were about 233 $m^3/d/km^2$ (160,000 gpd/mi^2) and yields from mined basins were about 509 $m^3/d/km^2$ (350,000 gpd/mi^2). The areas investigated also had land-subsidence features and, in one case, water levels in one well apparently had fallen as much as 15 m (50 ft) below pre-mining levels. In addition, the water levels fluctuate substantially in response to storm events, indicating rapid infiltration and discharge.

Parizek (1979) reports that in western Pennsylvania cases of both reduced baseflow and major gains of ground-water inflow are known. In Clinton County, in Kettle Creek, Crawley Run, and Cooks Run watersheds, small tributary streams that existed prior to deep mining have become intermittent.

The findings of those studies showing increased in baseflow are different from those which might be expected if one were to assume a simple hydrologic system consisting of recharge from the surface, relatively rapid percolation to the mine, and direct discharge of mine drainage to a stream. Such a system would presumably reduce recharge to underlying strata and, thereby, result in reduced baseflow in the stream. Possible explanations for an increase rather than a decrease in baseflow are: (1) more rapid movement of mine waters through and around mines, as well as above mines, as a result of fracturing of surrounding rocks, (2) shifts in ground-water divides caused by disturbance of strata resulting in increased capture of precipitation, and (3) reduction of evapotranspiration because of a decline of the water table.

TABLE 5. CLASSIFICATION OF LEGAL CASES INVOLVING COAL MINING AND RELATED WATER PROBLEMS*

State	Type of problem: Spring or well-water quality	Spring flow or water levels in wells	Surface water
Alabama	2	5	8
Illinois	0	0	4
Kentucky	1	4	3
Maryland	0	0	0
Ohio	0	0	3
Pennsylvania	0	7	4
Virginia	1	5	3
West Virginia	0	0	3

* Cases reviewed are those from courts rendering written opinions and in general only include State Supreme Courts, District Appellate Courts, and Federal Circuit Courts.

Growitz (1978) found that streamflow is significantly affected by underground mines in the eastern middle and western middle anthracite fields of Pennsylvania. He noted that streams typically lose water in their headwater reaches in the coal fields and gain water along their lower reaches where most of the mine water is discharged. Overall, streamflow per unit area of drainage basin downstream from some anthracite fields is significantly higher than unit streamflow from adjacent unmined areas. This may be attributed to a reduction in evapotranspiration and, consequently, an increase in infiltration of precipitation in the mined area (Growitz, 1978).

SECTION 8

EFFECTS OF UNDERGROUND MINING ON GROUND-WATER QUALITY

GENERAL GEOCHEMICAL RELATIONSHIPS

The geochemical reactions which occur in the formation of coal mine drainage have been studied extensively. The commonly postulated equation for the oxidation of pyrite in the presence of atmospheric or dissolved oxygen in water is:

$$2\ FeS_2 + 7\ O_2 + 2\ H_2O = 2\ FeSO_4 + 2\ H_2SO_4$$

Despite the body of knowledge that exists on the subject, there are still some uncertainties over the exact mechanisms which cause or accelerate this reaction. Factors such as iron- or sulfide-reducing bacteria, available alkalinity, presence of water, oxygen partial pressure and the nature of pyrite have been cited as factors influencing the formation of mine waters.

Sulfide minerals oxidized according to the above reaction will produce acid and will result in high concentrations of sulfate and ferrous iron. Such acid waters moving in natural environments will accelerate the breakdown of clay, other silicate minerals, and carbonate minerals. This increases concentrations of silica, aluminum, calcium, magnesium, and manganese. Levels of iron, aluminum, and manganese are affected by the pH of the solution. In addition, trace elements such as zinc, copper, nickel, cobalt, and chromium can be far above normal background concentrations in acid waters.

As the time and distances over which water travels in the subsurface become longer, mineral-water relationships become increasingly important. For instance, acid ground waters passing through sediments containing calcareous materials become increasingly hard and may eventually become a near neutral calcium sulfate solution.

Studies in the Clarion River/Redbank Creek Watershed of northwestern Pennsylvania indicate the significance of geochemical controls in the occurrence of major and minor chemical constituents in ground water (Gang and Langmuir, 1974). The solubility of iron was determined to be largely controlled by precipitation of oxyhydroxides. In those ground waters that contain measurable alkalinity, the solubility of siderite (iron carbonate) commonly limits ferrous iron concentrations. Alkalinity, in turn, is controlled in part by the presence of calcareous sediments. Consequently, in areas where limestone is relatively abundant, ferrous iron concentrations are much lower in the ground water than in areas where limestone is scarce or absent.

Both silica and aluminum in ground water associated with mining are derived from the leaching of clay by acid mine water. Gang and Langmuir (1974) point out the importance of limestone in controlling aluminum concentrations. They observed that in subbasins with abundant limestone, very low aluminum concentrations and relatively high pH were found both in the ground water and streams. Zinc, nickel, copper, and cadmium, which are normally the most abundant trace elements in acid mine drainage, were also low (Gang and Langmuir, 1974).

Natural Water Quality

An important consideration in assessing the potential impact of mining on ground-water quality is the natural background quality prior to mining. An important consideration in assessing the potential impact of mining on ground-water quality is the natural background quality prior to mining. A detailed discussion of natural ground-water quality and variability is beyond the scope of this report. However, a study of the nation's ground-water quality recently completed by Pettyjohn, et al (1979) provides state-wide summaries of key constituents in ground water. Selected maps from this study are given in Appendix B of this report. These maps are necessarily very generalized and also tend to present the most favorable picture of water quality, since the samples collected were taken primarily from water-supply type wells. Table 6 gives mean concentrations of key constituents in coal counties of states where information was available.

A review of maps presented in Appendix B indicates that on the scale of these maps and for the constituents analyzed that the percentage of ground water presently being sampled in the coal regions of the eastern United States that meets EPA Interim Secondary Drinking Water Standards is variable from state to state. The standards for the constituents analyzed are chloride, 250 mg/l; sulfate, 250 mg/l; and total dissolved solids, 500 mg/l. Most states have at least one area within their respective coal regions that yields poor quality ground water. These areas include most of southern Illinois, much of southwestern Indiana, much of western Kentucky and a small part of eastern Kentucky, most of southeastern Ohio, a small section of southwestern Pennsylvania, a portion of central Tennessee, and parts of northern West Virginia.

Highly mineralized shallow ground water occurs naturally at depths of less than 91 m (300 ft) in parts of every state studied, especially in Illinois and western Kentucky. As noted in a previous section, brine or salt water is pumped from many mines in Illinois during active mining. In western West Virginia, mineralized ground water (chloride content greater than 250 mg/l) is commonly found at depths of less than 91 m (300 ft) beneath the Ohio, Kanawha, Big Sandy, Coal, and Little

TABLE 6. MEAN CONCENTRATIONS OF KEY CONSTITUENTS IN THE GROUND WATER OF COAL BEARING COUNTIES OF SELECTED STATES*

(Source: Pettyjohn, et al., 1979)

County	Ca & Mg	Na & K	Cl	SO_4	Dissolved Solids	Hardness
ALABAMA						
Fayette	28	44	41	5	86	16
Jefferson	42	4	4	9	130	108
KENTUCKY						
Bell	34	35	11	23	196	94
OHIO						
Jefferson	89	32	18	92	N/A	254
PENNSYLVANIA						
Allegheny	61	48	23	97	414	180
WEST VIRGINIA						
Boone	47	79	32	40	147	99
Logan	17	82	17	45	253	73
Marion	83	6	11	19	245	216

* Concentrations in milligrams per liter

Kanawha Rivers (Wilmoth, 1975). Generally where major streams serve as natural discharge zones for deep ground water, the depth to the fresh-water/salt-water contact is less beneath these streams than elsewhere in the region.

Davis (1973) found that water from shallow bedrock aquifers in southern Illinois is poorest in an area between the southeastern edge of the Salem-Woodlawn fault block and the Wabash Valley Fault Zone (Figure 55). This large area of poor quality water is probably being contaminated by upswelling brines. Ring averages of total dissolved solids range from 500 to 2,500 mg/l. Sulfate averages range as high as 1,400 mg/l. Ring averaging is a method of computing running averages for the three-dimensional case.

Upward migration of naturally salty ground water can also result from man's activities. During petroleum exploration, deep mineralized ground water may move up locally into shallower aquifers through unplugged oil and gas wells. Large withdrawals of shallow fresh ground water can also induce upconing of salty water. In the Charleston area of Kanawha County, West Virginia, mineralized ground water has moved upward as much as 23 to 30 m (75 to 100 ft) below major pumping centers. The chloride content of water in the principal sandstone aquifer at Charleston has increased from less than 100 mg/l to more than 300 mg/l in several well fields, and to more than 1,000 mg/l in water from individual wells (Wilmoth, 1975). Similar case histories have been reported in almost every state in the study area. Although the effect of mine pumping or drainage on the movement of the fresh-water/salt-water interface has not been specifically investigated, it is conceivable that prolonged discharge of large amounts of water from mines could cause upward movement of salty ground water.

Influence of Paleoenvironments on Water Quality

Recent studies indicate that the depositional environment of coal and the amount of calcareous material in strata associated with coal are major factors in determining the quality of drainage from coal mines (Williams and Keith, 1963; Caruccio, 1968, 1970; Ferm and Caruccio, 1974; and Caruccio and others, 1977).

Regional sedimentation during the middle to upper Pennsylvanian Period in the central Appalachians has been characterized as being controlled largely by transgressing and regressing deltaic systems. Marine limestones represent offshore carbonate tidal islands and barriers, and the associated red and green shales were derived from oxidized clay deposited on the floor of lagoons. Sandstone and orthoquartzite were apparently derived from bay barrier deposits; these rocks commonly grade into dark gray shales or back barrier sediments. Coal-bearing strata, dark shales, and graywacke sandstone represent fluviodeltaic depositional environments (Ferm, 1974).

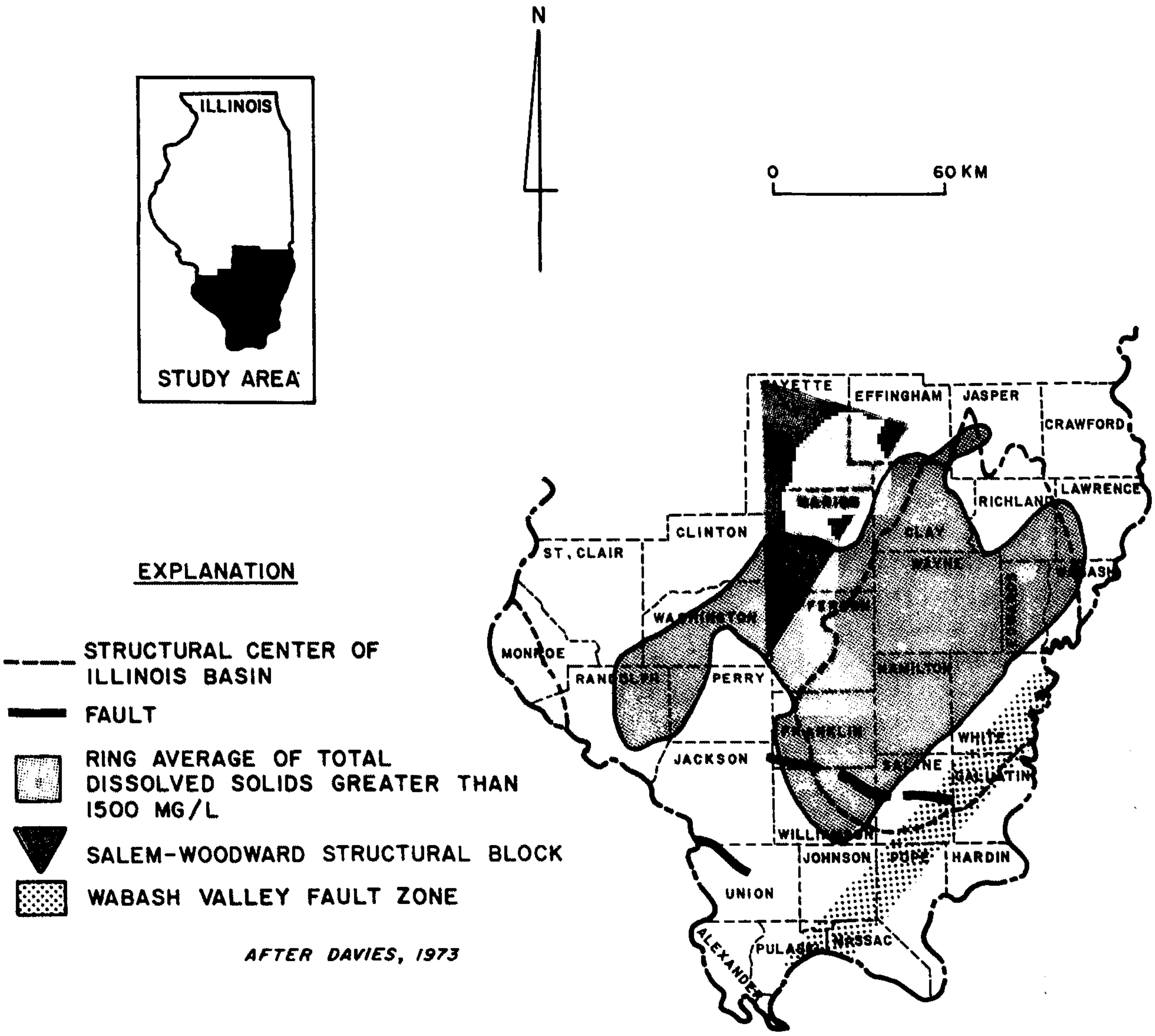


Figure 55. Poor quality shallow ground water in southern Illinois.

Detailed studies by Williams (1960) in western Pennsylvania provide an illustration of a transgressive paleoenvironment in the Allegheny Group of Pennsylvanian age. In that geologic unit, the rocks grade upward from marine to continental in origin. In addition, Ferm and Williams (1960) showed that this transgression is expressed as lithologic facies changes that grade from marine deposits in western Pennsylvania to brackish water deposits farther east.

Studies by Caruccio (1968 and 1970) show that the mode of occurrence of pyrite within coal strata is of greater importance than the total sulfur content in determining the production of acid by leaching. Framboidal pyrite, attributable to primary deposition, was found to result in much higher acid production than secondary pyrite. It is believed that coals associated with lower deltaic or marine environments contain pyrite of a more reactive nature than that in coals deposited in a fluviodeltaic or more continental environment.

These observations on acid mine drainage production are supported by the results of laboratory leaching and field studies in eastern Kentucky (Caruccio and others, 1977). The following is a summary of the key findings of those studies:

(1) Reactive pyrite (framboidal pyrite) is associated with back barrier/lower delta plain coals. Lower amounts of pyrite are present in seams within strata of upper delta plain origin.

(2) Where water in sediments deposited in an upper delta plain paleoenvironment is in contact with calcareous material, the sediments generate sufficient alkalinity to effectively neutralize the acidity produced. Therefore, mine drainage from these deposits is typically nearly neutral (about pH 7.0) and has a high specific conductance and sulfate content.

(3) The quality of mine drainage is related to the geochemistry of the natural water and to the occurrence of framboidal pyrite. Both may vary in relation to the occurrence of coals and strata in different environments.

A comprehensive regional water-quality study by Hornberger (in progress, 1978) deals with similar geochemical concepts in southwestern Pennsylvania. The study was based on chemical analyses of water leached through coals and on the quality of surface and ground water in the vicinity of coal mines. In addition, water-quality data from the outcrop area of the Pottsville and Allegheny Groups were evaluated with respect to the type of environment under which the coal was deposited, and to the presence or absence of carbonate strata. Preliminary findings of Hornberger's study indicate that:

(1) Although the total sulfur content of the Lower Kittanning coal from various mines tends to vary with the type of paleoenvironment (more in marine coals and less in fresh-water coals), there is little or no relationship between the degree of acidity in mine-water discharges and

the paleoenvironment. A number of discharges from mines in fresh-water coals were acidic in character. The rocks associated with the Lower Kittanning coal generally have a low calcareous content. Leaching studies indicate that there are wide ranges in the acidity of mine water from individual mines. This makes it difficult to make valid comparisons between mine types without a large number of samples.

(2) In general, there is good correlation between mines with little or no acid mine discharge and the presence of calcareous sediments in the overburden.

(3) Where mining operations are sparse, surface- and ground-water quality are better than in heavily mined areas.

Although the studies by Caruccio and others (1977) and by Hornberger (1978) agree on several important points, the latter study concluded that the paleoenvironment, in itself, is not a prime factor in controlling the severity of acid mine drainage. Both studies concur that where strata containing significant amounts of calcareous material are associated with coal, acid drainage is usually not produced.

EFFECTS OF MINING OPERATIONS

Regional studies dealing with water quality in mined areas are concerned largely with effects on stream quality and not with ground-water quality. This is primarily because the effects of acid mine drainage on stream quality and biota are obvious, and because of the use of streams for public supply and industrial purposes. Changes in ground-water quality are usually reflected in changed stream quality especially during low-flow period. During this time, surface runoff is minimal, and the percentage contribution of ground water is highest. However, stream quality during low-flow periods is also affected by mine discharges. On a regional basis, it is difficult to separate out how much of the quality changes in streams are due to the inflow of naturally mineralized ground water, how much is due to inflow of ground water that is chemically altered by mining, and how much is due to mine discharges. Detailed watershed studies are necessary to separate out the importance of each of these components. In most parts of the study area, including mined and unmined areas, the dissolved solids in streams generally increase as the baseflow increases. Examples of these conditions are found in the Guyandotte River in southern West Virginia (mostly underground mines), Tobey Creek in northwestern Pennsylvania (mostly surface mines), and in the Big Muddy River in southwestern Illinois (surface and underground mines).

The transport of dissolved constituents in ground water is considerably more complex than their transport in drainage from mine portals. As previously discussed, the potential for ground-water flow in and around mines depends on local geologic characteristics, the physical setting of the mine, and the method of mining. Ground-water quality further depends not only on geologic and hydrologic controls but also on various geochemical factors.

In a study of ground water in the Monongahela River basin, Ward and Wilmoth (1968) report that acid sulfate water from underground mines adds chemical contaminants to some aquifers. However, such contamination apparently is not widespread at present according to the results of analyses from the small number of existing wells.

Carlston (1958) found a wide range in concentrations of dissolved solids in ground water in Monongalia County, West Virginia, a heavily mined area. The concentrations ranged from 42 to 2,530 mg/l and pH values were near neutral in most samples. The highly mineralized ground water was attributed in part to widespread contamination by salt water from oil wells as well as to mining. A more recent study in this same county by Hilgar and Rauch (1979) concluded that wells and springs located within about 200 ft of a polluted stream or within about 800 ft of a mine are commonly contaminated with mine drainage, whereas more distant water supplies are of good quality (with less than 100 mg/l sulfate).

Gang and Langmuir (1974) also found that one of the major sources of contamination in two subbasins of the Clarion River/Redbank Creek drainage basin in Pennsylvania was contaminated water from flowing abandoned oil and gas wells. In this case, however, the source of poor quality water was not deep brines, but shallow ground water whose quality had been altered by mining and transported upward through uncased wells.

As part of the present investigation, the ground water in Allegheny, Greene, Indiana, and Washington Counties in southwestern Pennsylvania was evaluated regionally for chemical-quality characteristics. This region was chosen because the area has been mined predominantly by underground methods for years, and Pennsylvania has the best ground-water quality records of any state in the Appalachian Basin. Analyses of water obtained from the Federal STORET system for more than 50 wells and springs in this part of Pennsylvania were evaluated to ascertain whether the mean, range, and variability of concentrations of selected chemical constituents might indicate effects of mining (Table 7). Most of the results are within the range of natural ground-water quality; but the maximum concentrations of constituents in several samples from Allegheny and Indiana Counties strongly suggest contamination. Specific site investigations would be needed to identify the causes of the contamination.

The majority of the ground-water analyses in the STORET system are from public water-supply wells which are generally in areas of high quality water. It is not likely, therefore, that a monitoring network based on data in the STORET system could readily detect ground-water contamination unless it were widespread.

An assessment of ground-water quality on a more detailed basis indicated quality degradation in a number of scattered locations in a nine-county area of southwestern Pennsylvania (Table 8 - Green International, Inc., 1976). Historically, most of the coal production in these counties has been by underground methods. Average concentrations

TABLE 7. STATISTICAL SUMMARY OF CONCENTRATIONS OF SELECTED CHEMICAL CONSTITUENTS IN GROUND WATER FROM FOUR HEAVILY UNDERGROUND MINED COUNTIES IN PENNSYLVANIA*

(All dissolved constituents in milligrams per liter)

Parameter	Minimum	Maximum	Mean	Standard deviation
Conductivity (µmhos)	141	2,800	627	372
pH	5.0	8.4	7.3	0.56
Total Hardness (as $CaCo_3$)	10	660	200	151
Calcium (Ca)	0.8	282	63	50
Magnesium (Mg)	0.3	370	19	50
Sulfate (So_4)	4.0	1,240	135	168
Iron (Fe)	0.01	9.1	2.0	2.2
Aluminum (Al)	0.05	1.2	0.38	0.42

* Data from STORET computer printout provided by Pennsylvania Bureau of Water Quality Management.

TABLE 8. AREAS OF SUSPECTED GROUND-WATER DEGRADATION DUE TO COAL MINING IN SOUTHWESTERN PENNSYLVANIA

(Source: Green International, Inc., 1976)

County and Township	Watershed	Average concentrations (mg/l; ph in units) Fe	SO_4	pH
ALLEGHENY COUNTY				
West Deer Township	W. Branch Deer Creek	6	922	3.83
Fawn Township	Bull Creek	50		
Frazer, Harmar and Springdale Townships	Little Deer Creek	22	1,307	6.71
Findley Township	Potato Garden Run	69	793	-
Penn, Patton, North Versailles, Mifflin & Pittsburgh Townships	Turtle Creek	21	433	-
ARMSTRONG COUNTY				
Madison and Washington Townships	Mahoning Creek and Allegheny River	34	665	3.05
Rayburn and East Franklin Townships	Cowanshannock Creek and Allegheny River	21	2,380	2.98
North and South Buffalo Townships	Allegheny River	82	2,631	3.45
BUTLER COUNTY				
Allegheny Township	Bear Creek	20	560	2.40
Connoquenessing Township	Little Connoquenessing Creek	14	475	3.60
Jackson Township	Connoquenessing Creek	7	560	-
FAYETTE COUNTY				
Springfield & Stewart Twps.	Trib. Youghiogheny River	25	3,312	4.07
INDIANA COUNTY				
Banks Township	South Brady Run	37	829	4.48
Cherryhill Township	Two Lick Creek	34	955	4.26
Brush Valley Township and Center Township	Little Yellow Creek, Brush Creek, Black Lick Creek, Two Lick Creek	64	1,714	3.86
WESTMORELAND COUNTY				
Sewickley, Rostraver, South Huntingdon Townships	Sewickley Creek	23	1,076	5.05
South Huntingdon Township	Sewickley Creek	-	3,075	4.02

of selected mining-related parameters indicate that the ground water in a number of townships is unmistakably contaminated. The data suggest that the more detailed a ground-water monitoring network becomes, the greater are the number of incidents of contamination that are identified.

Instances of contaminated ground water in aquifers below underground mines are not well documented. However, where mines are located in upland topographic settings and the coal crop is exposed along valley walls, acid discharges have been observed below the mines. These discharges can be 50 to 300 ft or lower in elevation when compared with the mine floor. Specific examples in Pennsylvania include mines in Kettle Creek and Cooks Run in Clinton County; mines near Philipsburg, Centre County; mines near Mederia and Pennfield, Clearfield County (Parizek, 1979).

Vertical hydraulic interconnection between aquifers and between mines and aquifers may be greatly increased where oil and gas wells have been drilled in the vicinity of underground mines. Such conditions are widespread in Pennsylvania and West Virginia. In Butler County, Pennsylvania, a combination of oil and gas wells, deep coal mines and strip mines provide a mechanism for upward movement of contaminated mine water (Thompson, 1972). Coal-waste wash water from strip mining and surface runoff enter deep mine pools through near-surface openings. The water moves through fractured rock into the lower parts of the mine where the head is sufficient to cause the ground water to rise to the surface through abandoned oil and gas wells and ultimately discharge to Slippery Rock Creek (Thompson, 1972).

In some cases, poor quality mine water may be induced into pumping cones of depression for wells completed above abandoned deep mines. Not uncommonly, good quality water may be pumped for a few hours to a month or more before mine water migrates toward the pumping center. This has been observed at the village of 6-mile Run and in Blue Spruce Park near Indiana, Pennsylvania (Parizek, 1979).

Hollyday and McKenzie (1973) have developed a hydrogeochemical model to explain the chemistry of ground water and mine drainage in the vicinity of abandoned underground mines in western Maryland. Geochemical diagrams in Figure 56 represent the variation in concentration of selected water-quality parameters in water sampled at various points within the flow system. Wells A and B yield water which is very low in dissolved solids. Water discharging at Site 1 is fresh ground water of a calcium bicarbonate type that has flowed through two unmined coal seams and is relatively unaltered. In the mine, the fresh ground water dissolves the iron and acid sulfates resulting from pyrite oxidation, loses its small bicarbonate alkalinity in neutralizing some of the acid, and becomes strongly acidic and enriched in iron sulfate. The acid in this mine water attacks minerals in the fallen roof rock and in the heaved underclay of the mine floor. Aluminum and silica are released from the clay, and calcium, magnesium, and bicarbonate are released from the carbonate rocks, along with significant concentrations of minor elements. Subsequent increases in pH are caused by loss of carbon dioxide to the

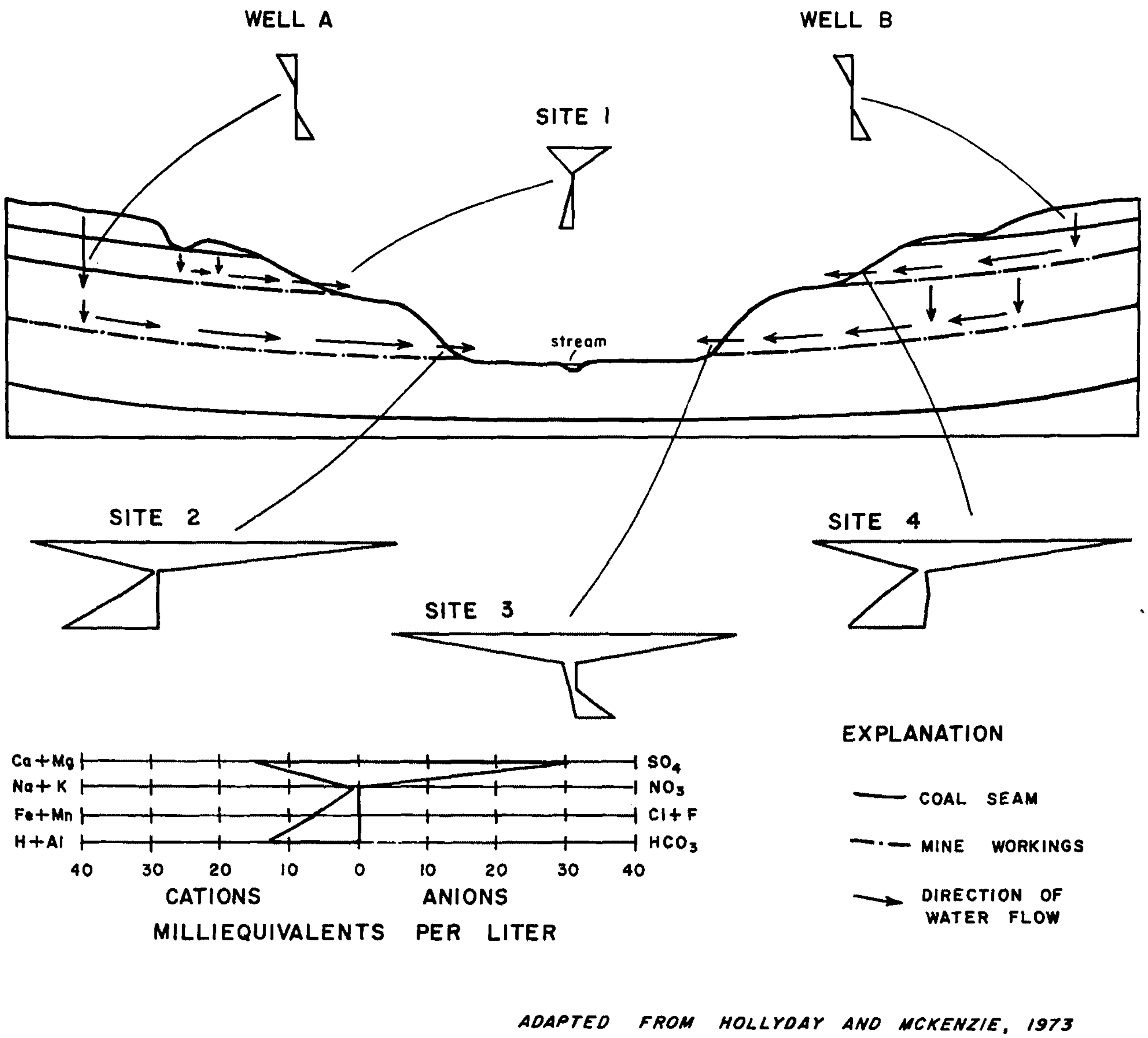


Figure 56. Schematic cross section showing water-quality differences near mine workings in western Maryland.

atmosphere, and by removal of ferric iron, aluminum, and minor metal ions by precipitation, and replacement by hydrogen ions (note diagrams at Sites 2 and 4).

Mine drainage at Site 3 is chemically similar to discharge water at Sites 2 and 4 except that the water is considerably more neutralized. In this model the ground water above the lowest seam is significantly altered chemically only where it is in contact with a mined-out seam. The model also indicates that passage of water through rock layers between mined-out seams helps neutralize poor quality water.

In studies of the anthracite fields of Pennsylvania, Growitz (1978) found that the dissolved solids in ground-water discharge (mostly mine drainage) is relatively low in regional upland areas and high in lowland areas. This situation is apparently related to the length of time the water has been in contact with the rocks. In contrast, the pH is lowest in recharge areas and highest in discharge areas. Growitz (1978) attributes this to the oxidizing conditions and reducing conditions in the respective hydrogeologic settings.

Ground-water quality in the Tom's Run drainage basin of Clarion County, Pennsylvania, has been degraded by mining activities (Emrich and Merritt, 1969). Although the mining is primarily by surface methods, the flow, geochemical mechanisms, and effects on water quality are similar to those produced by drift mines in the same setting. As seen in Figure 57, seepage through the mine floor travels to deeper aquifers along joints, fractures, and especially old abandoned gas and oil wells. The water then discharges at the land surface through springs and flowing abandoned wells formerly used for oil and gas exploration or production.

Similar results were obtained near Kylertown in Clearfield County, Pennsylvania (Brown and Parizek, 1971), where samples of ground water between the Lower Kittanning "B" Coal Seam and the top of the Connoquenessing Sandstone ranged in pH from 4 to 6 (Figure 57). Mine water and nearby stream water had a pH of 3. Because the strata in the study area contain little or no carbonate, either sufficient dilution is occurring to raise the pH or the deeper flow system is partly isolated.

A study of an active deep mine (152 m or 500 ft) in Alabama included sampling of ground-water inflow and outflow from the mine. Table 9 gives the mean values of the chemical and physical parameters of six samples of influent ground water, as well as the mean values for three samples of the effluent. Changes in concentrations of dissolved constituents between the influent and effluent are not significant. Other data from the same mine suggest that the bulk of the aluminum, iron, manganese, potassium, and much of the sodium in the effluent is in the suspended solids (clay, etc.) and is not in the form of dissolved ions in the water (Shotts and others, 1978). In addition, all the chromium, copper, lead, and nickel are in the suspended solids. The fact that neither the mine influent nor the effluent is acidic is probably the main reason for the low concentrations of trace elements in solution.

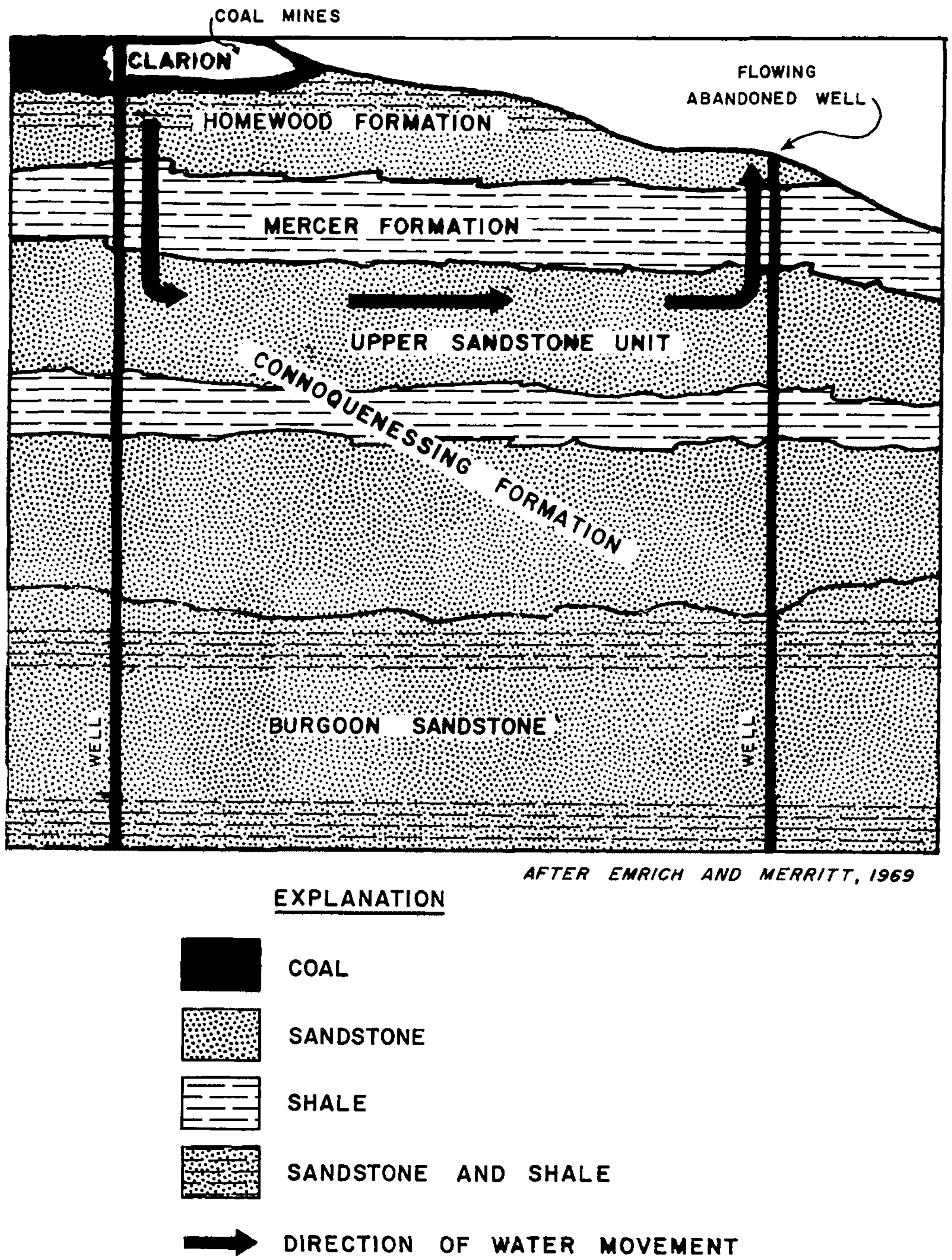


Figure 57. Influence of mines and abandoned wells on ground-water flow patterns in Clarion County, Pennsylvania.

TABLE 9. RESULTS OF CHEMICAL ANALYSES OF MINE WATER INFLUENT AND EFFLUENT IN ALABAMA*

(Source: Shotts and others, 1978)

Parameter	Mean value from influent†	Mean value from effluent‡
Acidity (as $CaCO_3$)	--	0
Alkalinity (as $CaCO_3$)	531	559
Ammonia (as N)	0.47	0.62
Chloride (Cl)	127	336
Hardness (as $CaCo_3$)	7	67
pH	8.96	8.62
Suspended Solids	507	3,713
Specific Conductance (μmhos/cm)	1,208	1,923
Temperature (°C)	22.5	23
Sulfate (SO_4)	--	14
Turbidity (JTU)	15	7,688

* Dissolved constituents in milligrams per liter; other constituents as shown.

† Based on six samples.

‡ Based on three samples.

In regions with naturally poor quality ground water, it may be difficult to assess the degree of water-quality alteration that results from mining. For example, in a study of the Tradewater River basin in western Kentucky, Grubb and Ryder (1972) hypothesized movement of mineralized water from surface and underground mines to fresh-water aquifers. However, no firm evidence was obtained to document the theoretical flow, owing to a lack of wells and the presence of high background concentrations of sulfate and dissolved solids, even in unmined areas, that prevented any reliable determination of the degree of contamination.

A review of legal cases in eight states suggests that the effect of underground mining on water well or spring quality is generally not the cause of lawsuits (see Table 5). The cases documented primarily dealt with domestic ground-water supplies whose quality was rather abruptly degraded by mining practices. Instances of gradual degradation, which are much more difficult to prove, are not evident from the data in legal case histories.

Significant changes in hydrologic conditions can occur in the vicinity of mines following their abandonment. These changes are related to the geologic conditions, types of openings, and methods of mining. They generally involve water-level elevations, mine drainage, and water-quality characteristics. Inundation of shaft, slope, and down-dip mines following abandonment should result in reduction of contamination according to commonly accepted geochemical models. This takes place mainly because pyritic oxidation is greatly reduced where direct contact with atmospheric oxygen is eliminated. Only a few such cases, however, have been quantitatively documented.

An investigation of mine drainage quality at two abandoned mines in Clearfield County (Mentz and Warg, 1975) lends support to the relationship described above. Other than a difference in orientation of the openings, the two mines had similar mining histories and geologic conditions. The acidity of the discharge from the freely draining up-dip mine was consistently higher than 2,200 mg/l, in contrast to the acidity of the discharge from the flooded down-dip mine, which was generally less than 100 mg/l. Concentrations of total iron ranged from 600 to 800 mg/l and sulfate averaged 2,000 mg/l in water from the up-dip mine, whereas concentrations of iron and sulfate were 55 mg/l and 600 mg/l, respectively, in water from the down-dip mine. Manganese and aluminum concentrations were also substantially higher in the up-dip mine.

A regional assessment of the relationship between mine-discharge quality and mine-sealing techniques provides some evidence of improvement in water quality as a result of the use of the most common sealing techniques (Bucek and Emel, 1977). Examination of data from 86 flooded abandoned mines in 11 states led to the following findings that are applicable to this study:

(1) Effluents from flooded shaft, slope, and drift mines generally have better quality than discharges from open-air, dry-sealed, or partly flooded up-dip drift mines.

(2) The group of mines with better overall water quality includes more sites where samples were taken from inundated shaft/slope or hydraulically sealed mines. However, the observed modification in the mine effluent quality of flooded and non-flooded mines, and for the range of closure types (including no closure at all), indicates that although mine closures affect the quality of the effluent to some extent, they are not the sole factor.

(3) The overall effect of the closures on water quality was beneficial in terms of reduced acidity and increased alkalinity.

(4) The sulfate concentration remained unaffected or increased, and total iron concentration increased in the majority of cases.

(5) Trend analyses of records before and after sealing show that the mine closures for more than half of the sites caused a reversal or reduction in contamination trends, and also a reduction in the variability of the water quality.

Improvement in mine drainage and ground-water quality in the anthracite fields of Pennsylvania may be related to the flooding of most mines since the 1940's. In 1941 coal production was still relatively high and many mines were kept dewatered by pumping. Growitz (1978) notes water-quality improvements over the last few decades. Between 1941 and 1975 the average pH of discharges from all anthracite fields increased from 3.0 to 4.0. In 1975, sulfate and acidity in discharges were reduced to a little more than half of the 1965 concentrations in the southern anthracite field. A decreasing supply of soluble minerals and a decreasing rate of mineral solution may be the principal factors controlling the improvement in water quality.

It is likely that the quality of ground water above and around abandoned underground mines will be least affected where exposure of coal to air is eliminated and where ground-water flow conditions are most similar to pre-mining conditions. This is most likely to occur where mines are deep, overlying rocks have very low permeability, roof fracturing is minimal, and mines are flooded. However, where ground-water circulation from the land surface has significantly increased over pre-mining conditions, exposed pyrite will be in contact with increased amounts of oxygen and water, and therefore will be susceptible to leaching. These conditions are most likely to occur above relatively shallow underground mines in areas of moderate to rugged topography.

SECTION 9

EFFECTS OF SURFACE DISPOSAL OF UNDERGROUND MINING WASTES ON GROUND-WATER QUALITY

DISPOSAL PRACTICES

Refuse Piles

Most of the approximately 72,000 abandoned and active underground coal mines in the United States have had refuse associated with the sites at some time during their history. The total number of sizeable active or abandoned refuse piles and surface impoundments in eastern coal fields is between 3,000 and 5,000, and these contain more than 2.7 billion metric tons (3 billion tons) of refuse (National Science Foundation, 1975). A reported 538 million metric tons (592 million tons) of bituminous coal were produced in 1973, 18 percent (97 million metric tons) of which was rejected as waste (National Science Foundation, 1975). With the production of an estimated 1.0 billion metric tons (1.1 billion tons) of coal by 1985, there will be at least 181 million metric tons (200 million tons) of waste produced annually. This will be in addition to the 2.7 billion metric tons (3 billion tons) of waste already accumulated, more than a third of which is in 863 anthracite processing waste banks (Falkie and others, 1974). The amount of refuse generated has increased over the years, primarily because of changes in mining methods, more efficient coal cleaning processes, and the demand for cleaner coals.

For the purpose of this report, refuse from coal preparation plants includes coarse material (gob) and fine sediment (slurry). A refuse pile is a permanent or long-term accumulation of mine, or cleaning plant refuse materials. Almost all the coal wastes from eastern underground mines are disposed of on the land surface. Disposal in worked-out underground mines is more costly than surface disposal and is not a common practice. Underground disposal is used primarily to control subsidence in abandoned mine workings under residential and commercial areas. Before 1920, coal was typically separated from refuse underground by hand, and the refuse was left in worked-out areas of the mine. With the advent of mechanization and full-seam mining, more of the refuse was brought to the surface, and economics dictated that it be disposed of there. More than half of the active piles in Pennsylvania are located within 0.4 km (0.25 mi) of stream banks. The percentage of abandoned piles this close to waterways is probably higher (Wewerka and others, 1976).

Refuse dumps vary considerably in size, ranging from less than a hectare to hundreds of hectares in area and from several meters to more than 100 m in height. Most piles are small, less than 402,308 m^3 (523,000 yd^3). However, the bulk of coal refuse is in very large piles, those containing over 1.2 million m^3 (1.5 million yd^3).

A comprehensive statewide survey of refuse from underground mines in Illinois identified approximately 2,024 ha (hectares) or 5,000 acres as problem areas (Nawrot, 1978). These areas met one or more of the following conditions: (1) exposed refuse materials (1,138 ha of gob and 270 ha of slurry), (2) abandoned tipple areas (160 ha), (3) contaminated water impoundments (95 ha), (4) adjacent affected terrestrial areas, (5) mine drainage to adjacent ditches, streams, and/or rivers (276 mine sites), and (6) potentially hazardous mine openings including openings with mine drainage. Documentation of this type was not available for other states, but it is likely that even greater volumes of refuse from underground mines exist in Pennsylvania, West Virginia, and Kentucky. In Illinois, refuse-oriented problem areas range in size from 0.05 to 83 ha (0.1 to 206 acres), and 43 percent of the areas are less than 0.8 ha (2 acres).

Waste Impoundments

The two types of underground coal mining wastes that are usually disposed of in impoundments are: sludge from the neutralization of acid mine drainage, and fine coal refuse (in slurry form) from the cleaning process. This report is concerned only with the slurry disposal practice. An impoundment is a permanent or long-term storage facility on the surface that is used for containment of mine water or cleaning-plant slurry. Most are used for the disposal or treatment (settling of fines) of waste, but some are used to store plant-processing water. Some impoundments are used for both purposes simultaneously. The quantities of effluent stored in impoundments at eastern United States coal-preparation plants are unknown and are typically not measured.

Slurry pipeline disposal of fine refuse causes a significant degree of differential settling to develop in the pond because the coarser fraction settles out close to the pipe opening into the pond and the finest fraction is deposited farthest away. Stratification also occurs in the fine refuse deposits, with beds ranging from a fraction of an inch to several inches in thickness.

RELATIONSHIP OF COAL WASTES TO GROUND-WATER QUALITY

Physical Characteristics of Coal Refuse

Coal refuse is mostly coal, slate, carbonaceous and pyritic shale, and clay that are associated with the coal seam. Most coal refuse is a soft clayey shale; the flat plate-like fragments of slate and shale degrade to clay upon exposure to weathering or mechanical handling and compaction. Both burned and unburned coal refuse tend to weather faster than most other alluvial or residual soils. Grain sizes range from colloidal to more than 31 cm (12 in). Coarse and fine wastes are usually disposed of separately, although in some places they are combined to form embankments for impounding slurry.

Moulton and others (1974) studied the suitability of coal refuse as an engineering material. Samples of representative refuse were selected from north-central and northwestern West Virginia. The samples consisted of both fresh refuse directly from the preparation plant hopper and aged refuse that had been exposed to the atmosphere from 18 months to 30 years. Weathering causes a decrease in the percentage of the coarsest fraction (gravel) and an increase in the finer fractions, especially in the silt and clay sizes, giving more surface area for air and water contact. The coarse fraction is more affected by weathering than the fine fraction. Using a common soil classification, some refuse samples classify as sand, but most are sandy or silty gravel. The materials look and behave very much like typical residual soils, similar to soils from the weathered zone immediately above rock. The specific gravity of the refuse is relatively low compared to typical West Virginia soils. Dry density tests show maximum and minimum void ratios to be higher than would normally be expected for natural alluvial materials with similar grain size characteristics. Standard Procter compaction tests indicate that the old refuse has a substantially higher optimum water content than fresh refuse.

The properties of refuse are influenced by procedures used in the preparation plant. Grain size is especially affected, which in turn dramatically influences refuse permeability. The permeability of coarse refuse commonly ranges from 0.01 to 0.000001 cm/s (210 to 0.021 gpd/ft^2), and is typically 0.0001 cm/s (2.1 gpd/ft^2) (W. A. Wahler and Assoc., 1978). Moulton and others (1974) found the permeability of representative coarse West Virginia refuse to range from 0.00001 cm/s to less than 0.0000001 cm/s (0.21 to less than 0.0021 gpd/ft^2). with older more densely compacted refuse having the lower values. The wide range is due mainly to differences in the age of the pile and/or the degree of compaction. Greater compaction results in smaller continuous voids and thereby reduces permeability. Poorly sorted refuse with a large percentage of fines has a very low permeability, while coarse refuse with little or no fines has a high permeability. The ratio of horizontal to vertical permeability of coarse refuse from ten West Virginia sites was less than 10:1 on the average, with many samples less than 2:1. The permeability of fine refuse ranges from 0.0003 to 0.000001 cm/s (6.4 to 0.0021 gpd/ft^2); the horizontal to vertical permeability ranges from 15:1 to 100:1, and averages 25:1 (W. A. Wahler and Assoc., 1978). The lower permeability of fine refuse retards the infiltration of precipitation where the refuse is not saturated. In cases where infiltration into a pile is low, surface runoff may be much more harmful to the environment than seepage is to ground water.

The common range for natural moisture content (ratio of the weight of the water to the weight of dry solids) in coarse West Virginia refuse is 4 to 16 percent, and averages 10 percent. The natural moisture content of 87 fine refuse samples ranged from 8 to 56 percent, and averaged

21 percent (Moulton and others, 1974). The bulk of both the coarse and fine refuse in this study was nonplastic. This means it did not contain a significant percentage of clay, which retains moisture but does not yield water readily.

Chemical Characteristics of Coal Wastes

Upon exposure to air, changes occur in both the mineral impurities associated with coal and the coal itself. Fresh coal reacts with oxygen to form peroxide groups on the surface of the pile. With time, more stable oxidation compounds and oxides of carbon form. Alternating sun and rain cause fresh surfaces to open up in the coal as a result of rapid changes in moisture content, which accelerates oxidation. Oxidation also speeds up with increasing temperature. The initial oxidation of coal is rapid, but this rate may drop to one-tenth the initial value after 30 hours (Wachter and Blackwood, 1978). Oxidation decreases the volatile matter and carbon content of coal.

Metal sulfides in refuse piles are also oxidized upon exposure, forming sulfuric acid and precipitating ferric compounds and sometimes eventually forming simple aromatic acids and oxalic acid. The surface of the refuse may become highly acidic (pH less than 3). The rate of pyrite oxidation is affected by oxygen concentration, particle size, temperature, moisture content, pH, redox potential of the reaction, and crystal size and form of pyrite (Wachter and Blackwood, 1978). It is less dependent on the amount of water present. Oxidation is essentially continuous, while precipitation (and hence drainage) is intermittent.

Low winter temperatures reduce the oxidation rate, and below-freezing temperatures preclude infiltration or runoff. Upon melting, the initial meltwaters from snow and ice leach the oxidation products and produce a large volume of moderately polluted water. In the summer, chemical reactions on waste piles are accelerated, and precipitation often occurs as high intensity storms. The reduced overall flow in summer and autumn produces a smaller volume of much more concentrated drainage. Thus, although the total amount of dissolved constituents produced is larger in winter and spring, water quality is generally less affected because of the lower oxidation rate. The worst quality drainage is produced in the first flush after a dry period when pollutants are released in slugs, e.g., during a summer thunderstorm.

The products of oxidized pyrite are washed away by subsequent precipitation, and new pyrite surfaces are exposed for oxidation. Drainage percolates through the pile and into the ground, or runs off the pile into trenches, pits, or streams. Trapped pools frequently contain high concentrations of pollutants. When storms occur, these pollutants are washed out as slugs.

The quantity of infiltration depends heavily on the amount of precipitation, the shape of the refuse pile, volume of refuse contained, degree of compaction, existence of soil and vegetative cover, degree of water control, and the terrain near the pile. The quality of drainage in turn depends on the composition of the coal refuse, the composition of the wash water (e.g., highly mineralized mine drainage), type of cleaning process used, the rate and degree of chemical weathering, the permeability of the pile materials, rate of water movement through the pile, and length of time water is in contact with soluble materials. In addition, different ions vary in their susceptibility to leaching, in their velocity within the aquifer, and in their different trends of dispersal migration. Libicki (1977) found that aluminum, chromium, and iron were present as contaminants in laboratory leachate of gob samples, but molybdenum, strontium, and cyanide were not. With only a 2-percent weight difference between volumes of leachate and uncontaminated ground water, gravitational mixing did not cause extensive vertical migration of contaminants. Contaminants exhibited a tendency to migrate near the surface of the water table, especially where the gob was saturated.

The loading of pollutants from fine refuse is generally greater because of the larger surface area available for contact with percolating water. The lower permeability of fine refuse reduces the velocity of percolating water and provides longer contact time. Fines generally have low pH and high calcium ion-exchange capacity (W. A. Wahler and Assoc., 1978). The main pollution concern with slurry ponds is the contribution of suspended solids to surface waters and not chemical degradation of ground waters. Libicki (1977) ranked the relative threat to ground-water quality of different preparation plant process wastes (from greatest to least):

Pile Unsaturated and Subject to Leaching by Percolating Water	Pile Saturated and Subject to Leaching by Ground Water
1. Wastes from water washer	1. Wastes from water washer
2. Wastes from heavy washer	2. Flotation wastes
3. Wastes from dry separation	3. Wastes from heavy washer
4. Flotation wastes	4. Wastes from dry separation

This ranking is based on relative grain size. For an unsaturated site that is subject to leaching by percolating water, wastes from the water washer contain grain sizes from very fine to medium coarse (silty to 80 mm), making for a fairly permeable waste with sufficient fines (increased surface area) to release significant contamination; wastes from the heavy washer are medium to coarse (20 to 250 mm), meaning higher permeability so a larger volume of water moves through them faster, and less fines giving a smaller total surface area that releases fewer contaminants; wastes from dry separation are coarse, giving high permeability and a large volume of leachate, but small surface area releasing a small amount of contaminants. Flotation wastes are very fine (silty to 2 mm), so that even though there is a very large surface area for release of contaminants, the permeability is greatly reduced over the other wastes and much less leachate percolates to the ground water.

The relative threat to ground-water quality is changed where the disposal site is below the water table, because the waste is always saturated. Wastes from the water washer are still the most serious because the wide gradation in particle size makes for high permeability and large surface area; very fine wastes from flotation processes provide even more surface area, but the overall permeability is greatly reduced, medium to coarse waste from the heavy washer has a higher permeability, but less surface area over which contaminants are released. Coarse waste from the dry separation process has the least surface area for contact with dissolving fluids.

The quantity of natural bases (alkali and alkaline-earth cations, commonly present as carbonates or as exchangeable cations on clays) in a refuse may be enough to neutralize the acid produced at a rate equal to or exceeding the rate of acid production. On the average, sedimentary rocks contain higher neutralizing potentials than the maximum acid potentials from pyritic sulfur (Smith and others, 1974). Smith and others (1974) found that, where sulfur is present only as pyrite, the total sulfur content accurately quantifies the acid-producing potential of the material. For a refuse containing 0.1-percent sulfur, all as pyrite, the complete oxidation of 907 metric tons (1,000 tons) of refuse would require 2,840 kg (6,250 lb) of calcium carbonate to neutralize the sulfuric acid produced. In parts of southern Illinois, there are enough alkaline salts in the refuse from some mines to neutralize acid production. However, chloride and total dissolved solids are still a problem (W. A. Wahler and Assoc., 1978; Gluskoter, 1965).

In a study of shallow ground-water quality below a refuse pile in Macoupin County, Illinois, Schubert and others (1978) found the concentrations of several metals to be very high less than 122 m (400 ft) from the gob pile. Concentrations of several metals from these wells exceeded recommended drinking-water limits by orders of magnitude. Beyond 122 m (400 ft), the levels of dissolved metals decreased significantly. Mechanisms of attenuation were not investigated but are probably attributable to hydraulic dispersion, dilution, adsorption, and cation-exchange reactions with the glacial till underlying the site. In addition, precipitation of metals may have occurred as a result of increased pH.

Wewerka and others (1975) estimated that an average of 1.7 to 2.2 kg/ha/d (1.5 to 2.0 lb/acre/d) of sulfuric acid and 0.56 to 0.79 kg/ha/d (0.5 to 0.7 lb/acre/d) of soluble iron are produced from refuse in eastern coals. However, in some highly mineralized areas, acid has formed at a rate exceeding 337 kg/ha/d (300 lb/acre/d). According to these figures, a single large pile has far more potential for acid and contaminant production than does an abandoned mine (Wewerka and others, 1976). This is primarily due to the fact that the refuse is finely divided and well exposed, making weathering and leaching processes more effective.

Recent studies indicate that most acid production occurs in the outer layer of waste (Martin, 1974). After the acid is produced, it can infiltrate the refuse pile, be stored temporarily while dissolving metals from the refuse, and reappear later in dry weather as an acid spring or seep. Peterson (1975) reports that comparisons of recently established refuse piles with those over 100 years old show that, in compaction, degradation of refuse takes place very slowly below depths of three feet. However, where new material is continually exposed to the atmosphere by erosion, oxidation will continue almost indefinitely.

Uncompacted refuse generally produces a larger quantity of much poorer quality drainage than compacted refuse. At one site studied by Wahler and Associates (1978), drainage from a refuse pile into a slurry pond typically did not seriously affect pond water quality. The refuse was usually compacted periodically. However, one winter several loose piles were dumped from trucks and not compacted. A heavy snowfall occurred and the meltwater percolated slowly through this part of the refuse. The resulting seepage carried such high contaminant loads that the entire pond required lime treatment to meet EPA point-source discharge regulations before the excess water could be discharged to a stream as usual.

Davidson (1974) studied the composition of wastes from underground bituminous coal mines in Pennsylvania. Over 300 samples were taken out of the major seams mined in distinct geographic areas of the state, and included both fresh and weathered refuse (from the same seams when possible). Analyses of the samples showed such wide divergence in physical and chemical characteristics that no trends or generalizations could be made for specific geographic area, coal seam, or even the depth from which the samples were collected (Table 10).

Chemical analyses were conducted on West Virginia refuse by the U. S. Bureau of Mines (Table 11). These analyses included a complete list of trace elements.

Hydrogeologic Settings of Disposal Sites

As in the case of underground mines, the hydrogeologic setting of a disposal site plays an important role in determining rates and direction of ground-water flow. The nature of flow, in turn, influences the extent and magnitude of water-quality changes. In an attempt to characterize the different types of coal-waste sites in the study area and describe their hydrogeologic setting, 23 active and abandoned piles were inspected in five states as part of this study. These locations are given in Figures 58 and 59. Table 12 lists the sites by type and indicates their general locations. The sites visited represent a good cross section of refuse types with regard to topography, geology, coal seam mined, size, age, relation to surface drainage, and water quality. The findings arrived at in this section are based primarily on observations made during these inspections.

On the average, coarse refuse allows the infiltration of precipitation at higher rates than do either bedrock or till in coal regions. Few estimates of rates of recharge on refuse could be found in the literature. A study at the New Kathleen refuse area in Illinois (Barthauer and others, 1971) found 20 to 50 percent of rainfall infiltrated

TABLE 10. CHEMICAL CHARACTERISTICS OF SAMPLES OF UNDERGROUND MINE REFUSE IN PENNSYLVANIA*

(Source: Davidson, 1974)

Parameter		Seam						
		A	B	C	C'	D	E	Pittsburgh
pH	average	3.1	3.4	3.0	3.5	3.8	3.8	3.6
	range	1.5	4.6	1.0	1.8	3.1	7.0	5.3
Acidity (meq H^+/100 gm)	average	8.5	9.8	6.4	5.1	6.4	8.0	8.8
	range	19.9	112.4	12.2	8.1	12.1	38.6	33.1
Conductance (μmhos/cm)	average	0.87	1.88	1.51	0.32	0.31	1.61	2.30
	range	2.01	20.08	4.79	1.20	1.63	8.45	6.63
Sulfate (SO_4)	average	1,209	3,395	12,097	873	739	4,643	10,953
	range	2,992	26,513	50,076	1,765	3,000	30,088	29,880
Phosphorus (P)	average	0.2	1.3	0.6	1.0	1.8	3.1	6.7
	range	1.0	15.5	0.8	1.9	16.5	16.5	20.3
No. of samples		10	88	8	16	26	50	70

* Dissolved constituents in milligrams per liter; other constituents as shown.

TABLE 11. CHEMICAL ANALYSES OF WEST VIRGINIA REFUSE*

(Source: Wewerka and others, 1975)

Parameter		Minimum	Maximum
Beryllium (Be)		0.2	3
Sodium (Na)		150	375
Magnesium (Mg)		500	8,000
Aluminum (Al)	(%)	> 2.5	--
Silica (Si)	(%)	> 2.5	--
Potassium (K)		500	1,200
Calcium (Ca)		50	2,000
Scandium (Sc)		3	25
Titanium (Ti)		300	3,000
Vanadium (V)		25	250
Chromium (Cr)		3	25
Manganese (Mn)		65	1,300
Iron (Fe)	(%)	0.75	4.1
Cobalt (Co)		3	25
Nickel (Ni)		25	250
Copper (Cu)		12	50
Zinc (Zn)		30	85
Gallium (Ga)		3	25
Yttrium (Y)		3	25
Zirconium (Zr)		3	25
Silver (Ag)		0.3	2.5
Cadmium (Cd)		0.25	1.0
Lead (Pb)		20	150

* Concentrations in milligrams per liter except where shown as percent.

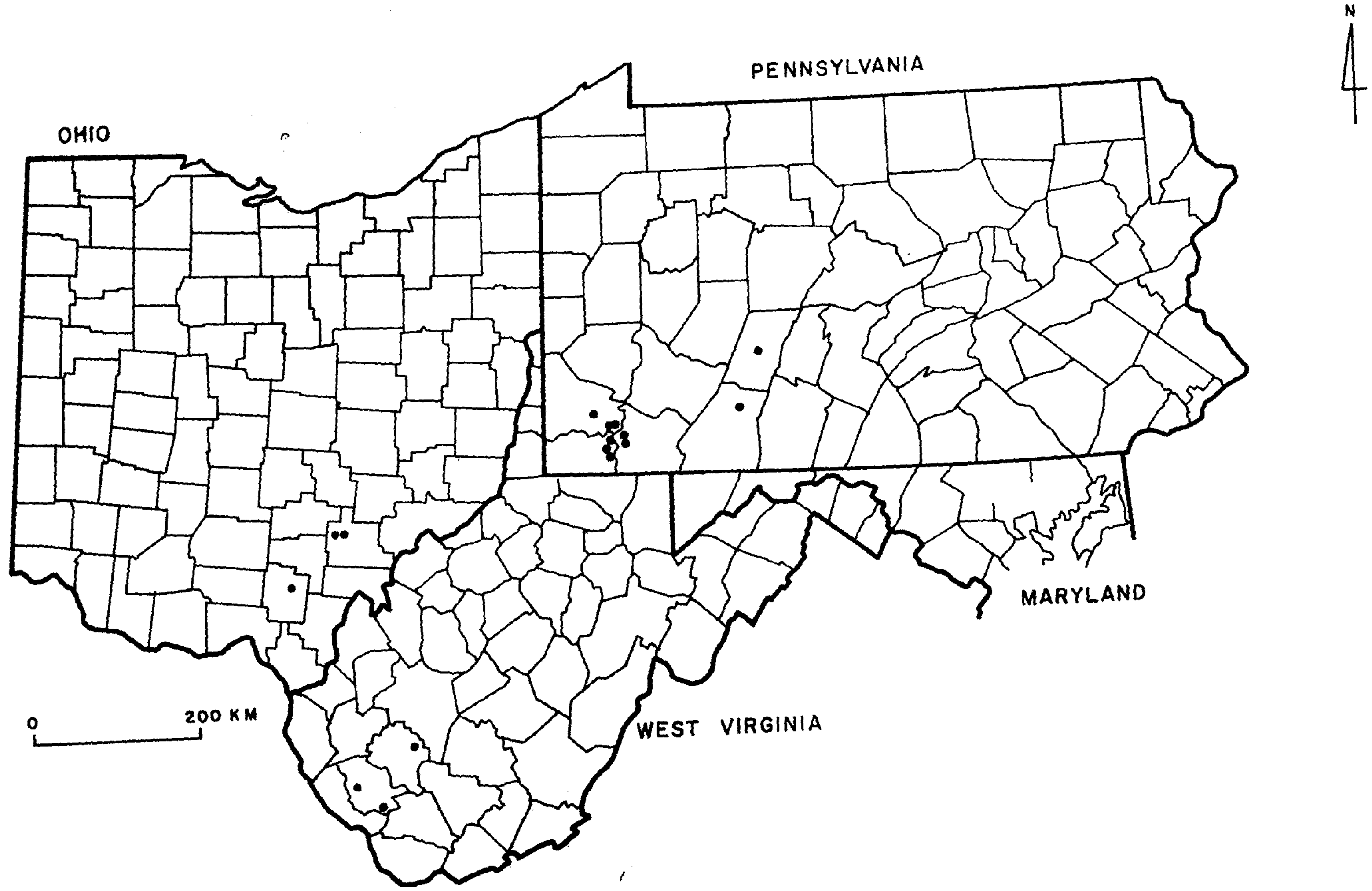


Figure 58. Coal-waste disposal sites inspected as part of this study in central Appalachians.

Figure 59. Coal-waste disposal sites inspected as part of this study in the Eastern Interior Basin.

TABLE 12. GENERAL LOCATION AND TYPE OF COAL-WASTE SITES VISITED DURING THIS STUDY

State and County	Nearest Town	Type of Refuse Pile
ILLINOIS		
Macoupin County	Carlinville	Waste heap (flat terrain)
Saline County	Harco	Waste heap
Franklin County	Valier	Waste heap
Franklin County	West Frankfort	Waste heap
KENTUCKY		
Ohio County	Beaver Dam	Highwall lake backfill
Union County	Morganfield	Waste heap
Ohio County	Centertown	Side hill and ridge
Union County	Morganfield	Waste heap
OHIO		
Athens County	Kimberly	Complex
Athens County	Chauncey	Cross valley and valley fill
Jackson County	Roads	Complex (hilltop and cross valley)
PENNSYLVANIA		
Greene County	Alicia	Complex
Fayette County	Isabella	Cross valley and valley fill
Greene County	Nemacolin	Complex
Washington County	Courtney	Cross Valley and valley fill
Greene County	Alicia	Side hill and ridge
Washington County	Vanceville	Complex
Fayette County	Fredericktown	Cross valley and valley fill
Washington County	Ginger Hill	Cross valley and valley fill
Cambria County	Colver	Side hill
Somerset County	Hooversville	Side hill

(continued)

TABLE 12 (continued)

State and County	Nearest Town	Type of Refuse Pile
WEST VIRGINIA		
Boone County	Keith	Cross valley and valley fill
Logan County	Peach Creek	Cross valley and valley fill
Logan County	Upper Whitman	Cross valley and valley fill

the pile. The permeability range of 0.000001 to 0.01 cm/s (0.02 to 212 gpd/ft^2) for coarse refuse corresponds to values representative of sandy silt or sandy clay to reasonably clean, medium to coarse sand. Typical recharge rates in such deposits for precipitation conditions typical of the Appalachian states range from 26 to 64 cm (10 to 25 in) per year. This range is probably twice that found in glacial till or in Pennsylvanian sedimentary rocks.

Perched water conditions can be expected to occur in a refuse pile if the pile is situated on sediments of lower permeability (Figure 60). A temporary recharge mound will slowly dissipate via seeps along the perimeter of the pile and by vertical leakage to the water table. The leakage to the natural sediments below the pile will be higher than normal infiltration rates to these sediments. Evapotranspiration will be virtually zero once water enters the refuse because vegetative cover will be insignificant on unreclaimed piles and because the perched water table will typically be deep within the pile. Assuming an average recharge rate of 38 cm (15 in) per year, a typical 40-ha (100-acre) pile would absorb approximately 377 m^3/d (0.10 mgd). Part of this degraded water will percolate to the shallow ground-water system. The actual amount reaching the ground-water system depends on the underlying sediments and the hydrogeologic setting. The total number of hectares of coal refuse in the study region and the leachate generated from such refuse is unknown. However, in Illinois approximately 1,620 ha (4,000 acres) of exposed refuse from underground mining has been surveyed (Nawrot, 1978). If 25 percent of the water entering this refuse eventually reaches ground water, the volume of refuse leachate would be about 3,785 m^3/d (1.0 mgd).

The close proximity of most refuse piles to stream courses results in the discharge of most contaminated ground water to nearby surface waters. Streams serve as ground-water discharge areas and thus help prevent or retard deep infiltration of contaminants. In addition, the travel distances are relatively short (tens to hundreds of meters) and the residence times may be only weeks to several years. Over these short distances, leachate from a given refuse area behaves as a rather distinct slug or plume flowing toward the nearest point of discharge. Thus, degradation within the subsurface occurs in a relatively well defined pathway which is limited in extent.

The size and shape of contaminated plumes in ground water are largely controlled by variations in porosity and hydraulic conductivity of the earth materials, fluid density, the attenuation capacity of the soil, direction of ground-water flow, the volume of leachate, and the time since start of infiltration. Physical processes that control the flux of solute into and out of the volume of contamination as it moves through the system are advection and dispersion. Advection is the component of solute movement attributed to transport by flowing ground water. This process is profoundly affected by geologic heterogeneities along the path of flow. Dispersion occurs as a result of mixing and molecular diffusion. Dispersion results in a spread of the plume both longitudinally and laterally as the body proceeds down-gradient.

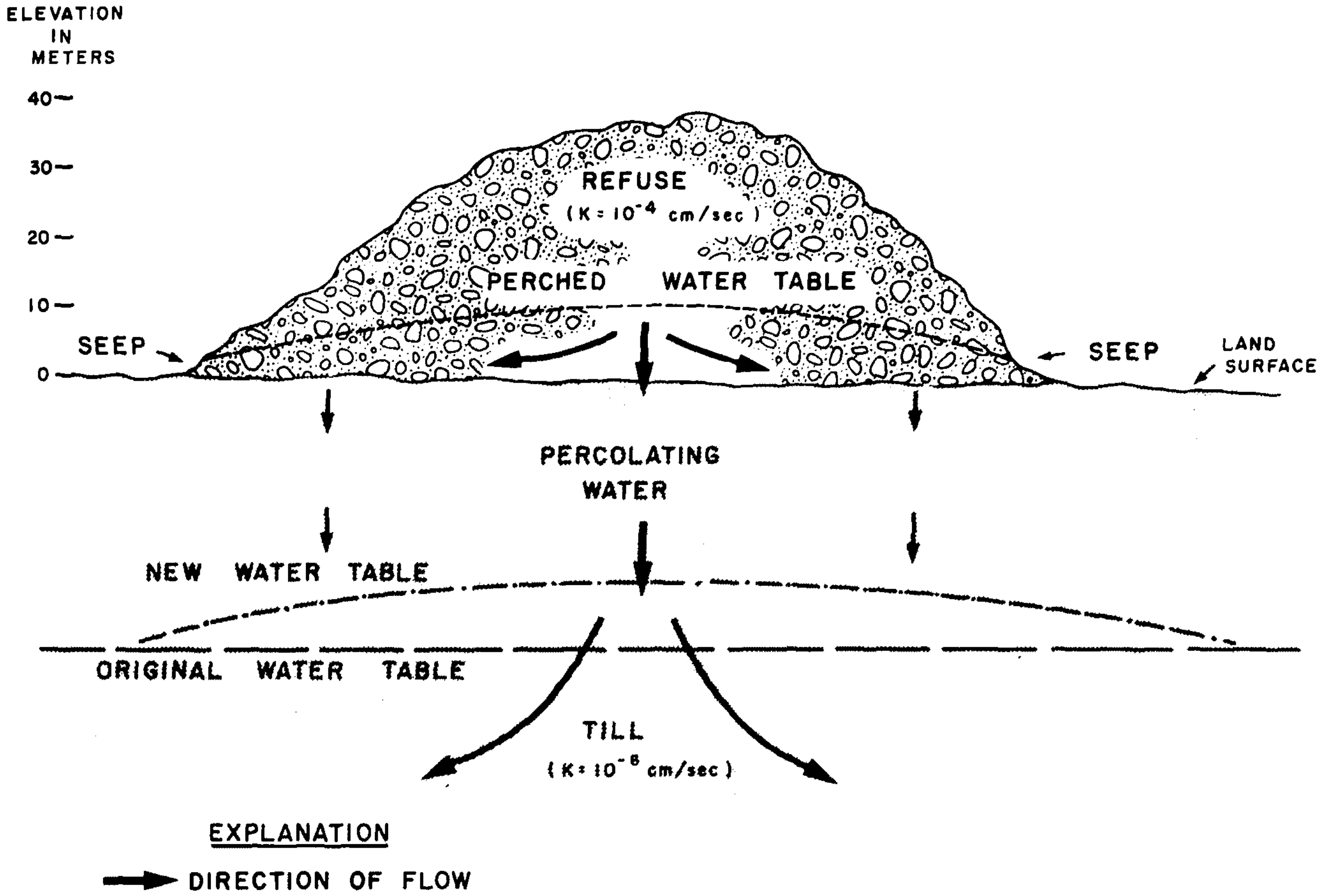


Figure 60. Schematic diagram of flow from a coal-waste heap in flat terrain.

Depending on grain size, grain-size distribution, and velocity of flow, dispersion can cause a significant lateral spread of contaminants in some cases. Figure 61 shows two situations where transverse dispersity is low in one case and high in the other.

The nature of ground-water flow leads to the conclusion that ground-water contamination from refuse is somewhat localized, except where bodies of coal are so numerous that contaminants enter over a very large area. There is no direct evidence that gross contamination is occurring, although it is to be expected in regions of extensive coal-waste disposal.

Piles and impoundments are located on a variety of landforms and assume various shapes depending on the original topography, the type of material disposed of, and the equipment used for disposal. Almost all fall into a general classification consisting of just a few categories. The major types of refuse piles and waste-water impoundments found in the eastern United States coal fields are as follows (W. A. Wahler and Assoc., 1978):

Refuse Piles	Impoundments
1. Waste heap	1. Diked pond and incised pond
2. Side-hill and ridge dumps	2. Side-hill impoundment
3. Cross-valley and Valley-fill dumps	3. Cross-valley impoundment
4. Complex dump	

Certain types of impoundments are usually associated with specific types of refuse piles, as shown in the above list. A brief description of the hydrogeologic setting of the various types of piles and impoundments is given in the following pages.

Waste Heap

This type of refuse pile is usually found in relatively flat terrain and is the typical setting for coal waste in Illinois and parts of western Kentucky. In this region, disposal piles usually are situated on ground moraine deposits (till). Figure 62 shows refuse piles of this type in southern Illinois. Because the location of this type of pile is not dependent on a natural hollow or valley, disposal sites are not necessarily near surface drainage features. The permeability of the till is very low, typically 0.00001 cm/s (0.21 gpd/ft^2). Much of the water percolating into the refuse piles in this setting is deflected in a horizontal direction upon contact with the ground surface. That is, the great majority of discharge off or through the pile is expressed as surface runoff or seeps around the perimeter. Some vertical infiltration to shallow ground-water systems can occur and may affect the quality of the water in the immediate vicinity of the pile (Figure 60).

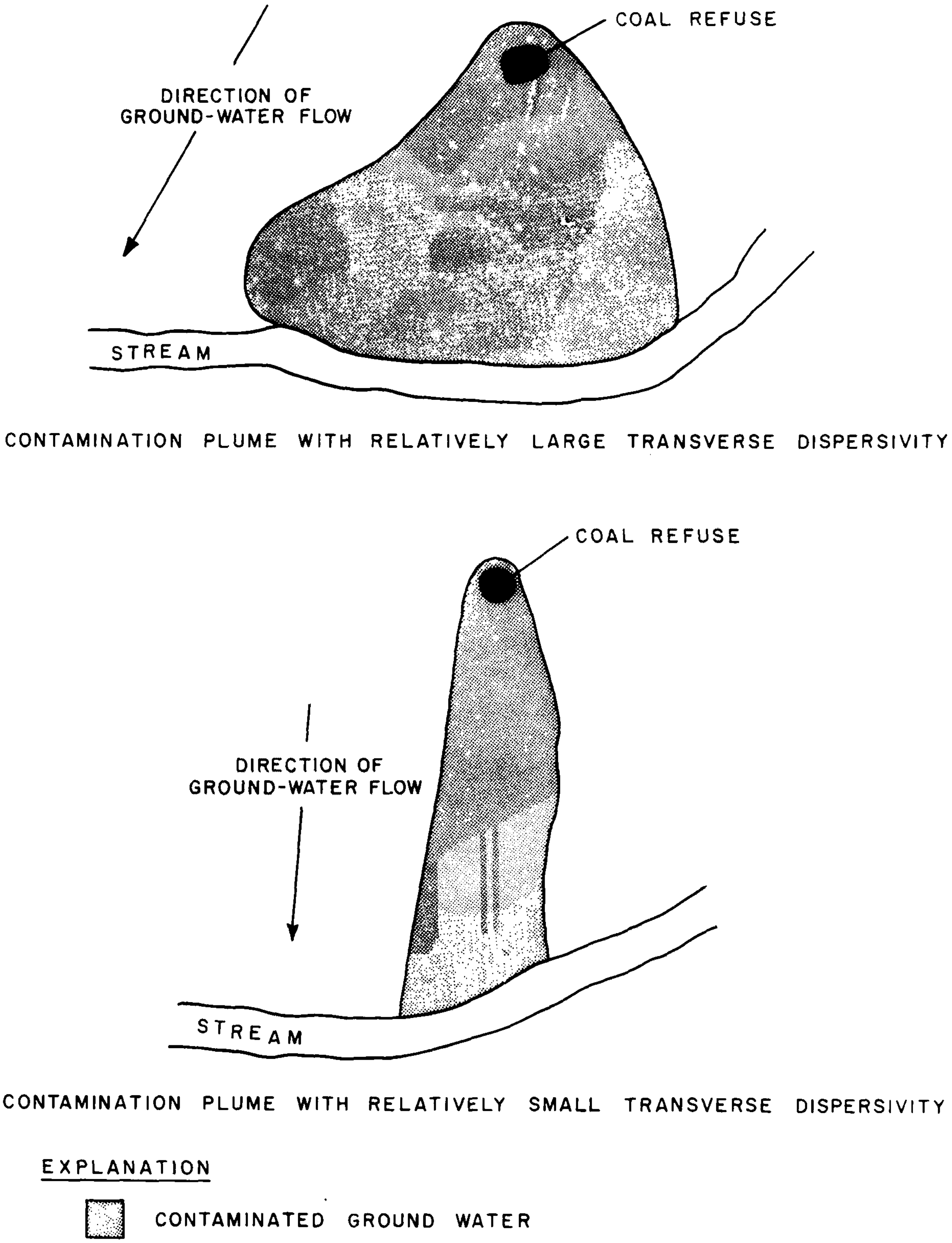


Figure 61. Plumes of Contaminated Ground Water Resulting From Coal Refuse Piles Near Streams.

Old Ben Coal Co. refuse area near Valier in Franklin County, Illinois.

Peabody Coal Co. refuse area near Harco in Saline County, Illinois.

Figure 62 - Coal-waste heaps and diked slurry ponds in southern Illinois.

Since typical waste heaps lack the capability to contain liquids, impoundments are usually constructed of earth or fine-grained refuse. Slurry water is contained in either diked ponds (above ground) or in incised ponds (below land surface). Where leakage under the dikes or through the pond floor occurs, it affects ground-water quality in a manner similar to infiltration through the pile. Slurry water ponds in flat terrain are shown in Figure 62.

Side-Hill and Ridge Dumps

In some cases, coal-refuse materials are dumped along ridge crests or on the side of a small hill. Large piles may become unstable and slide across a drainage course becoming a valley-type refuse dump, as discussed in the next section. Side-hill and ridge dumps are more prevalent in the Appalachian Basin and in parts of western Kentucky than in Illinois. Figure 63 shows side-hill dumps in Pennsylvania and West Virginia.

Since the refuse typically reaches its angle of repose when dumped, surface runoff is high. Side-hill impoundments usually develop on top of older sections of side-hill dumps that have reached the crest of the ridge or hill. Therefore, any leakage through the bottom of the pond will percolate through the underlying coarse refuse. Where the slope of the hill is moderate or steep and the underlying material is low-permeability bedrock, most of the water will discharge as springs along the toe of the dump. If the hill is covered by a mantle of colluvium or weathered bedrock, seepage into this material will occur, with subsequent movement into valley-floor alluvial deposits (Figure 64).

Cross-Valley and Valley-Fill Dumps

Cross-valley and valley-fill dumps are varieties of a disposal procedure that involves dumping directly in a stream valley. This is the most common form of disposal of large amounts of refuse in the moderate to rugged topography of the Appalachian states. Cross-valley refuse is built across a valley or stream course and may allow the stream to continue flowing through it without blockage. Where the refuse acts as a dam and slurry is piped behind it, cross-valley impoundments are created. Figure 65 illustrates two examples of such sites. If the size of a cross-valley dump increases and completely fills a valley, it is called a valley-fill dump.

In the dissected Appalachian Plateau, the floors and walls of valleys are composed of bedrock of low to moderate permeability. As detailed in a previous section, average permeabilities of such rocks range from 0.000001 to 0.0001 cm/s (0.021 to 2.1 gpd/ft^2). Because streams serve as points of ground-water discharge, disposal within valleys probably results in little or no ground-water quality problems since leakage into underlying rocks is negligible. This type of disposal tends to have its maximum effect on surface waters.

Duquesne Light Co. Warwick #2 refuse pile along the Monongahela River near Alicia in Greene County, Pennsylvania.

Island Creek Coal Co. #20 refuse pile near Upper Whitman in Logan County, West Virginia.

Figure 63 - Side-hill and ridge refuse dumps in West Virginia and Pennsylvania.

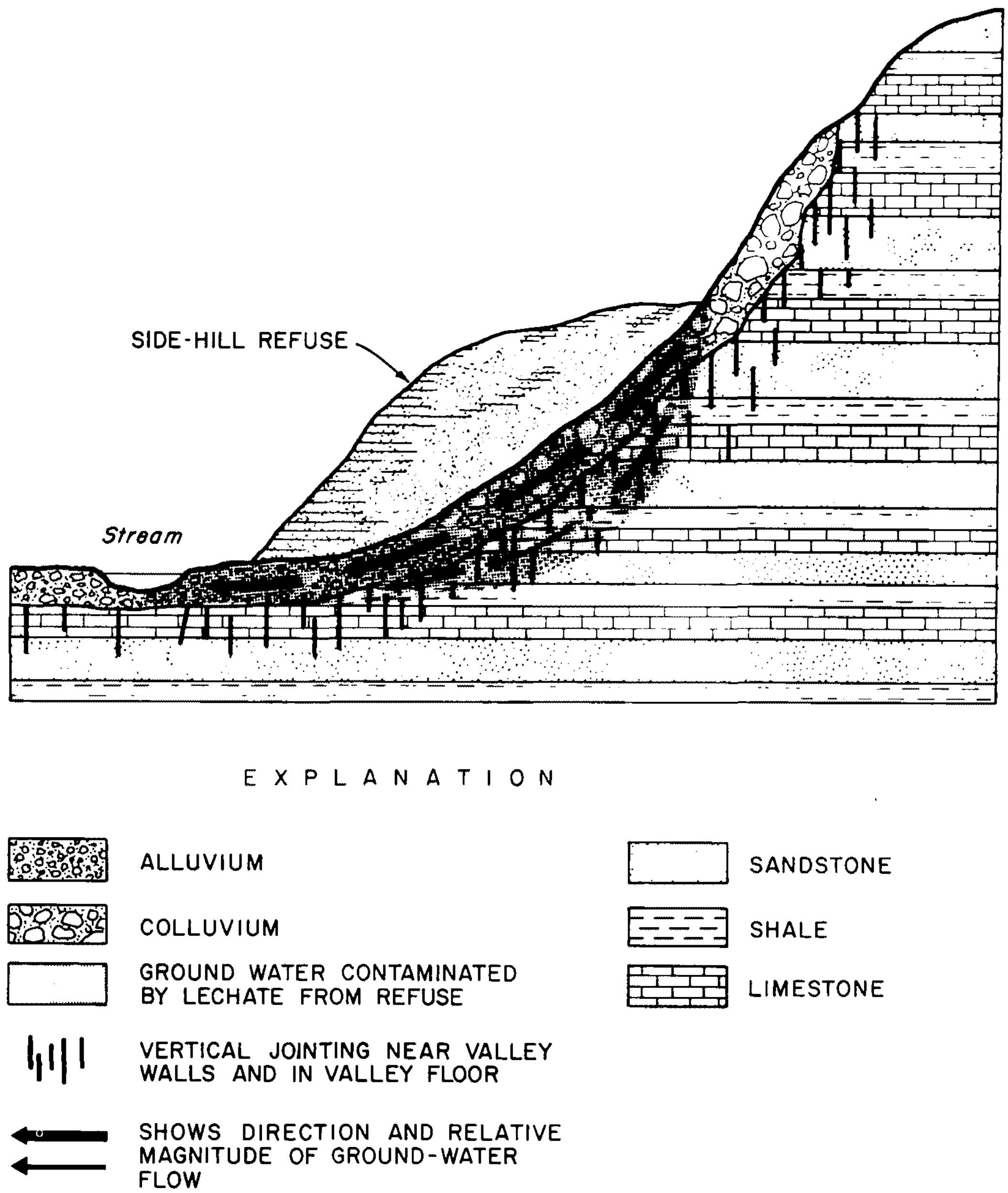


Figure 64 Schematic diagram of flow from a side-hill refuse disposal area.

National Mines Corp. Isabella refuse area near Isabella in Fayette County, Pennsylvania.

Duquesne Light Co. Warwick #2 refuse area near Alicia in Greene Co., Pennsylvania.

Figure 65 - Cross-valley dumps and impoundments in moderately rugged terrain.

When a cross-valley pile becomes large and blocks natural drainage or is used to impound slurry, water levels in the valley can reach a height sufficient to reverse the natural direction of ground-water flow (Figure 66). In time, a portion of the valley may become a recharge area for flow to adjacent valleys. Suggestions of this type of development were found at several of the sites investigated. In these cases, previously non-existent springs developed in walls of valleys adjacent to valleys filled with coal waste after the latter were dammed. The quality of the spring waters is poor and reflects the influence of water passing through coal refuse.

Complex Dumps

This category of dump is used for a deposit that consists of more than one of the basic types of dump and has an irregular shape. Complex dumps develop where the mode of operation has changed and disposal techniques are modified as the dump is enlarged or where a very large amount of material must be spread over an irregular landscape (W. A. Wahler and Assoc., 1977). These piles may be developed wherever refuse covers a large area, but are most common in regions of moderate relief such as Ohio, western Kentucky, and Alabama. In these areas, a single pile may cover hilltops, ridges, and valleys. The effects of complex dumps on ground water is a combination of the effects of the more limited type of piles. This composite effect is obviously more difficult to predict and can involve degradation of shallow or deep ground waters.

Nature of Pollutants and Ground-Water Contamination

Contaminants from coal refuse can be organic or inorganic. The inorganic fraction is always present and can be inherent (confined within the coal structure), or extraneous (foreign to the plant material that formed the coal). The inherent elements are primarily iron, phosphorus, sulfur, calcium, potassium, and magnesium, and typically comprise two percent or less by weight of the coal. Extraneous minerals are deposited contemporaneously with the peat or later through cracks in the solidified peat; these minerals form the ash. Ash content depends on the quality of the coal and the degree of cleaning; it ranges from 3 to 20 percent by weight, and averages 10 percent. Table 13 is a list of trace inorganic elements in coal. Heavier metallic elements, such as arsenic, zinc, and lead, are typically in inorganic combination with coal and hence more susceptible to leaching. Lighter elements, such as beryllium, germanium, and boron, tend to be in organic combination (Grube and others, undated).

Low levels of organic contaminants in water from coal waste or stockpiles are likely since coal is primarily organic. The constituents determined in most studies are carbon, hydrogen, oxygen, nitrogen, sulfur, ash, and volatiles. In a nationwide study of runoff and drainage from coal stockpiles stored outdoors, Wachter and Blackwood (1978) analyzed for six organic compounds. Although these results are not directly applicable to coal waste, strong similarities would be expected between waters passing through coal and refuse mined from the same seam.

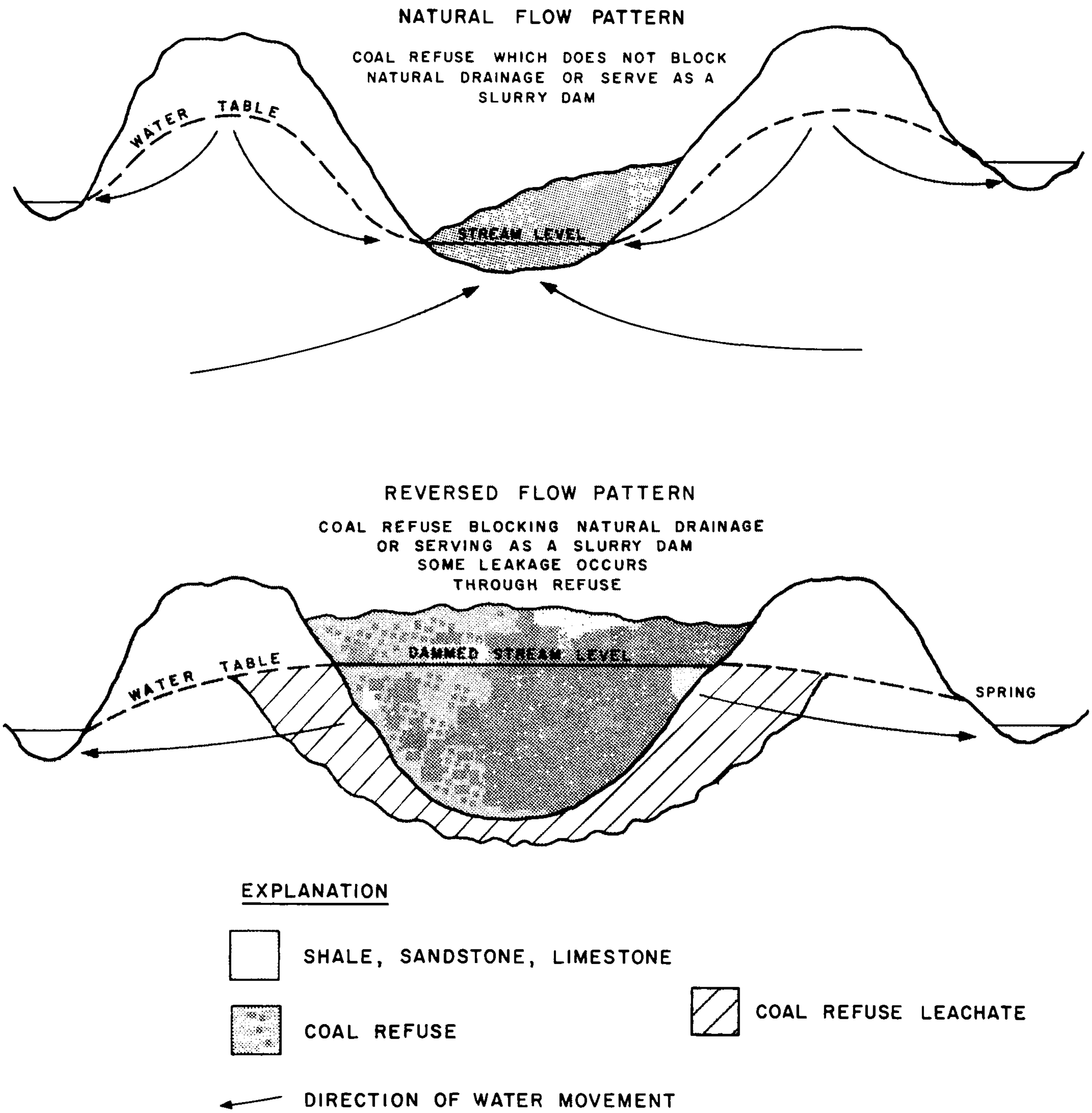


Figure 66 Effect of cross-valley refuse disposal on ground-water flow patterns.

TABLE 13. TRACE INORGANIC ELEMENTS IN COAL
(Source: Wachter and Blackwood, 1978)

Trace inorganic elements (about 0.1% or less, on ash)		
Beryllium	Chromium	Lanthanum
Fluorine	Cobalt	Uranium
Arsenic	Nickel	Lithium
Selenium	Copper	Scandium
Cadmium	Zinc	Manganese
Mercury	Gallium	Strontium
Lead	Germanium	Zirconium
Boron	Tin	Barium
Vanadium	Yttrium	Ytterbium
Bismuth		

The organics were found in the leachate in tens of parts per billion, which was 3 to 8 times higher than background concentrations (Table 14). The leachate sample was obtained by subjecting coal stockpile materials to "rainfall" from an artificial simulator. The background sample was the same water as that used for the "rain," and the source level in Table 14 is the calculated concentration expected in actual stockpile drainage.

Ground-water quality data from the vicinity of refuse piles and impoundments are very sparse. Ground-water monitoring of disposal sites is virtually non-existent in the states studied. For instance, in southwestern Pennsylvania where underground mining is prevalent and the concern for ground water is relatively high, only two of sixty active refuse sites have any monitoring (Higbee, 1978). In the course of this study, only one coal-waste disposal area was found with a comprehensive ground-water monitoring system. This site is presently under study by Argonne National Laboratory near Staunton in southern Illinois.

Schubert and others (1978) reported ground-water quality findings from the Illinois site mentioned above. This area contains an unreclaimed refuse pile which has been abandoned for 53 years. Twenty-two shallow monitoring wells were installed in the glacial till surrounding the refuse pile and the slurry area. The pH of the ground water was low (2 to 4), and concentrations of acidity, sulfate, and several metals were extremely high in the immediate vicinity (less than 122 m or 400 ft) of the gob pile. Acidity was 8,370 mg/l and sulfate was 5,739 mg/l in one well. Concentrations of several metals from these wells exceeded recommended drinking-water limits by orders of magnitude. At distances greater than 122 m (400 ft), acidity and dissolved metals decreased greatly. Specific conductance and concentrations of manganese were relatively high in a few wells 183 m (600 ft) from the pile.

In 1973, a joint project between POLTEGOR (Central Research & Design Institute for Opencast Mining, Wroclaw, Poland) and the U. S. EPA was undertaken to study the influence of gob disposal on ground-water quality. A test disposal site was constructed, consisting of 14 monitoring wells around a pit filled with 500,000 m^3 of gob from underground coal mines. Analyses were performed on uncontaminated ground water, contaminated ground water, and leachate derived in laboratory leaching tests. Libicki (1977) reports that the results show the "unquestionably deteriorating influence of gob storage on ground-water quality." A 20-m (66-ft) thick gob deposit produced "measurable" pollution 60 m (196 ft) away after approximately 15 months. The main body of the contaminants was transported at about the velocity of ground-water flow.

W. A. Wahler and Associates (1978) studied refuse piles and slurry ponds at five sites in the eastern United States, analyzing inlet and outlet waters for comparison. They found that wherever oxidation of pyrite had occurred or was occurring, there generally were increases in contaminant concentrations in discharge waters. If acidic water was in contact with the refuse, there were increased concentrations of sulfate

TABLE 14. ORGANIC EFFLUENT CONCENTRATIONS
(Source: Wachter and Blackwood, 1978)

Compound	Concentration, 10^{-3} g/m^3*		
	Coal leachate	Background	Source level
2-Chloronaphthalene	16	2	14
Acenaphthene	22	7	15
Fluorene	21	7	14
Fluoranthene	24	8	16
Benzidine	18	4	14
Benzo(ghi)perylene	52	8	44

* 10^{-3} g/m^3 = µg/l = ppb.

and heavy metals (iron, copper, manganese, nickel, and zinc), and depending on the mineral content of the refuse, increased chloride and lighter metals, too (aluminum and magnesium). Table 15 shows an analysis of seepage water from an impoundment containing low sulfur metallurgical coal refuse. The embankment is constructed of coarse refuse.

Martin (1974) described the results of an EPA study of effluents from refuse piles in Illinois, Kentucky, Pennsylvania, and West Virginia to determine the pollutants to be expected. Samples included seeps and direct runoff from piles, ponds in and around piles, and receiving streams both above and below the waste disposal sites. Generally, the metal and sulfate concentrations varied directly with acidity, but there was no overall correlation between ion concentration and acidity. Acidity values ranged from alkaline to 7,020 mg/l, and values as high as 34,300 mg/l are reported in the literature.

TABLE 15. CHEMICAL QUALITY OF AN ALKALINE SEEP FROM A COAL REFUSE SITE*

(Source: W. A. Wahler & Associates, 1978)

Parameter	Concentration
Flow (liter/min)	20
pH	7.30
Dissolved Oxygen (O_2)	--
Acidity (as $CaCo_3$)	--
Alkalinity (as $CaCO_3$)	202
Conductivity (µmhos/cm)	2,920
Aluminum (Al)	1.7
Cadmium (Cd)	0.001
Chloride (Cl)	--
Copper (Cu)	0.001
Ferrous iron (Fe)	0.1
Total iron (Fe)	4.9
Lead (Pb)	0.014
Magnesium (Mg)	118
Manganese (Mn)	4.8
Mercury (Hg)	0.0002
Nickel (Ni)	0.037
Sulfate (SO_4)	2,600
Zinc (Zn)	0.04

* Concentrations in milligrams per liter unless indicated otherwise.

SECTION 10

GROUND-WATER PROBLEMS FROM FUTURE MINING

The previous sections of this report have set a background for determining, on a regional basis, where underground mining may have the greatest effects on ground-water resources. As mentioned in the Introduction, the relative value of a ground-water resource is taken to be a combination of use and availability. The purpose of this section is to identify, in general, where future underground mining will likely impact ground water. This is done by a series of maps, Figures 67 through 72, which show areas estimated to have the most significant ground-water effects from future underground mines, based on the following criteria:

(1) counties with coal reserves for underground mining of greater than 455 million metric tons (501 million short tons);

(2) counties with over 4×10^3 m^3/d (1.06 mgd) ground-water pumpage;

(3) areas within these counties which have sufficient ground water for small industrial and public supplies (reported well yields generally over 1.6 l/s, or 25 gpm, unless otherwise stated).

In addition, Figures 67 through 72 show the location of underground coal mines that are planned for the next few years in the counties with large coal reserves. As expected, there is good correlation between planned new underground mines and counties with greatest coal reserves. Tennessee was not included because it contained no areas that satisfied the coal reserve and ground-water availability criteria used. Virginia and Maryland did not have sufficient data on ground-water availability. Indiana and eastern Pennsylvania did not have adequate data on ground-water use.

The results of this analysis show that there are very limited areas in the Eastern Interior Basin and in the southern Appalachians that have the potential for significant ground-water problems from future underground mining. In fact, in these two regions, only nine counties met the criteria of large reserves and significant ground-water usage. The central Appalachians, and in particular parts of western Pennsylvania and southern West Virginia, are indicated as having a greater potential for significant impacts. In this region, 31 counties have been identified as satisfying the first two criteria.

The type and degree of problems from state to state and even from area to area are not the same. This analysis does not take into account

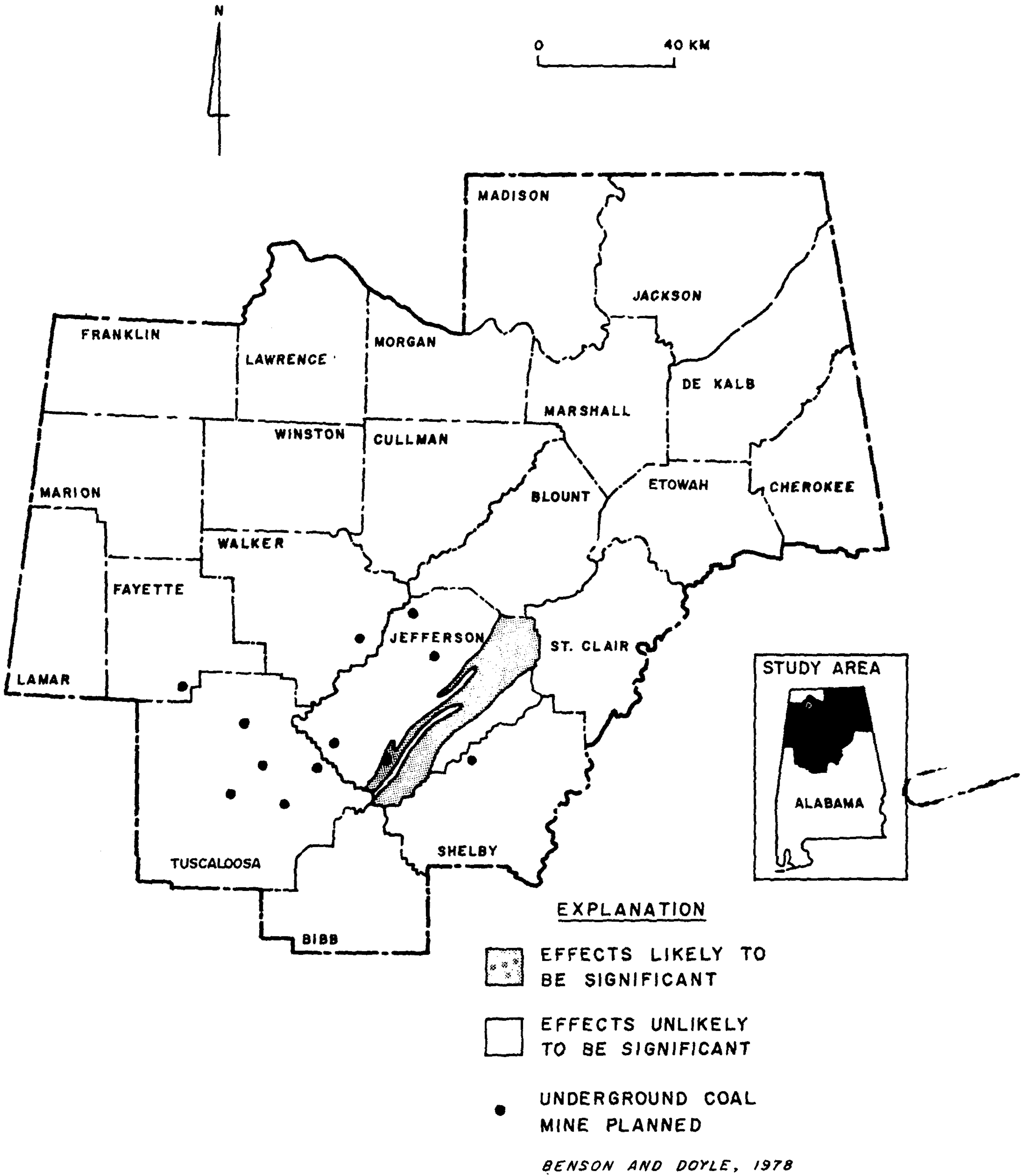


Figure 67. Areas in Alabama with potential for significant ground-water effects from future underground mining.

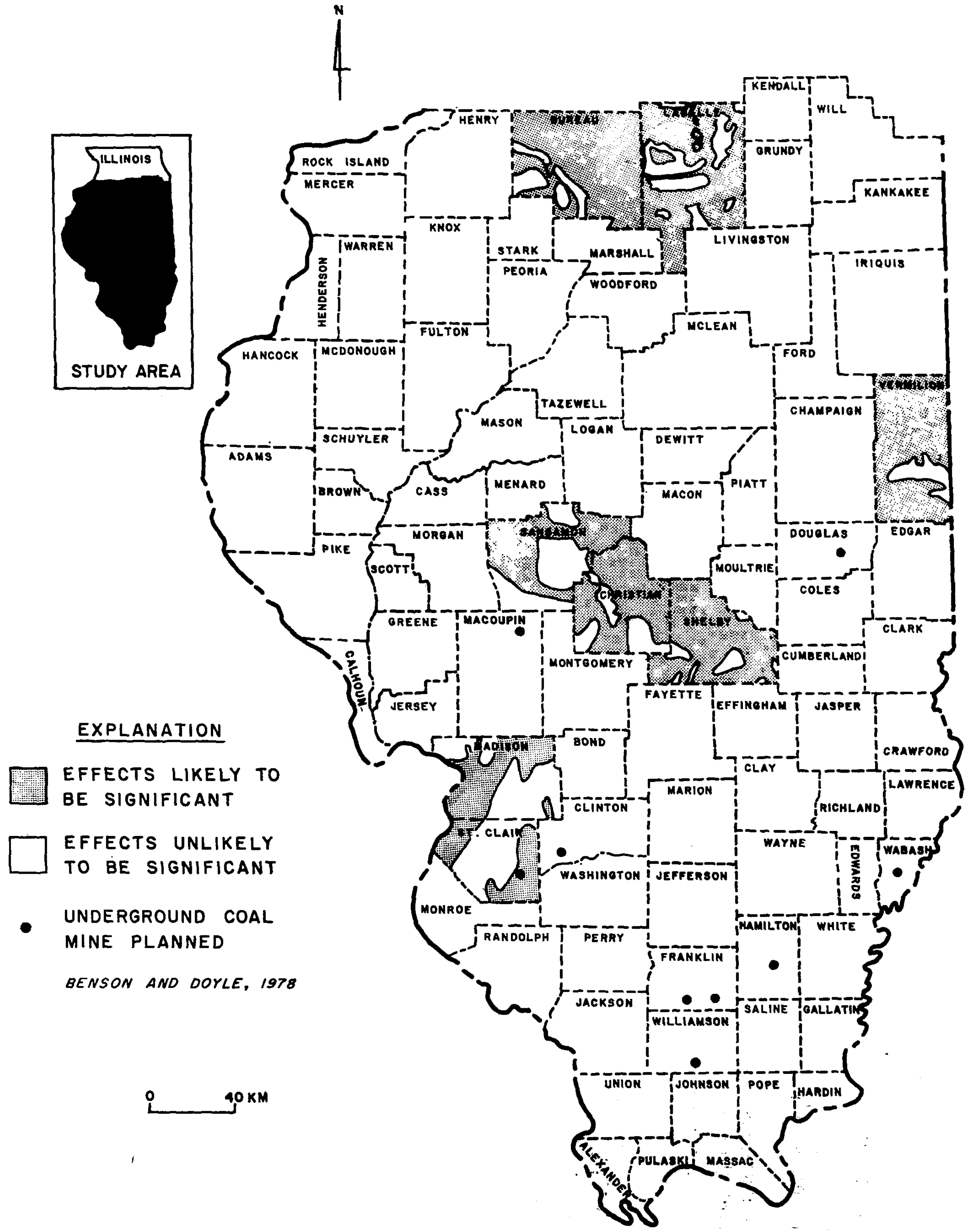


Figure 68 Areas in Illinois with potential for significant ground-water effects from future underground mining.

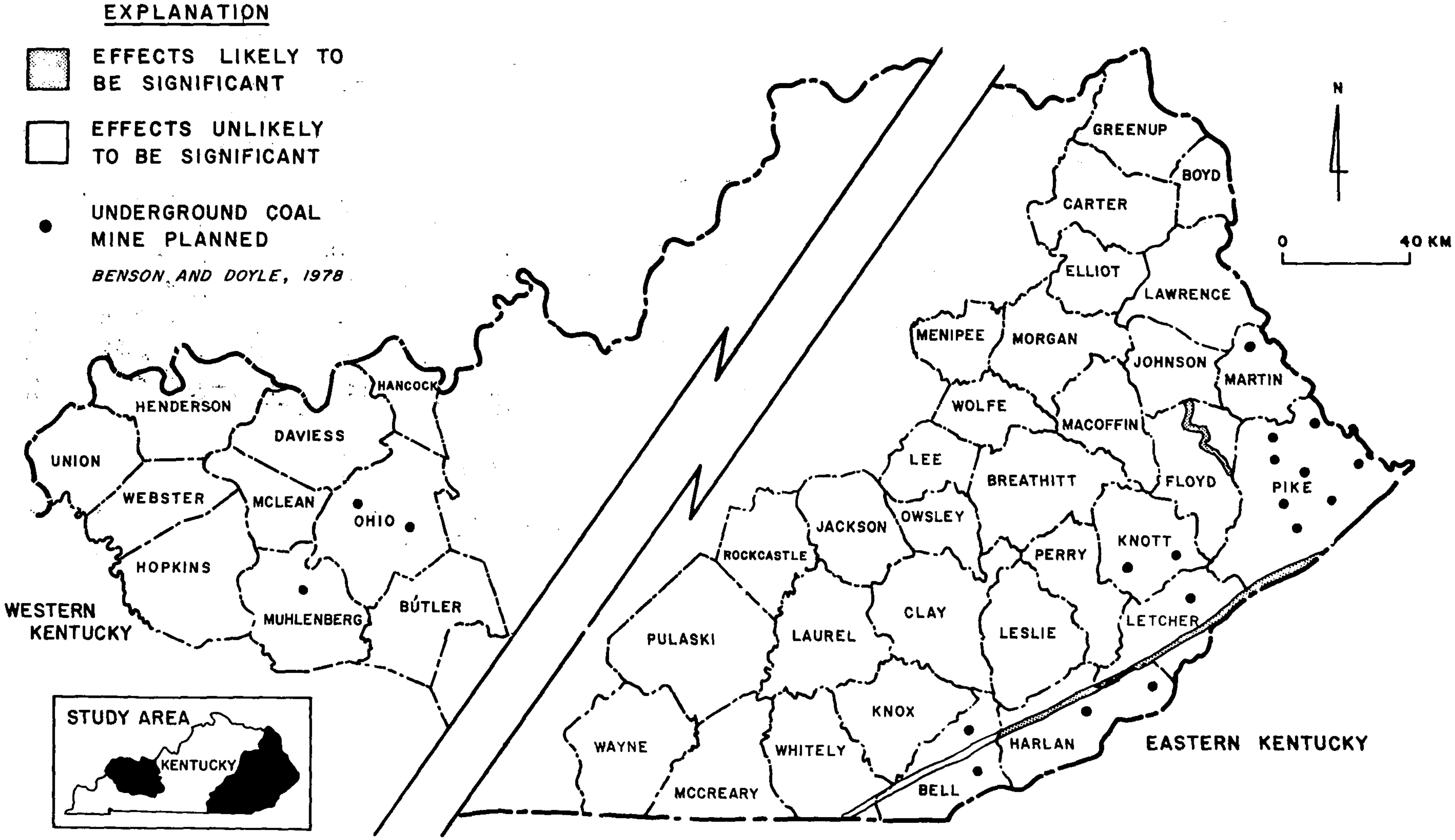


Figure 69. Areas in Kentucky with potential for significant ground-water effects from future underground mining.

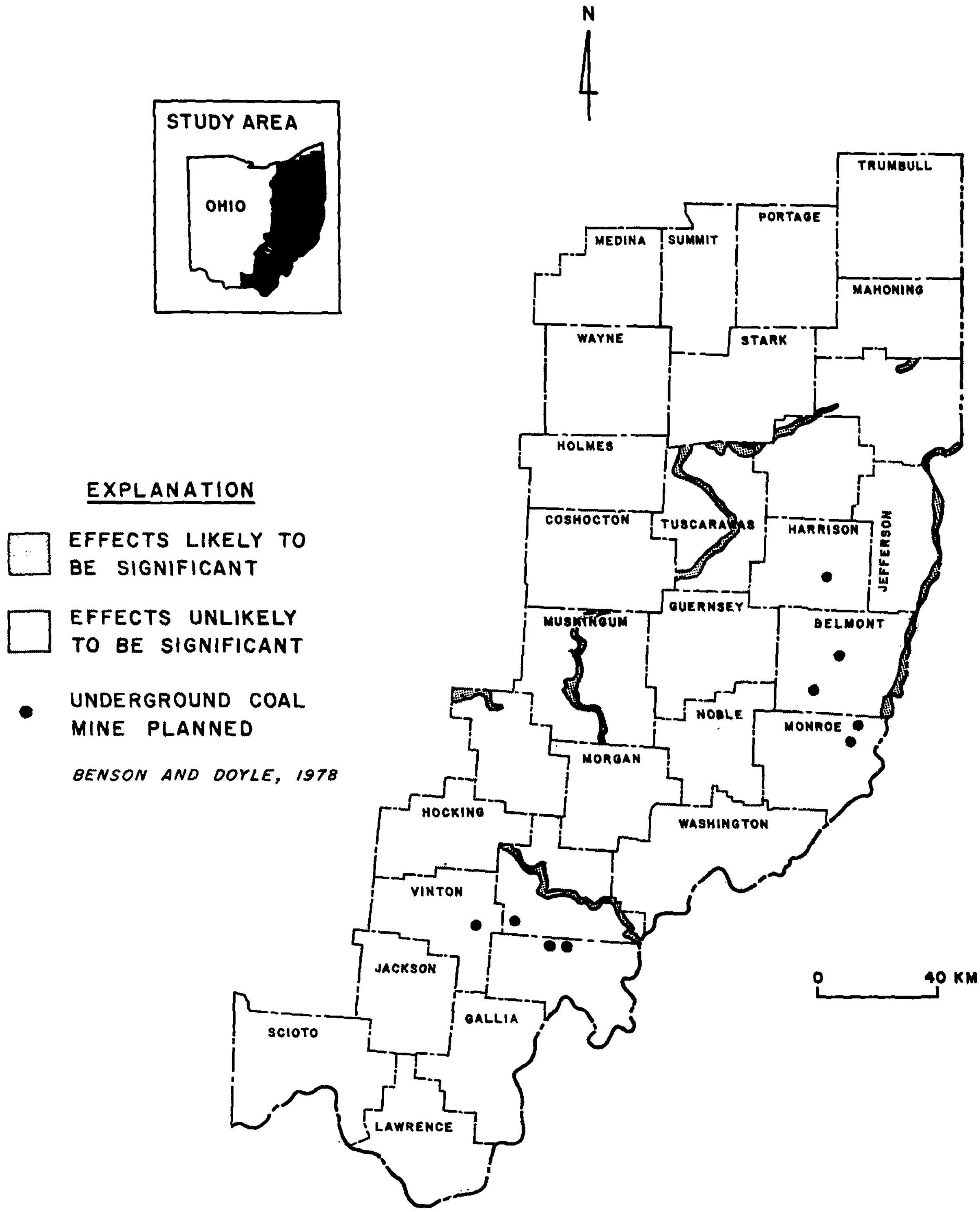


Figure 70. Areas in Ohio with potential for significant ground-water effects from future underground mining.

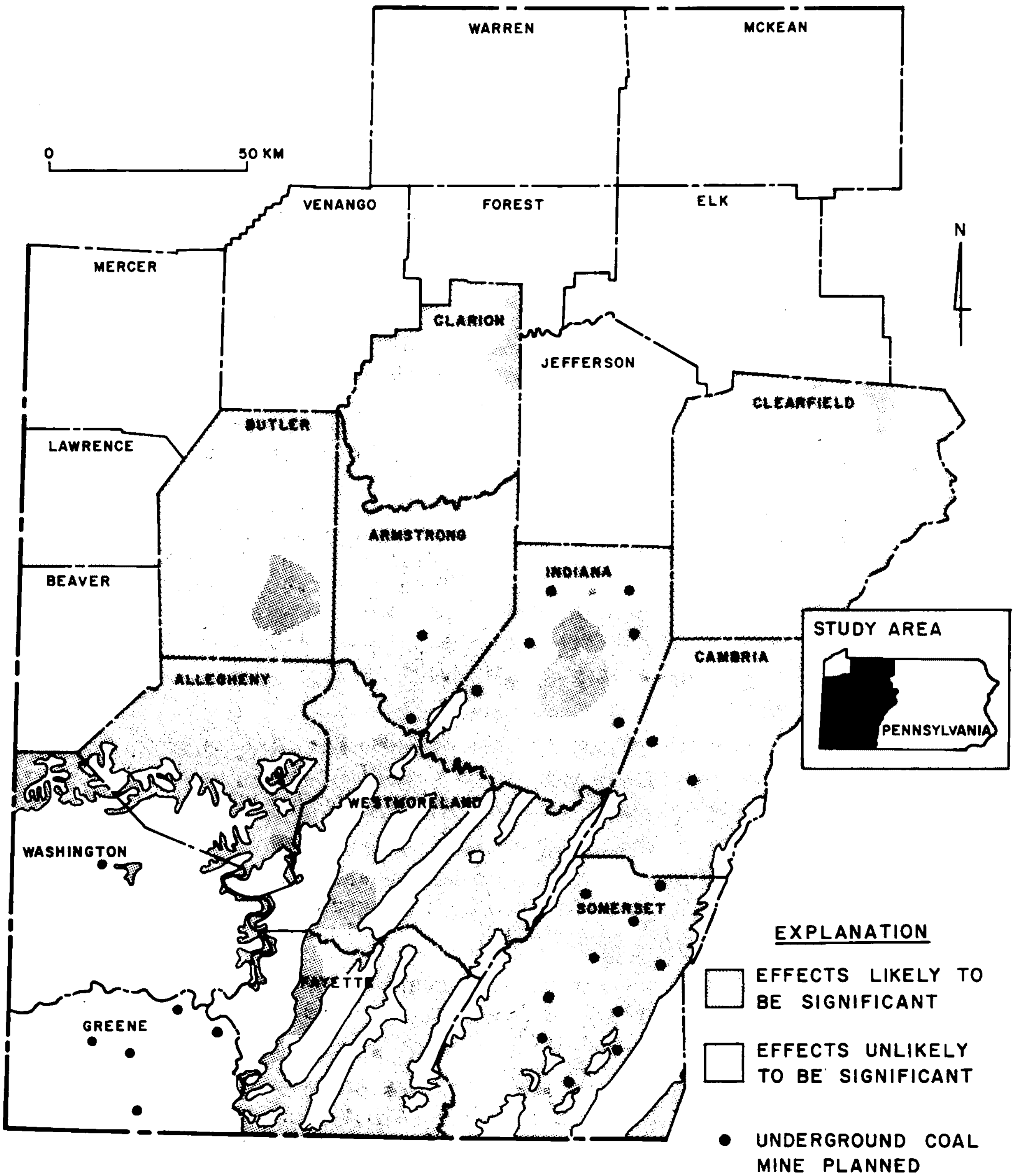


Figure 71. Areas in western Pennsylvania with potential for significant ground-water effects from future underground mining.

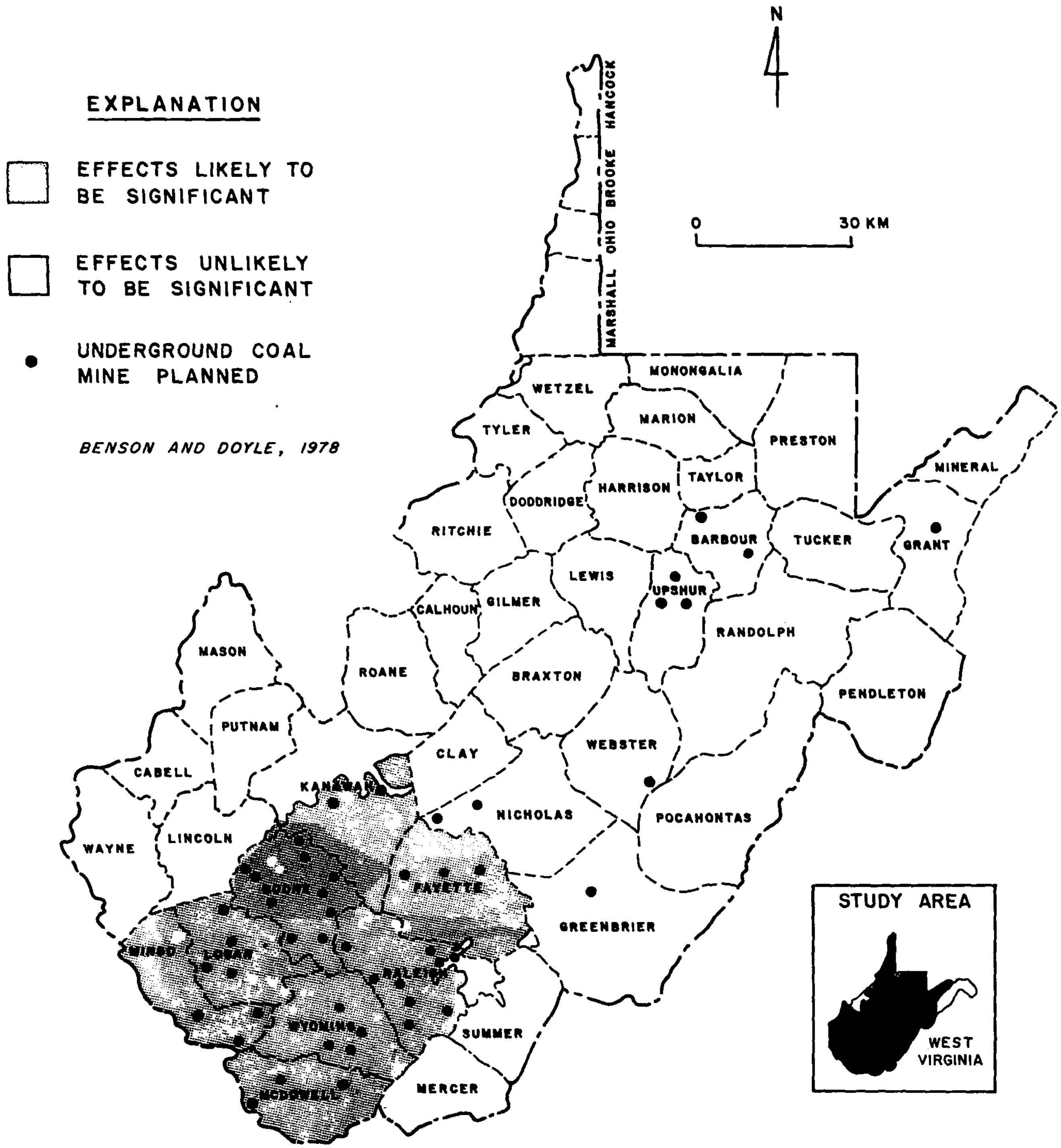


Figure 72 Areas in West Virginia with potential for significant ground-water effects from future underground mining.

differences in water quality as related to type of seam mined or the nature of the adjacent strata. For instance, ground-water quality effects even in the identified areas of the southern Appalachian Basin may be restricted to only minor increases in dissolved solids with no increase in acidity or reduction of pH. However, changes in water levels and ground-water flow may still occur. Ground-water quality effects in parts of the northern Appalachians and the Eastern Interior Basin may, on the other hand, be more significant in terms of dissolved solids, acidity, and pH. The figures do not assess potential problems in the vertical dimension. For instance, although refuse disposal on top of thin surficial sand deposits in Illinois may cause water-quality degradation of shallow ground waters, deep mining in the same area may cause little or no hydrologic problems. In addition, there may be areas not designated on the figures where ground water may be affected significantly by mining. This is true because, locally, domestic supplies may be severely impacted even in areas with poor aquifers and limited mining. In general, the maps presented in this section are meant to serve as a guide to areawide problems and are not meant to be utilized for detailed planning or evaluation of individual mines.

SECTION 11

METHODS FOR MITIGATING HYDROLOGIC AND WATER-QUALITY EFFECTS

PRE-MINING PLANNING AND MINING TECHNIQUES

Site Investigations

In the pre-mining stage, an assessment of the hydrogeology and ground-water geochemistry of a proposed site can help operators plan the mining activities so as to minimize adverse effects on ground-water resources. One purpose of such an assessment is to alert the mining company and the environmental regulatory agencies of possible water problems. This could help in the development of a more cost-effective program of water handling and treatment and provide the best data base for long-term water-level and water-quality monitoring. A comprehensive assessment might include evaluation of the following factors:

- Presence of calcareous sediments near a coal seam to be mined. In many areas, major carbonate units have been identified and mapped. Such strata help minimize water-quality degradation by buffering or neutralizing ground waters.

- Definition of pre-mining ground-water quality above and below a proposed mine. Knowledge of natural quality provides a baseline for detection of later changes. For example, the buffering capacity of the ground water provides a guide for predicting possible production of acid ground waters.

- Definition of pre-mining ground-water flow systems. Water-level and permeability data can be used to determine direction and rates of natural flow, as well as the potential for changes in water inflow and outflow from the mine.

- Relation of mine site to fracture systems. Investigations using aerial photography, remote sensing, and, in some cases, geophysical methods have been shown to be effective in locating permeable zones in bedrock aquifers.

- Presence of nearby water-supply wells, springs, and aquifers. Field surveys and review of existing reports provide information on the present and future importance of ground water in an area. Such data constitute a basis for evaluating potential impacts on present or future neighbors.

- Potential for mine roof instability, subsidence, and induced fracturing of overlying rocks. The depth and type of mining contemplated and the nature of the roof rocks can be used to assess the potential for rock failure. Such failures can greatly alter flow patterns in and around a mine. This should be especially evaluated with regard to overlying surface-water bodies and nearby aquifers.

- Mineralogy of overlying rocks and reactive nature of pyrite. Studies in West Virginia and elsewhere have shown it is possible to assess the relative acid leaching potential of coals by laboratory tests. In addition, a knowledge of the paleoenvironment of coal deposition may be indicative of the reactive nature of pyrite in certain circumstances.

- Suitability of preparation plant sites and coal-waste disposal sites. Where these facilities are placed on very low permeability surficial deposits and/or in ground-water discharge areas, there is very little potential for ground-water contamination. In other hydrogeologic environments, it may be more difficult to assess the potential for contamination, so that a field study may be required. Such a study would involve the installation of borings to determine the potential for leakage from coal waste to the ground-water system and subsequent move ment of contaminants toward nearby streams, wells, and springs.

Upon consideration of these factors, it may become apparent that a particular site is environmentally or economically not suitable for mining or disposal, or that certain mining techniques or engineering controls will be necessary. The basic technology needed to make hydrogeologic assessments of mine-site suitability exists, although existing information is usually inadequate to address all the factors listed above. In most cases, site-specific studies would be required to make the pre-mining assessment.

Regulations authorized under the Surface Mining Control and Reclamation Act (PL 95-87) require collection of ground-water data necessary to evaluate the hydrogeologic systems around new and existing underground mining operations. These regulations require the collection of background data on ground-water quality and flow, the installation of monitor wells, and an assessment of how hydrogeologic conditions will change as a result of mining activities.

Site-specific studies generally require drilling of test wells in the vicinity of the mine. In many instances, it may be possible to increase the utility of exploratory boreholes by using them for hydrologic testing and monitoring. This could greatly minimize the total costs of the hydrogeologic assessment. This combining of functions, however, would require detailed planning and designing prior to any drilling. Boreholes not designed for monitoring or hydrologic testing are typically not suitable for such purposes.

Mining Techniques

The orientation of a mine, method of mining, and location of portals all can cause changes in ground-water and surface-water quality. As previously discussed, up-dip drift mines generally have the greatest discharges. In addition, available data suggest that the quality of drainage from up-dip mines is poorer than from other methods of mining.

Down-dip mining eventually results in the inundation of more areas within the mine, which may result in a better quality of the discharged water. After the abandoned down-dip mine is flooded, a new ground-water flow system is established, and water will begin flowing around, through, and below the abandoned seam.

High-extraction techniques which result in roof caving and, in some cases, subsidence, tend to increase alterations in flow, especially where mining is extensive. Low-extraction techniques in which substantial support pillars are left in place minimize changes in flow. However, such techniques require that 30 to 60 percent of the coal be left in place. The tradeoffs involved in selecting the most cost-effective and environmentally sound methods have not been rigorously analyzed in the past. It may be that low-extraction techniques may only be justified where alternative mining methods are likely to disturb surface drainage or where water levels in an important aquifer will be lowered.

ENGINEERING AND HYDROLOGIC CONTROLS

Most of the engineering and hydrologic controls currently in use to minimize effects on ground water generally fall into the class of cure rather than prevention and are seldom fully effective. For example, the majority of existing controls are aimed at mitigating adverse effects on surface-water quality or flow. Most of these address ground water only as an incidental item.

Mine Sealing

Mine seals fall into three general categories: hydraulic (dry), water, and air seals. Hydraulic (dry) seals are designed to prevent precipitation of surface runoff from entering deep mine workings, but are not designed to withstand hydraulic heads. Water seals are constructed to retain water behind them and withstand head pressure. Air seals, of course, are meant to prevent the flow of air into the mine via old entryways or ventilation shafts. Seals are typically constructed of clay, earth, concrete, limestone, and/or various precipitates. Constructing effective, long-lasting seals poses numerous engineering problems. Wherever joints or fractures extend from the mine to the surface, seepage can occur in the form of springs or seeps. Such situations are most likely where there is extensive underground mining above or near local surface drainage. Problems can be especially serious in drift mines, where large hydraulic heads build up behind water seals.

The objective of water seals is to stop mine drainage from portals and to flood most of the mined-out area. From a ground-water standpoint, there is probably little advantage in using water seals except where the surrounding rocks have been virtually undisturbed by mining activities. Under such conditions, natural water levels will be re-established and ground-water quality may improve after mine abandonment. Under other conditions, the effectiveness of water seals as a method of improving ground-water quality varies according to the technique used and to the geology around the site where it is employed. This is borne out by the conclusions of a study of mine-effluent quality from a large number of recently abandoned mines using several different closure techniques (Bucek and Emel, 1977). In that study, double bulkhead seals were effective in reducing acidity of mine discharges at 80 percent of the sites. However, the reductions ranged from 32 to 100 percent, and changes in sulfate and iron concentrations were erratic.

Air seals are another common mine-drainage control technique which may indirectly affect ground-water quality near underground mines. During the 1930's, the Federal Work Projects Administration closed thousands of mines (mostly with air seals) throughout the eastern United States to abate acid-mine drainage (Thompson and Emrich, 1969). Less extensive programs have continued to the present time. In this method, air-tight seals are constructed at all portals and ventilation shafts in an attempt to reduce oxygen and the resulting production of sulfuric acid in the mine. Results of a demonstration project near Elkins, West Virginia, indicate that air sealing at a small underground mine did reduce acidity, iron, and sulfate concentrations (Industrial Environmental Research Laboratory, 1977). However, at a large underground mine nearby, air seals were found to be impractical because of the many adits, shafts, and rock fractures that introduced air into the mine. Available data from this and other studies suggest that air seals will have a beneficial effect on ground-water and surface-water quality only if mines are deep and subsidence and fracturing are minimal.

Surface-Water Diversion

As noted in a previous section, loss of surface water to underground mines or to rocks affected by mining will tend to increase the deterioration of ground-water and mine-drainage quality. A variety of techniques exist for preventing or reducing the amount of surface water that enters a mine. These include canals, gravity drains, interceptor trenches, and stream-channel liners. Unfortunately, many of these can deteriorate after relatively short periods of use. In addition, most techniques are suitable only for intermittent or small drainageways. By far, the most effective technique for preventing surface-water seepage to underground mines is to take precautions prior to and during mining. In most cases, this involves leaving sufficient rock supports below water courses, especially where subsidence is likely or where rock fractures may reach the surface.

Ground-Water Diversion

A hydrologic control which deals more specifically with ground water involves diversion of ground water in the vicinity of a mine by means of pumping wells. Variations of this technique have been described by several investigators (Parizek and Tarr, 1972; Schubert, 1978). The basic concept involves interception of ground water which would normally enter a mine and become contaminated by solution of exposed sulfide minerals. The pumped water is disposed of either by direct discharge to a stream or by gravity drainage through connector wells to deeper aquifers. Computer simulation of a connector well scheme in Clearfield County, Pennsylvania, showed that negligible reduction in leakage into the study mine would occur where the permeability of the overlying sandstone was less than 0.00035 cm/s (7.4 gpd/ft^2) (Schubert, 1978). Higher sandstone permeabilities, more than 0.0035 cm/s (74.2 gpd/ft^2), are more conducive to dewatering. According to the model, 25 connector wells would potentially decrease vertical leakage in the mine investigated by only 2.4 percent, assuming a sandstone permeability of 0.0035 cm/s (74.2 gpd/ft^2).

A recent pilot study near Carrolltown, Pennsylvania, investigated the feasibility of dewatering at a specific site (W. A. Wahler & Assoc., 1978). Wells placed on a line near the working face of the mine pumped about 5 l/s (80 gpm) and caused an average reduction of 34 percent in inflow to the mine. Data indicate that about 45 percent of the water pumped from the wells would otherwise have entered the mine. Projection of data from a pumping test of several weeks duration indicated that after 120 days of pumping, up to 80 percent of the water pumped would have been diverted from mine flows. Discharge water from the pumped wells was of good quality and was diverted to a nearby stream without treatment.

The concept of ground-water diversion around underground mines by pumping wells is technically sound under certain circumstances. The efficient application of this method requires a good understanding of the ground-water system around the mine. Conditions most favorable for diversion seem to be at wet mines where the mine discharge quality is acidic and the surrounding ground-water quality is good. Under these conditions, because lime treatment would be expensive, the costs of pumping water may be justified.

Underdrains or Liners

A method commonly used for controlling leachate from solid-waste sites is the construction of underdrains or liner systems which collect the waste fluids and allow treatment at a central location prior to discharge. This method, however, is only suitable for new sites or expanding sites where the system can be installed prior to waste disposal. Liners and/or underdrains have been suggested for use at coal refuse piles, although no sites with operating systems were encountered in the course of this study.

Assuming that an underdrain or liner would function as designed, this method would virtually eliminate the potential for ground-water contamination. However, engineering difficulties have been encountered in installations at solid-waste facilities. In addition, the overall cost-effectiveness of such techniques for coal refuse has not been evaluated. It is difficult to envision such methods being employed universally. However, in areas where ground water is an important resource and may be threatened by proposed refuse disposal, liners or underdrains may be practical.

Reclamation

The most common form of refuse reclamation, where it is practiced, is the revegetation of the pile surface. This commonly involves the placement of some form of topsoil, which is typically seeded with selected legumes and hardy species of grasses including fescue, rye, reed canary, and barley. The purpose of such reclamation is partly to improve aesthetics and partly to reduce sediment loading to nearby surface waters. Probably the only beneficial effect on ground water that might be expected as a result of standard reclamation techniques is a reduction in infiltration because of increased evapotranspiration. Reduced infiltration would, presumably, result in some reduction in leachate production and subsequent reduction in contamination of nearby ground waters and surface waters.

Several demonstration reclamation projects relating to the disposal of highly acidic refuse have involved the addition of lime to the pile surface in an effort to buffer infiltrating waters. A study by Waddell (1978) evaluated the effect of coal waste flue dust and limestone fragments placed on the surface of acidic highway embankments in Centre County, Pennsylvania, on ground- and surface-water quality. These embankments are similar to coal refuse piles in that exposed rock is heavily laden with iron sulfide minerals. Detailed monitoring of the water quality of seeps downgradient from the embankment spoil indicate that some improvement in water quality did occur, apparently as a result of treatment. pH levels increased from about 4 to approximately 5.5 at one sampling point. At the same site, sulfate and acidity concentrations also decreased. The treatment was not effective in reducing the acid load of nearby surface waters because the volume of pyrite-bearing rock in the treated area was small compared to that in the highway embankment and spoil piles.

Work at the New Kathleen Mine in Illinois included test plots covered with soil and treated with agricultural limestone. A vegetative cover was established and surface-runoff quality was monitored for one year. The average rate of acid formation for runoff from the entire restored refuse pile was estimated at 2.9 kg (6.5 lb.) acid as $CaCO_3$/ha/d, representing a reduction of over 91 percent as compared with the original unrestored pile. No ground-water samples were taken to see if ground-water quality also showed improvement.

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APPENDIX A

TABLE A-1. QUADRANGLES IN PENNSYLVANIA
(cited in Figures 18 and 20)

1.	Sugar Lake	39.	Cranberry
2.	Dempseytown	40.	Kossuth
3.	Titusville - South	41.	Fryburg
4.	Pleasantville	42.	Lucinda
5.	West Hickory	43.	Cooksburg
6.	Kellettville	44.	Sigel
7.	Mayburg	45.	Munderf
8.	Lynch	46.	Corman
9.	Russell City	47.	Brandy Court
10.	James City	48.	Kersey
11.	Wilcox	49.	Weedville
12.	Glen Hazel	50.	Dents Run
13.	Wildwood Fire Tower	51.	Sharon West
14.	Kinsman	52.	Sharon East
15.	Greenville West	53.	Greenfield
16.	Greenville East	54.	Mercer
17.	Hadley	55.	Grove City
18.	New Lebanon	56.	Barkeyville
19.	Utica	57.	Eau Claire
20.	Franklin	58.	Emunton
21.	Oil City	59.	Knox
22.	President	60.	Clarion
23.	Tionesta	61.	Strattonville
24.	Tylersburg	62.	Corsica
25.	Marienville - West	63.	Brookville
26.	Marienville - East	64.	Hazen
27.	Hallton	65.	Falls Creek
28.	Portland Mills	66.	Sabula
29.	Ridgway	67.	Penfield
30.	St. Marys	68.	Huntley
31.	Rathbun	69.	The Knobs
32.	Orangeville	70.	Devils Elbow
33.	Sharpsville	71.	Pottersdale
34.	Fredonia	72.	Campbell
35.	Jackson Center	73.	Edinburg
36.	Sandy Lake	74.	New Castle - North
37.	Polk	75.	Harlansburg
38.	Kennerdell	76.	Slippery Rock

(continued)

TABLE A-1 (continued)

77.	West Sunbury	126.	Marion Center
78.	Hilliards	127.	Rochester Mills
79.	Parker	128.	Burnside
80.	Rimbersburg	129.	Westover
81.	Sligo	130.	Irvona
82.	New Bethlehem	131.	Ramey
83.	Summerville	132.	Houtzdale
84.	Coolspring	133.	Sandy Ridge
85.	Reynoldsville	134.	East Liverpool - North
86.	Du Bois	135.	Midland
87.	Luthersburg	136.	Beaver
88.	Elliott Park	137.	Baden
89.	Clearfield	138.	Mars
90.	Lecontes Mills	139.	Valencia
91.	Frenchville	140.	Curtisville
92.	Karthaus	141.	Freeport
93.	New Middletown	142.	Leechburg
94.	Bessemer	143.	Whitesburg
95.	New Castle - South	144.	Elderton
96.	Portersville	145.	Ernest
97.	Prospect	146.	Clymer
98.	Mt. Chestnut	147.	Commodore
99.	East Butler	148.	Barnesboro
100.	Chicora	149.	Hastings
101.	East Brady	150.	Coalport
102.	Templeton	151.	Blandburg
103.	Distant	152.	East Liverpool - South
104.	Dayton	153.	Hookstown
105.	Valier	154.	Aliquippa
106.	Punxsutawney	155.	Ambridge
107.	McGees Mills	156.	Emsworth
108.	Mahaffey	157.	Glenshaw
109.	Curwensville	158.	New Kensington - West
110.	Glen Richey	159.	New Kensington - East
111.	Wallaceton	160.	Vandergrift
112.	Philipsburg	161.	Avonmore
113.	Black Moshannon	162.	McIntyre
114.	East Palestine	163.	Indiana
115.	New Galilee	164.	Brush Valley
116.	Beaver Falls	165.	Strongstown
117.	Zelienople	166.	Colver
118.	Evans City	167.	Carrolltown
119.	Butler	168.	Ashville
120.	Saxonburg	169.	Altoona
121.	Worthington	170.	Weirton
122.	Kittanning	171.	Burgettstown
123.	Mosgrove	172.	Clinton
124.	Rural Valley	173.	Oakdale
125.	Plumville	174.	Pittsburgh - West

(continued)

TABLE A-1 (continued)

175. Pittsburgh - East
176. Braddock
177. Murrysville
178. Slickville
179. Saltsburg
180. Blairsville
181. Bolivar
183. New Florence
183. Vintondale
184. Nanty Glo
185. Ebensburg
186. Cresson
187. Steubenville East
188. Avella
189. Midway
190. Canonsburg
191. Bridgeville
192. Glassport
193. McKeesport
194. Irwin
195. Greensburg
196. Latrobe
197. Derry
198. Wilpen
199. Rachelwood
200. Johnstown
201. Geistown
202. Beaverdale
203. Blue Knob
204. Kethany
205. West Middletown
206. Washington West
207. Washington East
208. Hackett
209. Monongahela
210. Donora
211. Smithton
212. Mt. Pleasant
213. Mammoth
214. Stahlstown
215. Ligonier
216. Boswell
217. Hooversville
218. Windber
219. Ogletown
220. Valley Grove
221. Claysville
222. Prosperity
223. Amity
224. Ellsworth
225. California
226. Fayette City
227. Dawson
228. Connellsville
229. Donegal
230. Seven Springs
231. Bakersville
232. Somerset
233. Stoystown
234. Central City
235. Schellsburg
236. Majorsville
237. Wind Ridge
238. Rogersville
239. Waynesburg
240. Mather
241. Carmichaels
242. New Salem
243. Uniontown
244. South Connellsville
245. Mill Run
246. Kingwood
247. Rockwood
248. Murdock
249. Berlin
250. New Baltimore
251. Cameron (W. Va.)
252. New Freeport
253. Halbrook
254. Oak Forest
255. Garards Fort
256. Masontown
257. Smithfield
258. Brownfield
259. Ft. Necessity
260. Ohiopyle
261. Confluence
262. Markleton
263. Meyersdale
264. Wittenberg
265. Fairhope

TABLE A-2. QUADRANGLES IN VIRGINIA
(cited in Figure 21)

1.	Majestic	44.	Wise
2.	Wharn Cliffe	45.	Toms Creek
3.	Jamboree	46.	St. Paul
4.	Hurley	47.	Carbo
5.	Panther	48.	Lebanon
6.	Hellier	49.	Elk Garden
7.	Elkhorn	50.	Saltville
8.	Harman	51.	Broadford
9.	Grundy	52.	Chatham
10.	Patterson	53.	Nebo
11.	Bradshaw	54.	Evarts
12.	War	55.	Pennington Gap
13.	Gary	56.	Keokee
14.	Anawalt	57.	Big Stone Gap
15.	Bramwell	58.	East Stone Gap
16.	Jenkins West	59.	Fort Blackmore
17.	Jenkins East	60.	Dungannon
18.	Clintwood	61.	Moll Creek
19.	Haysi	62.	Hansonville
20.	Prater	63.	Brumley
21.	Vansant	64.	Hayters Gap
22.	Keen Mtn.	65.	Varilla
23.	Jewell Ridge	66.	Ewing
24.	Amonate	67.	Rose Hill
25.	Tazewell North	68.	Hubbard Springs
26.	Tiptop	69.	Ben Hur
27.	Cove Creek	70.	Stickleyville
28.	Whitesburg	71.	Duffield
29.	Flat Gap	72.	Clinchport
30.	Pound	73.	Gate City
31.	Caney Ridge	74.	Hilton
32.	Nora	75.	Mendota
33.	Duty	76.	Wheeler
34.	Big A Mtn.	77.	Coleman Gap
35.	Honaker	78.	Back Valley
36.	Richlands	79.	Sneedville
37.	Pounding Mill	80.	Kyles Ford
38.	Tazewell South	81.	Looneys Gap
39.	Hutchinson Rock	82.	Plum Grove
40.	Garden Mtn.	83.	Church Hill
41.	Benham	84.	Kingsport
42.	Appalachia	85.	Indian Springs
43.	Norton	86.	Blountville

TABLE A-3. QUADRANGLES IN WEST VIRGINIA
(cited in Figure 22)

1.	Weirton	48.	Aurora
2.	Steubenville	49.	Table Rock
3.	Tiltonsville	50.	Gorman
4.	Bethany	51.	Mount Storm
5.	Wheeling	52.	West Milford
6.	Valley Grove	53.	Mount Clare
7.	Businessburg	54.	Brownton
8.	Moundsville	55.	Philippi
9.	Majorsville	56.	Nestorville
10.	Powhatan Point	57.	Colebank
11.	Glen Easton	58.	Lead Mine
12.	Littleton	59.	Davis
13.	Wadestown	60.	Mt. Storm Lake
14.	Blacksville	61.	Greenland
15.	Osage	62.	Pomeroy
16.	Morgantown North	63.	Vadis
17.	Lake Lynn	64.	Camden
18.	Bruceton Mills	65.	Weston
19.	Brandonville	66.	Berlin
20.	Glover Gap	67.	Century
21.	Mannington	68.	Audra
22.	Grant Town	69.	Belington
23.	Rivesville	70.	Mozark Mtn.
24.	Morgantown South	71.	Addison
25.	Masontown	72.	Cheshire
26.	Valleypoint	73.	Glenville
27.	Cuzzart	74.	Gilmer
28.	Wallace	75.	Roanoke
29.	Shinnston	76.	Adrian
30.	Fairmont West	77.	Junior
31.	Fairmont East	78.	Elkins
32.	Gladesville	79.	Bowden
33.	Newburg	80.	Cedarville
34.	Kingwood	81.	Burnsville
35.	Terra Alta	82.	Walkersville
36.	Oakland	83.	Rock Cove
37.	Kitzmiller	84.	Sago
38.	Westernport	85.	Beverly West
39.	Keyser	86.	Beverly East
40.	Salem	87.	Elmwood
41.	Wolf Summit	88.	Sutton
42.	Clarksburg	89.	Hacker Valley
43.	Rosemont	90.	Goshen
44.	Grafton	91.	Bancroft
45.	Thornton	92.	Sissonville
46.	Fellowsville	93.	Ivydale
47.	Rowlesburg	94.	Little Birch

(continued)

TABLE A-3 (continued)

95.	Diana	146.	Radnor
96.	Skelt	147.	Kiahsville
97.	Pickens	148.	Big Creek
98.	Durbin	149.	Mud
99.	Saint Albans	150.	Madison
100.	Pocatalico	151.	Williams Mtn.
101.	Big Chimney	152.	Sylvester
102.	Blue Creek	153.	Eskdale
103.	Clendenin	154.	Powellton
104.	Elkhurst	155.	Beckwith
105.	Clay	156.	Fayetteville
106.	Swandale	157.	Winona
107.	Widen	158.	Corliss
108.	Tioga	159.	Quinwood
109.	Cowen	160.	Duo
110.	Webster Springs	161.	Lobelia
112.	Bergoo	162.	Webb
113.	Lavalette	163.	Trace
114.	Winslow	164.	Chapmanville
115.	Charleston West	165.	Henlawson
116.	Charleston East	166.	Clothier
117.	Quick	167.	Wharton
118.	Mammoth	168.	Whitesville
119.	Bentree	169.	Dorothy
120.	Lockwood	170.	Pax
121.	Gilboa	171.	Oak Hill
122.	Summersville	172.	Thurmond
123.	Carigsville	173.	Danese
124.	Camden on Gauley	174.	Rainelle
125.	Webster Springs - SW	175.	Rupert
126.	Webster Springs - SE	176.	Kermit
127.	Mingo	177.	Naugatuck
128.	Prichard	178.	Myrtle
129.	Wayne	179.	Holden
130.	Nestlow	180.	Logan
131.	Branchland	181.	Amherstdale
132.	Hager	182.	Lorado
133.	Griffithsville	183.	Pilot Knot
134.	Julian	184.	Arnett
135.	Racine	185.	Eccles
136.	Belle	186.	Beckley
137.	Cedar Grove	187.	Prince
138.	Montgomery	188.	Meadow Creek
139.	Gauley Bridge	189.	Meadow Bridge
140.	Ansted	190.	Asbury
141.	Summersville Dam	191.	Williamson
142.	Mt. Nebo	192.	Delbarton
143.	Nettie	193.	Barnabus
144.	Richwood	194.	Man
145.	Louisa	195.	Mallory

(continued)

TABLE A-3 (continued)

196.	Oceana
197.	Matheny
198.	McGraws
199.	Lester
200.	Crab Orchard
201.	Shady Spring
202.	Hinton
203.	Fort Spring
204.	Matewan
205.	Majestic
206.	Wharncliffe
207.	Gilbert
208.	Baileysville
209.	Pineville
210.	Mullens
211.	Rhodell
212.	Odd
213.	Flat top
214.	Panther
215.	Iaeger
216.	Davy
217.	Welch
218.	Keystone
219.	Crumpler
220.	Matoaka
221.	Patterson
222.	Bradshaw
223.	War
224.	Gary
225.	Anawalt
226.	Bramwell
227.	Bluefield
228.	Amonate
229.	Tazewell - North

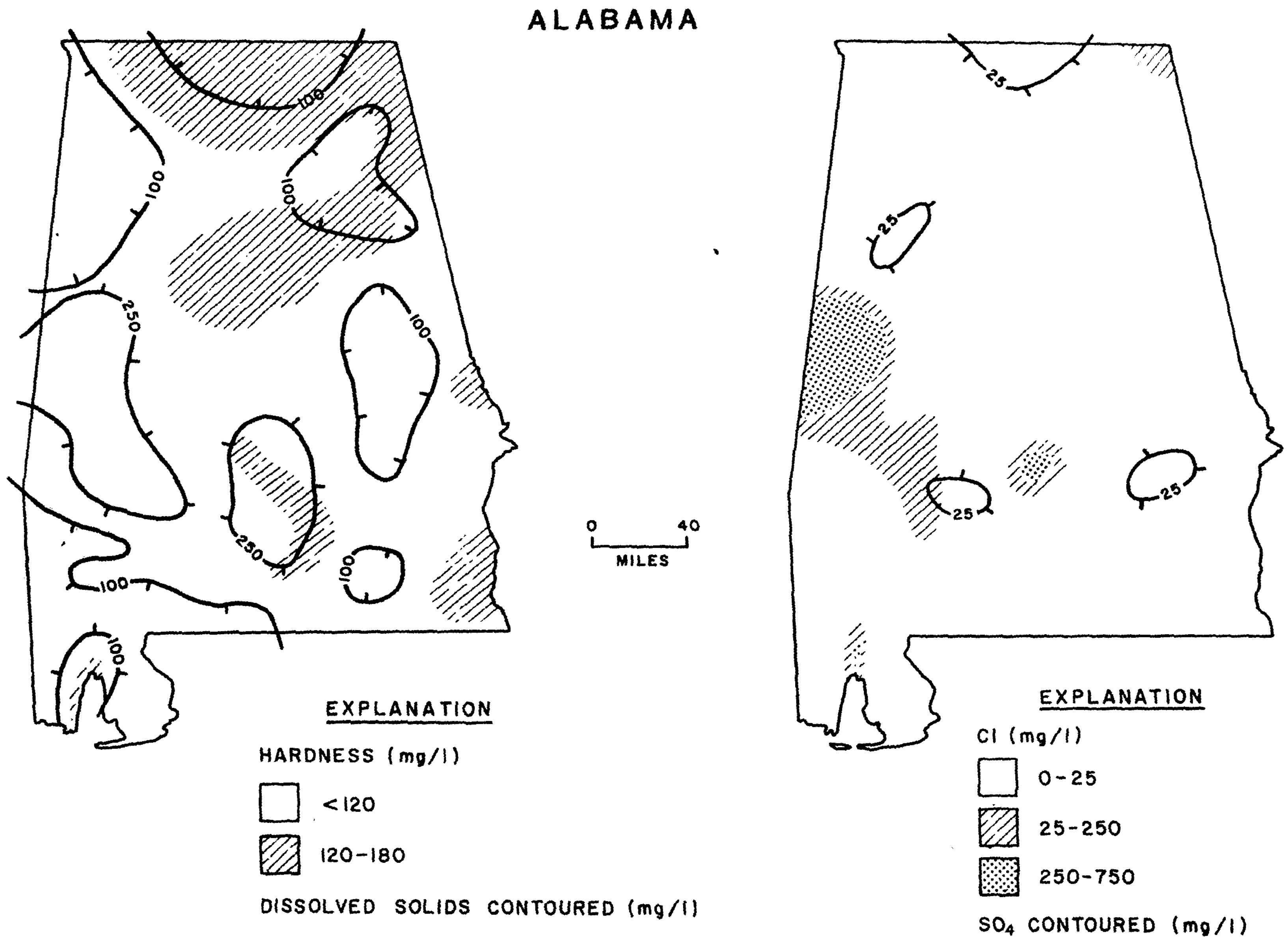

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Figure B-1. Ground-water quality in Alabama.

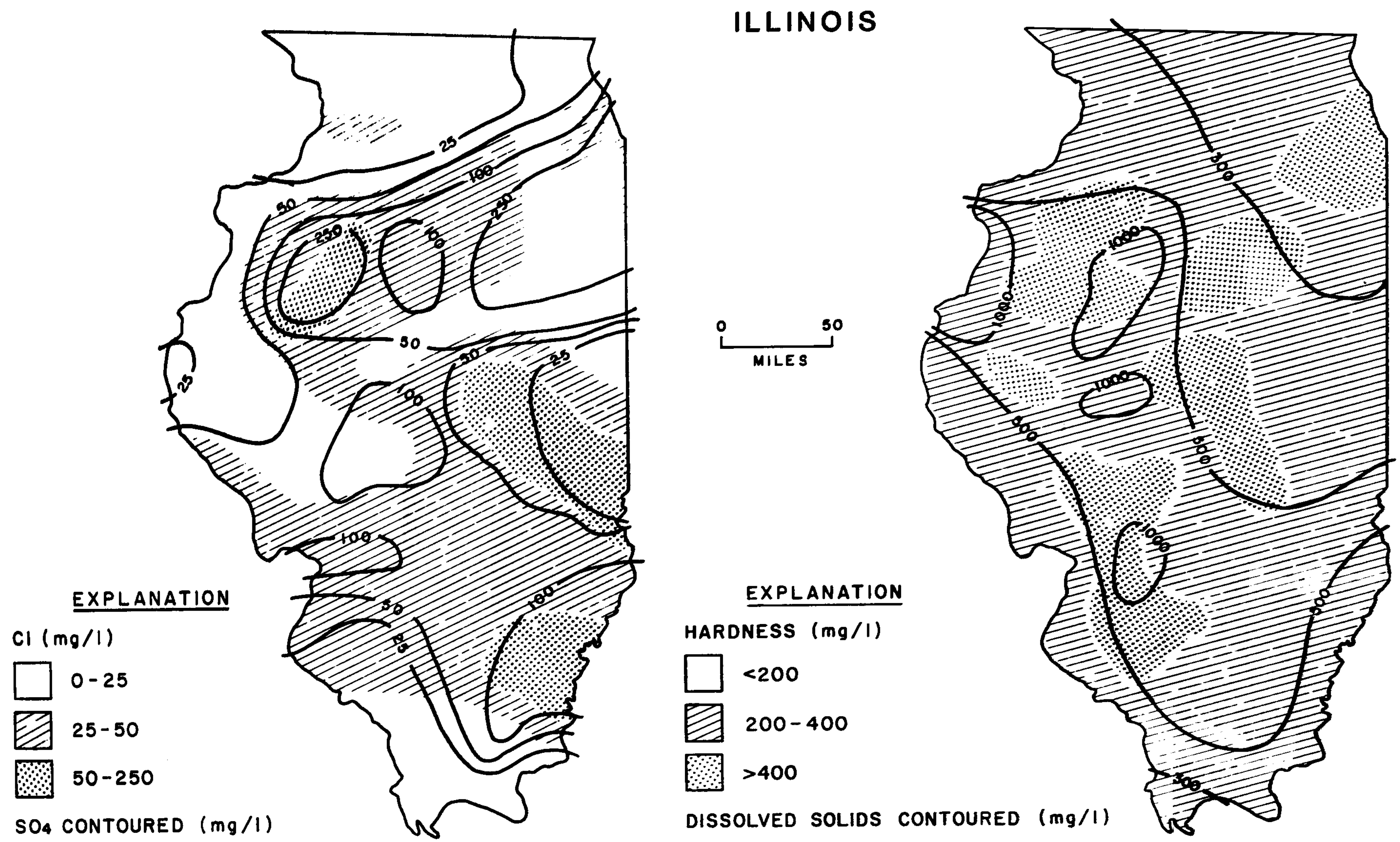

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Figure B-2. Ground-water quality in Illinois.

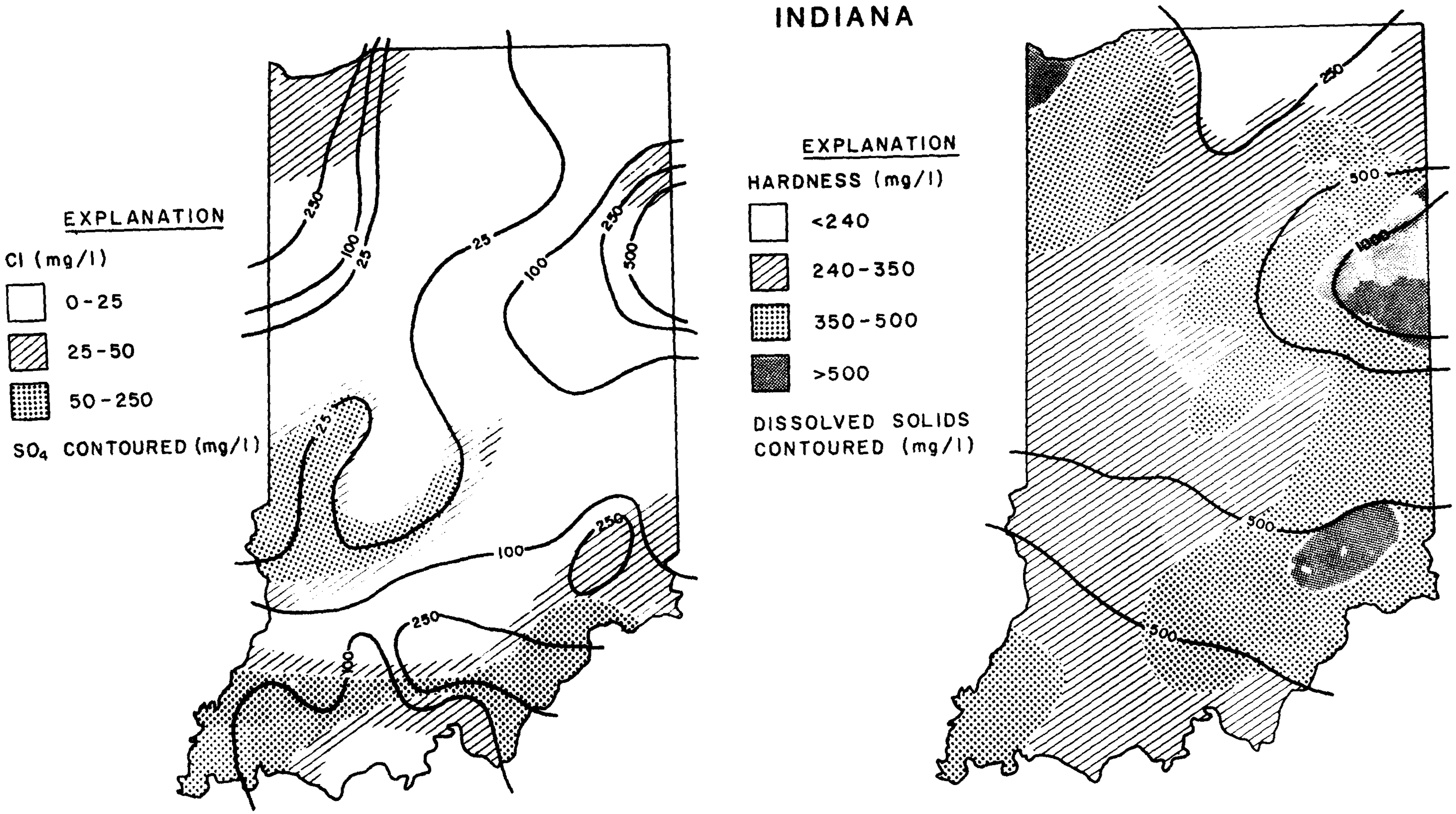

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Figure B-3. Ground-water quality in Indiana.

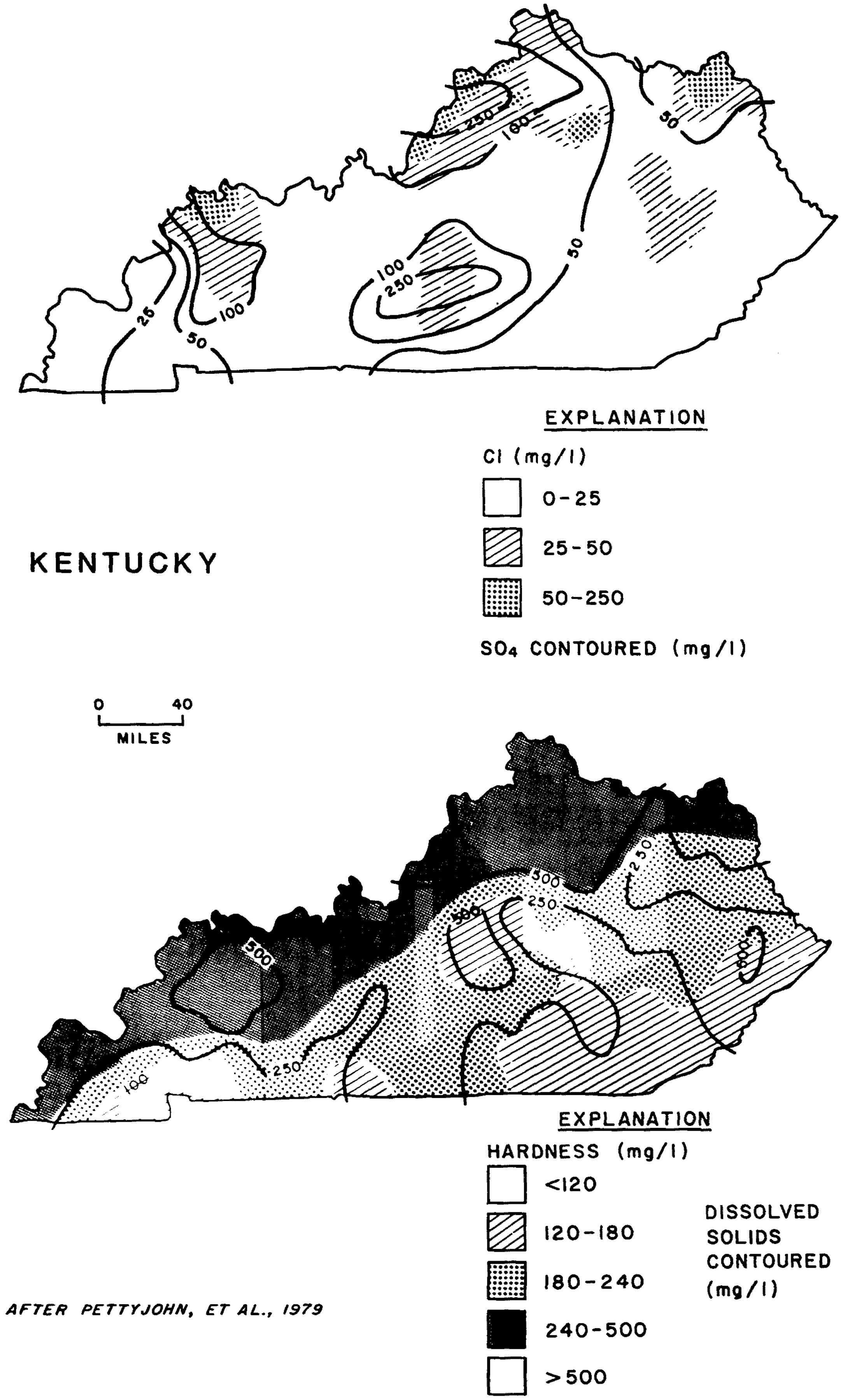

Figure B-4. Ground-water quality in Kentucky.

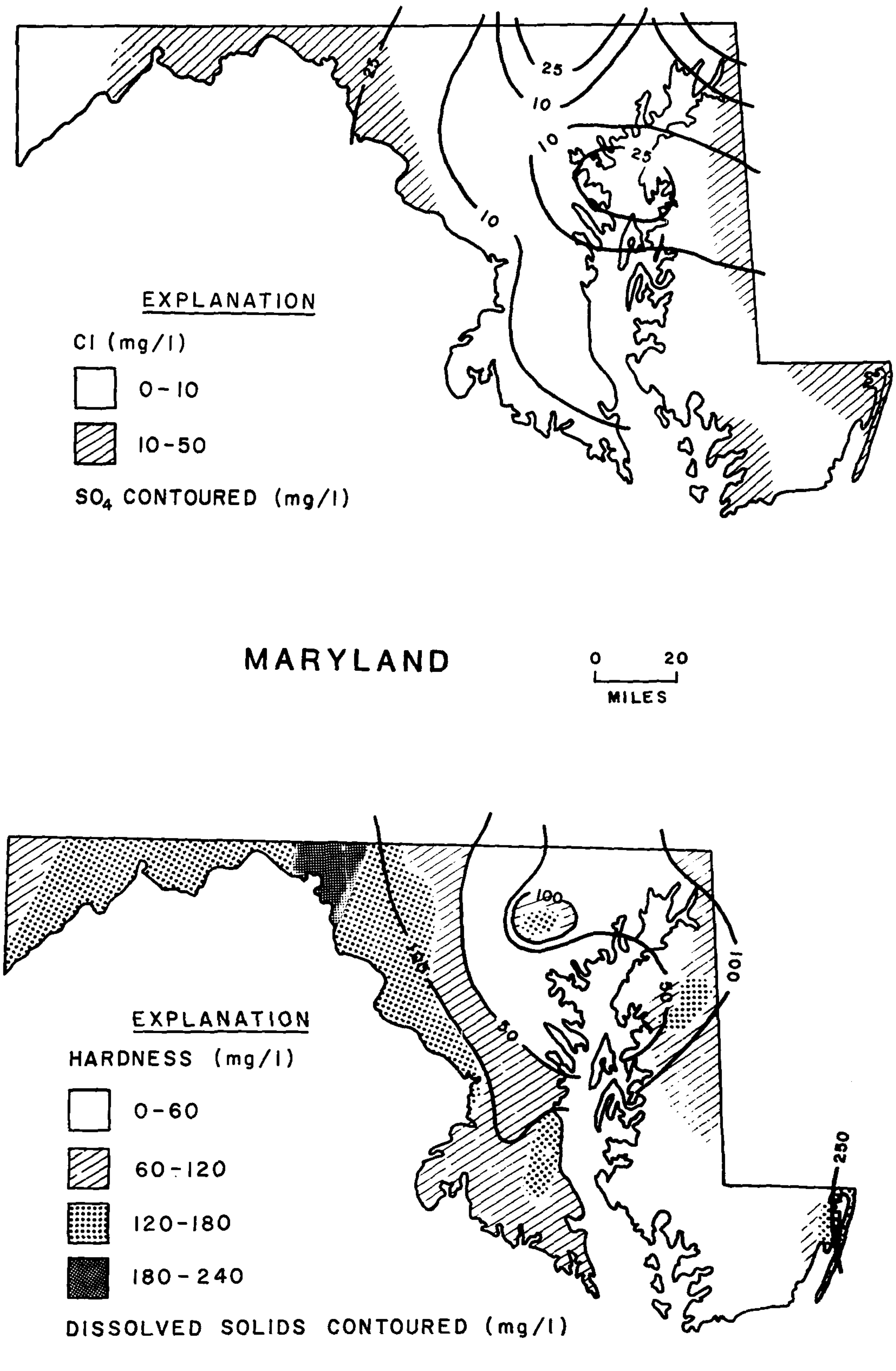

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Figure B-5. Ground-water quality in Maryland.

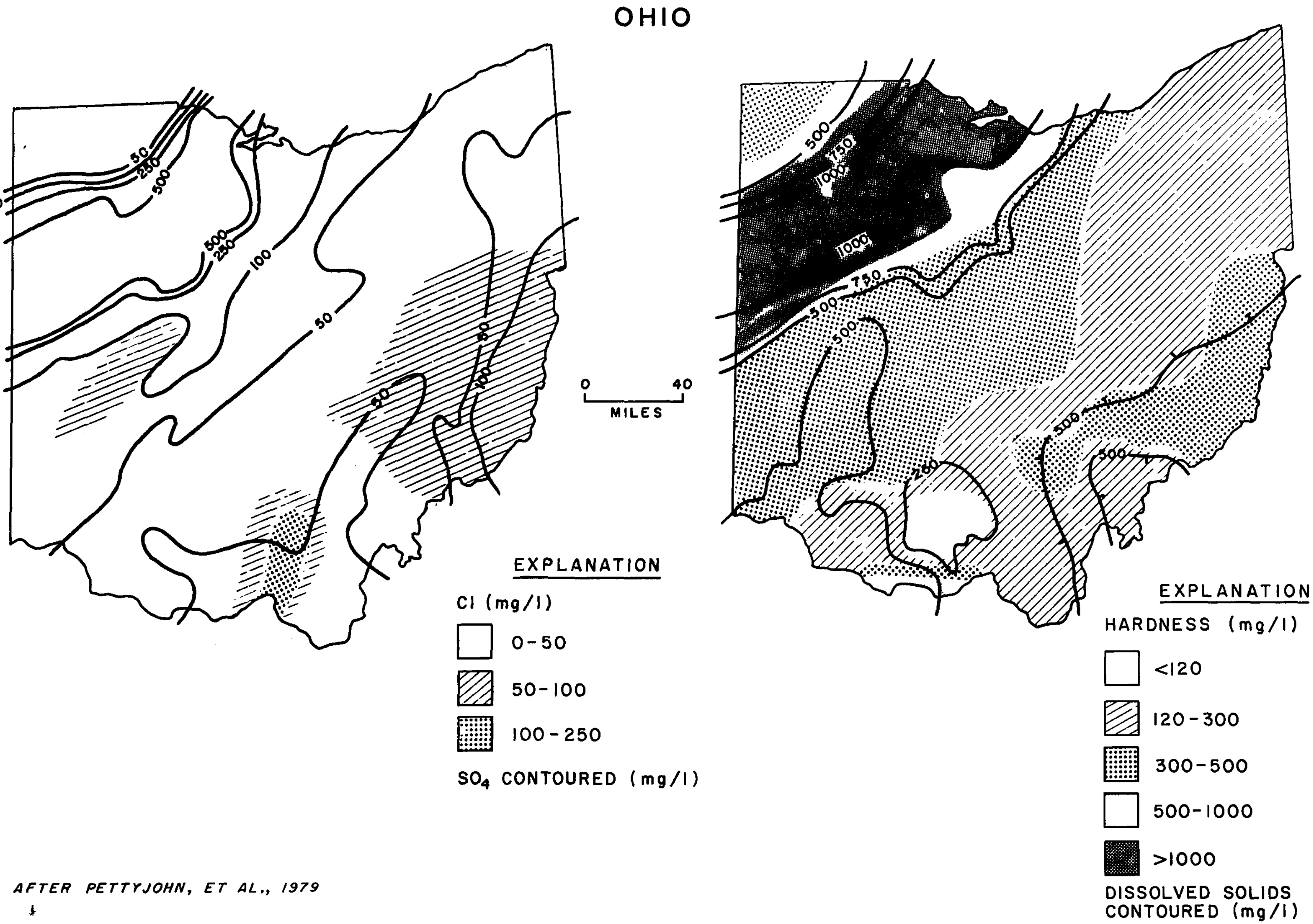

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Figure B-6. Ground-water quality in Ohio.

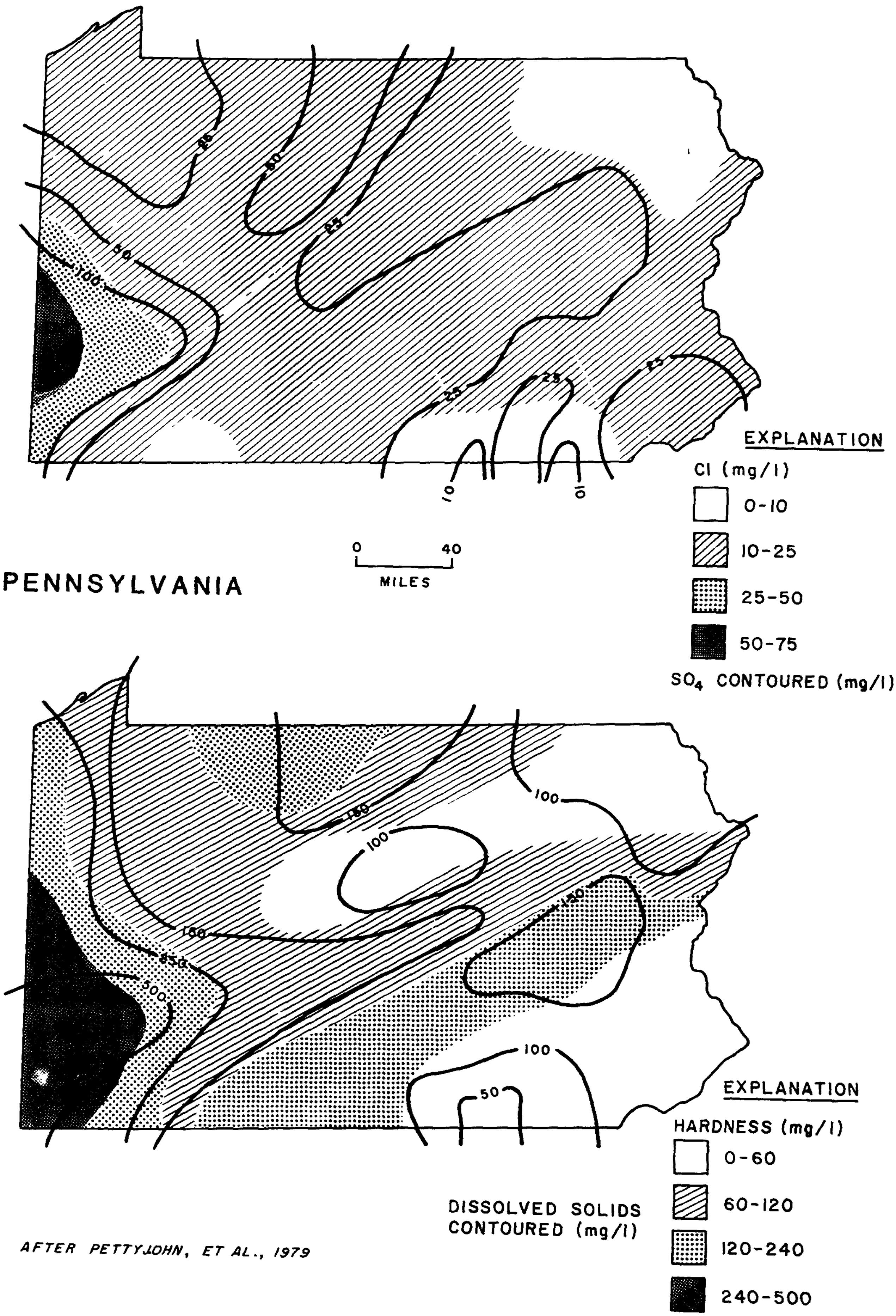

Figure B-7. Ground-water quality in Pennsylvania.

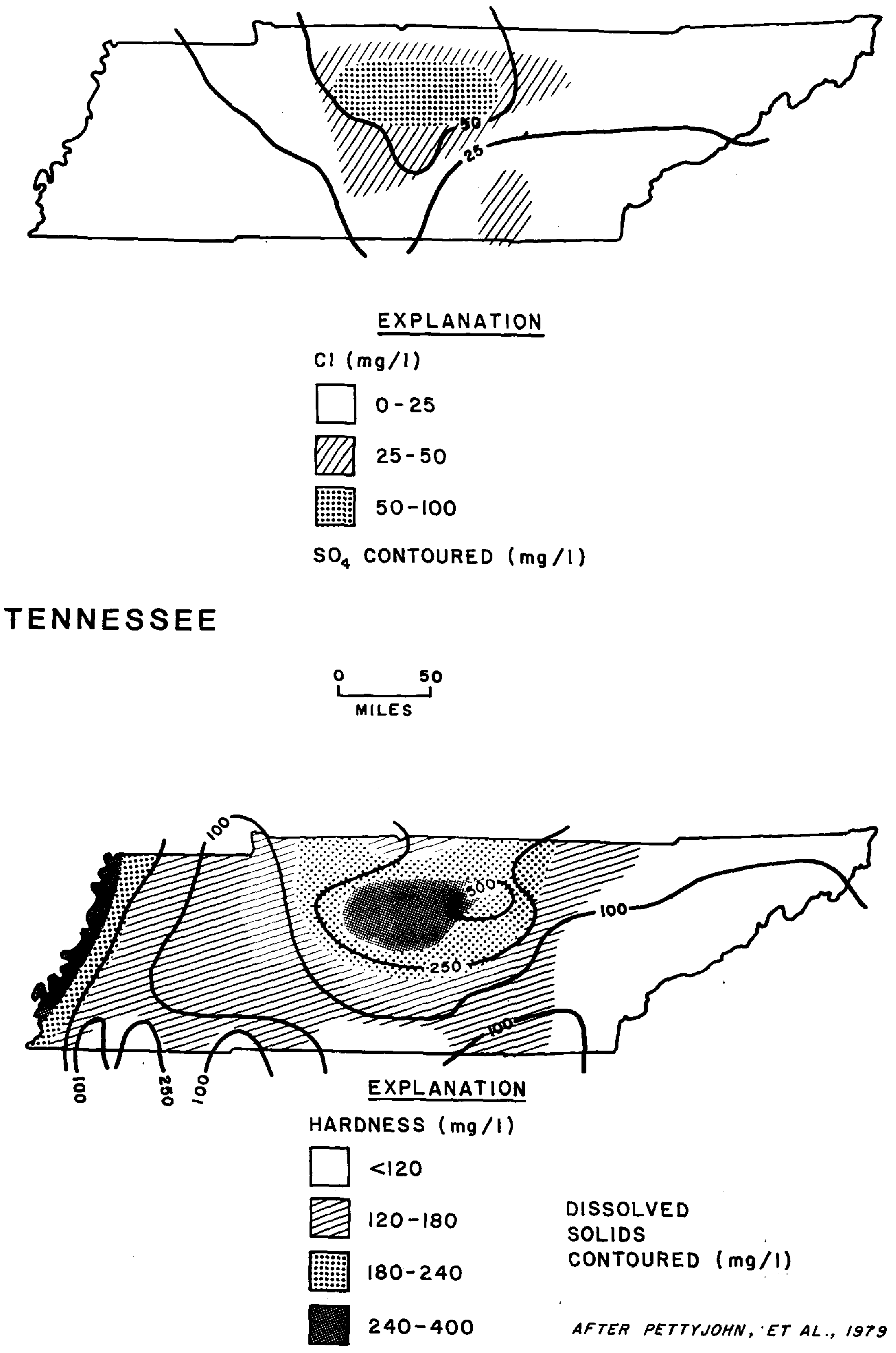

Figure B-8. Ground-water quality in Tennessee.

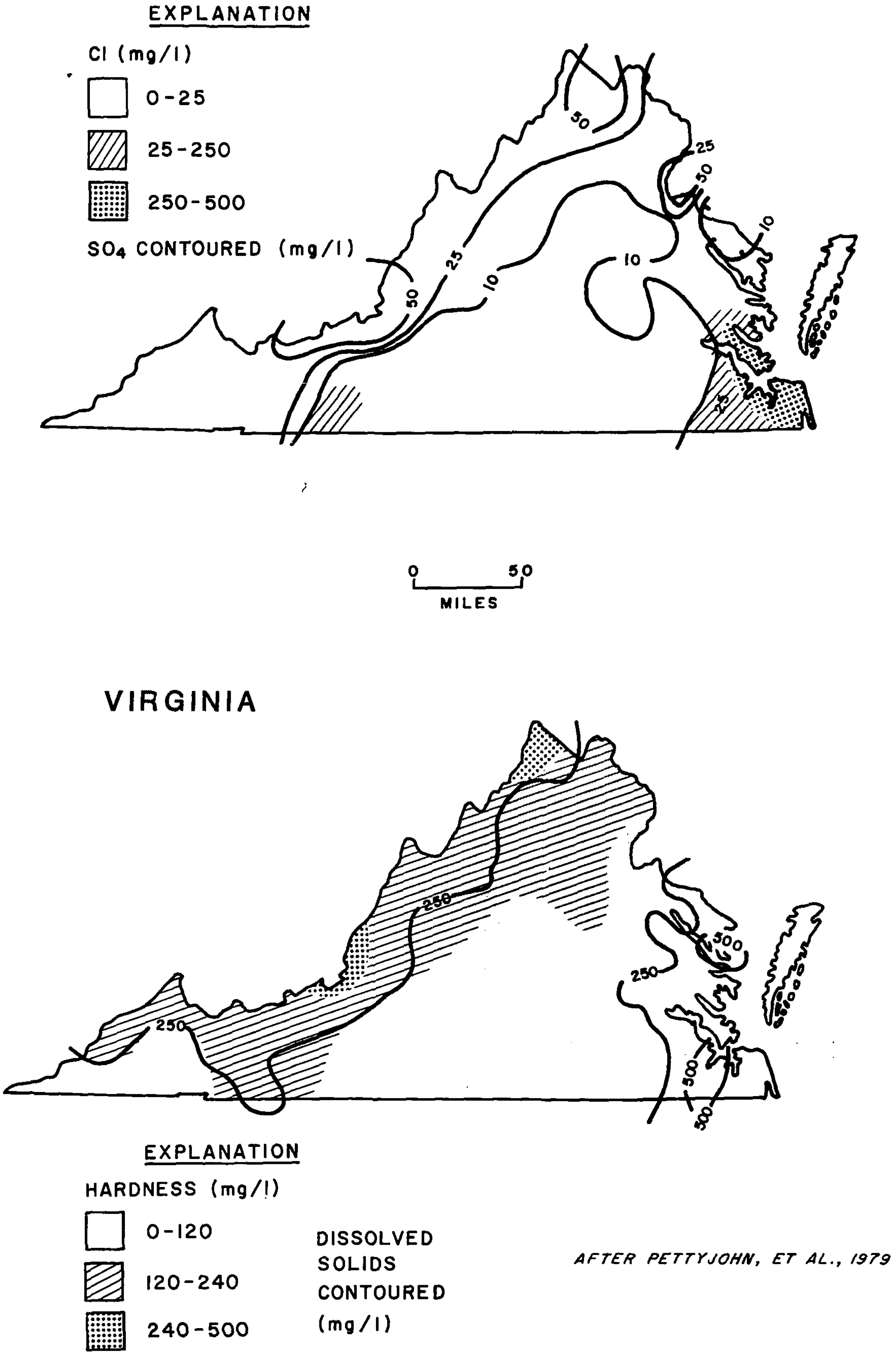

Figure B-9. Ground-water quality in Virginia.

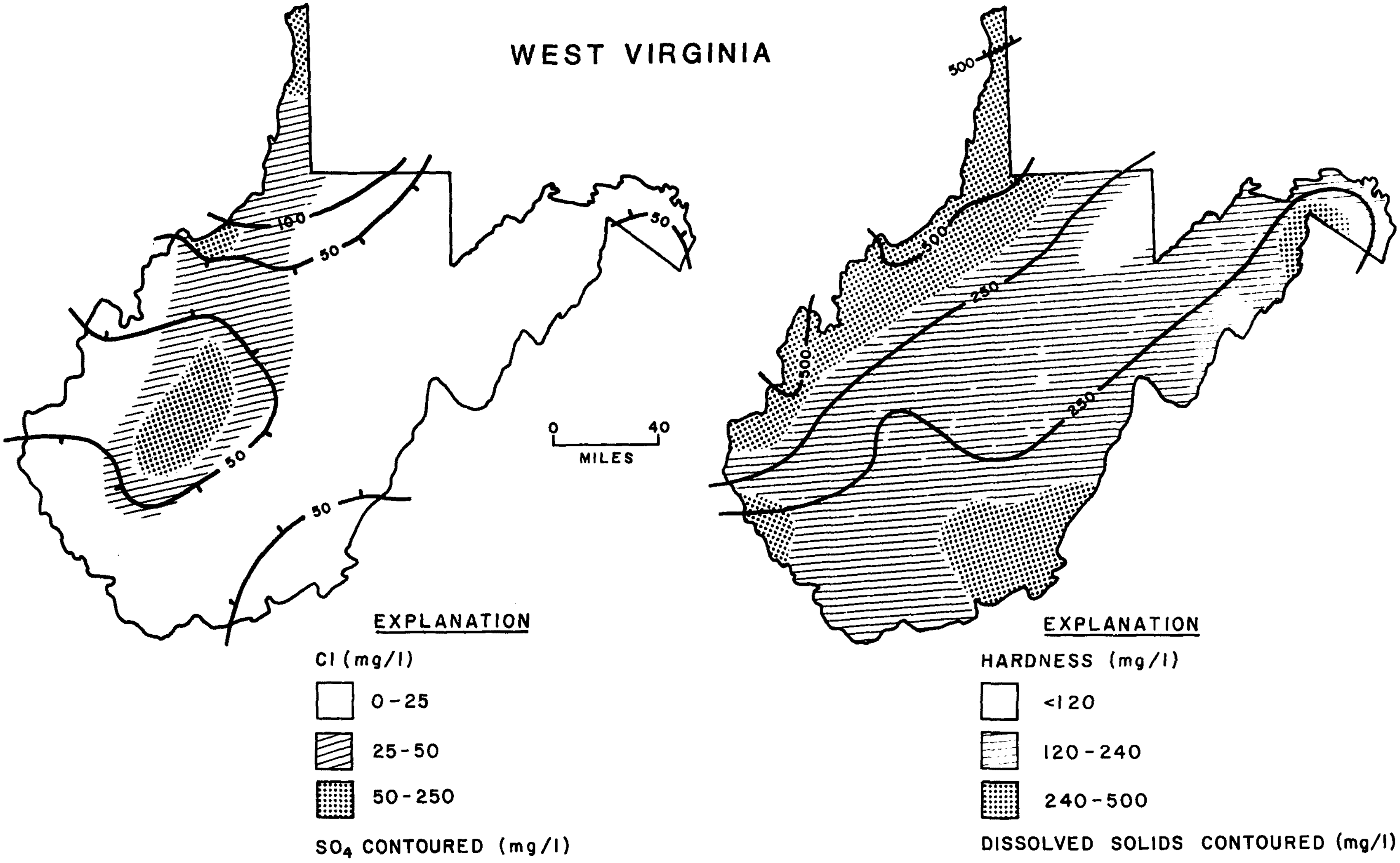

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Figure B-10. Ground-water quality in West Virginia.

TECHNICAL REPORT DATA

(Please read Instructions on the reverse before completing)

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16. ABSTRACT This report addresses the past effects and the possible future effects of underground coal mining activities on ground-water resources in the region east of the 100th meridian. Such effects are highly dependent on the location of the mine with respect to natural flow system. Recharge-discharge relationships in the vicinity of active mines may be altered, and lowered ground-water levels may not recover to pre-mining conditions after closure.

Studies indicate that contamination of ground water exists in many places in the immediate vicinity of coal mines. Many refuse piles and impoundments likely affect stream and shallow ground-water quality. However, on a regional basis, there is little evidence from the scanty data on hand of gross ground-water contamination in heavily mined areas.

From the viewpoint of the value of ground-water resources, it is most likely that future underground mining in the Eastern Interior Basin and the southern Appalachians will result in adverse ground-water effects in only very limited areas. The central Appalachians, and in particular parts of western Pennsylvania and southern West Virginia, have a greater potential for such impacts. Pre-mining planning based on knowledge of local hydrogeology and geochemistry can lead to changes in mining techniques or planning that will help to minimize adverse effects on ground water.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
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