United States
Environmental Protection
Agency

Office of
Solid Waste and
Emergency Response

9285.9-15C
EPA/540/R-95/060
PB95-963240

Superfund

INTRODUCTION TO GROUNDWATER INVESTIGATIONS

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FOREWORD

This manual is for reference use of students enrolled in scheduled training courses of the U.S. Environmental Protection Agency (EPA). While it will be useful to anyone who needs information on the subjects covered, it will have its greatest value as an adjunct to classroom presentations involving discussions among the students and the instructional staff.

This manual has been developed with a goal of providing the best available current information; however, individual instructors may provide additional material to cover special aspects of their presentations.

Because of the limited availability of the manual, it should not be cited in bibliographies or other publications.

References to products and manufacturers are for illustration only; they do not imply endorsement by EPA.

Constructive suggestions for improvement of the content and format of the manual are welcome.

INTRODUCTION TO GROUNDWATER INVESTIGATIONS

(165.7)

3 Days

This introductory course is designed to provide participants with information concerning hydrogeological processes and the necessary elements of a sound groundwater site investigation. It is intended for personnel who are involved in groundwater contamination investigations but have **little prior hydrogeological experience**. This course is **not** designed for geologists or hydrogeologists.

Topics that are discussed include hydrogeological definitions and concepts; rock cycle; soil formation; depositional environments; geochemistry; geophysics; drilling, construction, and placement of monitoring wells; groundwater sampling considerations; and groundwater modeling.

Instructional methods include lectures, group discussions, case studies, and class problem-solving exercises.

After completing the course, participants will be able to:

- Identify the components of a groundwater system.
- List the primary hydrogeological parameters to be considered in a site investigation.
- Construct a flow net and calculate the hydraulic gradient at a site.
- Discuss the primary advantages and disadvantages of the most common geophysical survey methods.
- Identify geochemical profiles in contaminated groundwater.
- Identify the different types of pumping tests and the information that can be obtained from each.
- Describe monitoring well drilling and sampling techniques.

CONTENTS

Acronyms and Abbreviations

Glossary

SECTION 1 STANDARD ORIENTATION AND INTRODUCTION

SECTION 2 ROCK CYCLE

SECTION 3 DEPOSITIONAL ENVIRONMENTS

SECTION 4 SOILS

SECTION 5 DRILLING METHODS

SECTION 6 HYDROGEOLOGY

SECTION 7 WELL INSTALLATION

SECTION 8 VADOSE ZONE

SECTION 9 GEOPHYSICAL METHODS

SECTION 10 GEOCHEMICAL MODELS

SECTION 11 GROUNDWATER MODELS

SECTION 12 PROBLEM EXERCISES
- Problem 1—Cross-section Exercise
- Problem 2—Sediment Analysis
- Problem 3—Groundwater Model Demonstration
- Problem 4—Hydrogeological Exercises
- Problem 5—Aquifer Stress Tests
- Problem 6—Groundwater Investigation

SECTION 13 APPENDICES
- Appendix A—Checklist for a Hydrogeological Investigation
- Appendix B—Sampling Protocols
- Appendix C—References
- Appendix D—Sources of Information
- Appendix E—Soil Profiles

ACRONYMS AND ABBREVIATIONS

ACS	American Chemical Society
AGI	American Geological Institute
ARAR	applicable or relevant and appropriate requirement
AST	aboveground storage tank
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
atm	atmosphere
BDAT	best demonstrated available technology
BM	Bureau of Mines
BNA	base/neutral/acid extractables
BOD	biochemical oxygen demand
BTEX	benzene, toluene, ethylbenzene, and xylenes
CAA	Clean Air Act
CDC	Centers for Disease Control
CE	current electrode
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
CERCLIS	CERCLA Information System
CERI	Center for Environmental Research Information
CFR	Code of Federal Regulations
CLP	Contract Laboratory Program
CFA	continuous flight auger
COC	chain of custody
COD	chemical oxygen demand
COE	U.S. Army Corps of Engineers
CWA	Clean Water Act
DO	dissolved oxygen
DOJ	U.S. Department of Justice
DOT	U.S. Department of Transportation
DQO	data quality objectives
DRI	direct-reading instruments
DNAPL	dense, nonaqueous phase liquid
Eh	oxygen-reduction potential
EM	electromagnetic
EMSL-LV	Environmental Monitoring Systems Laboratory - Las Vegas
EP_{tox}	toxicity-extraction procedure toxicity
EPA	U.S. Environmental Protection Agency
EPIC	Environmental Photographic Interpretation Center
ERP	Emergency Response Plan

ERT	EPA Emergency Response Team
ERTS	Earth Resources Technology Satellite
EROS	Earth Resources Observation Systems
ESB	EPA Environmental Services Branch
ESD	Environmental Services Division
eV	electron volt
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FIT	field investigation team
FRP	fiberglass reinforced plastic
FS	feasibility study
FSP	field sampling plan
GAC	granular activated carbon
GC	gas chromatography
GC/MS	gas chromatography/mass spectrometry
gpm	gallons per minute
GPR	ground-penetrating radar
GWA	Ground Water Act of 1987
HASP	health and safety plan (see also site safety plan)
HAZMAT	hazardous materials team
HRS	hazard ranking system
HSL	hazardous substance list (previous term for target compound list)
HSA	hollow-stem auger
HSO	health and safety officer (see also SSC)
HSWA	Hazardous and Solid Waste Amendments (to RCRA, 1984)
HWS	hazardous waste site
ICS	incident command system
IDL	instrument detection limit
IDLH	immediately dangerous to life and health
IP	ionization potential
IR	infrared (spectroscopy)
K	hydraulic conductivity
LEL	lower explosive limit
LNAPL	light, nonaqueous phase liquid
LUST	leaking underground storage tank
MCL	maximum contaminant level
MCLG	maximum contaminant level goal
MDL	method detection limit
MSL	mean sea level
MHz	megahertz
MS	mass spectrometry

MS/MS	mass spectrometry/mass spectrometry
NCIC	National Cartographic Information Center
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NEIC	National Enforcement Investigation Center
NIOSH	National Institute of Occupational Safety and Health
NOAA	National Oceanic and Atmospheric Administration
n.o.s.	not otherwise specified (used in shipping hazardous material)
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
NRC	Nuclear Regulatory Commission
NSF	National Sanitation Foundation
NTIS	National Technical Information Service
NWS	National Weather Service
OERR	EPA Office of Emergency and Remedial Response
OHMTADS	Oil and Hazardous Materials Technical Assistance Data System
OSHA	Occupational Safety and Health Administration
OSWER	EPA Office of Solid Waste and Emergency Response
OVA	organic vapor analyzer (onsite organic vapor monitoring device)
OWPE	EPA Office of Waste Programs Enforcement
PAC	powdered activated carbon
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyls
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PCP	pentachlorophenol
PEL	permissible exposure limit
PID	photoionization detector
POHC	principle organic hazardous constituent
POTWs	publicly owned treatment works
ppb	parts per billion
PPE	personal protective equipment
ppm	parts per million
PRP	potentially responsible party
psig	pounds per square inch gauge
PVC	polyvinyl chloride
QA	quality assurance

QA/QC	quality assurance and quality control
QAMS	quality assurance management staff
QC	quality control
RA	remedial action
RAS	routine analytical services
RCRA	Resource Conservation and Recovery Act of 1978
RI/FS	remedial investigation and feasibility study
ROD	record of decision
RPM	EPA remedial project manager
RQ	reportable quantity
SARA	Superfund Amendments and Reauthorization Act of 1986
SCBA	self-contained breathing apparatus
SCS	Soil Conservation Service
SDL	sample detection limit
SDWA	Safe Drinking Water Act
SI	site inspection
SITE	Superfund Innovative Technology Evaluation
SOP	standard operating procedure
SP	spontaneous potential
SVOC	semivolatile organic compound
SWDA	Solid Waste Disposal Act
TAT	technical assistance team
TCLP	toxicity characteristic leaching procedure
TEGD	Technical Enforcement Guidance Document
TDS	total dissolved solids
TLV	threshold limit value
TOC	total organic carbon
TOX	total organic halides
TSCA	Toxic Substances Control Act
TSDF	treatment, storage, and disposal facility
UEL	upper explosive limit
UMTRCA	Uranium Mill Tailing Radiation Control Act
USCG	United States Coast Guard
USCS	Unified Soil Classification System
USGS	U.S. Geological Survey
UST	underground storage tank
UV	ultraviolet
VOA	volatile organic analysis
VOC	volatile organic compound

GLOSSARY

acre-foot	enough water to cover 1 acre to a depth of 1 foot; equal to 43,560 cubic feet or 325,851 gallons
adsorption	the attraction and adhesion of a layer of ions from an aqueous solution to the solid mineral surfaces with which it is in contact
advection	the process by which solutes are transported by the bulk motion of the flowing groundwater
alluvium	a general term for clay, silt, sand, gravel, or similar unconsolidated material deposited during comparatively recent geologic time by a stream or other body of running water as a sorted or semisorted sediment in the bed of the stream or on its floodplain or delta, or as a cone or fan at the base of a mountain slope
anisotropic	*hydraulic conductivity* ("K"), differing with direction
aquifer	a geologic formation, group of formations, or a part of a formation that contains sufficient permeable material to yield significant quantities of groundwater to wells and springs. Use of the term should be restricted to classifying water bodies in accordance with stratigraphy or rock types. In describing hydraulic characteristics such as transmissivity and storage coefficient, be careful to refer those parameters to the saturated part of the aquifer only.
aquifer test	a test involving the withdrawal of measured quantities of water from, or the addition of water to, a well (or wells) and the measurement of resulting changes in *head* (water level) in the aquifer both during and after the period of discharge or addition
aquitard	a saturated, but poorly permeable bed, formation, or group of formations that does not yield water freely to a well or spring
artesian	confined; under pressure sufficient to raise the water level in a well above the top of the aquifer
artificial recharge	recharge at a rate greater than natural, resulting from deliberate or incidental actions of man
artesian aquifer	see *confined aquifer*
bedload	the part of the total stream load that is moved on or immediately above the stream bed, such as the larger or heavier particles (boulders, pebbles, gravel) transported by traction or saltation along the bottom; the part of the load that is not continuously in suspension or solution

capillary zone	negative pressure zone just above the water table where water is drawn up from saturated zone into soil pores due to cohesion of water molecules and adhesion of these molecules to soil particles. Zone thickness may be several inches to several feet depending on porosity and pore size.
capture	the decrease in water discharge naturally from a *ground-water reservoir* plus any increase in water recharged to the reservoir resulting from pumping
coefficient of storage	the volume of water an aquifer releases from or takes into storage per unit surface area of the aquifer per unit change in head
cone of depression	depression of heads surrounding a well caused by withdrawal of water (larger cone for confined aquifer than for unconfined)
confined	under pressure significantly greater than atmospheric throughout and its upper limit is the bottom of a bed of distinctly lower hydraulic conductivity than that of the material in which the confined water occurs
confined aquifer	geological formation capable of storing and transmitting water in usable quantities overlain by a less permeable or impermeable formation (confining layer) placing the aquifer under pressure
confining bed	a body of "impermeable" material stratigraphically adjacent to one or more aquifers
diffusion	the process whereby particles of liquids, gases, or solids intermingle as a result of their spontaneous movement caused by thermal agitation
discharge area	an area in which subsurface water, including both groundwater and water in the *unsaturated zone*, is discharged to the land surface, to surface water, or to the atmosphere
discharge velocity	an apparent velocity, calculated from Darcy's law, which represents the flow rate at which water would move through the aquifer if it were an open conduit (also called specific discharge)
dispersion	the spreading and mixing of chemical constituents in groundwater caused by diffusion and by mixing due to microscopic variations in velocities within and between pores
drawdown	the vertical distance through which the water level in a well is lowered by pumping from the well or a nearby well
effective porosity	the amount of interconnected pore space through which fluids can pass, expressed as a percent of bulk volume. Part of the total porosity

	will be occupied by static fluid being held to the mineral surface by surface tension, so effective porosity will be less than total porosity.
evapotranspiration	the combined loss of water from direct evaporation and through the use of water by vegetation (transpiration)
flow line	the path that a particle of water follows in its movement through saturated, permeable rocks (synonym: *streamline*)
fluid potential	the mechanical energy per unit mass of water or other fluid at any given point in space and time, with respect to an arbitrary state of datum
gaining stream	a stream or reach of a stream whose flow is being increased by inflow of groundwater (also called an effluent stream)
groundwater	water in the zone of saturation
groundwater divide	a ridge in the *water table* or other *potentiometric surface* from which groundwater moves away in both directions normal to the ridge line
groundwater model	simulated representation of a groundwater system to aid definition of behavior and decision-making
groundwater reservoir	all rocks in the *zone of saturation* (see also *aquifer*)
groundwater system	a groundwater reservoir and its contained water; includes hydraulic and geochemical features
head	combination of elevation above datum and pressure energy imparted to a column of water (velocity energy is ignored because of low velocities of groundwater). Measured in length units (i.e., feet or meters).
heterogeneous/geological formation	characteristics varying aerially or vertically in a given system
homogeneous	geology of the aquifer is consistent; not changing with direction or depth
hydraulic conductivity "K"	volume flow through a unit cross-section area per unit decline in head (measured in velocity units and dependent on formation characteristics and fluid characteristics)

hydraulic gradient — change of head values over a distance

$$\frac{H_1 - H_2}{L}$$

where:

H = head
L = distance between head measurement points

hydrograph — graph that shows some property of groundwater or surface water as a function of time

impermeable — having a texture that does not permit water to move through it perceptibly under the head difference that commonly occurs in nature

infiltration — the flow or movement of water through the land surface into the ground

interface — in hydrology, the contact zone between two different fluids

intrinsic permeability — pertaining to the relative ease with which a porous medium can transmit a liquid under a hydrostatic or potential gradient. It is a property of the porous medium and is independent of the nature of the liquid or the potential field.

isotropic — *hydraulic conductivity ("K")* is the same regardless of direction

laminar flow — low velocity flow with no mixing (i.e., no turbulence)

losing stream — a stream or reach of a stream that is losing water to the subsurface (also called an influent stream)

mining — in reference to groundwater, withdrawals in excess of natural replenishment and capture. Commonly applied to heavily pumped areas in semiarid and arid regions, where opportunity for natural replenishment and capture is small. The term is hydrologic and excludes any connotation of unsatisfactory water-management practice (see, however, *overdraft*).

nonsteady state-nonsteady shape — (also called unsteady state-nonsteady shape) the condition when the rate of flow through the aquifer is changing and water levels are declining. It exists during the early stage of withdrawal when the water level throughout the cone of depression is declining and the shape of the cone is changing at a relatively rapid rate.

nonsteady state-steady shape	is the condition that exists during the intermediate stage of withdrawals when the water level is still declining but the shape of the central part of the cone is essentially constant
optimum yield	the best use of groundwater that can be made under the circumstances; a use dependent not only on hydrologic factors but also on legal, social, and economic factors
overdraft	withdrawals of groundwater at rates perceived to be excessive and, therefore, an unsatisfactory water-management practice (see also mining)
perched	unconfined groundwater separated from an underlying body of groundwater by an unsaturated zone
permeability	the property of the aquifer allowing for transmission of fluid through pores (i.e., connection of pores)
permeameter	a laboratory device used to measure the intrinsic permeability and hydraulic conductivity of a soil or rock sample
piezometer	a nonpumping well, generally of small diameter, that is used to measure the elevation of the water table or potentiometric surface. A piezometer generally has a short well screen through which water can enter.
porosity	the ratio of the volume of the interstices or voids in a rock or soil to the total volume
potentiometric surface	imaginary saturated surface (potential head of confined aquifer); a surface that represents the static head; the levels to which water will rise in tightly cased wells
recharge	the processes of addition of water to the *zone of saturation*
recharge area	an area in which water that is absorbed eventually reaches the *zone of saturation*
safe yield	magnitude of yield that can be relied upon over a long period of time (similar to *sustained yield*)
saturated zone	zone in which all voids are filled with water (the water table is the upper limit)

slug-test — an aquifer test made by either pouring a small instantaneous charge of water into a well or by withdrawing a slug of water from the well (when a slug of water is removed from the well, it is also called a bail-down test)

specific capacity — the rate of discharge from a well divided by the drawdown in it. The rate varies slowly with the duration of pumping, which should be stated when known.

specific yield — ratio of volume of water released under gravity to total volume of saturated rock

steady-state — the condition when the rate of flow is steady and water levels have ceased to decline. It exists in the final stage of withdrawals when neither the water level nor the shape of the cone is changing.

storage — in groundwater hydrology, refers to 1) water naturally detained in a groundwater reservoir, 2) artificial impoundment of water in groundwater reservoirs, and 3) the water so impounded

storage coefficient "S" — volume of water taken into or released from aquifer storage per unit surface area per unit change in head (dimensionless) (for confined, S = 0.0001 to 0.001; for unconfined, S = 0.2 to 0.3)

storativity — the volume of water an aquifer releases from or takes into storage per unit surface area of the aquifer per unit change in head (also called coefficient of storage)

sustained yield — continuous long-term groundwater production without progressive storage depletion (see also *safe yield*)

transmissivity — the rate at which water is transmitted through a unit width of an aquifer under a unit hydraulic gradient

vadose zone — the zone containing water under pressure less than that of the atmosphere, including soil water, intermediate vadose water, and capillary water. Some references include the capillary water in the saturated zone. This zone is limited above by the land surface and below by the surface of the zone of saturation (i.e., the water table). Also called the unsaturated zone or zone of aeration. According to Freeze and Cherry (1979):

1. It occurs above the water table and above the capillary fringe.
2. The soil pores are only partially filled with water; the moisture content θ is less than the porosity n.
3. The fluid pressure p is less than atmospheric; the pressure head ψ is less than zero.
4. The hydraulic head h must be measured with a tensiometer.

5. The hydraulic conductivity K and the moisture content θ are both functions of the pressure head ψ.

water table surface of saturated zone area at atmospheric pressure; that surface in an unconfined water body at which the pressure is atmospheric. Defined by the levels at which water stands in wells that penetrate the water body just far enough to hold standing water.

Introduction to Groundwater Investigations (165.7)

Orientation and Introduction

INTRODUCTION TO GROUNDWATER INVESTIGATIONS

(165.7)

Presented by:

Halliburton NUS Corporation
EPA Contract No. 68-C2-0121

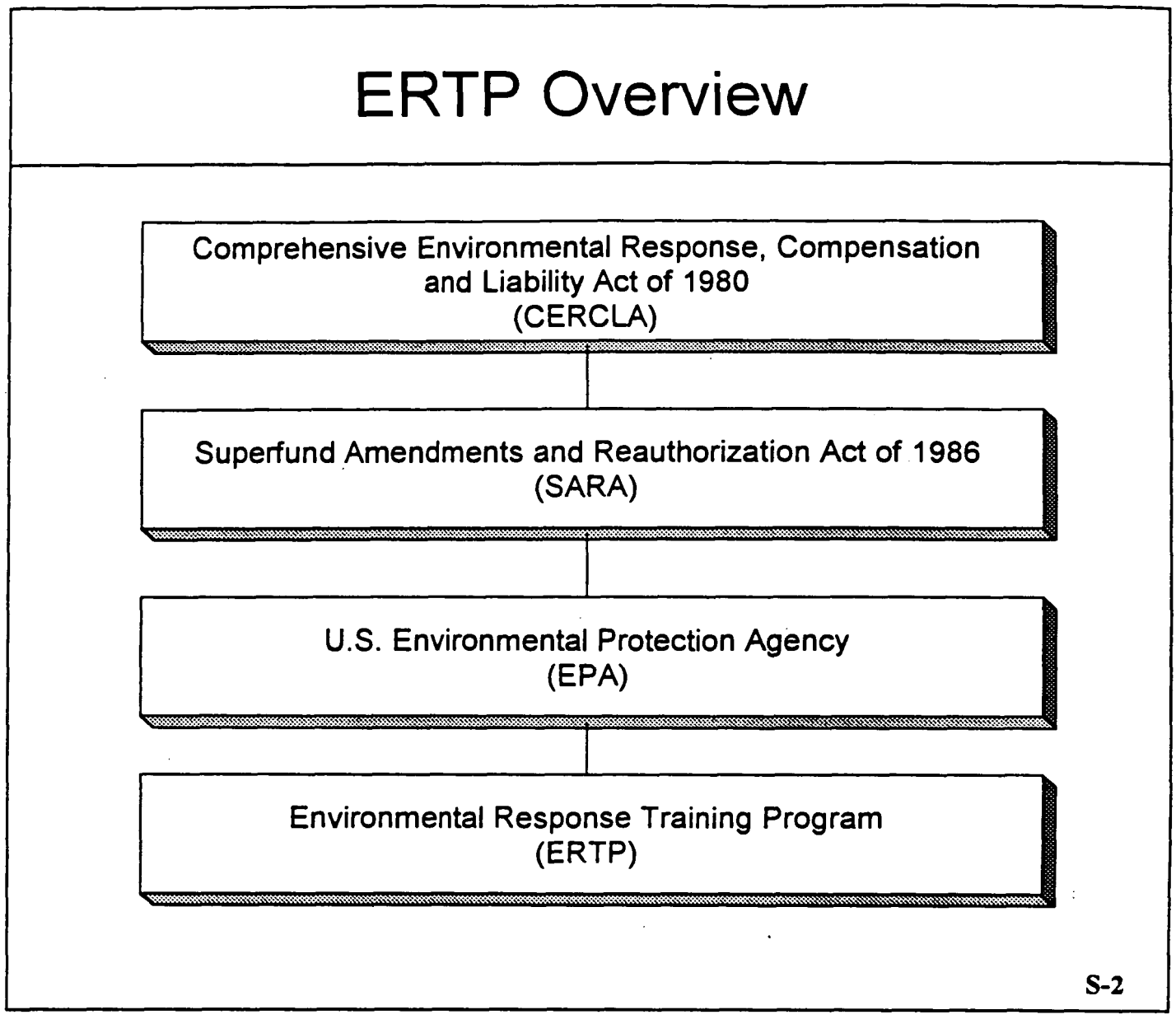
S-1

Orientation and Introduction

Agenda:

- Environmental Response Training Program (ERTP) overview
- Synopsis of ERTP courses
- Course layout and agenda
- Course materials
- Facility information

Notes

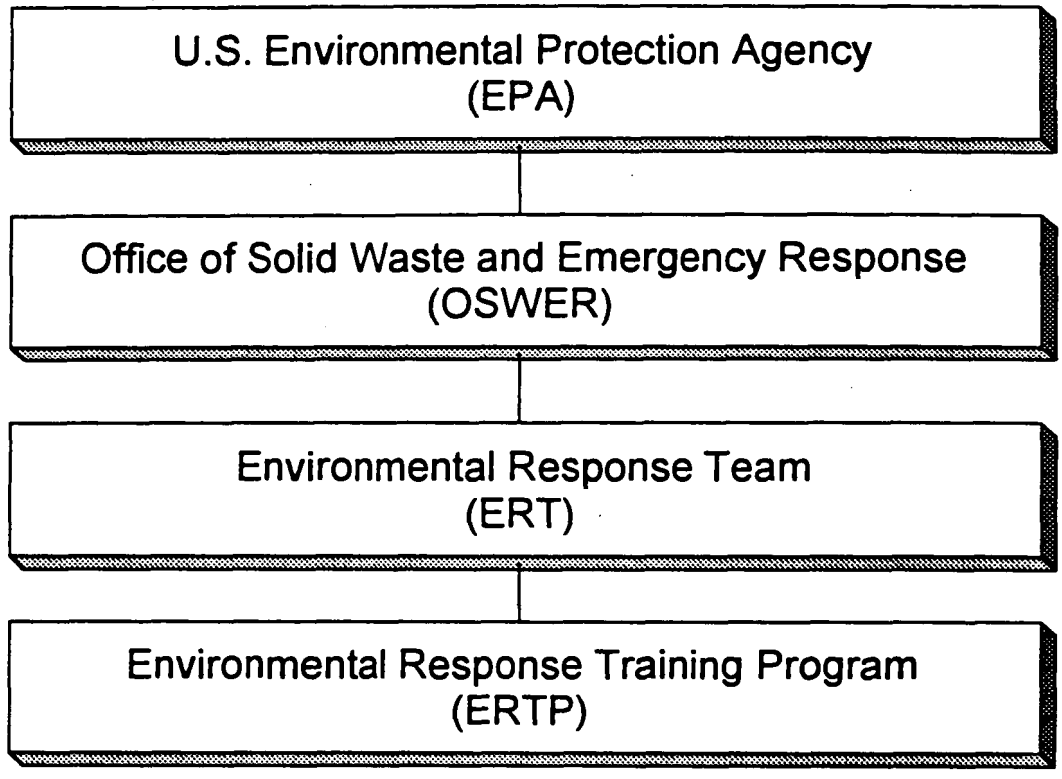


ERTP Overview

In 1980, the U.S. Congress passed the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), also known as Superfund. In 1986, the Superfund Amendments and Reauthorization Act (SARA) was passed. This act reauthorized CERCLA. CERCLA provides for liability, compensation, cleanup, and emergency response for hazardous substances released into the environment and for the cleanup of inactive waste disposal sites. The U.S. Environmental Protection Agency (EPA) allocated a portion of Superfund money to training. EPA's Environmental Response Team (ERT) developed the Environmental Response Training Program (ERTP) in response to the training needs of individuals involved in Superfund activities.

Notes

ERTP Overview



S-3

ERTP Overview

ERTP is administered by ERT, which is part of OSWER. ERT offices and training facilities are located in Cincinnati, Ohio, and Edison, New Jersey. ERT has contracted the development of ERTP courses to Halliburton NUS Corporation (EPA Contract No. 68-C2-0121). The ERTP program provides education and training for environmental employees at the federal, state, and local levels in all regions of the United States. Training courses cover areas such as basic health and safety and more specialized topics such as air sampling and treatment technologies.

Notes

Types of Credit Available

CEU

Continuing Education Units
(1.9 CEUs)

CEU Requirements

- 100% attendance at this course.
- ≥70% on the exam.

Notes

ERTP Courses

S-4

Health and Safety Courses

- *Hazardous Materials Incident Response Operations (165.5)*
- *Safety and Health Decision-Making for Managers (165.8)*
- *Emergency Response to Hazardous Material Incidents (165.15)*

S-5

Technical Courses

Now called ERT → Remediation activities at Sites

TAKE THIS →

- *Treatment Technologies for Superfund (165.3)*
- *Air Monitoring for Hazardous Materials (165.4)*
- *Risk Assessment Guidance for Superfund (165.6)*
- *Introduction to Groundwater Investigations (165.7)*
- *Sampling for Hazardous Materials (165.9)*
- *Radiation Safety at Superfund Sites (165.11)*

S-6

Special Courses

- *Health and Safety Plan Workshop (165.12)*
- *Design of Air Impact Assessments at Hazardous Waste Sites (165.16)*
- *Removal Cost Management System (165.17)*
- *Inland Oil Spills (165.18)*

S-7

Courses Offered in Conjunction with Other EPA Offices

✓ *Chemical Emergency Preparedness and Prevention Office (CEPPO)*
- *Chemical Safety Audits (165.19)*

S-8

✓ *Site Assessment Branch*
- *Preliminary Assessment*
- *Site Investigation*
- *Federal Facilities Preliminary Assessment/Site Investigation*
- *Hazard Ranking System*
- *Hazard Ranking System Documentation Record*

Notes

ALSO:

- GEOPHYSICS COURSE IN CHICAGO
- CHEMISTRY FOR ENV. PROFESSIONALS
- ADVANCED GROUNDWATER → FOR Hydros.

Course Goals

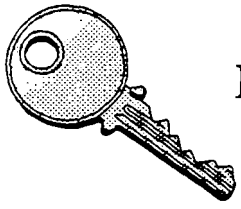
S-9

- Identify the components of a groundwater system.
- List the primary hydrogeological parameters to be considered in a site investigation.
- Construct a flow net and calculate the hydraulic gradient at a site.

S-10

- Discuss the primary advantages and disadvantages of the most common geophysical survey methods.
- Identify geochemical profiles in contaminated groundwater.
- Identify the different types of pumping tests and the information that can be obtained from each.
- Describe monitoring well drilling and sampling techniques.

Notes



Key Points:

- Agenda times are only approximate. Every effort is made to complete units, and finish the day, at the specified time.
- Classes begin promptly at 8:00 am. Please arrive on time to minimize distractions to fellow students.
- Breaks are given between units.
- Lunch is 1 hour.
- Each student must take the examination given on Thursday.
- Direct participation in field or laboratory exercises is optional. Roles are randomly assigned to ensure fairness.
- Attendance at each lecture and exercise is required in order to receive a certificate.

Notes

Training Evaluation

- The *Training Evaluation* is a tool to collect valuable feedback from YOU about this course.

- We value YOUR comments!! Important modifications have been made to this course based on comments of previous students.

DO

- Write in your comments at the end of each unit!

- Tell us if you feel the content of the course manual is clear and complete!

- Tell us if you feel the activities and exercises were useful and helpful!

- Tell us if you feel the course will help you perform related duties back on the job!

- Complete the first page at the end of the course before you leave!

- Write comments in ink.

DON'T

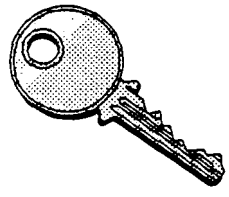
- Hold back!

- Focus exclusively on the presentation skills of the instructors.

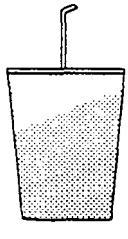
- Write your name on the evaluation, if it will inhibit you from being direct and honest.

Notes

Facility Information



Please put beepers in the vibrate mode and turn off radios. Be courteous to fellow students and minimize distractions.



Emergency Telephone Numbers

Emergency Exits

Alarms

Sirens

Notes

Section 2

ROCK CYCLE

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Define the Doctrine of Uniformitarianism

2. Describe the three basic rock types and their textures within the rock cycle

3. Identify the media responsible for the erosion and transport of sediments

4. Describe the process of lithification and cementation as related to sedimentary rocks

5. Describe how sedimentary particles become rounded, sorted, and stratified.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

ROCK CYCLE

S-1

Doctrine of Uniformitarianism

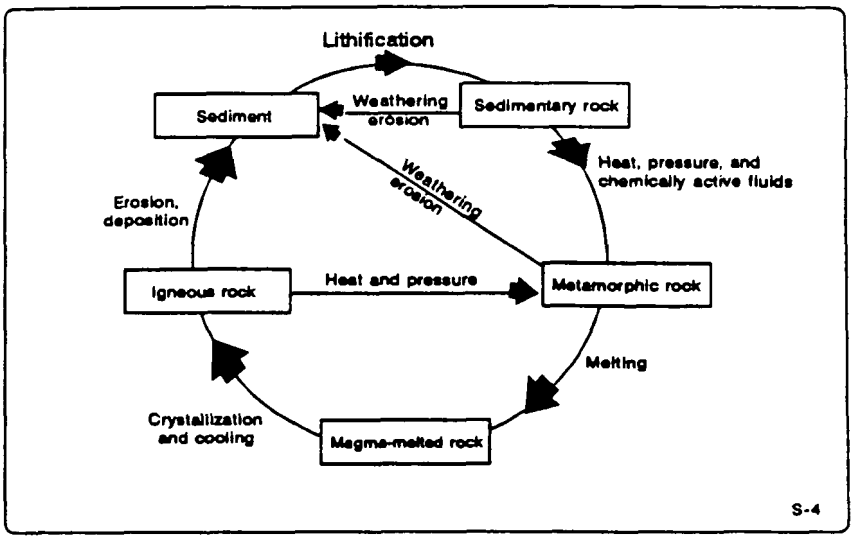
S-2

"The Present is the Key to the Past"

James Hutton, 1785

S-3

NOTES



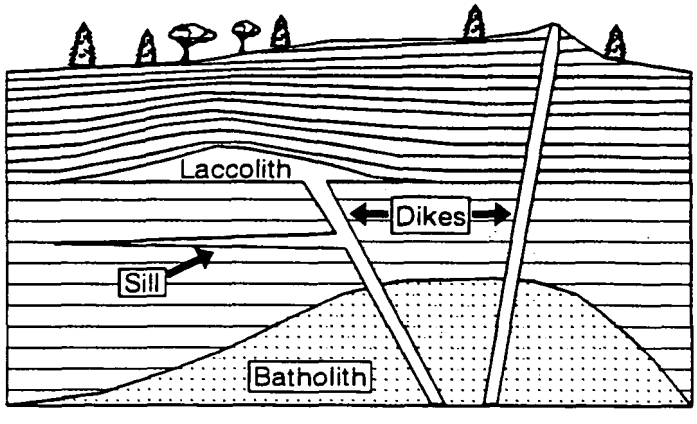
IGNEOUS ROCKS

- Solidified from molten liquid (magma)
- Volcanic rocks/extrusive rocks
  - Obsidian, lava, pumice, tuff
- Plutonic rocks/intrusive rocks
  - Batholiths, sills, laccoliths

S-5

IGNEOUS = FIRE FORMED

INTRUSIVE IGNEOUS ROCK BODIES



S-6

IGNEOUS ROCKS
Texture

- Rocks are composed of interlocking mineral grains
- Minerals form in a liquid or magma
- Size of minerals based on cooling rate of liquid

S-7

IGNEOUS ROCKS
Texture

- Intrusive: coarse-grained rock
- Visible minerals form in slow cooling liquid
- Examples: granite and gabbro
- Found in batholiths, laccoliths, and sills

S-8

IGNEOUS ROCKS
Texture

- Extrusive rocks: fine-grained or glassy rocks
- Small minerals form in fast cooling liquid
- Lava flows, volcanoes
- Examples: basalt and rhyolite

S-9

NOTES

COARSE GRAINED or FINE GRAINED } 2 TYPES OF TEXTURES

FINE GRAINED - cooled quickly

COARSE GRAINED - cooled slowly

IGNEOUS + Metamorphic rocks don't usually have pore spaces for contaminants to "hideout"

NOTES

marble, schist ⇒ contaminants don't have placed to hide.

Composed of same minerals, but different textures

IGNEOUS ROCKS
Equivalent Chemistry

Coarse-grained Texture		Fine-grained Texture
Gabbro	=	Basalt
Granite	=	Rhyolite

S-10

METAMORPHIC ROCKS
"Changed Form" Rocks

- Heat
- Pressure
- Chemically active fluids
- Recrystallization

S-11

foliated = "leaf like"

METAMORPHIC TEXTURES

- Interlocking crystals; marble
- Layers of platy minerals; schist (foliation)

S-12

NOTES

METAMORPHIC ROCKS
Metamorphism Process

Original Rock	Metamorphic Rock
Limestone	Marble
Sandstone	Quartzite
Basalt	Amphibolite
Siltstone/shale	Slate
	Phyllite
	Schist
Granite	Gneiss

S-13

heat
pressure
chemically active fluids

metamorphise something

SEDIMENTARY ROCKS
"Most Abundant Surficial Rock Type"

- Derived from preexisting rocks
- Composed of individual grains cemented together or chemically precipitated
- Most form in water environment
- Make up most rock aquifers

S-14

most often dealt w/ @ Sites

TYPICAL SEDIMENTARY ROCKS

Limestone	Dolomite
Shale	Siltstone
Sandstone	Conglomerate
Coal	Evaporites

S-15

gypsum
halite

Coal does not contain minerals but forms in sed. env.

NOTES

Contamination will most likely follow bedding planes

Geology controls fate + transport.

RECIPE FOR SEDIMENTARY ROCKS

- Erosion processes
- Deposition
- Lithification

S-16

SEDIMENTARY ROCKS Erosion Process

- Wind
- Water
- Ice
- Gravity
- Biology

S-17

SEDIMENTARY ROCKS Deposition

- Wind
- Water
- Ice
- Gravity

S-18

NOTES

SEDIMENTARY ROCKS
Lithification

- "Making into stone"
- Cementation: natural cements dissolved in and transported by groundwater

S-19

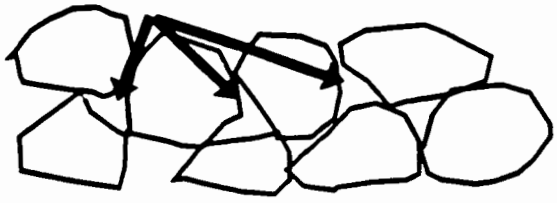
SEDIMENTARY ROCKS
Types of Cement

- Silica (types of quartz)
- Iron oxides (hematite/limonite)
- Clay mineral groups
  - Kaolinite, vermiculite, montmorillonite, illite
- Carbonates (calcite/aragonite)

S-20

SEDIMENTARY ROCKS

- Composed of particles of any rock type
  - "Pores" form during deposition



- Most aquifers are sedimentary rocks

S-21

clay → high porosity
low permeability

clay used for landfill liners in old days
→ but when organics come in contact w/ clay, it cracks and allows permeability, etc.

NOTES

Porosity = void space
holes

PRIMARY POROSITY

A measure of the total void space within a rock at the time it was formed

It is generally higher in sedimentary rocks and lower in igneous and metamorphic rocks

S-22

SECONDARY POROSITY

Void spaces that form after the rock has been formed (e.g., faults, joints, fractures, and conduits)

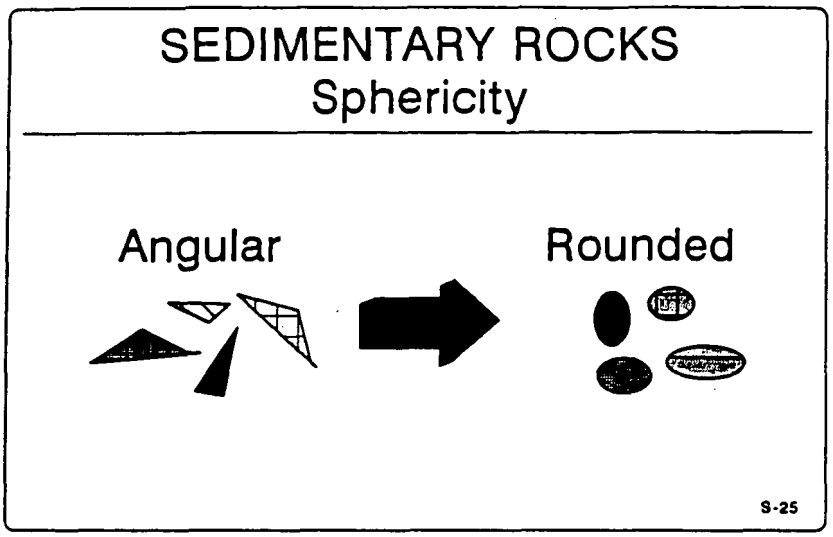
S-23

PERMEABILITY

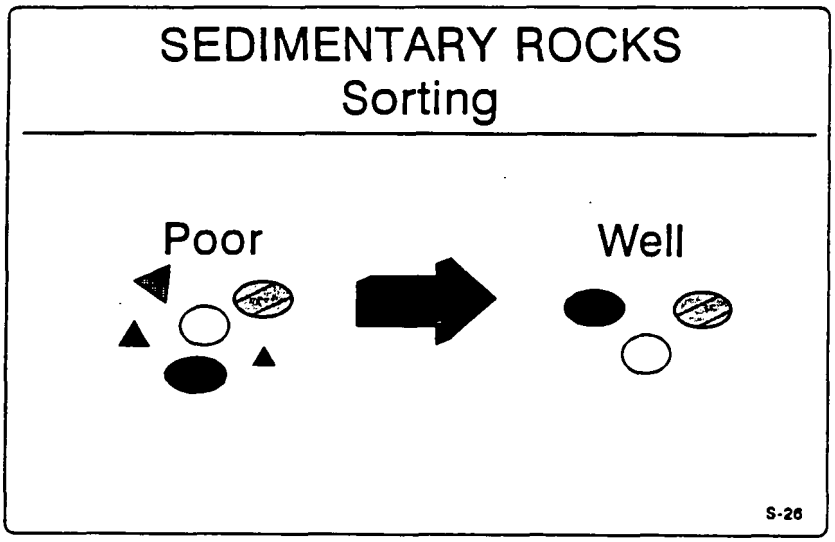
The ease with which liquid will move through a porous medium

S-24

NOTES



originally angular when break off → then become rounded gradually



Section 3

DEPOSITIONAL ENVIRONMENTS

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Describe the following depositional environments:

    a. Alluvial fans

    b. Braided streams

    c. Meandering streams

    d. Coastal (deltaic and barrier island complexes)

    e. Wind-blown deposits

    f. Carbonates

    g. Evaporites

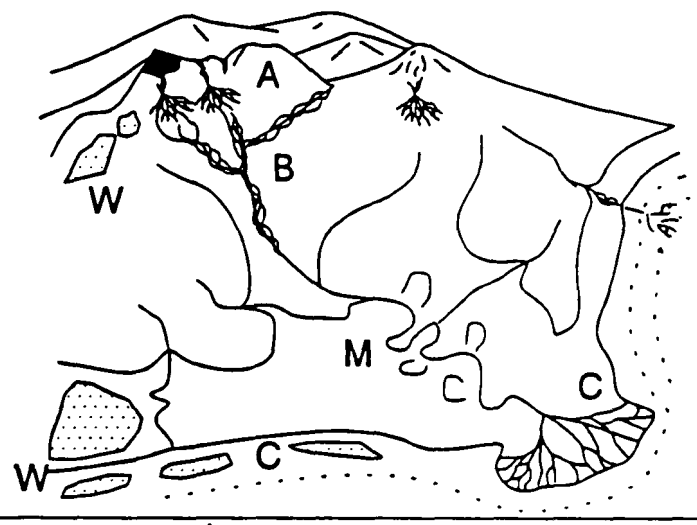
    h. Glaciers (continental and alpine).

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

DEPOSITIONAL ENVIRONMENTS

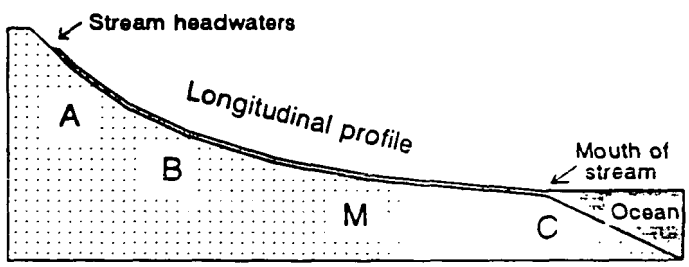
S-1



S-2

LONGITUDINAL PROFILE

A Alluvial and landslide
B Braided stream
M Meandering stream
C Coastal



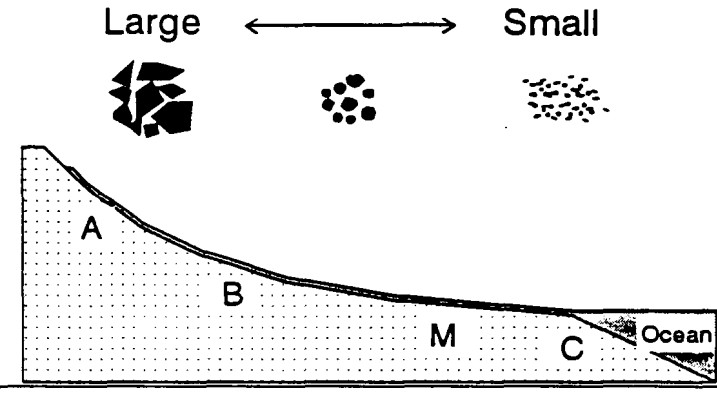
S-3

NOTES

ROCK TYPE	ENVIRONMENT
Conglomerate	Landslide, alluvial fan
Sandstone	Rivers, streams, beaches, deltas, dunes, sand bars
Clay/shale	Lagoon, lake, flood plain, deeper ocean
Limestone	Coral reef, atoll, deeper ocean

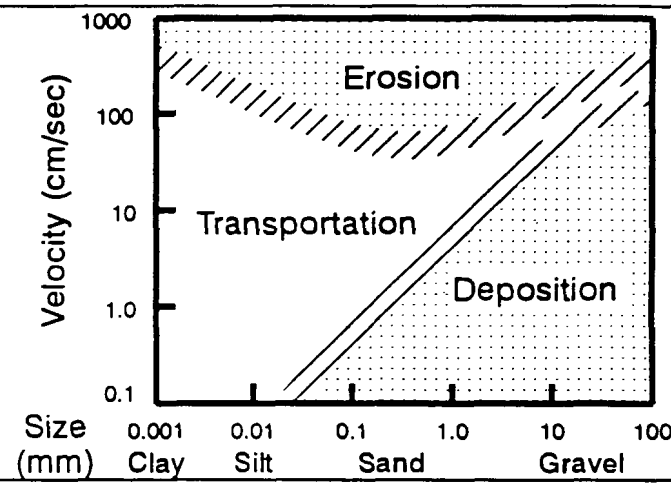
S-4

MEDIAN CHANNEL
Grain Size



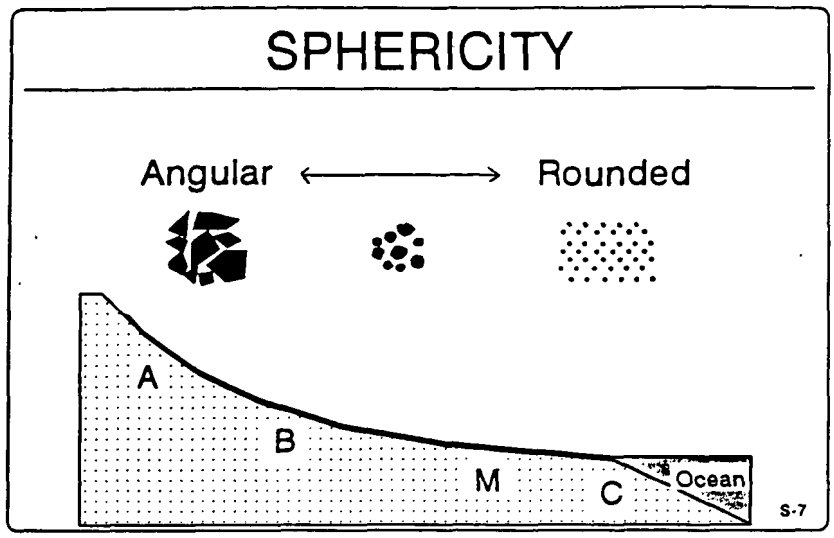
S-5

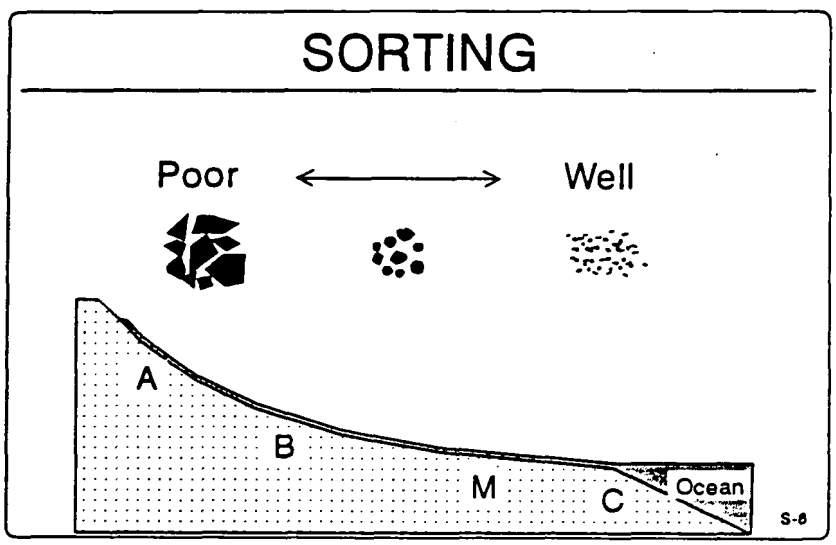
RELATIONSHIP OF STREAM VELOCITY

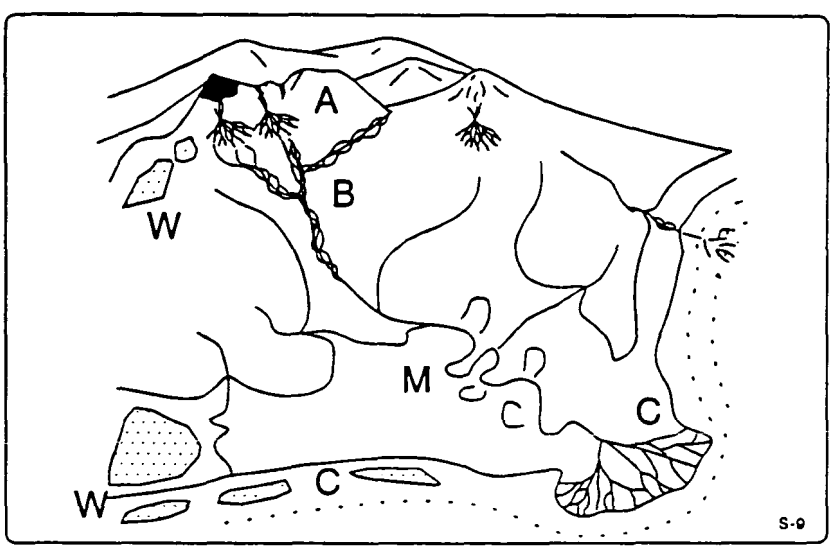


S-6

NOTES


SPHERICITY
Angular
Rounded
A
B
M
C
Ocean
S-7


SORTING
Poor
Well
A
B
M
C
Ocean
S-8


A
B
W
M
C
C
W
C
S-9

NOTES

DEPOSITIONAL ENVIRONMENTS

- Alluvial fan
- Braided stream
- Meandering stream
- Coastal deposits

S-10

DEPOSITIONAL ENVIRONMENTS (cont.)

- Wind-blown deposits
- Carbonates (Karst)
- Evaporites
- Glacial deposits

S-11

Alluvial Fan

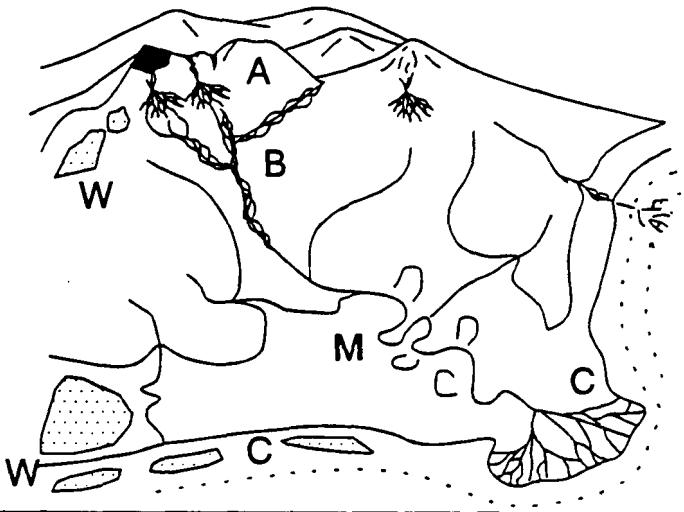
S-12

CHARACTERISTICS OF ALLUVIAL FANS

Depositional environments:

- Poor sorting and rounding
- High gradients
- Shallow and intermittent streams
- Hand-shaped

S-13



S-14

Braided Stream

S-15

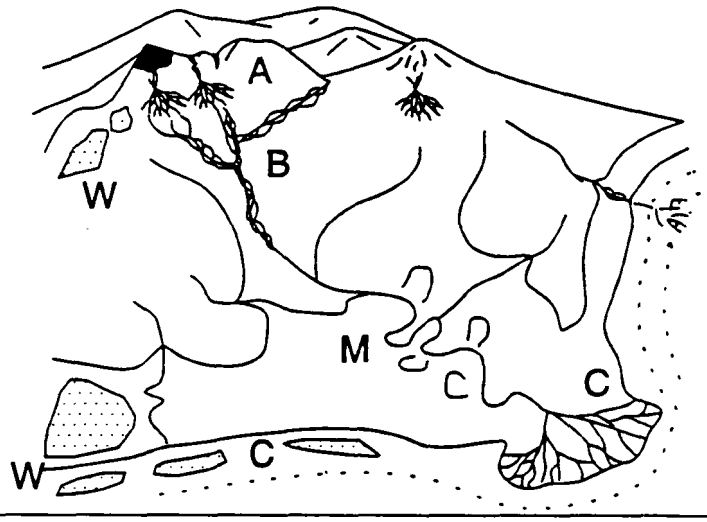
NOTES

CHARACTERISTICS OF BRAIDED STREAMS

Depositional environments:

- Resembles braided hair
- High to low gradients
- Shallow streams
- Poor to medium sorting
- Angular to subangular grains

S-16



S-17

Meandering Stream

S-18

CHARACTERISTICS OF MEANDERING STREAMS

Depositional environments:

- Low gradients
- Deep streams
- Grain size variations

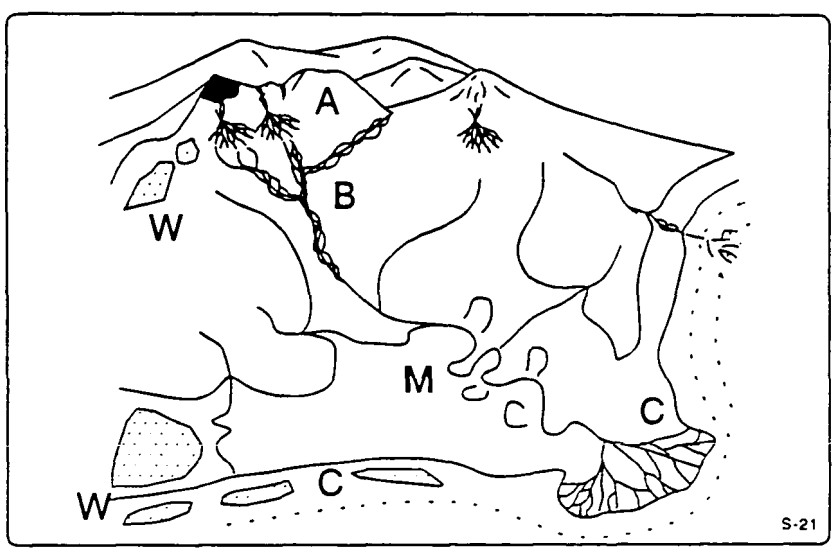
S-19

CHARACTERISTICS OF MEANDERING STREAMS (cont.)

Depositional environments:

- Oxbow lakes
- Levees and floodplains
- Point bars and cut banks

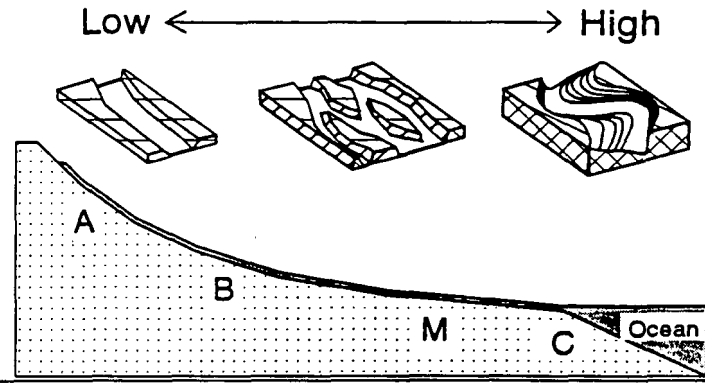
S-20



S-21

NOTES

STREAM CHANNEL
Sinuosity



S-22

Coastal Deposits

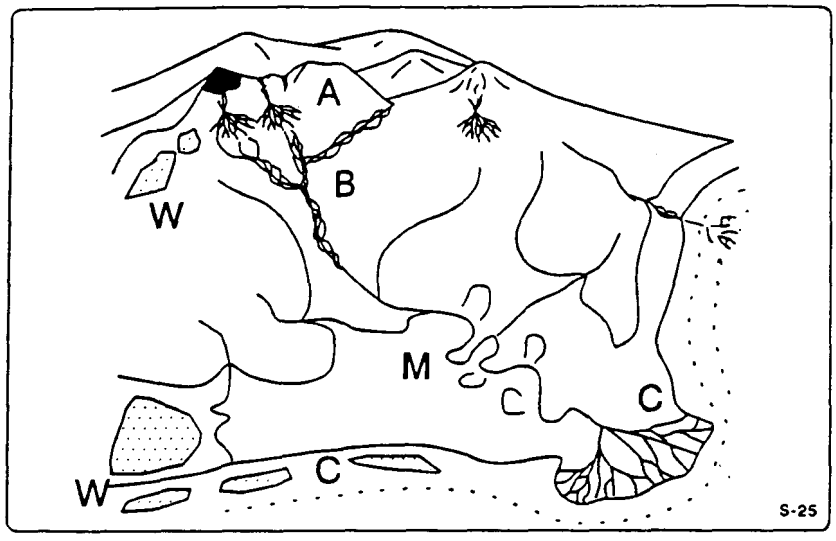
S-23

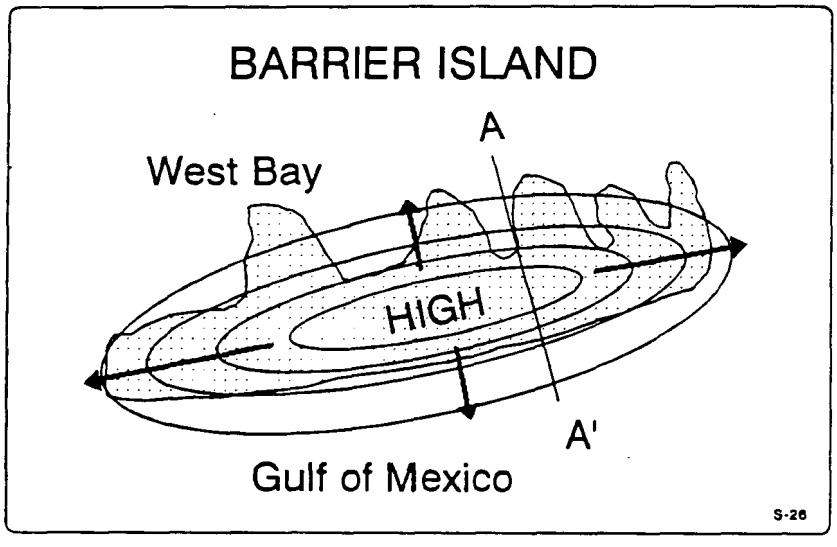
TYPICAL COASTAL DEPOSITS
Depositional Environments

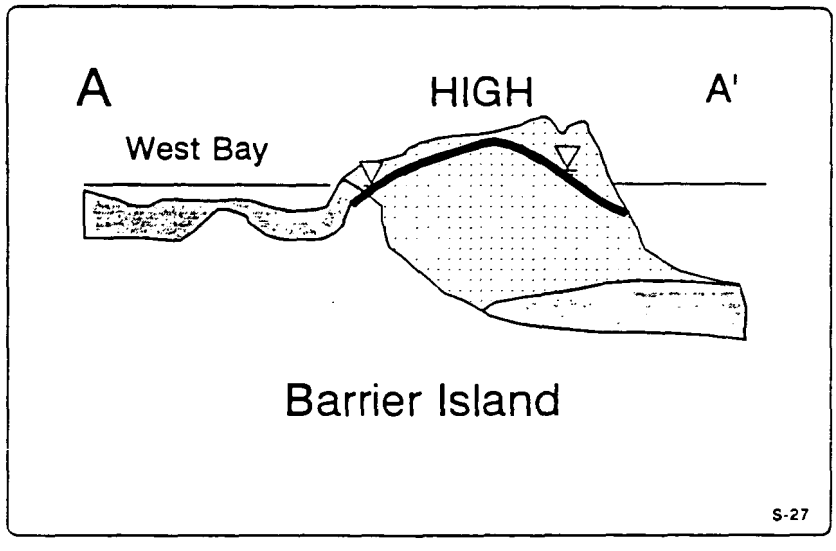
- Barrier islands
- Offshore bars
- Deltas
- Spits
- Tidal flats
- Reefs/cays

S-24

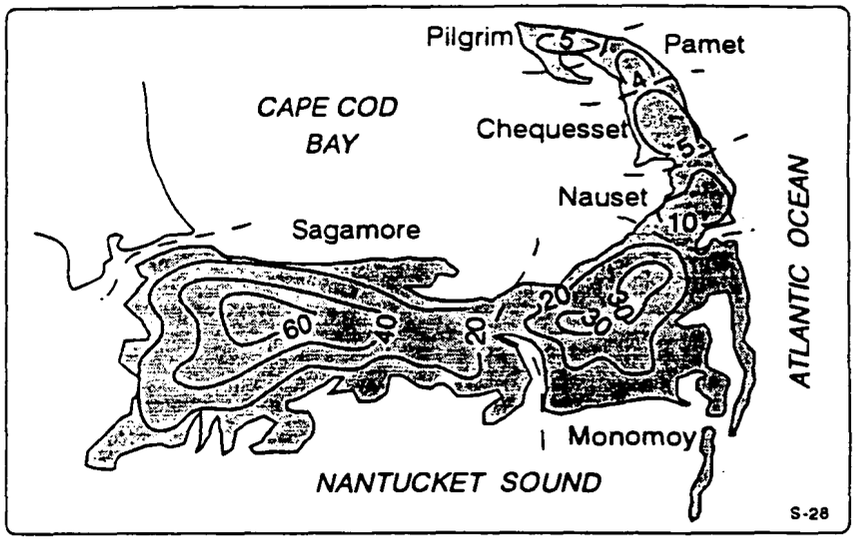
NOTES

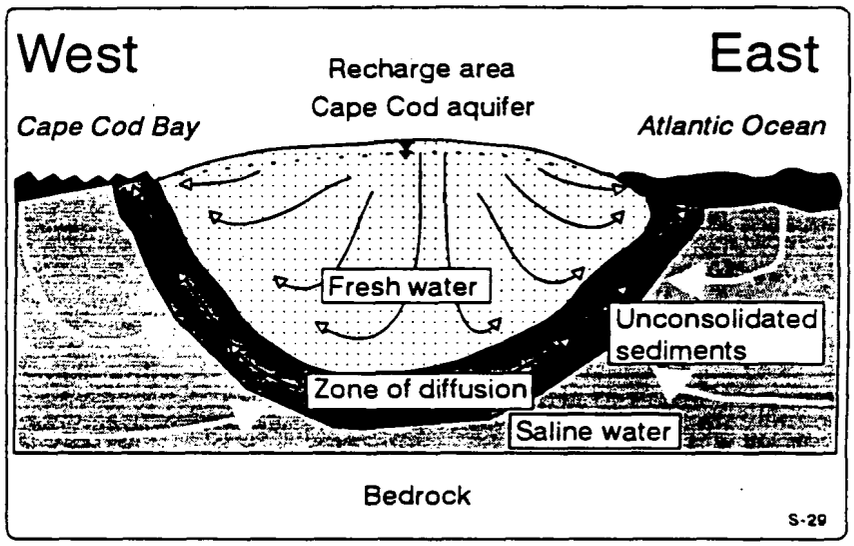

A
B
W
M
C
C
W
C
S-25

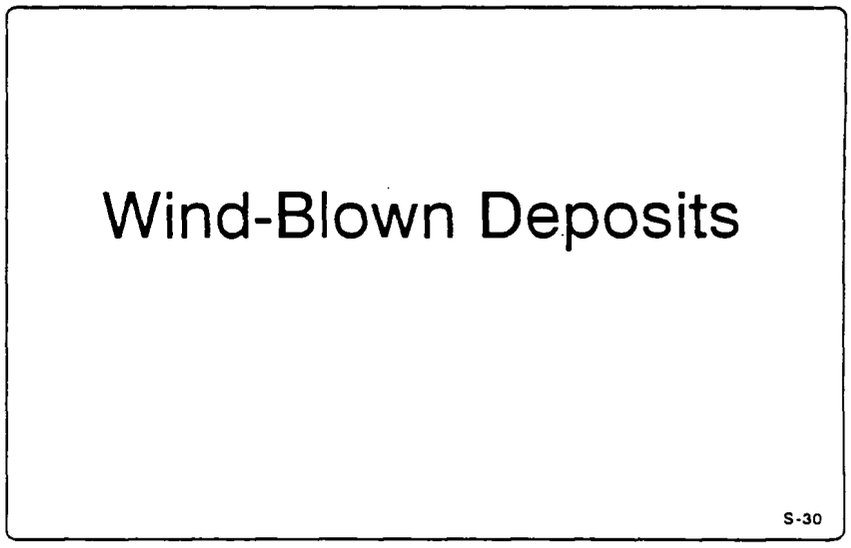

BARRIER ISLAND
A
West Bay
HIGH
A'
Gulf of Mexico
S-26


A
HIGH
A'
West Bay
Barrier Island
S-27

NOTES


Pilgrim
Pamet
CAPE COD
BAY
Chequesset
Nauset
Sagamore
ATLANTIC OCEAN
Monomoy
NANTUCKET SOUND
S-28

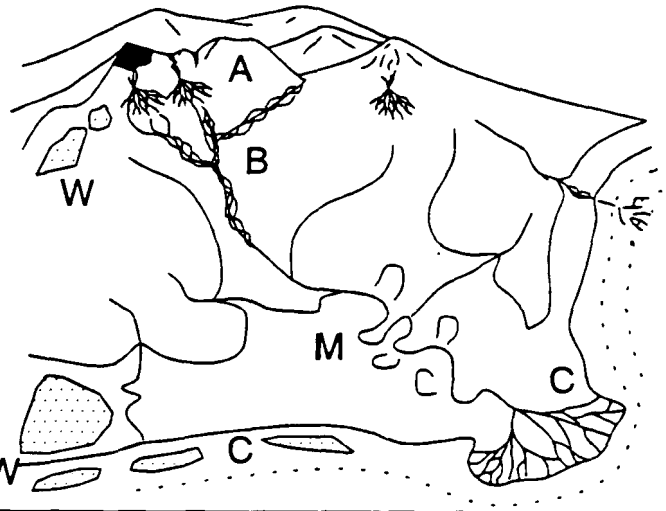

West
Recharge area
Cape Cod aquifer
East
Cape Cod Bay
Atlantic Ocean
Fresh water
Unconsolidated sediments
Zone of diffusion
Saline water
Bedrock
S-29


Wind-Blown Deposits
S-30

WIND-BLOWN DEPOSITS
Depositional Environments

- Dunes: continental and coastal
- Volcanic dust and ash
- Glacial till dust (loess)

S-31



S-32

Carbonate Rocks

S-33

NOTES

Calcium
Magnesium } hard water

CARBONATES

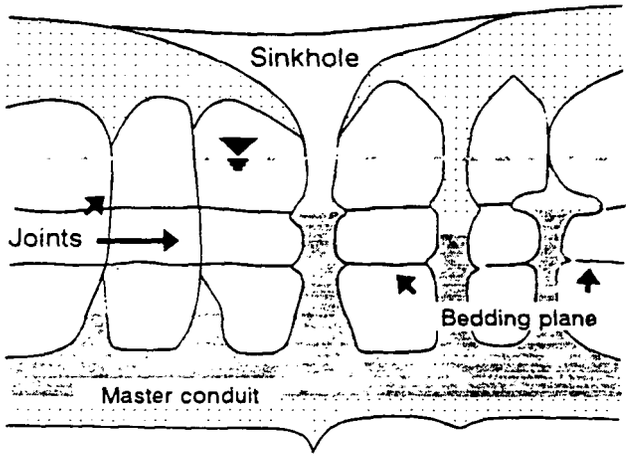
- Limestones
- Dolomites

S-34

KARST TOPOGRAPHY
Depositional Environment

- Soluble rocks at or beneath surface (carbonates, sulfates, chlorides)
- Chemical solution of soluble rocks
- Closed depressions (sink holes, swallets)
- Little or no surface drainage
- Caves, springs, disappearing streams

S-35



S-36

Evaporites

S-37

EVAPORITES

- Carbonates
- Sulfates
- Chlorides

S-38

Glaciation

S-39

NOTES

PROCESSES OF GLACIATION

- Erosion
- Transportation
- Deposition

S-40

GLACIERS/FREEZE-THAW

- Weathering and transport
- Large-scale changes
- Poor to excellent sorting (e.g., glacial till and outwash)

S-41

GLACIAL DEPOSITS Depositional Environments

- Outwash and till
- Moraines
- Drumlins
- Eskers
- Kettle holes
- Kames

S-42

SOILS

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Discuss the factors that influence soil formation processes

2. Differentiate between physical and chemical weathering

3. Describe the factors that influence soil morphology

4. Define the following physical and chemical properties of soil:
   a. Porosity
   b. Permeability
   c. Cation exchange capacity
   d. Bulk density
   e. Capillarity

5. Describe a common soil profile and the interaction of its component units.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

SOILS

S-1

WHY STUDY SOILS?

- First media encountered by spills and leaks
- Contaminant fate and transport
  - Interaction with air, water, microbes, and soil
  - Soil variability

S-2

ADSORB - onto surface
ABSORB - into material

CONTAMINANT FATE IN SOIL

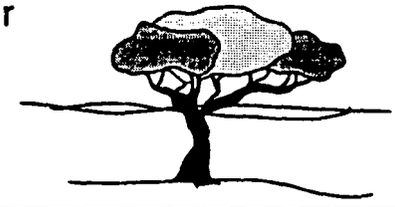
- How much:
  - Adsorbed to clay?
  - Adsorbed to organic matter?
  - Volatilized?
  - Consumed by microbes?
  - Entered water table?

S-3

NOTES

SOIL
Definition

- Material that supports the growth of plants
- Consists of :
  - Rock and mineral fragments
  - Organic matter
  - Water
  - Air



S-4

SOIL FORMATION
Controls

- Parent rock or sediment
- Climate
- Topography
- Presence/abundance of organisms
- Time

S-5

SOIL FORMATION
Soil from Bedrock

- Forms in place
- Derived from underlying bedrock
- Retains original bedrock structure
- Example: saprolitic soil

Soils that form in place

S-6

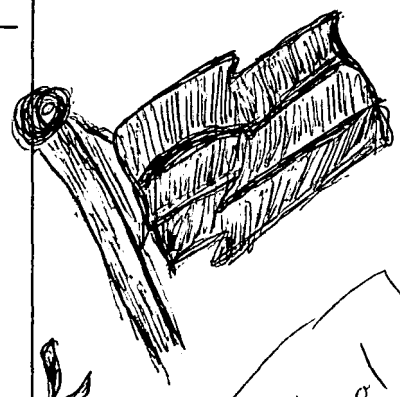
NOTES

SOIL FORMATION
Soil from Sediment

- River deposits
- Till deposits
- Outwash deposits
- Loess deposits

wind blown glacial material

S-7



SOIL FORMATION
Physical Weathering

- Breaks "big rocks" into "small rocks"
- Increases weathered surface
- Influenced by climate and topography
- Time

S-8

Porosity - stored material
permeability - material moved

SOIL FORMATION
Chemical Weathering

- Influenced by water and dissolved gases
- Acidic water
- Minerals are either gained or lost

water can change chemical make-up of materials(?)

S-9

NOTES

changes hard water to softer water since removing ca for sodium

SOIL FORMATION
Chemical Reactions in Soil

- Hydration/dehydration
- Oxidation/reduction (Eh potential)
- pH
- Ion exchange (calcium for sodium)
- Chelation (soil colloids)

S-10

SOIL FORMATION
Humid Climate

- High temperature
  - Rapid development of soil profile
  - Rapid oxidation and breakdown of organics
- Cold temperature
  - Slow oxidation
  - High accumulation of organic materials

S-11

SOIL FORMATION
Arid Climate

- High temperature
  - No organic horizon
  - Slow soil profile development
  - Rapid oxidation
- Low temperature
  - No organic horizon
  - Sterile soil
  - Slow oxidation

S-12

NOTES

SOIL MORPHOLOGY

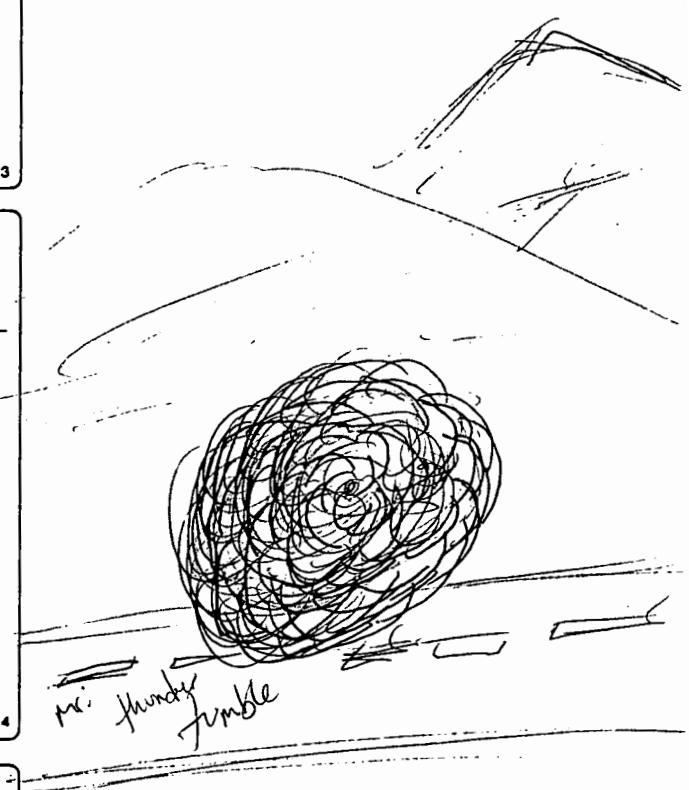
- Color
- Texture
- Structure
- Consistency
- Horizon boundaries

S-13

SOIL MORPHOLOGY
Color

- Moisture content of soil
- Parent rock type
- Abundance of organic matter
- Degree of oxidation/reduction

S-14



SOIL MORPHOLOGY
Examples of Soil Color

- Black-brown: organic material, Mn-minerals
- Reddish: iron oxides, oxidized
- Yellow-brown: iron oxides, poorly drained
- White: Ca-carbonates, silica/bauxite/clay

S-15

NOTES

SOIL MORPHOLOGY
Examples of Soil Color (cont.)

- Greenish or bluish gray: wetlands, gleyed soil
- Mottled soil: moving water table, oxidized and reduced

S-16

SOIL MORPHOLOGY
Soil Texture

- Percentage of sand, silt, and clay
- Water holding capacity
- Soil classification systems

S-17

DETERMINATION OF GRAIN SIZES

Particle Type	Particle Size (mm)	Familiar Example
Boulder	>305	Basketball
Cobble	76.0 – 305	Grapefruit
Gravel	4.76 – 76.0	Pea to orange
Sand	0.074 – 4.76	Rock salt to sugar
Silts and clay	<0.074	Talcum powder

S-18

NOTES

SOIL MORPHOLOGY
Structure

- Grade
- Shape
- Size

S-19

STRUCTURE
Grade

- Structureless
- Weak
- Moderate
- Strong

S-20

STRUCTURE
Shape

- Platy
- Prismatic
- Blocky
- Granular

S-21

NOTES

SOIL MORPHOLOGY
Consistency

- Cementation in soil
- Plasticity
- Strength
- Stickiness

S-22

SOIL PROPERTIES

- Infiltration
- Permeability
- Runoff
- Available water capacity
- pH / Eh

S-23

SOIL PROPERTIES (cont.)

- Cation exchange capacity
- Base saturation
- Mineralogy
- Bulk density

S-24

NOTES

SOIL PROPERTIES
Permeability

- Ability to transmit water and contaminants
- Depends on linkage of pore spaces

S-25

SOIL PROPERTIES
Cation Exchange Capacity

- Negative charge on soil particles
- High in clayey soils
- Low in sandy soils

S-26

SOIL PROPERTIES
Bulk Density

- Ratio of the mass to total volume of soil (g/cm^3)
- Volume includes air, liquid, and solid phases
- Particle density, solid phase only

S-27

NOTES

SOIL PROPERTIES
Bulk Density Examples

- Sandy soil 2.0 g/cm^3
- Silty soil 1.9 g/cm^3
- Clayey soil 2.2 g/cm^3

S-28

SOIL PROPERTIES
Porosity

- Ratio of open space to total volume
- Ability to hold or store water
- High in sedimentary rocks
- Low in crystalline rocks

S-29

SOIL PROPERTIES
Porosity Values (High)

- Styrofoam > 90 %
- Gravel 40 %
- Clay 70 %
- Shale < 20 %
- Limestone (karst) 50 %
- Fractured rock 50 %

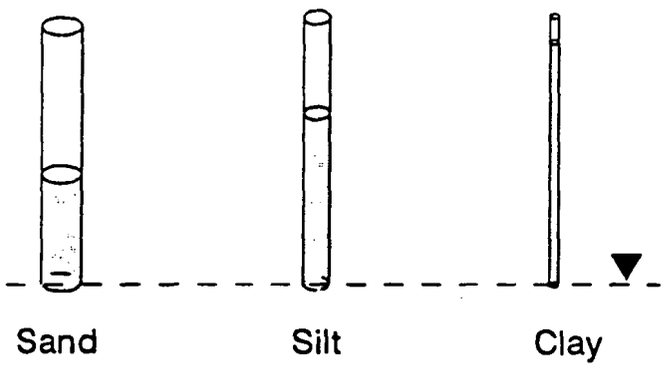
S-30

SOIL PROPERTIES
Capillarity

- Capillary fringe
- Height water rises above the water table
- Depends on size of the pores
- Less water content than saturated zone
- Does not yield water

S-31

CAPILLARY FRINGE



S-32

SOIL TYPE VARIABILITY

- Moisture content
- Organic content
- Thickness
- Mineral composition
- Microbe population
- pH and Eh

S-33

NOTES

SOIL PROFILE

Vertical succession of various soil layers to bedrock

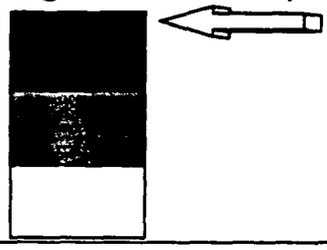
- O-horizon
- A-horizon
- B-horizon
- C-horizon

Most soils have formed during the last 2 million years

S-34

O-HORIZON
Characteristics

- Mainly organic matter (>20%) mixed with rock and mineral fragments
- Contains decaying animal and plant matter (humus)

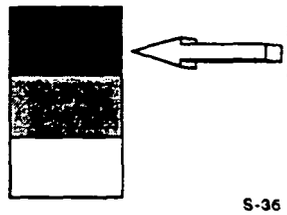


S-35

Not found in Most Cases
Best found in marsh or forest

A-HORIZON
Characteristics

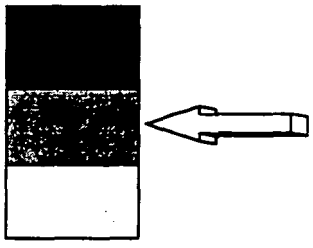
- Rock/mineral fragments mixed with organic matter
- Commonly known as "topsoil"
- Zone of leaching
- Contains large-sized pores



S-36

B-HORIZON
Soil Profile

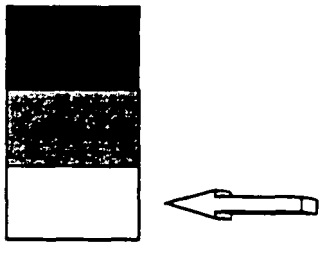
- Zone of accumulation (illuviation)
- Insoluble minerals leached from A-horizon



S-37

C-HORIZON
Soil Profile

- Partially decomposed bedrock
- Grades into unweathered bedrock



S-38

DRILLING METHODS

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Describe the following drilling methods:
   a. Cable tool
   b. Hollow-stem auger
   c. Mud rotary
   d. Air rotary
   e. Rotasonic

2. List the advantages and disadvantages of the following drilling methods:
   a. Cable tool
   b. Hollow-stem auger
   c. Mud rotary
   d. Air rotary
   e. Rotasonic

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

DRILLING METHODS

S-1

USES FOR WELLS

- Water supply
- Monitoring
- Remediation
- Lithology
- "Ground truthing"
- Hydraulic properties

S-2

SELECTION CRITERIA

- Hydrogeologic environment
  - Type of formation
  - Depth of drilling
- Type of pollutant
- Location
- Availability
- Cost

S-3

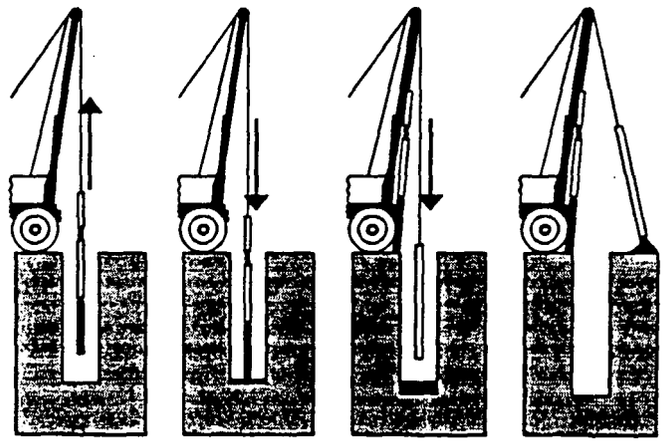
NOTES

DRILLING METHODS

- Cable tool
- Hollow-stem auger
- Mud rotary
- Air rotary
- Rotasonic

S-4

CABLE TOOL DRILLING METHOD



S-5

CABLE TOOL
Advantages

- Good sample recovery
- Good delineation of water-bearing zones during drilling
- Highly mobile
- Good drilling in most formations
- Inexpensive

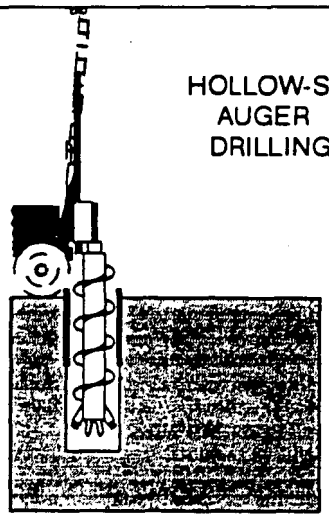
S-6

NOTES

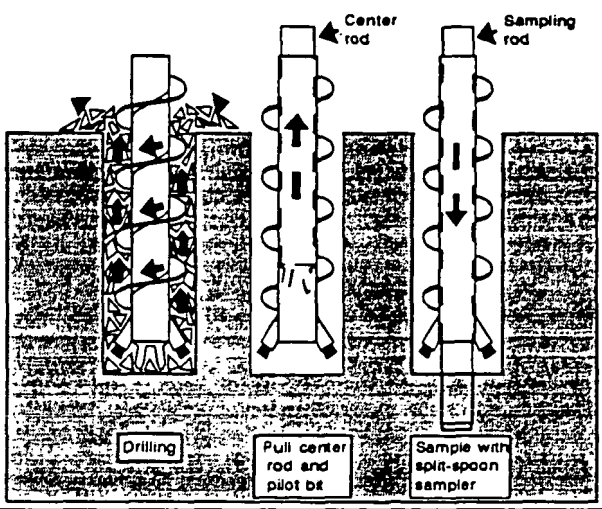
CABLE TOOL
Disadvantages

- Slow
- Requires driving casing in unconsolidated formations

S-7



S-8



S-9

NOTES

HOLLOW-STEM AUGER
Advantages

- Highly mobile
- No drilling fluid required
- Problems of hole caving minimized
- Soil sampling relatively easy

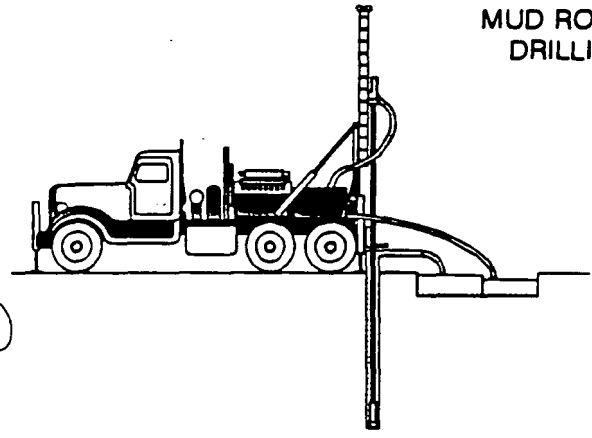
S-10

HOLLOW-STEM AUGER
Disadvantages

- Cannot be used in consolidated formations
- Limited depth capability (~150 feet)
- Cross contamination of permeable zones is possible
- Limited casing diameter

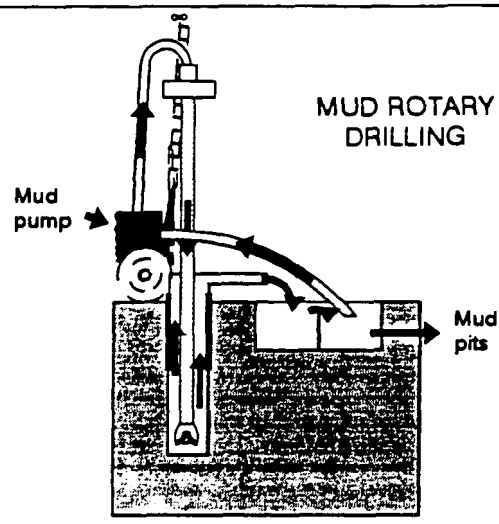
S-11

MUD ROTARY DRILLING



S-12

used in oil drilling more than in env. drilling



S-13

MUD ROTARY
Advantages

- Availability
- Satisfactory drilling in most formations
- Good depth capability

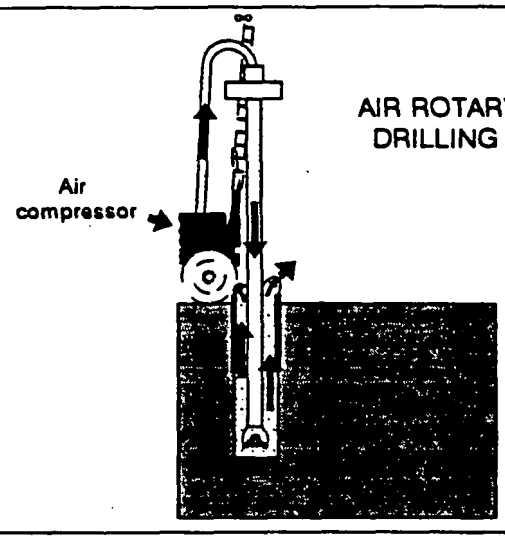
S-14

MUD ROTARY
Disadvantages

- Requires drilling fluid
  - Difficult to remove
  - May affect sample integrity
- Circulates contaminants
- Mobility may be limited – not necessarily
- Poor rock or soil sample recovery

S-15

NOTES



S-16

AIR ROTARY
Advantages

- No drilling fluid required
- Excellent drilling in hard rock
- Good depth capability
- Excellent delineation of water-bearing zones

S-17

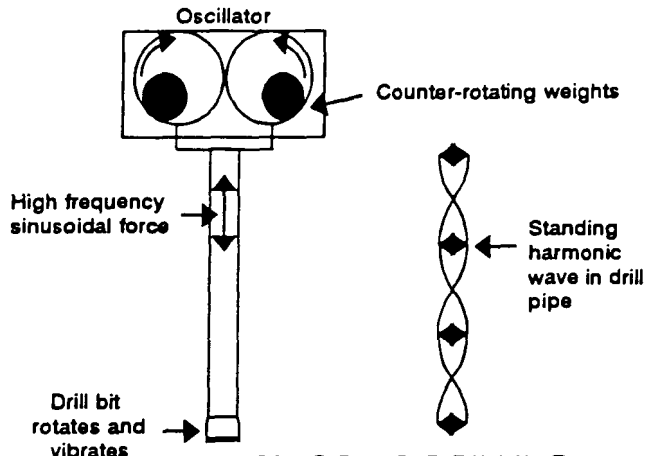
AIR ROTARY
Disadvantages

Not necessarily

- Casing may be required during drilling
- Cross contamination of different formations possible
- Mobility may be limited
- Difficult formation sampling

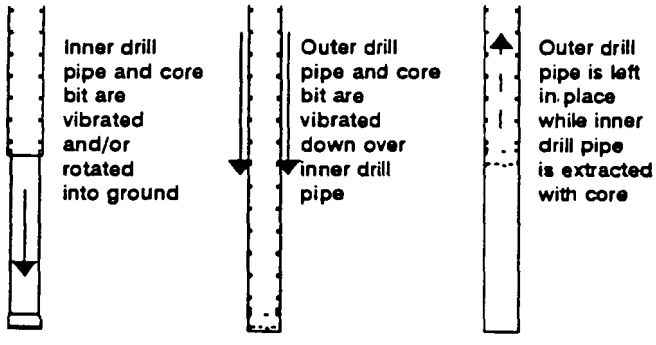
S-18

NOTES



ROTASONIC DRILLING

S-19



ROTASONIC DRILLING METHOD

S-20

ROTASONIC
Advantages

- Fast (20 shallow boreholes/day)
- Versatile (easily penetrates cobbly materials)
- Drills into consolidated and unconsolidated material
- Clean (cuttings and fluid minimized)
- Excellent sampling (quality cores)

S-21

NOTES

ROTASONIC
Disadvantages

- Cost
- Availability
- Dense or cobbly materials are heated by vibration (loss of volatiles)

S-22

HYDROGEOLOGY

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Describe the hydrologic cycle
2. Differentiate between porosity and permeability
3. Describe the difference between confined and unconfined aquifers
4. Evaluate the components of Darcy's Law, including hydraulic conductivity
5. Describe the differences between Darcian velocity and seepage velocity.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

HYDROGEOLOGY

S-1

HYDROGEOLOGY

The study of the interactions of geologic materials and processes with water, especially groundwater

S-2

WATER USES

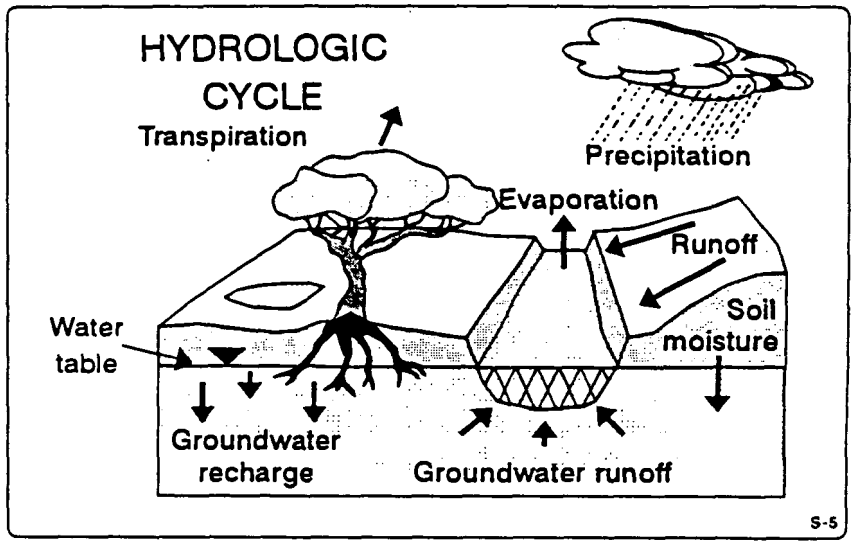
- Drinking
- Irrigation
- Fisheries
- Industrial
- Transportation
- Waste disposal

S-3

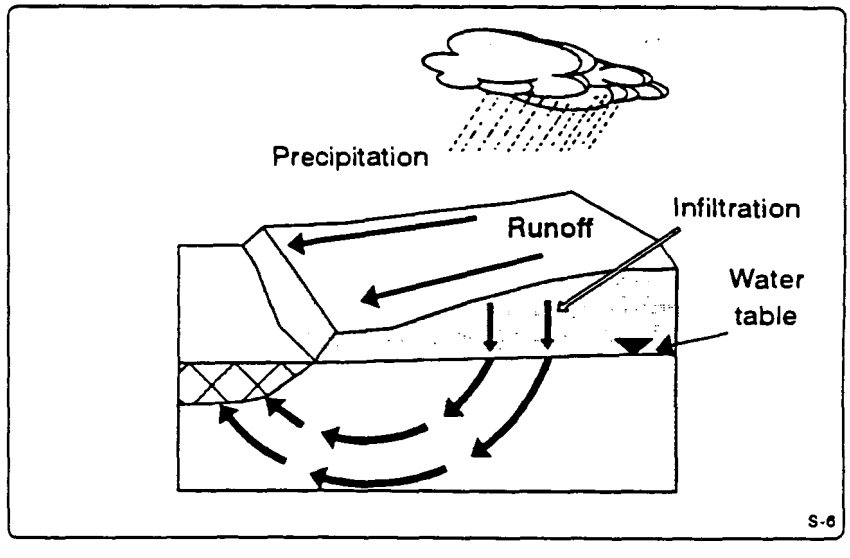
NOTES

HYDROLOGIC CYCLE

S-4

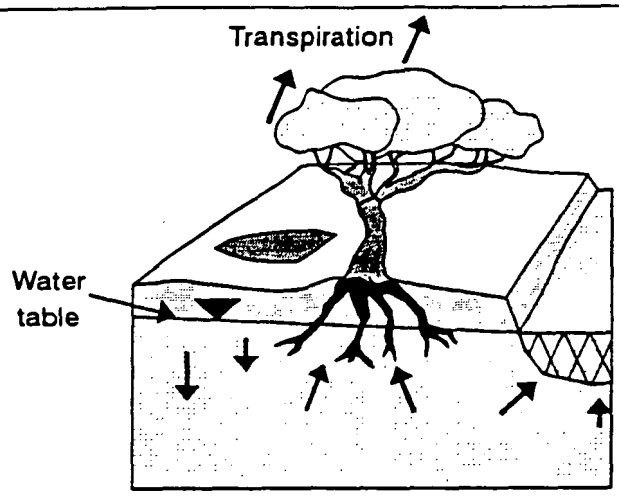


S-5



S-6

NOTES

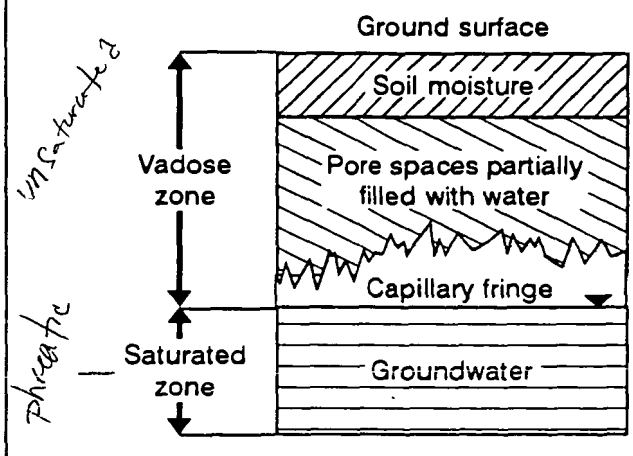


S-7

CONTROLS ON INFILTRATION

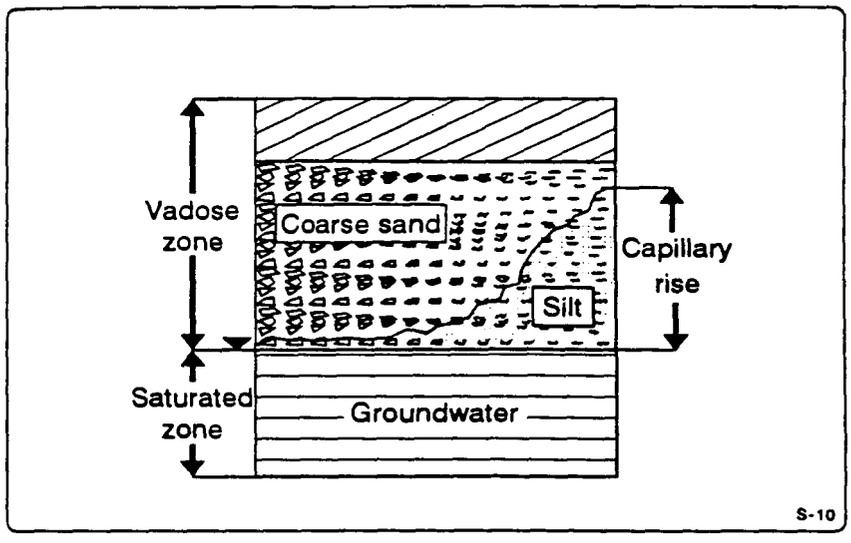
- Soil moisture
- Compaction of soil
- Micro- and macrostructures in the soil
- Vegetative cover
- Temperature
- Topographic relief

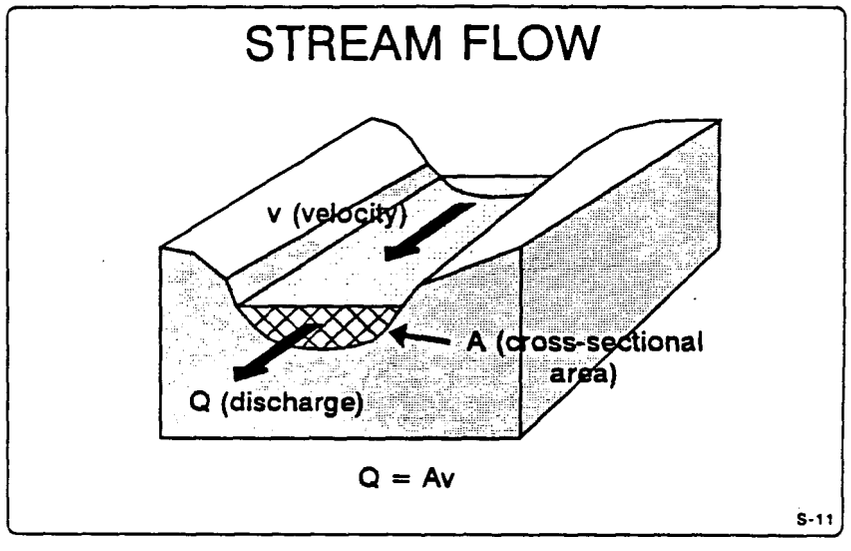
S-8

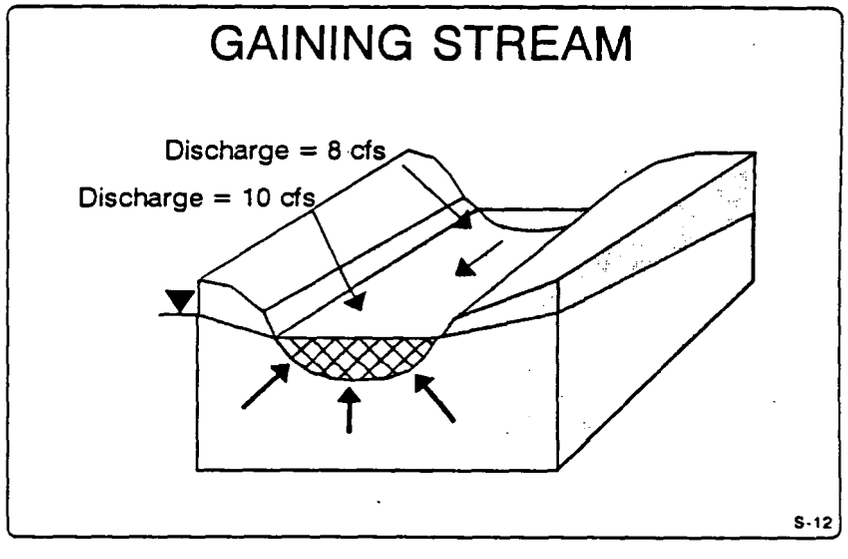


S-9

NOTES



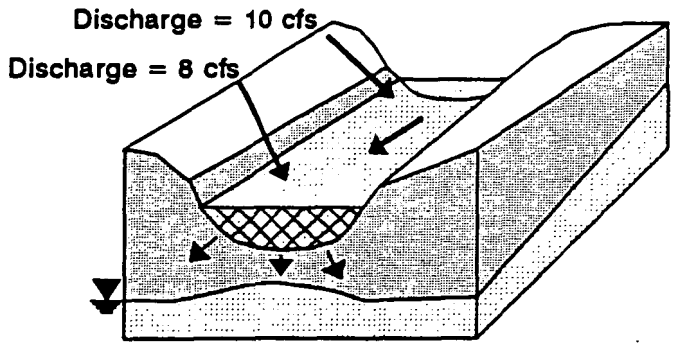




Most often happens after precipitation event, when GW is above/near GW table

NOTES

LOSING STREAM



S-13

Surface water supplying water to gw.

POROSITY
(N_T)

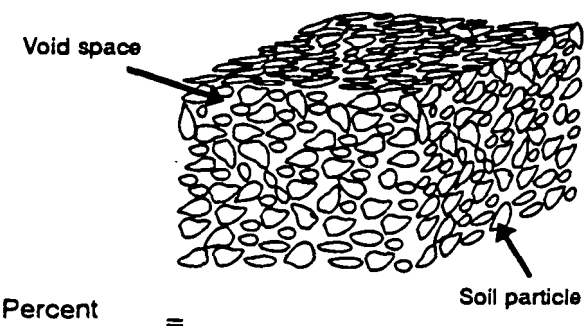
The volumetric ratio between the void spaces (V_v) and total rock (V_t):

$$N_T = \frac{V_v}{V_t} \quad ; \quad N_T = S_Y + S_R$$

S_Y = specific yield — H_2O THAT DRAINS

S_R = specific retention — H_2O THAT WILL NOT DRAIN

S-14

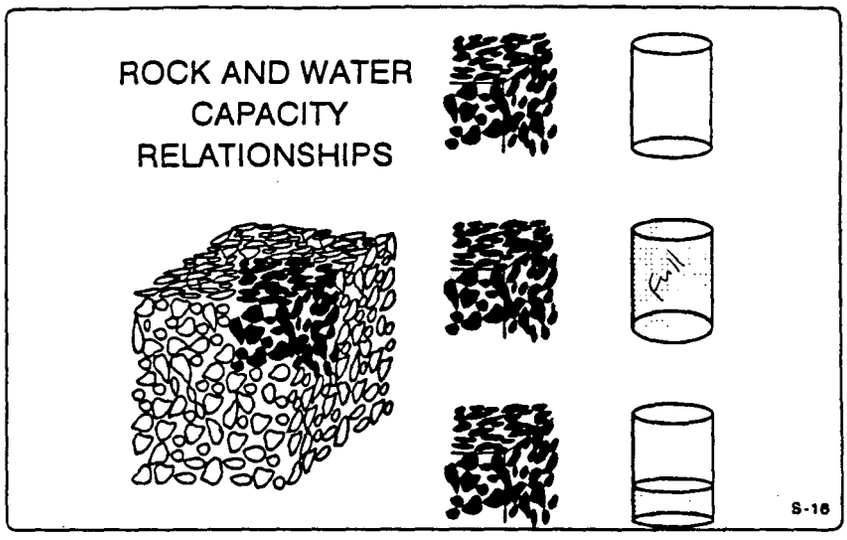


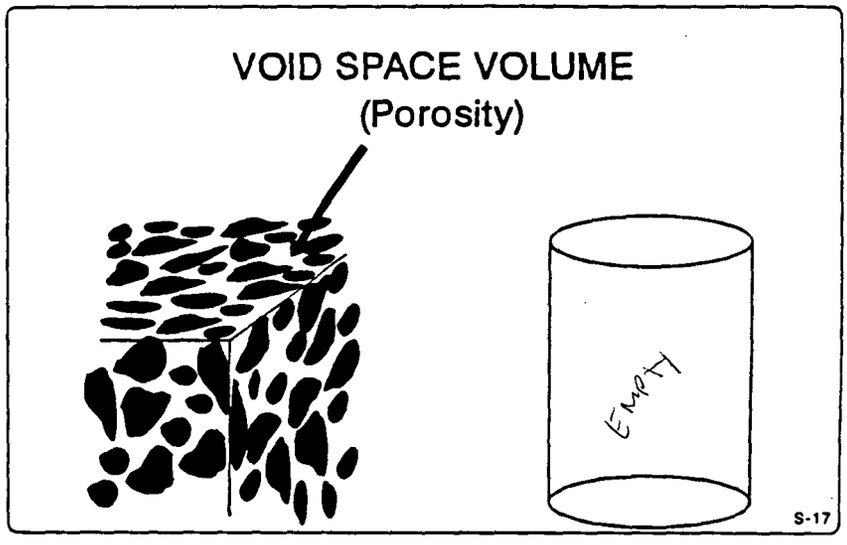
$$\text{Percent Porosity} = \frac{\text{Total Volume - Volume Soil Particles}}{\text{Total Volume}} \times 100$$

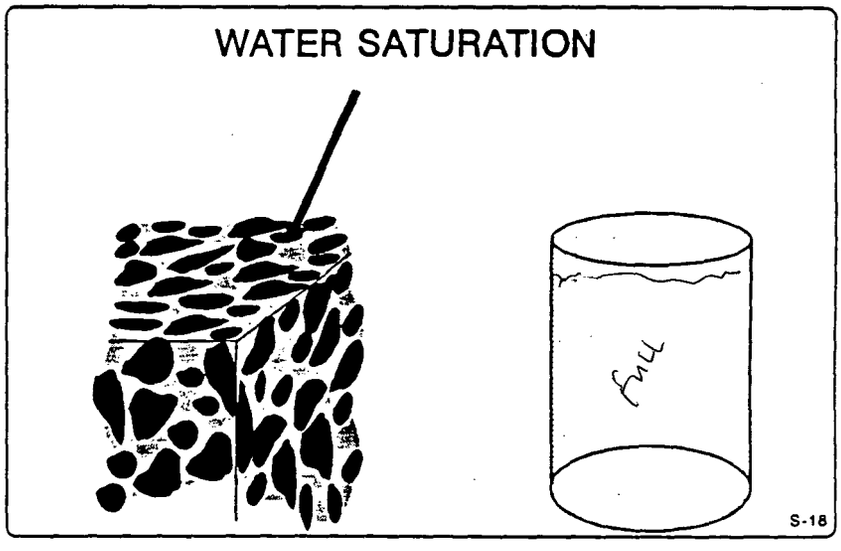
S-15

Clays up to 85% total porosity

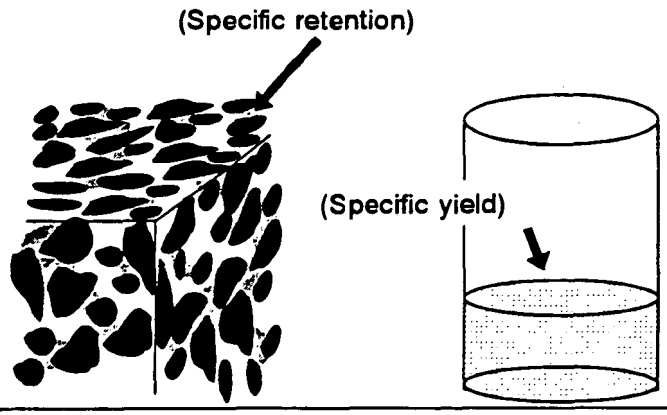
NOTES


ROCK AND WATER
CAPACITY
RELATIONSHIPS
Full
S-16


VOID SPACE VOLUME
(Porosity)
EMPTY
S-17


WATER SATURATION
FULL
S-18

WATER RETAINED AFTER GRAVITY DRAINAGE



S-19

PRIMARY POROSITY

Refers to voids formed at the time the rock or sediment formed

S-20

POROSITY

	Total Porosity (N_T)	Effective Porosity (n_e)
Clay	40–85%	1–10%
Sand	25–50%	10–30%
Gravel	25–45%	15–30%

S-21

NOTES

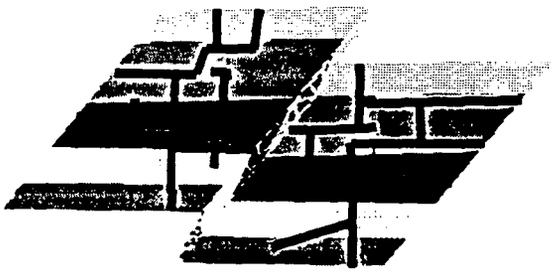
Ex: created by earthquakes, faults, etc
caves

SECONDARY POROSITY

Refers to voids that were formed after the rock was formed

S-22

SECONDARY POROSITY



S-23

PERMEABILITY

The ease with which liquid will move through a porous medium

S-24

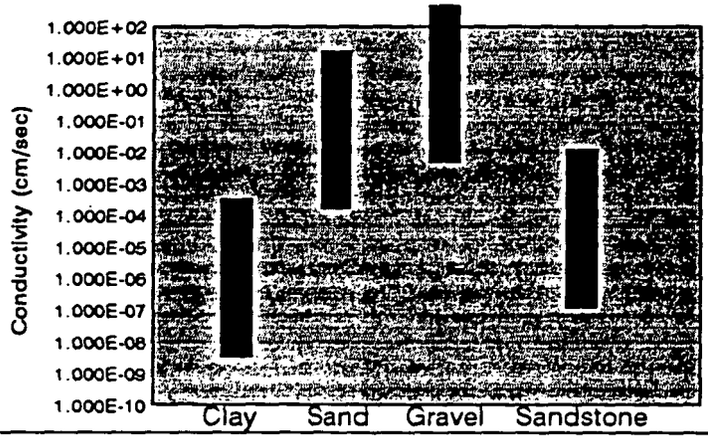
NOTES

HYDRAULIC CONDUCTIVITY

The capacity of a porous medium to transmit water

S-25

CONDUCTIVITY



S-26

AQUIFER

A permeable geologic unit with the ability to store, transmit, and yield water in "usable quantities"

DEPENDS ON WHO IS USING IT. IE: Household, INDUSTRY, etc.

→ NOT DEFINED

S-27

NOTES

HOMOGENEOUS

Having uniform sediment size and orientation throughout an aquifer

S-28

HETEROGENEOUS

Having a nonuniform sediment size and orientation throughout an aquifer

S-29

ISOTROPIC

is the same

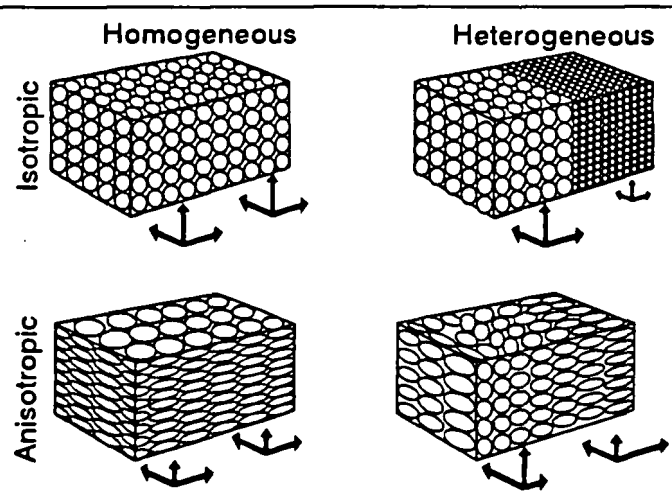
Hydraulic conductivity is independent of the direction of measurement at a point in a geologic formation

S-30

ANISOTROPIC

Hydraulic conductivity varies with the direction of measurement at a point in a geologic formation

S-31



S-32

AQUITARD

A layer of low permeability that can store and transmit groundwater from one aquifer to another

S-33

NOTES

AQUICLUDE

An impermeable confining layer

S-34

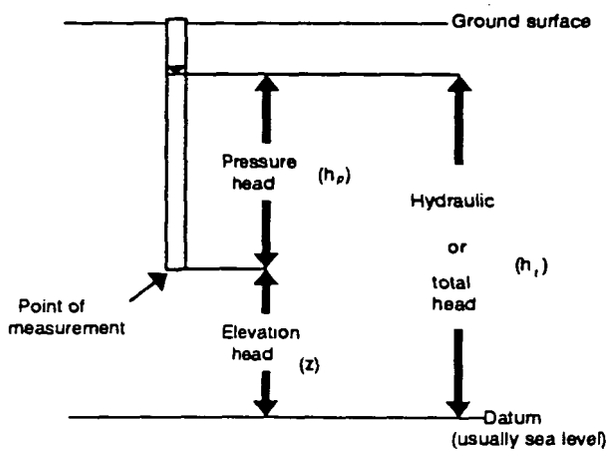
TOTAL HEAD (h_t)

- Combination of elevation (z) and pressure head (h_p)

$$h_t = z + h_p$$

- Total head is the energy imparted to a column of water

S-35



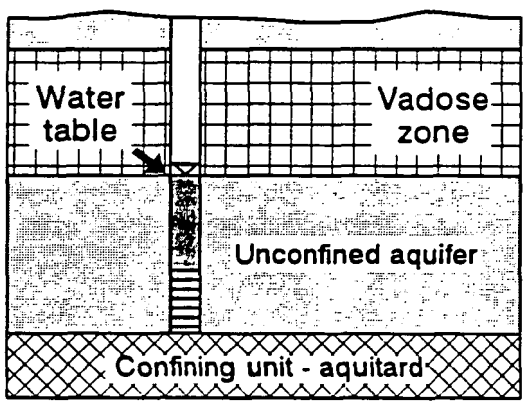
S-36

UNCONFINED AQUIFER (Water Table)

A permeable geologic unit having the ability to store, transmit, and yield water in usable quantities

S-37

UNCONFINED AQUIFER



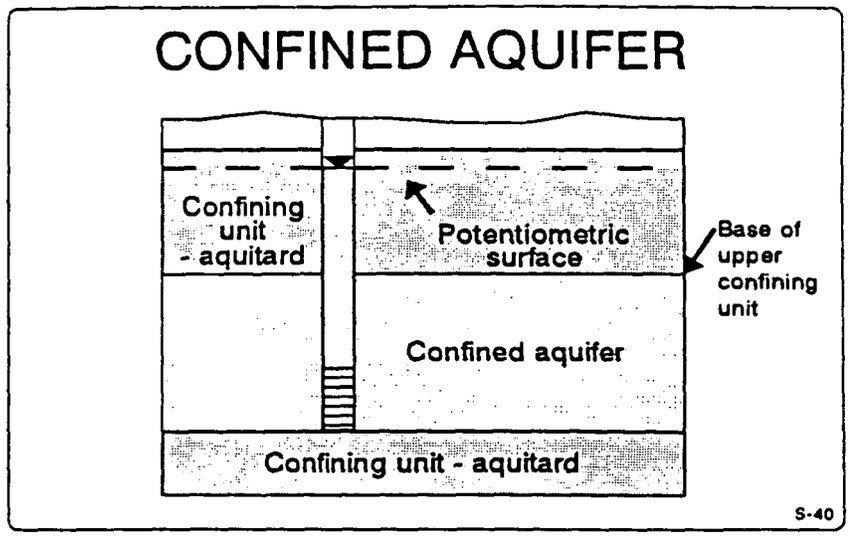
S-38

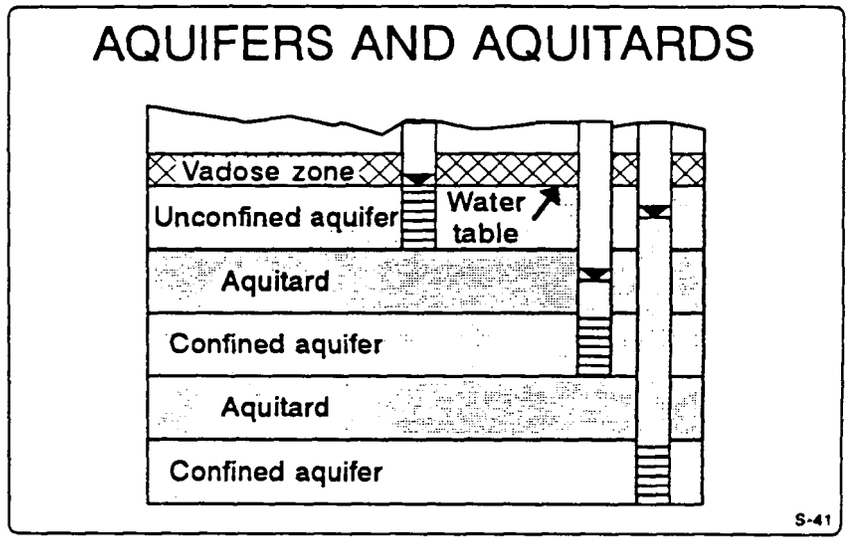
CONFINED AQUIFER (Artesian)

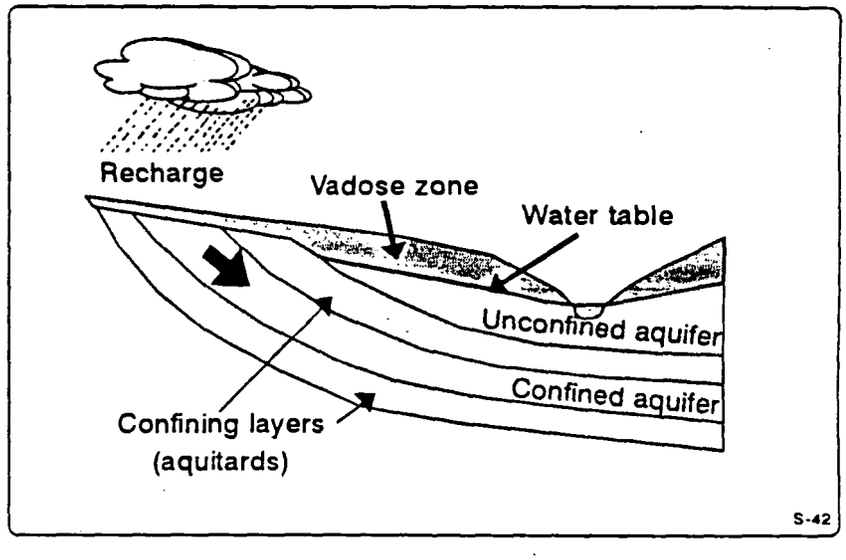
An aquifer overlain by a confining layer whose water is under sufficient pressure to rise above the base of the upper confining layer if it is perforated

S-39

NOTES


CONFINED AQUIFER
Confining unit - aquitard
Potentiometric surface
Base of upper confining unit
Confined aquifer
Confining unit - aquitard
S-40


AQUIFERS AND AQUITARDS
Vadose zone
Unconfined aquifer
Water table
Aquitard
Confined aquifer
Aquitard
Confined aquifer
S-41

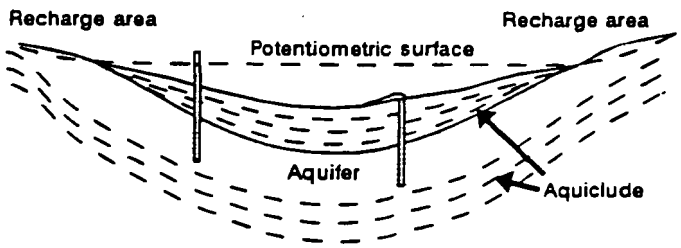

Recharge
Vadose zone
Water table
Unconfined aquifer
Confined aquifer
Confining layers
(aquitards)
S-42

POTENTIOMETRIC SURFACE

The level to which water will rise in an opening (well) if the upper confining layer of a confined aquifer is perforated

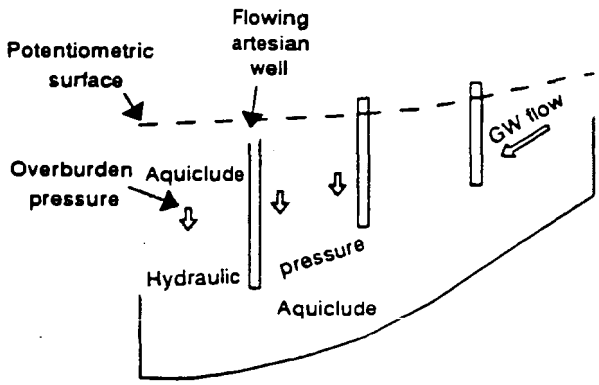
S-43

ARTESIAN GROUNDWATER SYSTEM



S-44

ARTESIAN GROUNDWATER SYSTEM



S-45

NOTES

DARCY'S LAW
Q = KIA

- Q = discharge
- K = hydraulic conductivity
- I = hydraulic gradient $\left(\frac{dh}{dl}\right)$
- A = area

S-46

DARCY'S LAW

- The flow rate through a porous material is proportional to the head loss and inversely proportional to the length of the flow path
- Valid for laminar flow
- Assume homogeneous and isotropic conditions

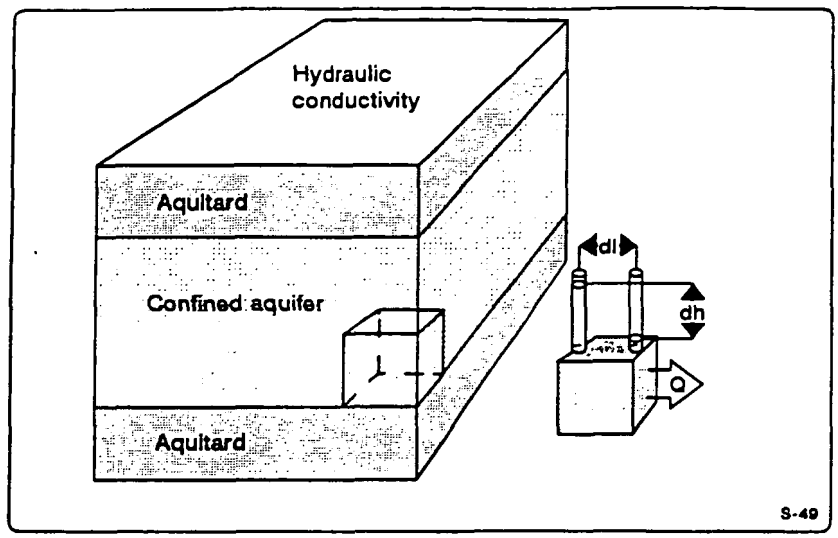
S-47

HYDRAULIC CONDUCTIVITY (K)

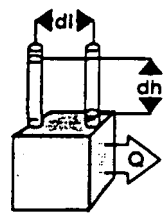
The volume of flow through a unit cross section of an aquifer per unit decline of head

S-48

NOTES



Hydraulic Conductivity



$$Q = KIA$$

$$K = \frac{Q}{IA}$$

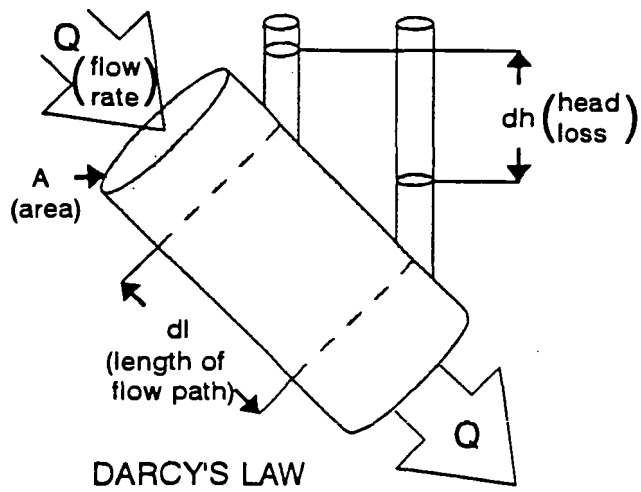
K = hydraulic conductivity

A = cross-sectional area

Q = rate of flow

I = hydraulic gradient $\left(\frac{dh}{dl}\right)$

S-50

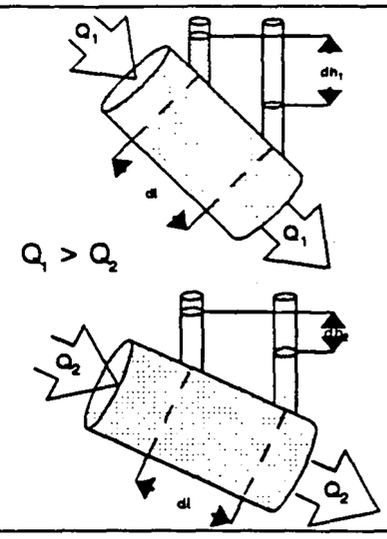


DARCY'S LAW

S-51

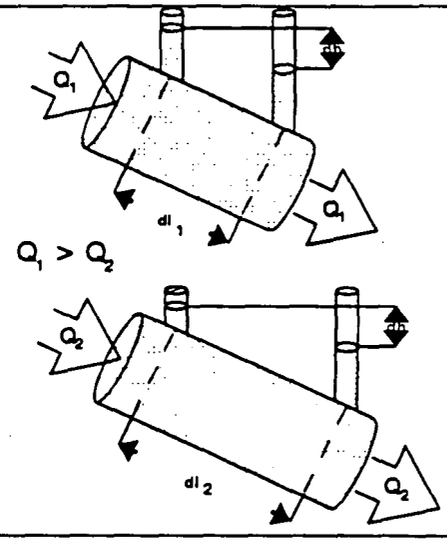
NOTES

Decreasing the hydraulic head decreases the flow rate



S-52

Increasing the flow path length decreases the flow rate



S-53

GROUNDWATER VELOCITY

- Darcy's Law $Q = KIA$ or $\frac{Q}{A} = KI$
- Velocity equation $Q = Av$ or $\frac{Q}{A} = v$

By combining, obtain:

- $v = KI$ Darcian velocity

S-54

GROUNDWATER VELOCITY

- Because water moves only through pore spaces that are connected, porosity is a factor

$N_T = \frac{V_v}{V_T}$ or $N_T = S_R + S_Y$

$n_e = S_Y = N_T - S_R$ ~ effective porosity

$v_s = \frac{Kl}{n_e}$ seepage velocity

S-55

TRANSMISSIVITY

- The capacity of the entire thickness of an aquifer to transmit water

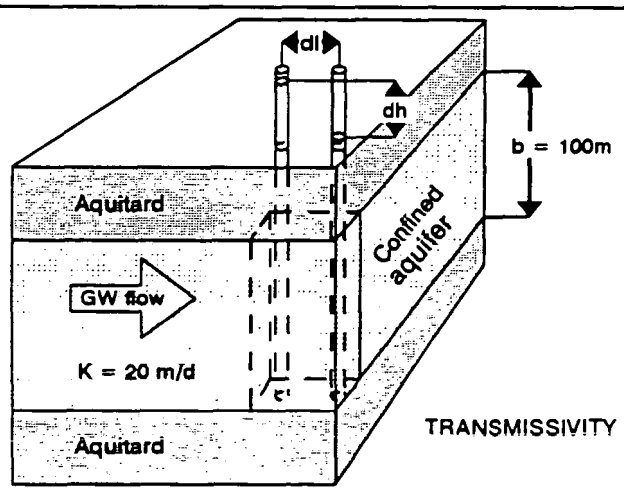
$$T = Kb$$

T = transmissivity

K = hydraulic conductivity

b = aquifer thickness

S-56



S-57

NOTES

TRANSMISSIVITY

$T = Kb$

$T = (20\ m/d)(100\ m)$

$T = 2000\ m^2/d$

S-58

STORATIVITY

Storage Coefficient

- The amount of water available for "use" in an aquifer (storage coefficient)
- "Specific yield" in an unconfined aquifer

S-59

WELL INSTALLATION

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. List the materials necessary for the installation of a well
2. Describe the installation of a well in an unconfined aquifer
3. Describe the installation of a well in a confined aquifer
4. Describe the concept behind nested wells
5. Describe the most common well sampling methods.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

WELL INSTALLATION

S-1

Selection of Filter Pack and Well Screen

S-2

WELL SCREEN

- Surrounded by filter pack
- Filter pack consists of:
  - Coarser materials
  - Uniform grain size
  - Higher permeability

S-3

NOTES

FILTER PACK Purpose

- To allow groundwater to flow freely into well
- To minimize or eliminate entrance of fine-grained materials

S-4

FILTER PACK Selection

- Multiply the 70-percent retained grain size of aquifer materials by 4 or 6
- Use 4 if formation is fine and uniform
- Use 6 if formation is coarser and nonuniform

S-5

FILTER PACK Uniformity Coefficient (UC)

$$\frac{\text{40 percent retained}}{\text{90 percent retained}} = \text{UC}$$

UC should not exceed 2.5

S-6

$UC = \frac{.020}{.016} = 1.38$

NOTES

WELL SCREEN Selection

Select screen slot opening to retain 90 percent of filter pack material

S-7

WELL MATERIALS

- Well screen/riser/well points
  - Teflon®
  - Stainless steel
  - PVC
- Gravel/filter pack
- Bentonite
- Grout/cement

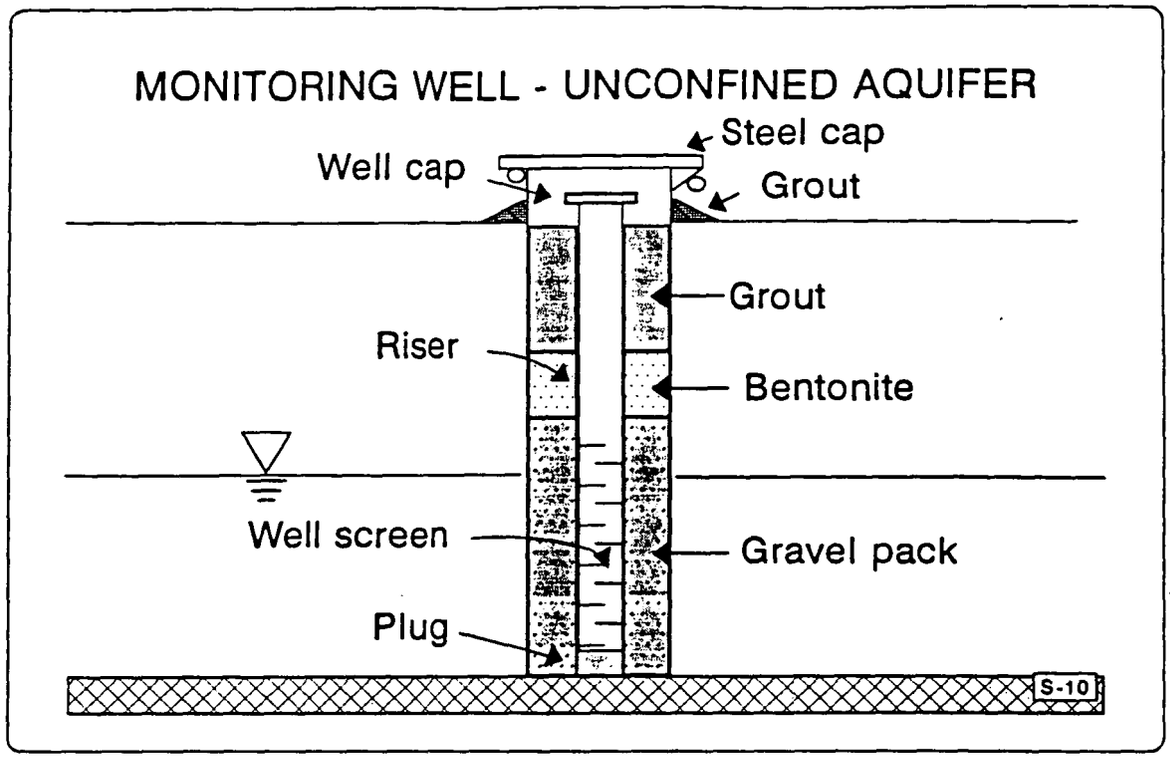
S-8

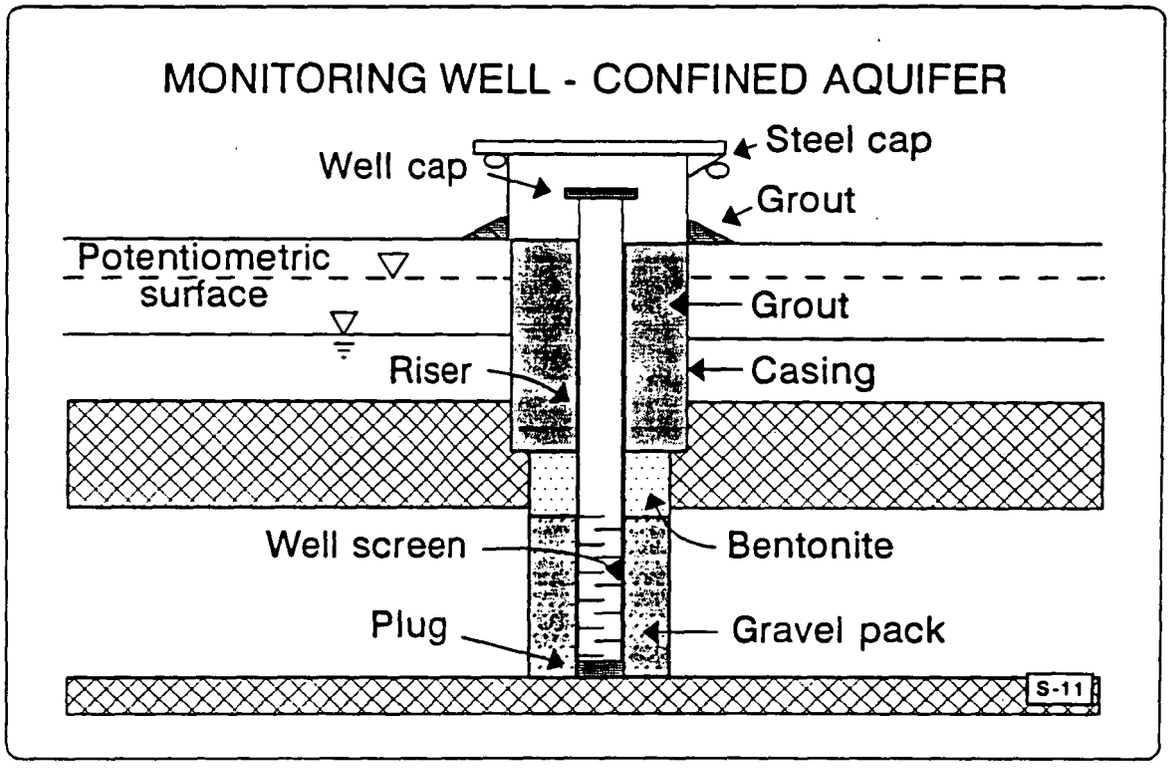
WELL INSTALLATION

- Unconfined aquifer
- Confined aquifer

S-9

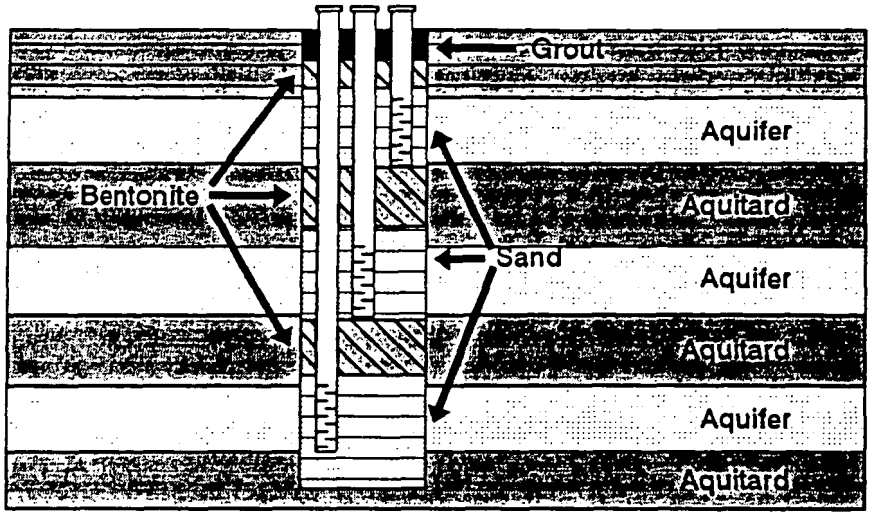
unconfined
water will rise to or above w/in hrs when drilling





NOTES

NESTED WELLS - MULTILEVEL SAMPLING



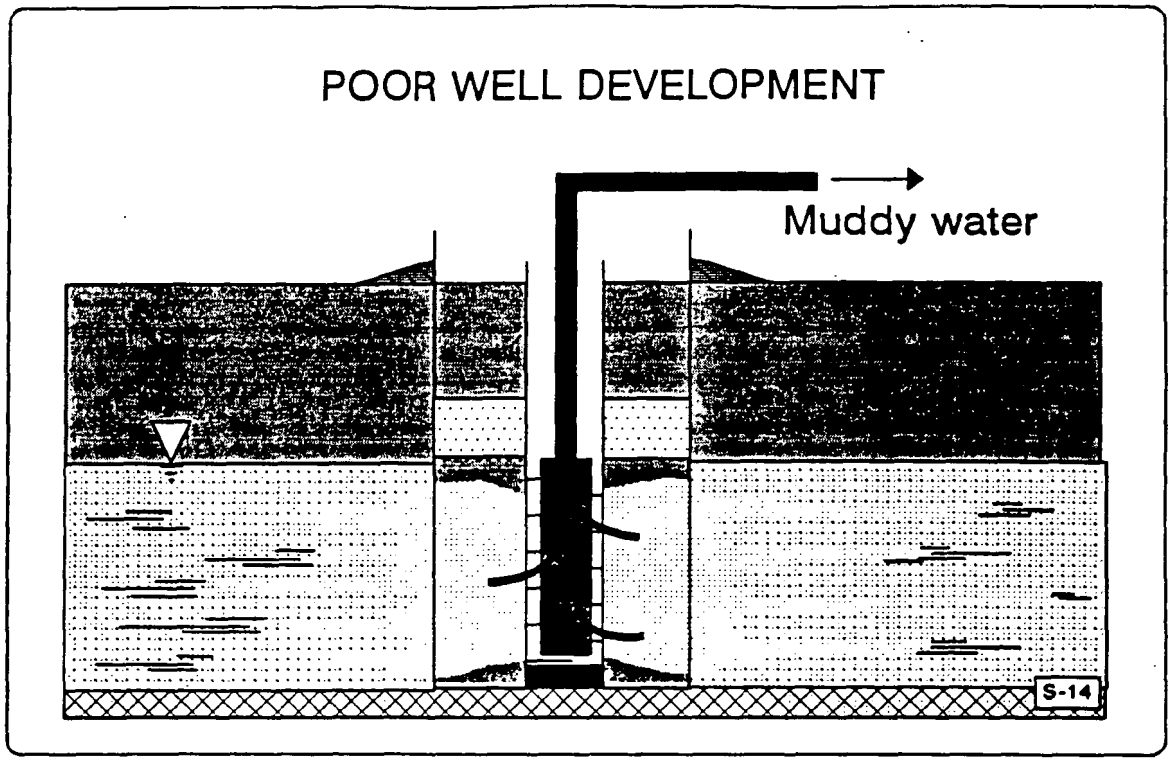
S-12

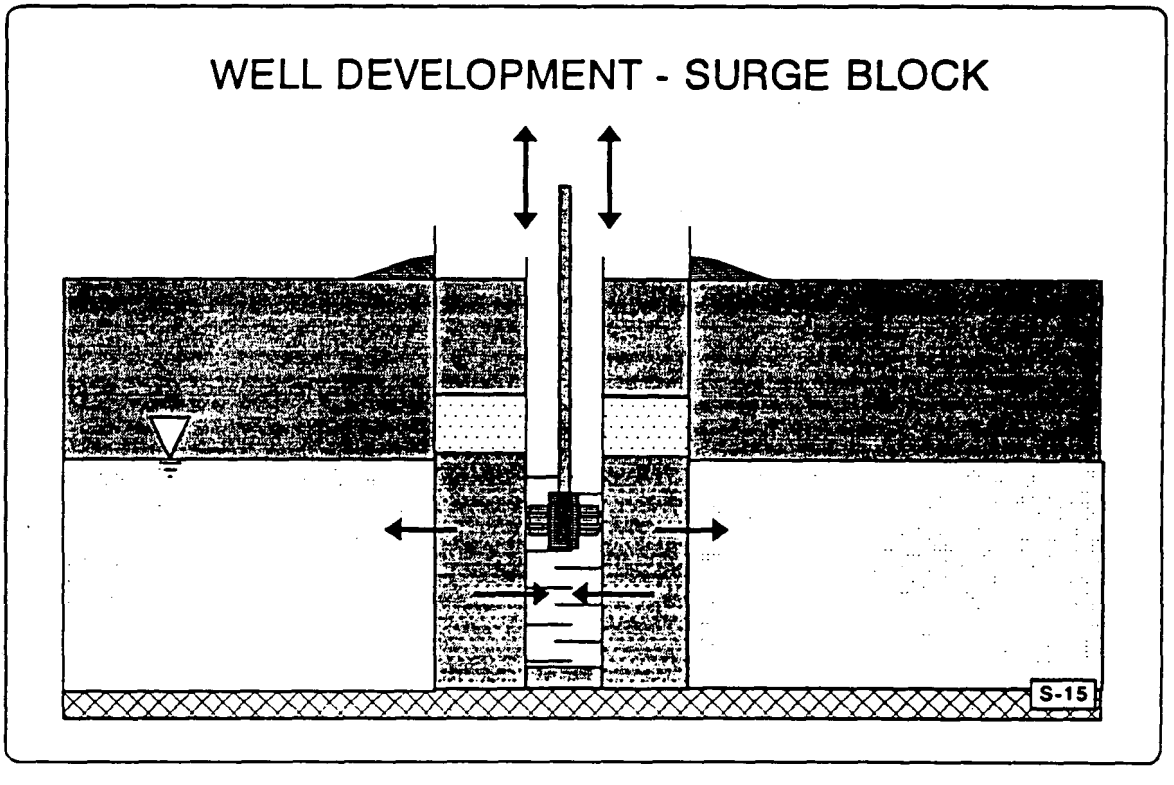
WELL AND AQUIFER DEVELOPMENT

- Surge block
- Bailer
- Pulse pumping
- Air surging

S-13

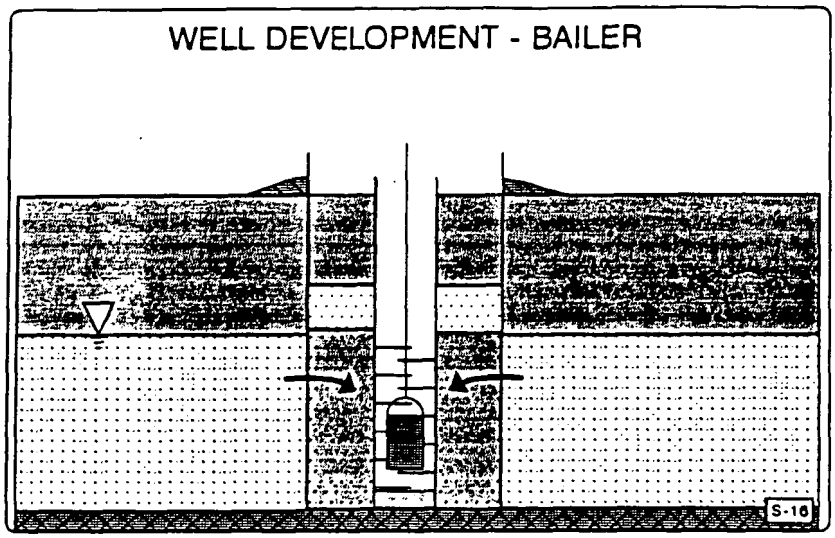
NOTES

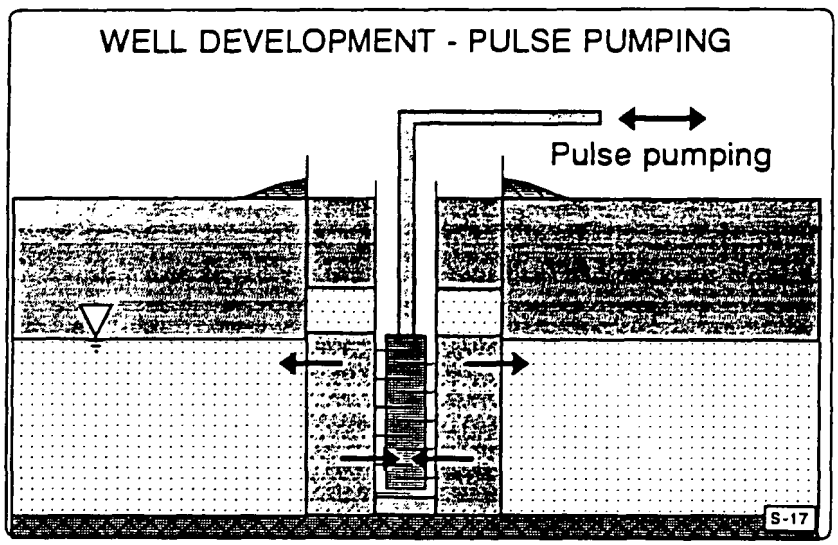

POOR WELL DEVELOPMENT
Muddy water
S-14

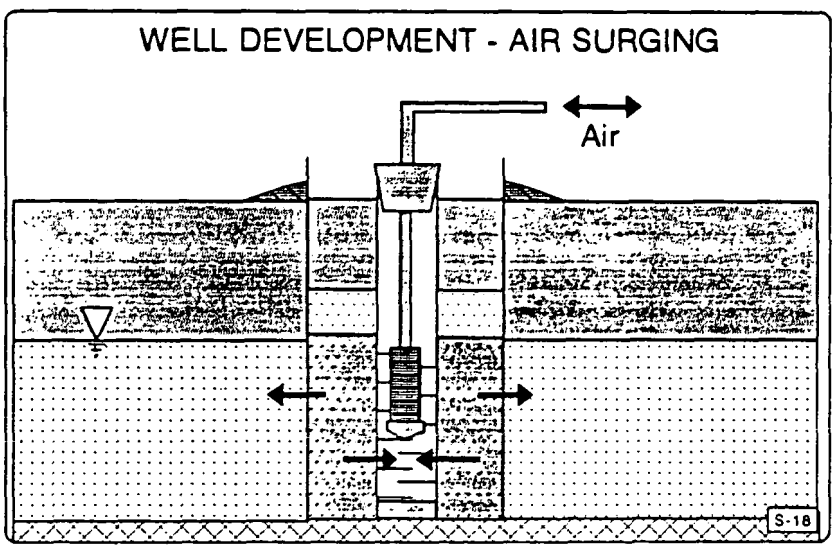

WELL DEVELOPMENT - SURGE BLOCK
S-15

NOTES

NOTES


WELL DEVELOPMENT - BAILER
S-16


WELL DEVELOPMENT - PULSE PUMPING
Pulse pumping
S-17


WELL DEVELOPMENT - AIR SURGING
Air
S-18

NOTES

SAMPLING METHODS

- Bladder pump
- Submersible pump
- Hand pump
- Bailer

S-19

GROUNDWATER SAMPLING PROTOCOL

- Flexible
- Written and defensible document to be used at all sites

S-20

Should Be Done Same way at the Same Time - Each Time Samples.

PURGING

- Volume is well specific
- Verified by temperature, pH, specific conductance, and turbidity

S-21

NOTES

DETERMINING WELL VOLUME

$$V = 0.041\ d^2 h$$

V = Volume of water in gallons

d = Diameter of well in inches

h = Depth of water in well in feet

S-22

FOR PURGING

Sampling for gw, not well water, so need to purge well water to get the good gw coming in.

VADOSE ZONE

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Describe the vadose zone
2. List three reasons why the vadose zone is important in groundwater investigations
3. Describe the operation of pressure vacuum lysimeters
4. Characterize the limitations of vacuum lysimeters
5. Describe the principles of soil gas wells
6. Characterize the limitations of soil gas wells.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

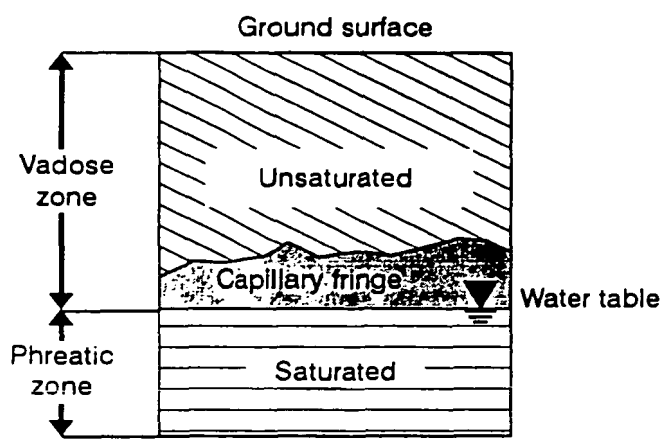
VADOSE ZONE

S-1

THE VADOSE ZONE

That part of the geologic profile between the ground surface and the water table, including the capillary fringe

S-2



S-3

NOTES

THE VADOSE ZONE

- Generally unsaturated
- <100% water content
- Capillary pressure predominant

S-4

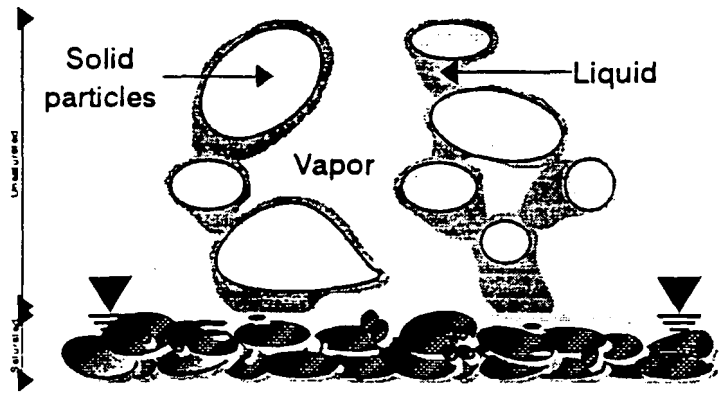
THE VADOSE ZONE

Consists of:

- Solid and particulate material
- Vapors in pore spaces
- Liquids on grain surfaces

S-5

THE VADOSE ZONE



S-6

CAPILLARY FRINGE

- Transition zone between saturated and unsaturated zones
- Result of capillary pressure pulling water into unsaturated zone

S-7

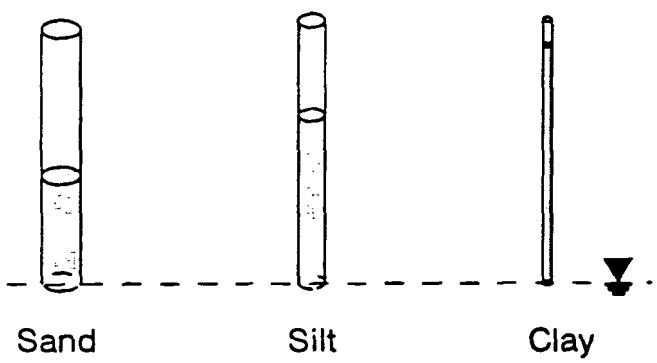
CAPILLARITY

The result of two forces:

- Attraction of water to the walls of the pore space (adhesion)
- Attraction of water molecules to each other (cohesion)

S-8

CAPILLARY FRINGE



S-9

NOTES

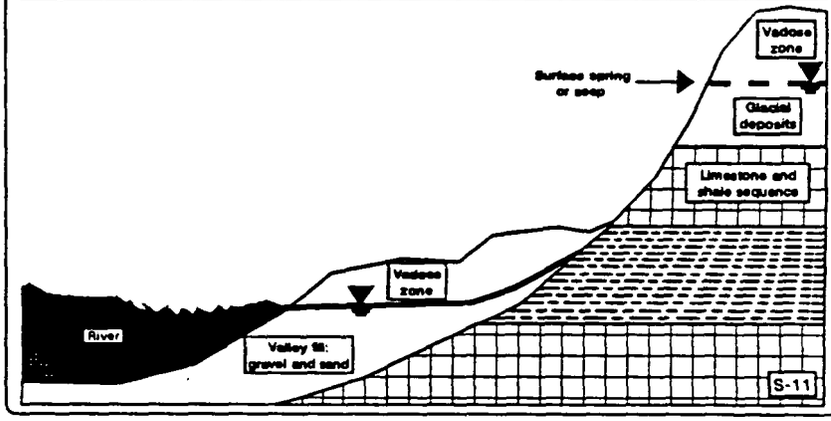
THE VADOSE ZONE
Physical Properties

Physical properties vary according to:

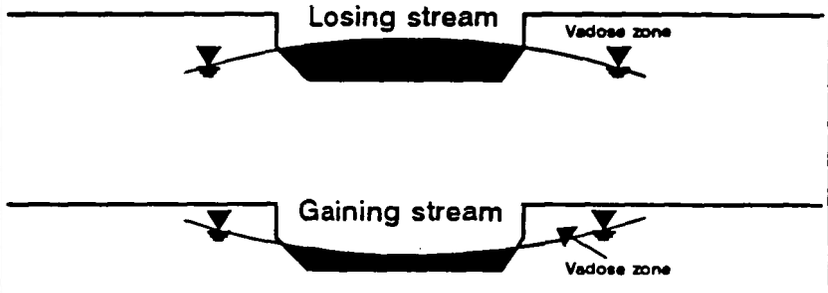
- Atmospheric conditions
- Hydrogeologic conditions
- Geologic conditions

S-10

THE VADOSE ZONE
Cross Section - Ohio River Valley

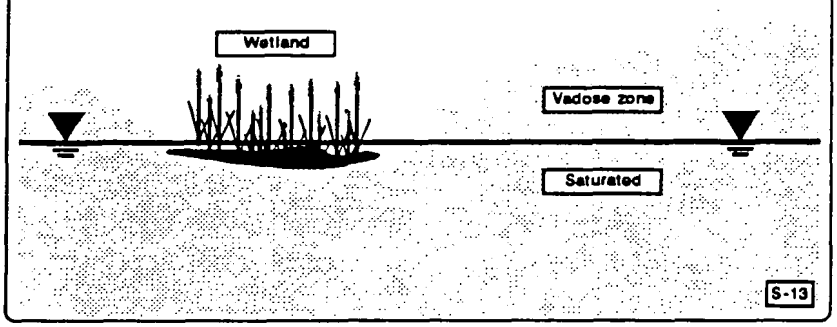


VADOSE ZONE ADJACENT TO STREAMS



S-12

VADOSE ZONE ADJACENT TO WETLAND



S-13

THE VADOSE ZONE Unsaturated Flow

Primarily affected by:

- Matric potential
- Osmotic potential
- Gravitational potential
- Moisture content

S-14

THE VADOSE ZONE Matric Potential

- Attraction of water to solid particles
- Responsible for upward flow of water or capillary pressure

S-15

NOTES

THE VADOSE ZONE
Osmotic Potential

Attraction of water to ions
or other solutes in the soil

S-16

THE VADOSE ZONE
Gravitational Potential

- Gravitational pull on water
- Encourages downward flow of water or infiltration

S-17

THE VADOSE ZONE
Soil Potential

- Combination of matric and osmotic potentials
- Impedes or binds the flow of water in the unsaturated zone

S-18

NOTES

THE VADOSE ZONE
Unsaturated Flow

Occurs when the gravitational potential is greater than the soil potential (matric + osmotic)

S-19

THE VADOSE ZONE
Moisture Content

Increased moisture content decreases soil potential (matric + osmotic), increasing the ability of water to flow

S-20

Devices for Measuring Moisture Content and Soil Potential in the Vadose Zone

S-21

NOTES

MOISTURE CONTENT

Measured by:

- Radioactive devices
- Time domain reflectometry

S-22

MOISTURE CONTENT
Radioactive Devices

- Neutron - Neutron
  - Directly measures soil or rock water content and porosity
- Gamma - Gamma
  - Determines soil or rock density
  - Indirectly measures water content and porosity

S-23

MOISTURE CONTENT
Radioactive Devices

- Advantages
  - In-situ measurements directly or indirectly related to water content
  - Average water content can be determined at depth
  - Accommodates automatic recordings
  - Near-surface water content measurements possible

S-24

MOISTURE CONTENT
Radioactive Devices

- Disadvantages:
  - Expensive
  - Radioactive source requires special care and license

S-25

MOISTURE CONTENT
Time Domain Reflectometry

- Measures an electromagnetic pulse emitted from one or more probes
- Determines moisture content

S-26

MOISTURE CONTENT
Time Domain Reflectometry

- Advantages
  - Accurate
  - Variable depth placement
  - Variety of sensor configurations
  - Remote and continual monitoring

S-27

NOTES

MOISTURE CONTENT
Time Domain Reflectometry

- Disadvantages
  - Probes must be placed properly
  - Long-term use untested
  - Cost of remote monitoring equipment relatively high

S-28

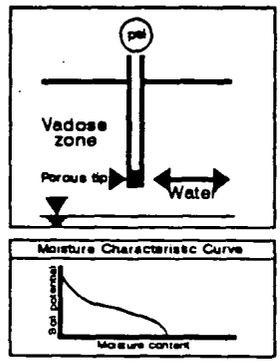
SOIL POTENTIAL

Measured by:

- Tensiometer
- Electrical resistance block
- Psychrometer

S-29

SOIL POTENTIAL
Tensiometer



S-30

SOIL POTENTIAL
Tensiometer

- Measures the matric potential in soil
- Advantages
  - Inexpensive
  - Durable
  - Easy to operate

S-31

SOIL POTENTIAL
Tensiometer

- Disadvantages
  - Ineffective under very dry conditions because of air entry
  - Sensitive to temperature changes
  - Sensitive to atmospheric pressure changes

S-32

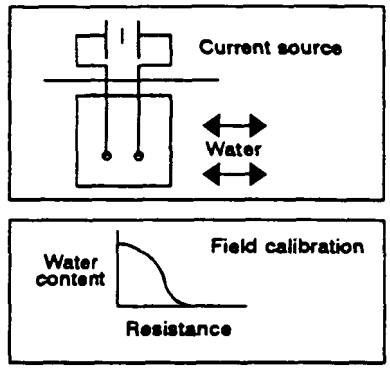
SOIL POTENTIAL
Tensiometer

- Disadvantages (cont.)
  - Sensitive to air bubbles in lines
  - Requires a long time to achieve equilibrium

S-33

NOTES

SOIL POTENTIAL
Electrical Resistance Blocks



S-34

SOIL POTENTIAL
Electrical Resistance Blocks

- Advantages
  - Suited for general use
  - Inexpensive
  - Can determine moisture content or soil potential
  - Requires little maintenance

S-35

SOIL POTENTIAL
Electrical Resistance Blocks

- Disadvantages
  - Ineffective under very dry conditions
  - Sensitive to temperature
  - Time-consuming field calibration
  - Affected by salinity
  - Ineffective in coarse or swelling/shrinking soils

S-36

SOIL POTENTIAL
Psychrometer

Measures soil potential under very dry conditions

S-37

SOIL POTENTIAL
Psychrometer

- Advantages
  - Continuous recording of pressures
  - Variable depth placement
  - Remote monitoring

S-38

SOIL POTENTIAL
Psychrometer

- Disadvantages
  - Very sensitive to temperature fluctuations
  - Expensive
  - Complex
  - Performs poorly in wet media

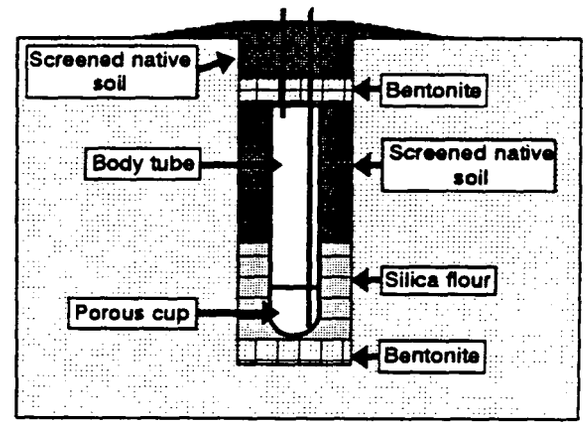
S-39

NOTES

SAMPLING FLUIDS AND VAPORS

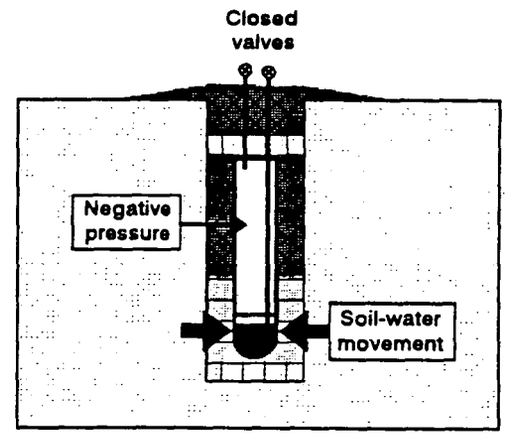
- Fluids
  - Pressure-vacuum lysimeter
- Vapors
  - Soil-gas probe

S-40



Pressure-Vacuum Lysimeter

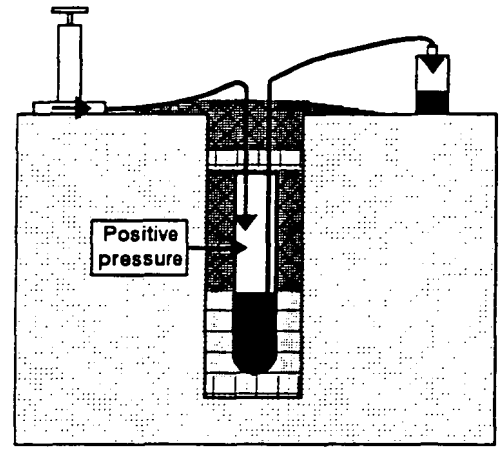
S-41



Collection of Pore Water

S-42

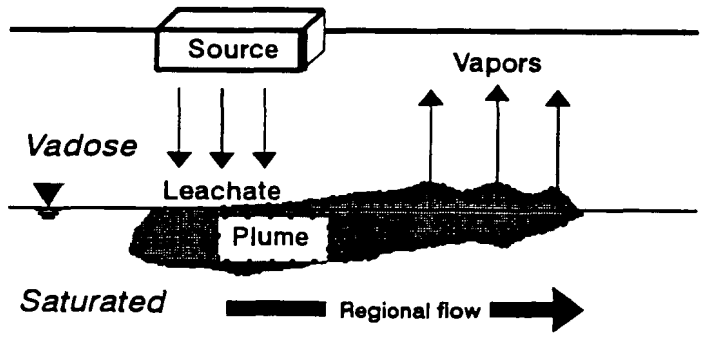
NOTES



Transfer to Sample Bottle

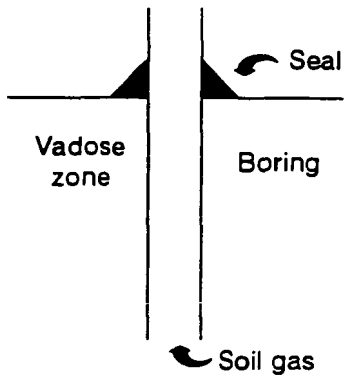
S-43

VADOSE ZONE VAPOR SAMPLING



S-44

SOIL GAS PROBE Schematic



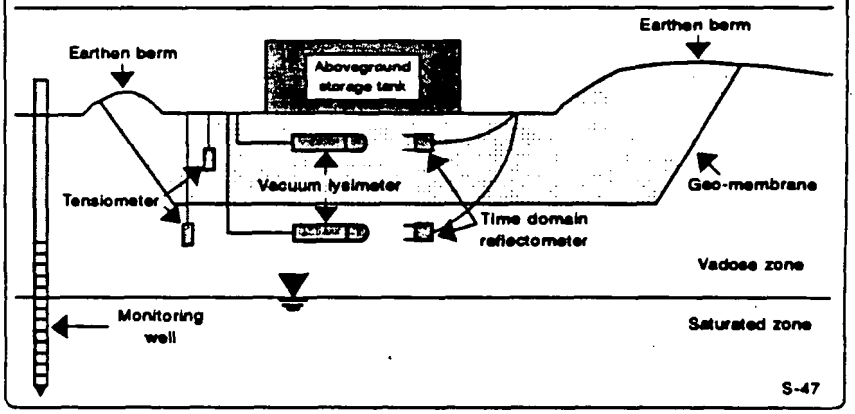
S-45

NOTES

Uses for Vadose Zone Monitoring Equipment

S-46

VADOSE ZONE MONITORING FOR NEW TANK FARM



S-47

GEOPHYSICAL METHODS

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Describe the basic principles of operation of the following surface geophysical methods:
   a. Magnetics
   b. Electromagnetics (EM)
   c. Electrical resistivity
   d. Seismic refraction
   e. Ground-penetrating radar

2. Identify the limitations of the following geophysical methods:
   a. Magnetics
   b. Electromagnetics (EM)
   c. Electrical resistivity
   d. Seismic refraction
   e. Ground-penetrating radar

3. Describe the basic principles of operation of the following borehole geophysical methods:
   a. Spontaneous potential
   b. Normal resistivity
   c. Natural-gamma

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

STUDENT PERFORMANCE OBJECTIVES (cont.)

d. Gamma-gamma
e. Neutron
f. Caliper
g. Acoustic
h. Temperature

4. Identify the limitations of the following borehole geophysical methods:
   a. Spontaneous potential
   b. Normal resistivity
   c. Natural-gamma
   d. Gamma-gamma
   e. Neutron
   f. Caliper
   g. Acoustic
   h. Temperature.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

GEOPHYSICAL METHODS

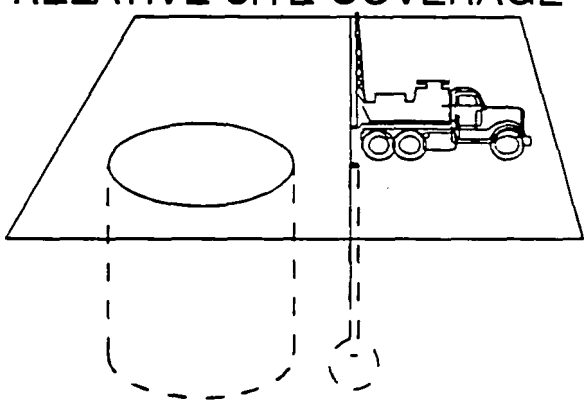
S-1

GEOPHYSICS

- Nonintrusive, investigative tool
- Site-specific methods
- "Ground truthed" data
- Professional interpretation

S-2

RELATIVE SITE COVERAGE



Volume of typical geophysical measurement

Volume of drilling or water sampling

S-3

NOTES

GROUND TRUTHING

Correlation of physical evidence (i.e., rock cores) to geophysical data

S-4

ANOMALY

Significant variation from background

S-5

GEOPHYSICAL TECHNIQUES

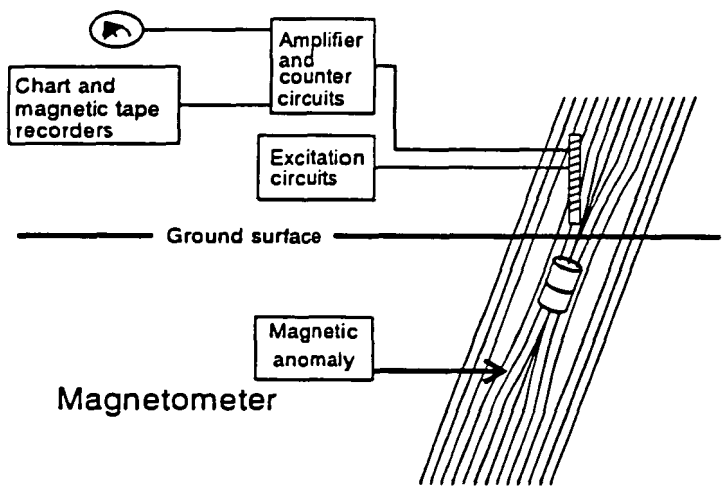
- Magnetics
- Electromagnetics (EM)
- Electrical resistivity
- Seismic refraction/reflection
- Ground-penetrating radar
- Borehole geophysics

S-6

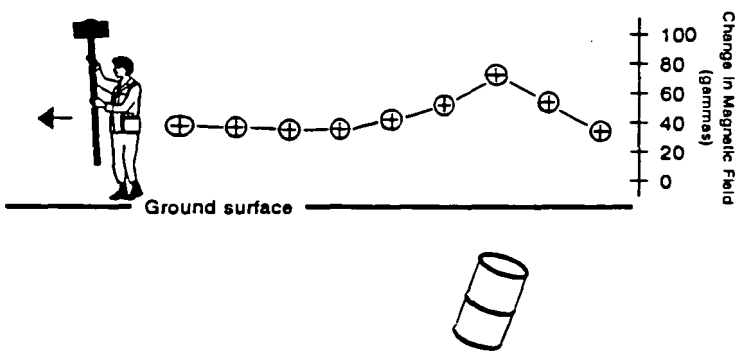
MAGNETICS

- Measurement of magnetic field strength in units of gammas
- Anomalies are caused by variations in magnetic field strength in the vicinity of the sensor

S-7



S-8



S-9

NOTES

MAGNETICS Advantages

- Relatively low cost (cost-effective)
- Short time frame required
- Little, if any, site preparation needed
- Simple survey sufficient (compass and tape)

S-10

MAGNETICS Disadvantages

- Cultural noise limitations
- Difficulty in differentiating between steel objects (i.e., 55-gallon drums and a refrigerator)

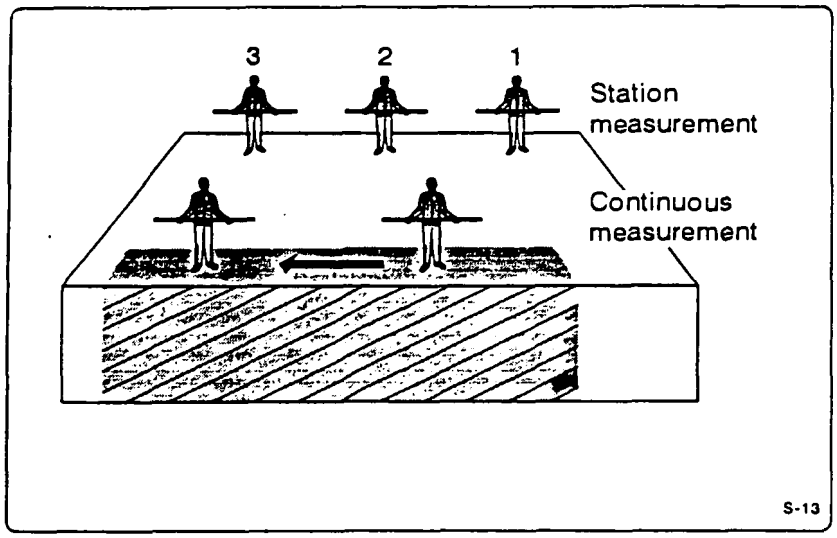
S-11

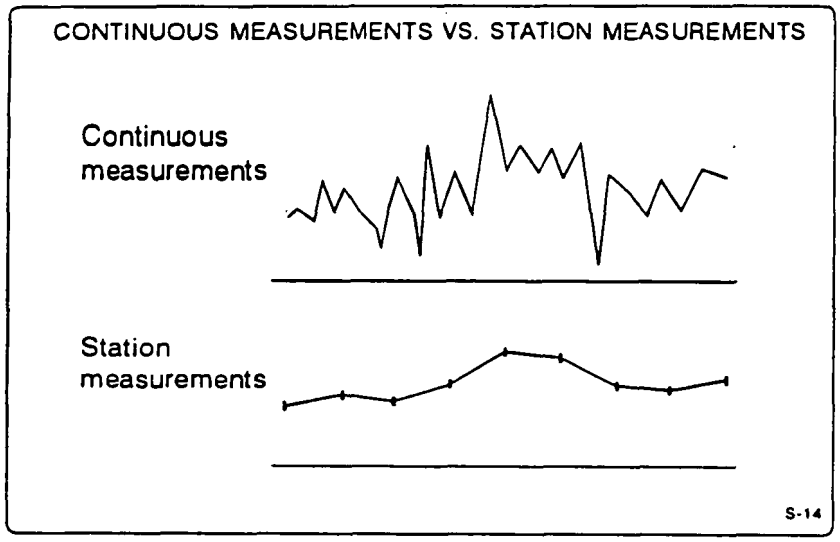
ELECTROMAGNETICS

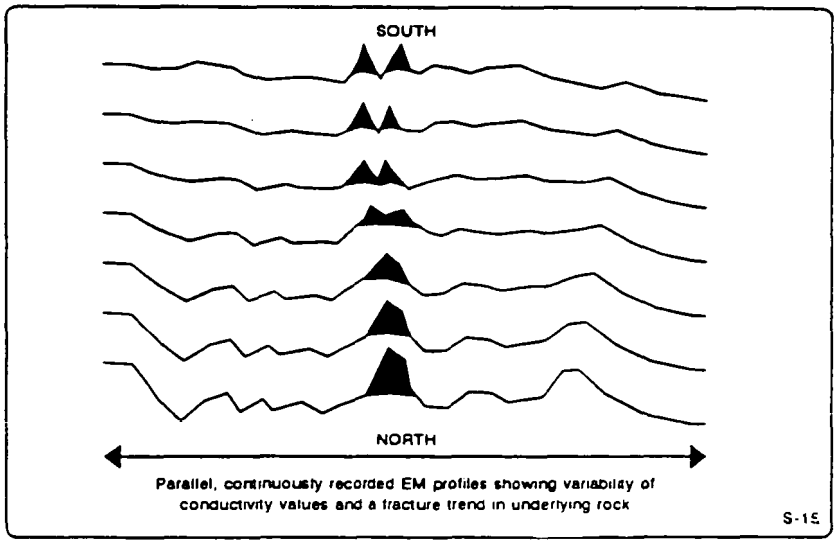
- Based on physical principles of inducing and detecting electrical flow within geologic strata
- Measures bulk conductivity (the inverse of resistivity) of geologic materials beneath the transmitter and receiver coils

S-12

NOTES


3
2
1
Station measurement
Continuous measurement
S-13


CONTINUOUS MEASUREMENTS VS. STATION MEASUREMENTS
Continuous measurements
Station measurements
S-14


SOUTH
NORTH
Parallel, continuously recorded EM profiles showing variability of conductivity values and a fracture trend in underlying rock
S-15

NOTES

ELECTROMAGNETICS Advantages

- Rapid data collection with minimum personnel
- Lightweight, portable equipment
- Commonly used in groundwater pollution investigations for determining plume flow direction

S-16

ELECTROMAGNETICS Disadvantages

- Cultural noise limitations (when used for hydrogeological purposes)
- Limitations in areas where geology varies laterally (anomalies can be misinterpreted as plumes)

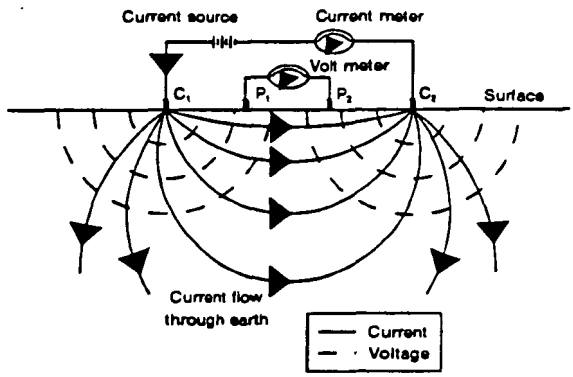
S-17

ELECTRICAL RESISTIVITY

- Measures the bulk resistivity of the subsurface in ohm-meter units
- Current is injected into the ground through surface electrodes

S-18

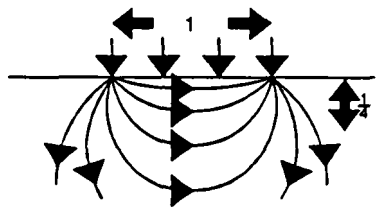
ELECTRICAL RESISTIVITY WENNER ARRAY



S-19

ELECTRICAL RESISTIVITY

Depth of investigation is equal to one-fourth of the distance between electrodes



S-20

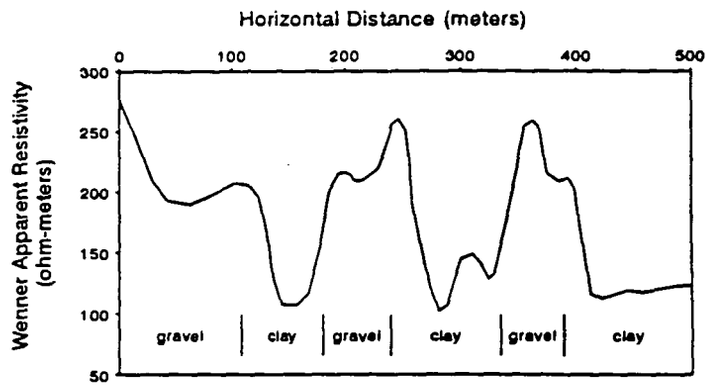
ELECTRICAL RESISTIVITIES OF GEOLOGIC MATERIALS

Function of:

- Porosity
- Permeability
- Water saturation
- Concentration of dissolved solids in pore fluids

S-21

NOTES



RESISTIVITY PROFILE ACROSS GLACIAL CLAYS AND GRAVELS

S-22

RESISTIVITY SURVEYS

- Profiling – lateral contacts using constant electrode spacing
- Sounding – stratigraphic changes measured with successively larger electrode spacings

S-23

ELECTRICAL RESISTIVITY Advantages

- Qualitative modeling of data is feasible
- Models can be used to estimate depths, thicknesses, and resistivities of subsurface layers

S-24

NOTES

ELECTRICAL RESISTIVITY (cont.) Advantages

- Layer resistivities can be used to estimate resistivity of saturating fluid
- Extent of groundwater plume can be approximated

S-25

ELECTRICAL RESISTIVITY Disadvantages

- Cultural noise limitations
- Large area free from grounded metallic structures required
- Level of effort/number of operational personnel

S-26

SEISMIC TECHNIQUES

- Refraction method
- Reflection method

S-27

NOTES

SEISMIC REFRACTION

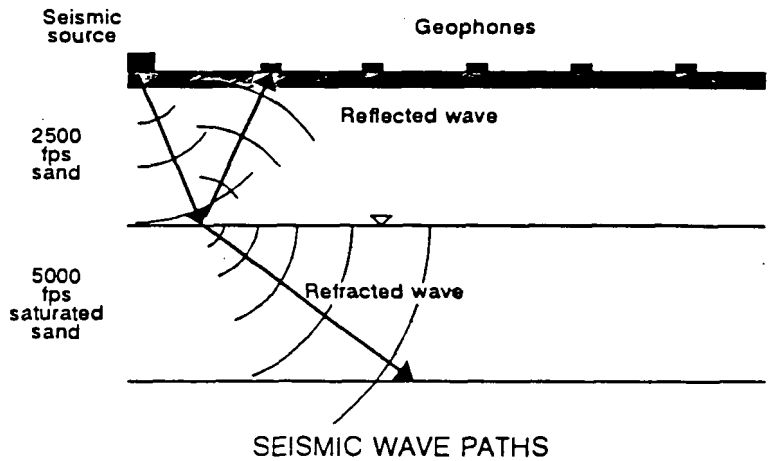
- Cheaper and easier
- Determination of velocity and depth of layers

S-28

SEISMIC REFLECTION

- More expensive and complex
- Resolution of thin layers

S-29



SEISMIC WAVE PATHS

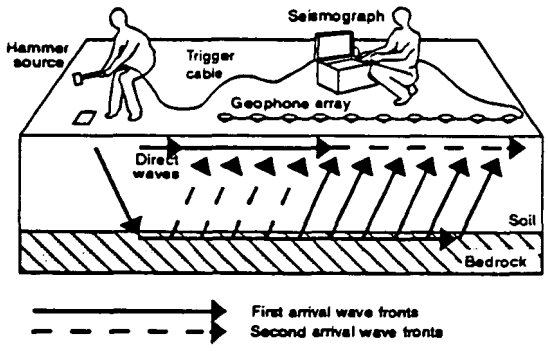
S-30

SEISMIC REFRACTION

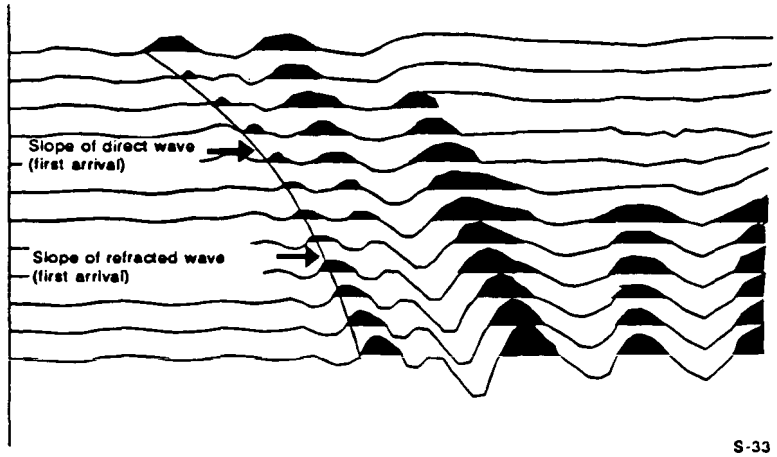
- Measures travel time of acoustic wave refracted along an interface
- Most commonly used at sites where bedrock is less than 500 ft below ground surface

S-31

SEISMOGRAPH FIELD LAYOUT SHOWING DIRECT AND REFRACTED WAVES



S-32



S-33

NOTES

SEISMIC REFRACTION Advantages

- Determine layer velocities
- Calculate estimates of depths to different rock or groundwater interfaces
- Obtain subsurface information between boreholes
- Determine depth to water table

S-34

SEISMIC REFRACTION Assumptions

- Velocities of layers increase with depth
- Velocity contrast between layers is sufficient to resolve interface
- Geometry of geophones in relation to refracting layers will permit detection of thin layers

S-35

SEISMIC REFRACTION Disadvantages

- Assumptions must be made
- Assumptions must be valid
- Data collection can be labor intensive

S-36

SEISMIC REFLECTION

- Measures travel time of acoustic wave reflected along an interface
- Precise depth determination cannot be made without other methods

S-37

SEISMIC REFLECTION (cont.)

- Magnitude of energy required is limiting factor
- Requires more complex data review

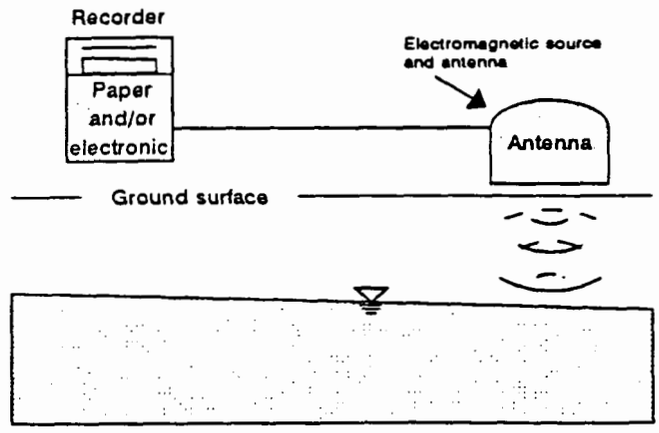
S-38

GROUND-PENETRATING RADAR

A transmitter emits pulses of high-frequency electromagnetic waves into the subsurface which are scattered back to the receiving antenna on the surface and recorded as a function of time

S-39

NOTES



GROUND-PENETRATING RADAR

S-40

GROUND-PENETRATING RADAR

- Depth penetration is severely limited by attenuation of electromagnetic waves into the ground

S-41

GROUND-PENETRATING RADAR (cont.)

- Attenuating factors
  - Shallow water table
  - Increase in clay content in the subsurface
  - Electrical resistivity less than 30 ohm-meters

S-42

GROUND-PENETRATING RADAR Advantages

- Continuous display of data
- High-resolution data under favorable site conditions
- Real-time site evaluation possible

S-43

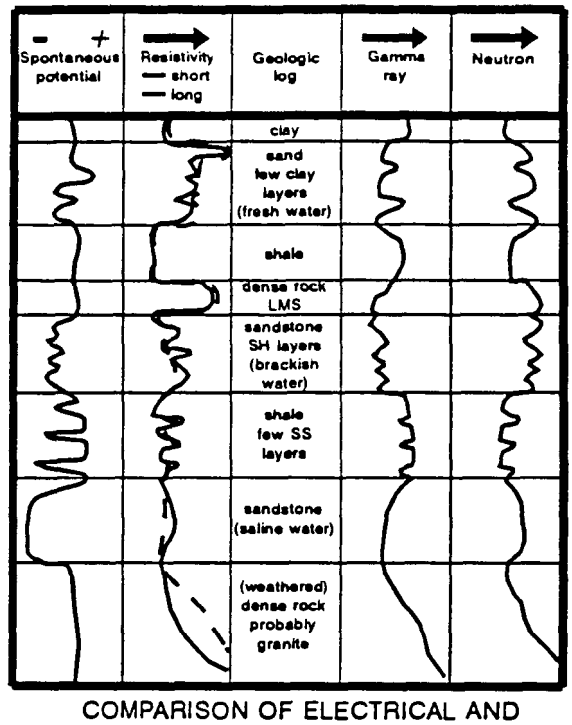
GROUND-PENETRATING RADAR Disadvantages

- Limitations of site-specific nature of technique
- Site preparation necessary for survey
- Quality of data can be degraded by cultural noise and uneven ground surface

S-44

Borehole Geophysics

S-45



COMPARISON OF ELECTRICAL AND RADIOACTIVE BOREHOLE LOGS

S-46

NOTES

NOTES

BOREHOLE GEOPHYSICS

- Spontaneous potential
- Normal resistivity
- Natural-gamma
- Gamma-gamma

S-47

BOREHOLE GEOPHYSICS (cont.)

- Neutron
- Caliper
- Acoustic
- Temperature

S-46

SPONTANEOUS POTENTIAL

- Records natural potential between borehole fluid and fluid in surrounding materials
- Can only be run in open, fluid-filled boreholes

S-49

NOTES

SPONTANEOUS POTENTIAL (cont)

Primary uses:

- Geologic correlation
- Determination of bed thickness
- Separation of nonporous rocks from porous rocks (i.e., shale-sandstone and shale-carbonate)

S-50

RESISTIVITY

- Measures apparent resistivity of a volume of rock or soil surrounding the borehole
- Radius of investigation is generally equal to the distance between the borehole current and measuring electrodes
- Can only be run in open, fluid-filled boreholes

S-51

GAMMA

- Measures the amount of natural-gamma radiation emitted by rocks or soils
- Primary use is identification of lithology and stratigraphic correlation
- Can be run in open or cased and fluid- or air-filled boreholes

S-52

GAMMA-GAMMA

- Measures the intensity of gamma radiation from a source in the probe after it is backscattered and attenuated in the rocks or soils surrounding the borehole

S-53

GAMMA-GAMMA (cont.)

- Primary use is identification of lithology and measurement of bulk density and porosity of rocks or soils
- Can be run in open or cased and fluid- or air-filled boreholes

S-54

NEUTRON

- Measures moisture content in the vadose zone and total porosity in sediments and rocks
- Neutron sources and detector are arranged in logging device so that output is mainly a function of water within the borehole walls

S-55

NOTES

NEUTRON (cont.)

- Can be run in open or cased and fluid- or air-filled boreholes

S-56

CALIPER

- Records borehole diameter and provides information on fracturing, bedding plane partings, or openings that may affect fluid transport
- Can be run in open or cased and fluid- or air-filled boreholes

S-57

ACOUSTIC

- A record of the transit time of an acoustic pulse emitted into the formation and received by the logging tool
- Response is indicative of porosity and fracturing in sediments or rocks
- Can be run in open or cased, fluid-filled boreholes

S-58

NOTES

TEMPERATURE

- A continuous record of the temperature of the environment immediately surrounding the borehole
- Information can be obtained on the source and movement of water and the thermal conductivity of rocks
- Can be run in open or cased, fluid-filled boreholes

S-59

GEOCHEMICAL MODELS

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. Evaluate the effect organic and inorganic contaminants have on groundwater chemistry

2. Identify chemical changes in groundwater from petroleum hydrocarbon contaminants

3. Identify chemical changes in groundwater from sewage and municipal landfill contaminants

4. Identify chemical changes in groundwater from acid, base, and ammonia spills and coal fly ash

5. Define the following chemical parameters:
   a. Hardness
   b. Alkalinity
   c. pH
   d. Eh

6. Describe how hardness, alkalinity, pH, and Eh affect water chemistry

7. Describe the effects of the carbonate buffering system on groundwater

STUDENT PERFORMANCE OBJECTIVES (cont.)

8. Define dense nonaqueous-phase liquids (DNAPLs) and light nonaqueous-phase liquids (LNAPLs)

9. Describe gas evolution in uncapped landfills.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

GEOCHEMICAL MODELS

S-1

PRIMARY DRINKING WATER STANDARDS

- Inorganics
- Microbiological
- Pesticides
- Volatile organic compounds
- Radioactivity

S-2

SECONDARY DRINKING WATER REGULATIONS

- Chloride
- Color
- Copper
- Corrosivity
- Fluoride
- Foaming agents
- Iron
- Manganese
- Odor
- pH
- Sulfate
- Total dissolved solids
- Zinc

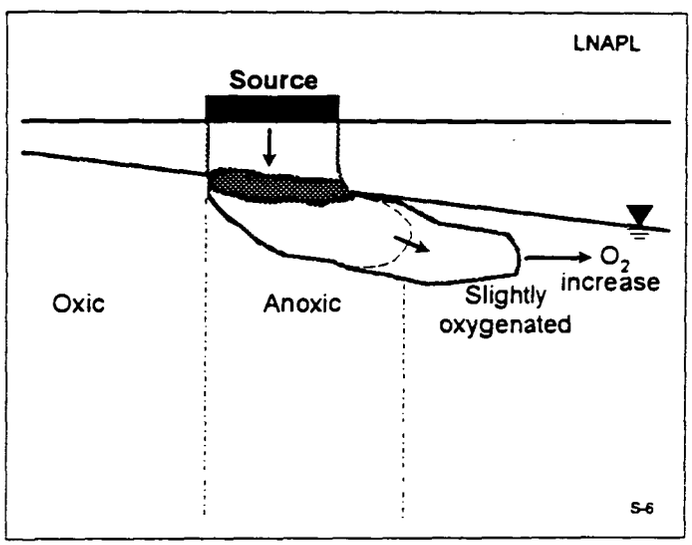
S-3

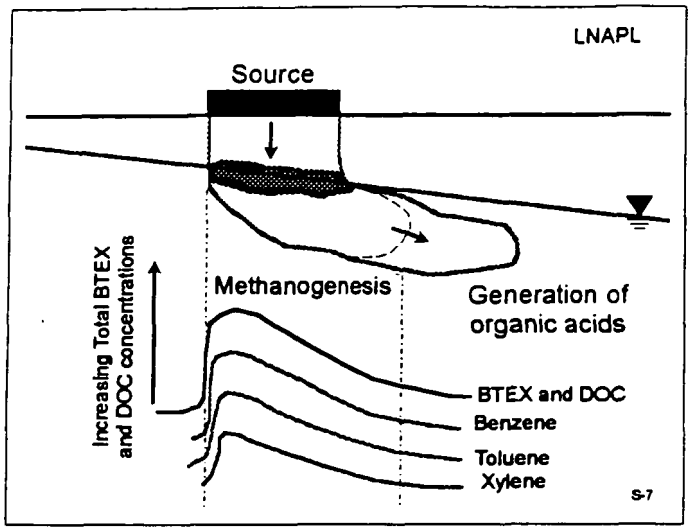
Anatomy of a Plume

S-4

Petroleum Contaminant

S-5



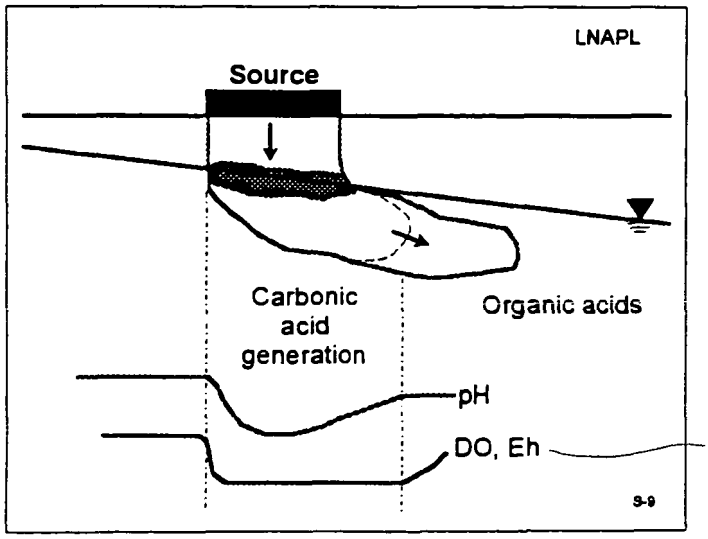


HYDROLYSIS OF ORGANIC CHEMICALS

$$C_6H_6 + O_2 + H_2O \longrightarrow HCO_3^- + \text{organic acids}$$

$$R - \overset{\overset{\displaystyle O}{\|}}{C} - OH$$

S-8



LNAPL

pH

$$pH = -\log [H^+]$$

S-10

LNAPL

CARBONATE BUFFERING SYSTEM

Organic chemicals $\longrightarrow CO_2$

$H_2O + CO_2 \rightleftharpoons H_2CO_3$
carbonic acid

$H_2CO_3 \rightleftharpoons H^+ + HCO_3^-$

$HCO_3^- \rightleftharpoons H^+ + CO_3^=$

Alkalinity is $HCO_3^- + CO_3^=$

S-11

Keep pH constant

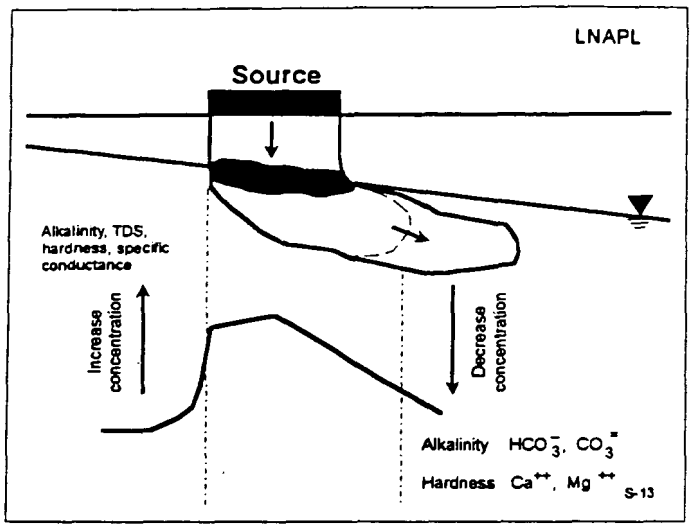
LNAPL

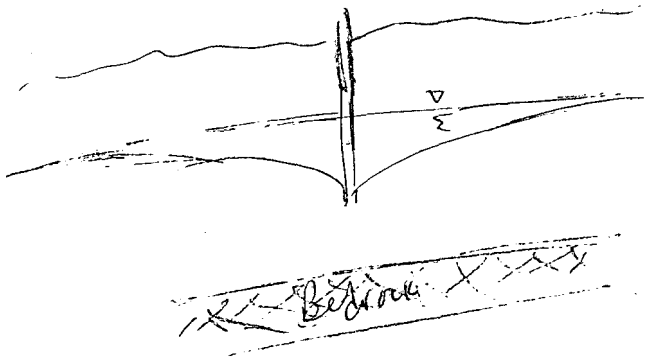
DISSOLUTION OF LIMESTONE

$(CaMg)CO_3 + H^+ \longrightarrow HCO_3^- + Ca^{++} + Mg^{++}$
Limestone

S-12

Increase
decrease





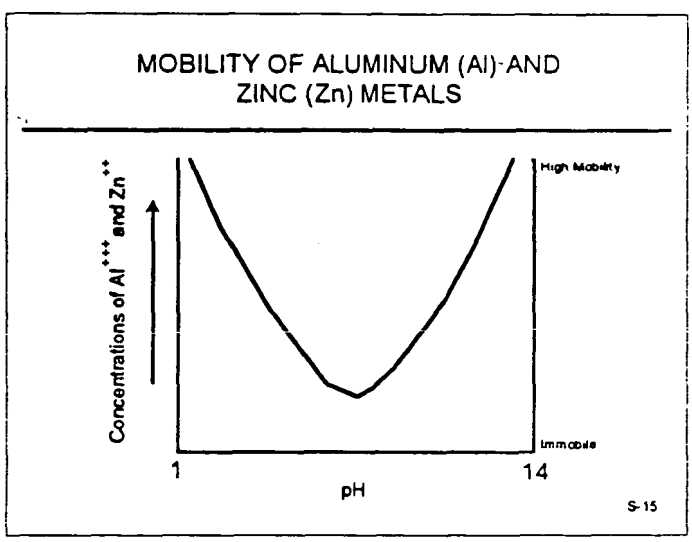
HARDNESS

Type	mg/L
Soft	0-60
Moderately hard	61-120
Hard	121-180
Very hard	>180

Calcium, Magnesium, WATER

S-14

chemical process
Chemical Process
Chemical process
chemical process

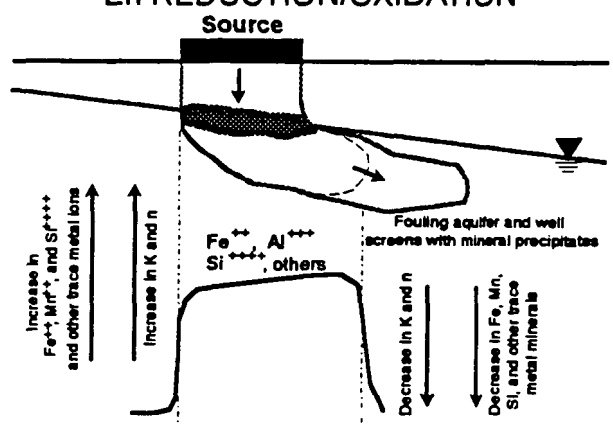


Redox Potential (Eh)

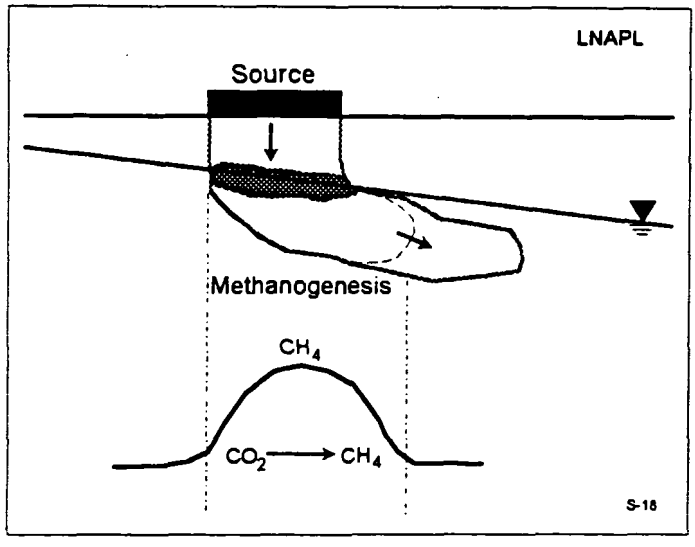
- Oxidizing or reducing environment
- High negative volts - reducing reactions
- High positive volts - oxidizing reactions
- Oxic vs. aerobic
- Anoxic vs. anaerobic

S-16

Eh REDUCTION/OXIDATION

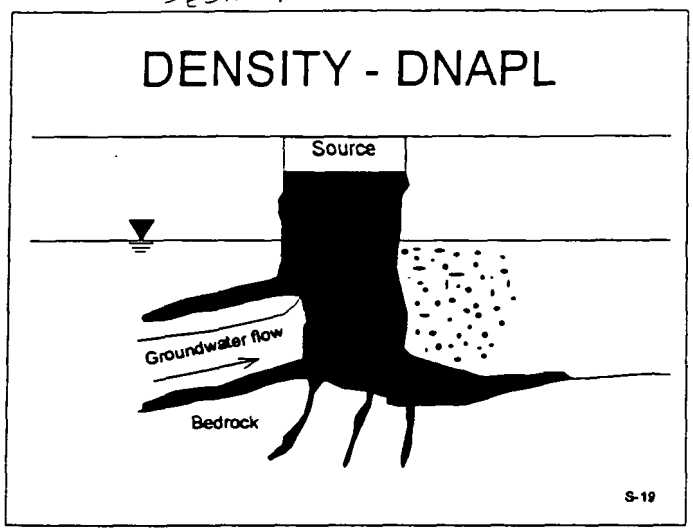


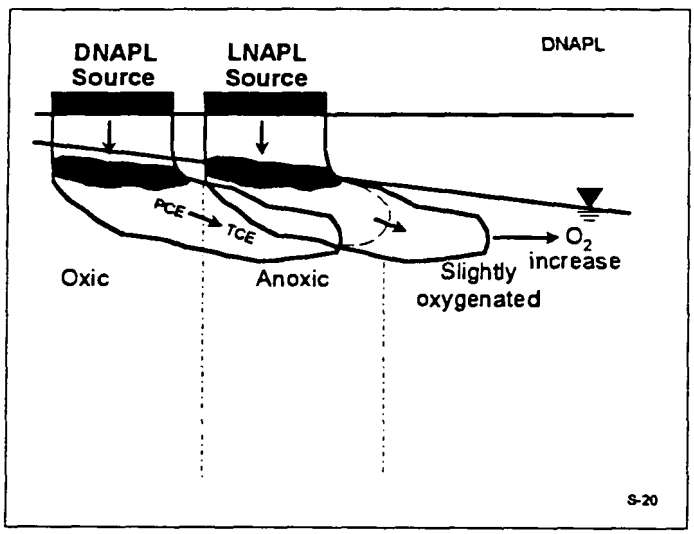
S-17



S-18

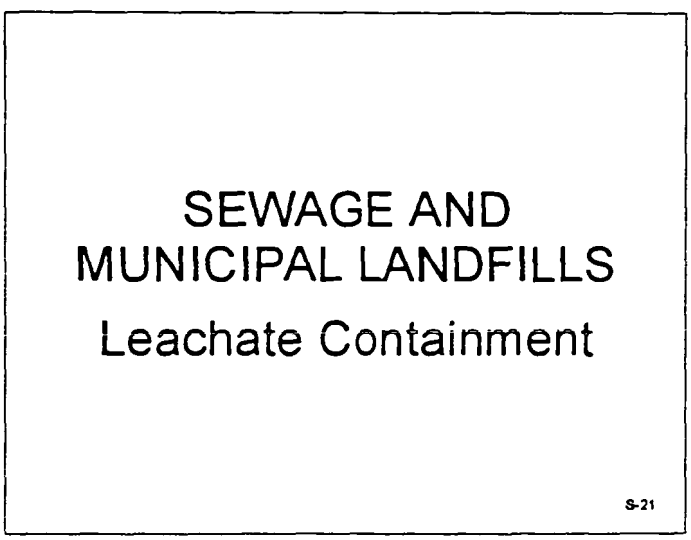
DESTINY

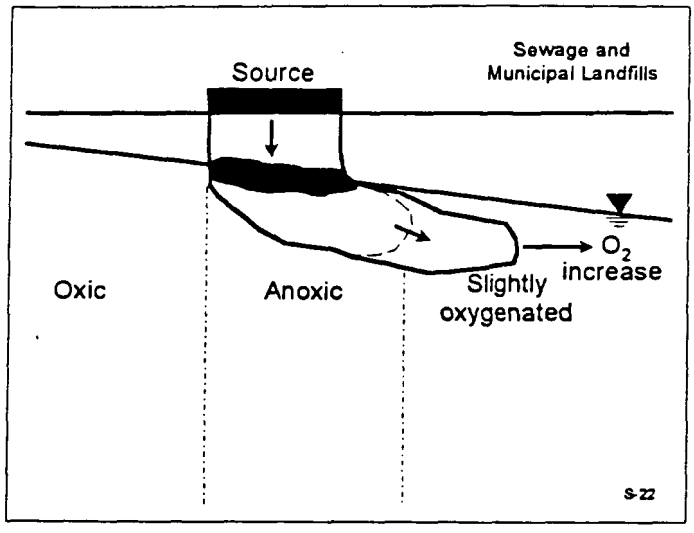


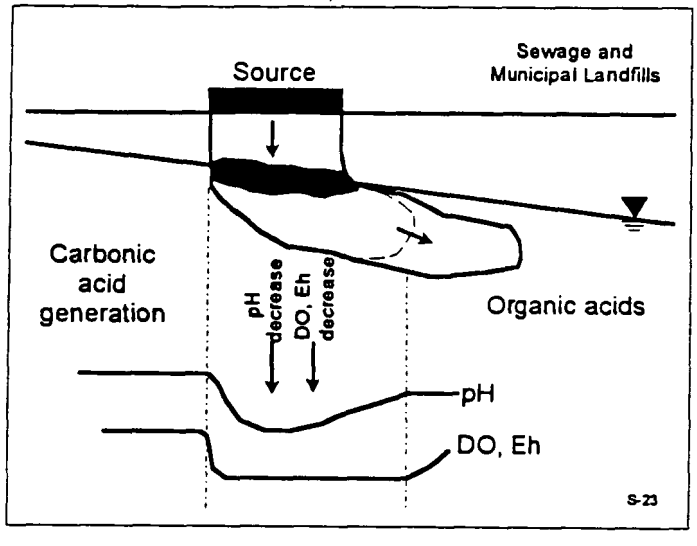


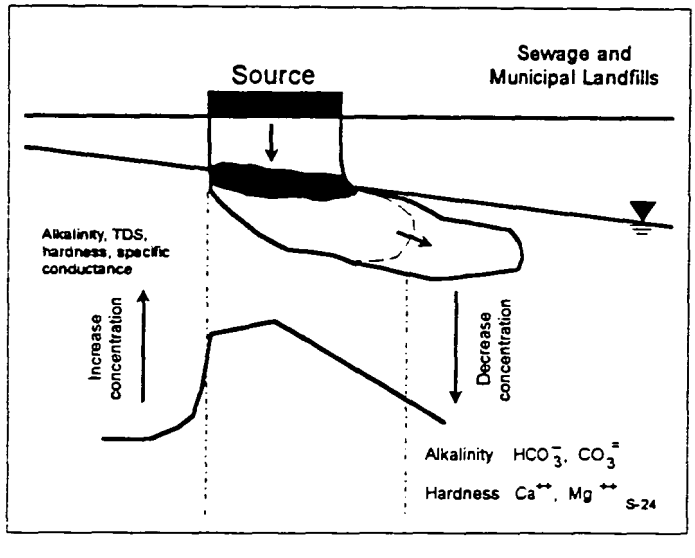
PHILADELPHIA

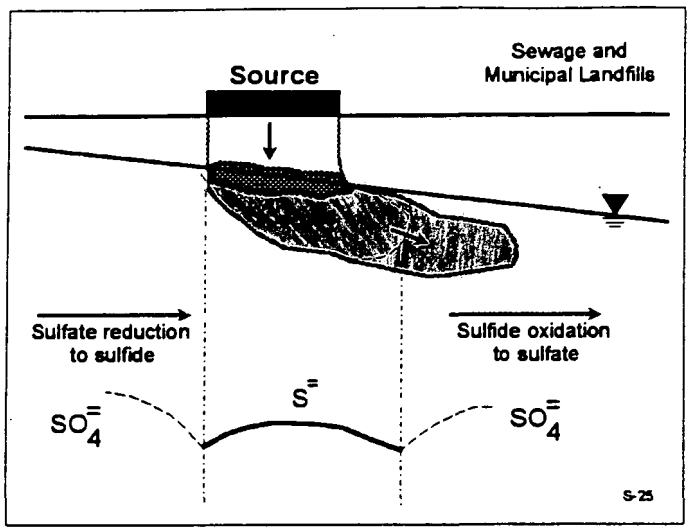
	AD	LAID
DELI	DIP	
LIP	IDEA	DIAL
PILL	IDLE	HIDE
HILL	ILL	PIE
DILL		HELP
HIP	LID	
HID	LIED	LAD
AIDE	LIE	LAP
PAL	LADEL	

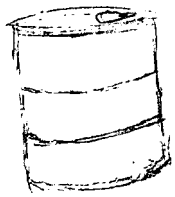



Source
Sewage and
Municipal Landfills
Oxic
Anoxic
Slightly
oxygenated
O_2
increase
S-22


Source
Sewage and
Municipal Landfills
Carbonic
acid
generation
pH
decrease
DO, Eh
decrease
Organic acids
pH
DO, Eh
S-23


Source
Sewage and
Municipal Landfills
Alkalinity, TDS,
hardness, specific
conductance
Increase
concentration
Decrease
concentration
Alkalinity HCO_3^-, $CO_3^=$
Hardness Ca^{++}, Mg^{++}
S-24





gypsum

Sewage and
Municipal Landfills

DISSOLUTION OF SHEET ROCK

$CaSO_4 \longrightarrow Ca^{++} + SO_4^=$

(sheet rock)

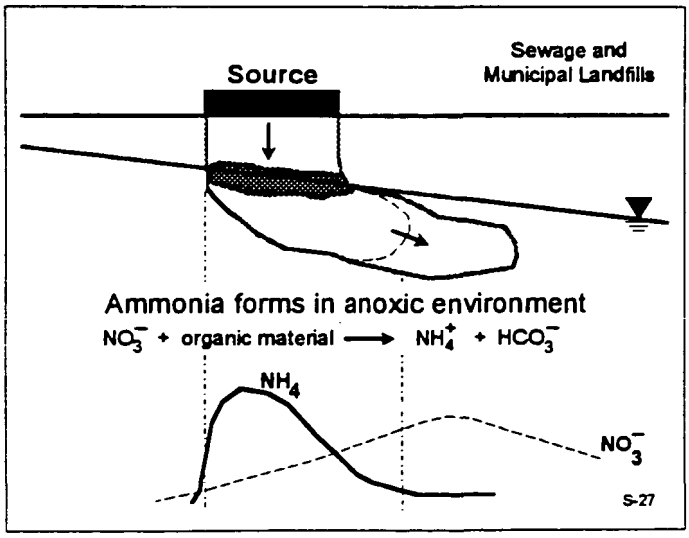
ION EXCHANGE

$Ca^{++}\downarrow \longrightarrow Na^+$ *("water softening")*

SULFATE REDUCTION

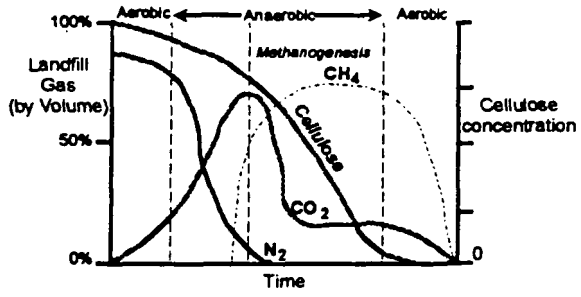
$SO_4^= \longrightarrow S^=$

S-26



Sewage and
Municipal Landfills

GEOCHEMICAL LIFE CYCLE OF A LANDFILL



S-28

Sewage and
Municipal Landfills

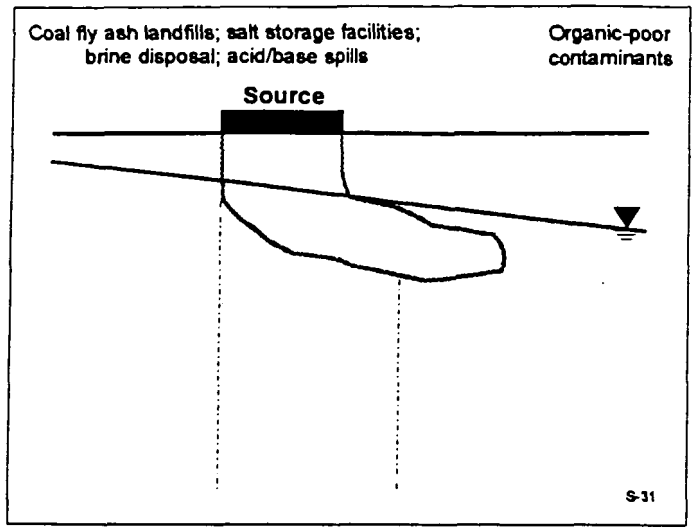
LANDFILL LEACHATE INDICATORS

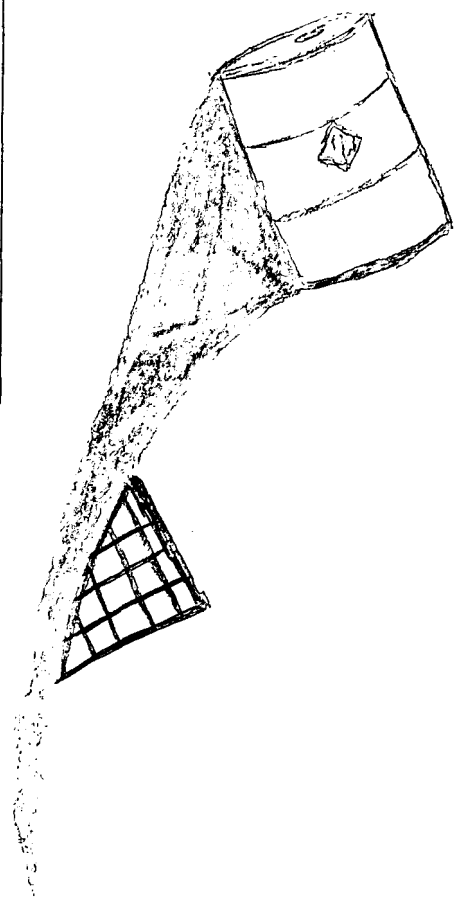
Excellent:	Good:
Ammonia (NH_4^+)	Low pH
DOC	High alkalinity (HCO_3^-)
Low Eh	Low sulfate
No O_2	Cl^-, Na^+, B^{+++}
High Fe^{++} and Mn^{++}	

S-29

Organic-Poor Contaminants

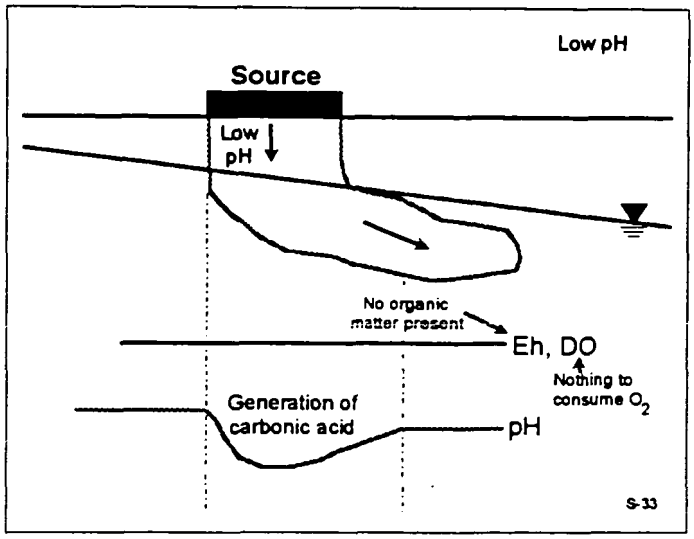
S-30

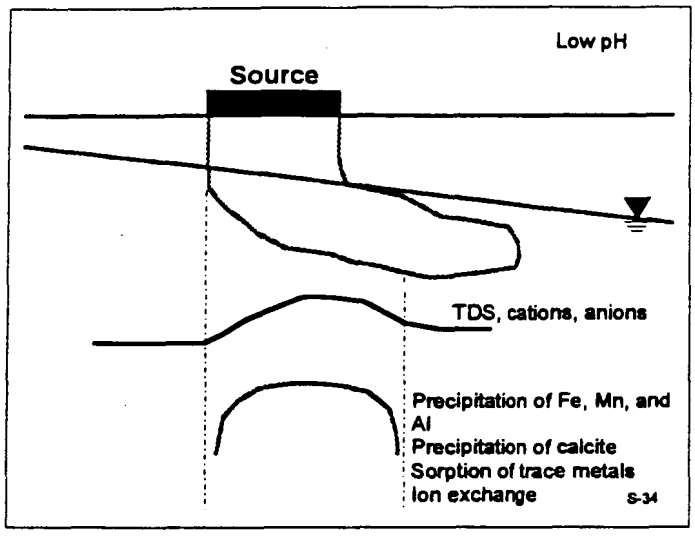




Low pH Contaminants

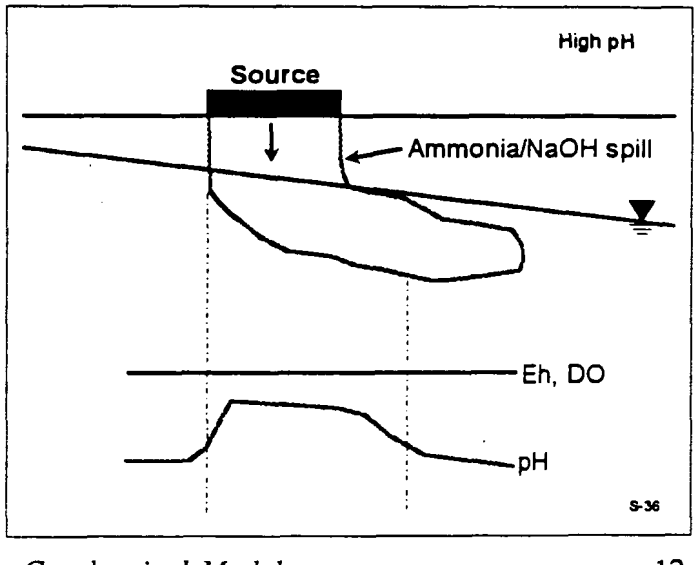
S-32

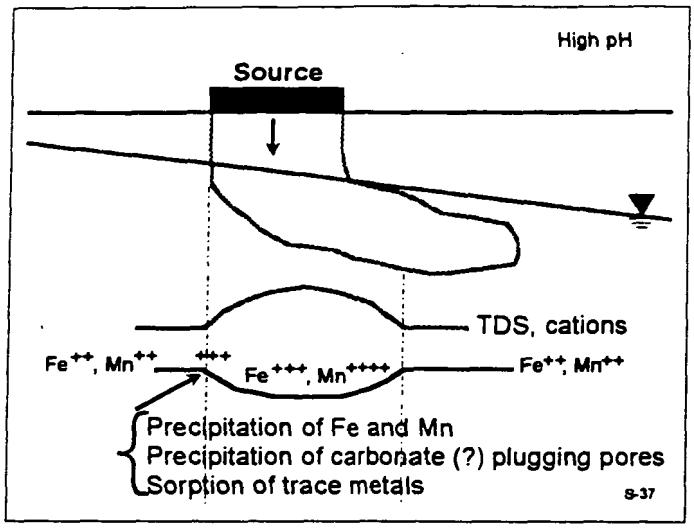




High pH Contaminants

S-35




High pH
Source
TDS, cations
Fe++, Mn++
Fe+++, Mn++++
Fe++, Mn++
Precipitation of Fe and Mn
Precipitation of carbonate (?) plugging pores
Sorption of trace metals
S-37

GROUNDWATER MODELS

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. List the physical processes that affect groundwater and contaminant flow

2. List the properties that are included in the retardation factor

3. List the parameters that are included in the basic equation used in groundwater computer programs

4. List the variables that groundwater models can be used to predict.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

GROUNDWATER MODELS

S-1

GROUNDWATER MODELS

An attempt to simulate groundwater flow conditions mathematically

- Used to predict groundwater levels (heads) over time
- Used to predict contaminant transport

S-2

PHYSICAL PROCESSES

- Advection
- Dispersion
- Density
- Immiscible phase
- Fractured media

S-3

NOTES

① $K = \frac{Q}{IA}$
capacity of porous medium to transmit water.

② POROSITY - N_T

P5-5 $N_T = \frac{V_v}{V_T} = S_y + S_r$

ADVECTION

- Average groundwater velocity
- Depends on:
  - Hydraulic conductivity (K)
  - Porosity (N_T)
  - Hydraulic gradient (I)

water stored

③ $I = \frac{dh}{dl}$.

S-4

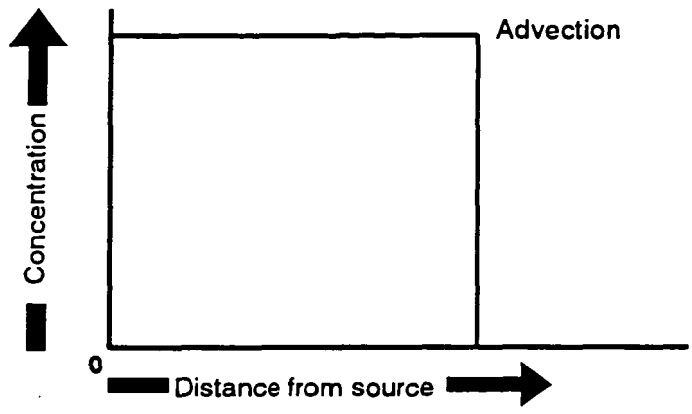
Q	=	Rate of flow
K	=	Hydraulic conductivity
A	=	Cross-sectional area of flow
I	=	Hydraulic gradient
n_e	=	Effective porosity

S-5

Q	=	KAI	Darcy's Law
Q	=	Av	Velocity equation
v	=	KI	Darcian velocity
v_s	=	$\bar{v} = \frac{KI}{n_e}$	Advective velocity*

* Seepage velocity or average linear velocity

S-6



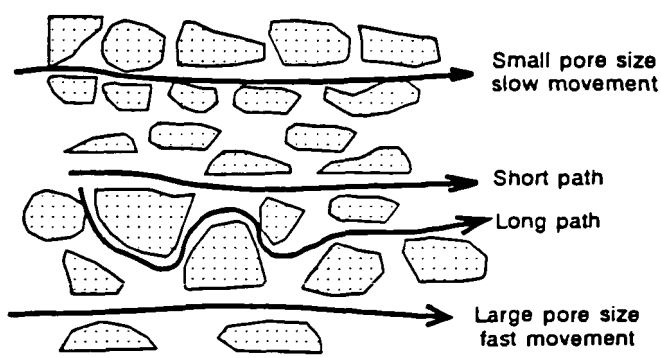
S-7

DISPERSION

- Tendency for solute to spread
- Caused by:
  - Mechanical mixing
  - Molecular diffusion

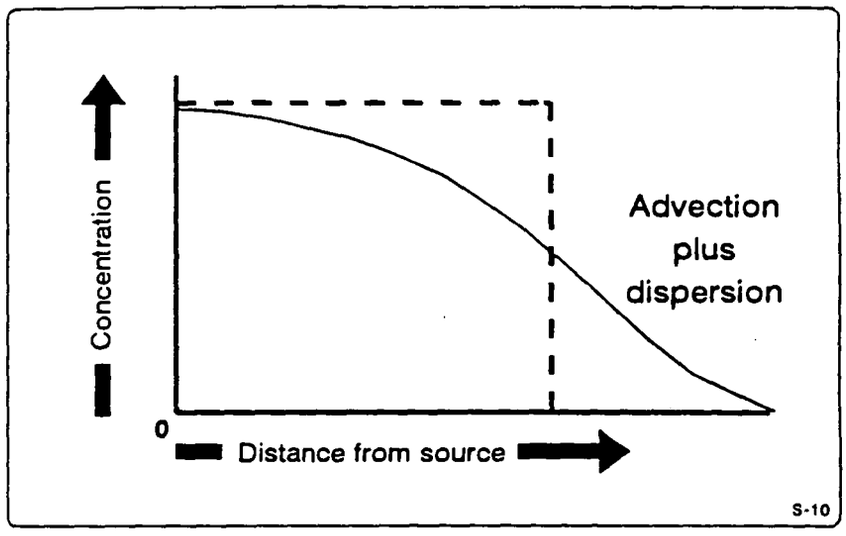
S-8

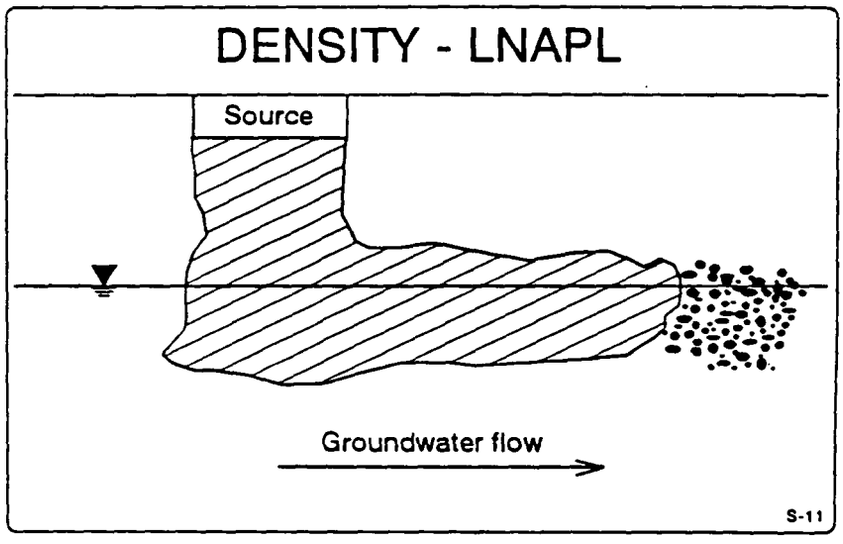
PATH LENGTH AND PORE SIZE AS FACTORS IN CONTAMINANT TRANSPORT

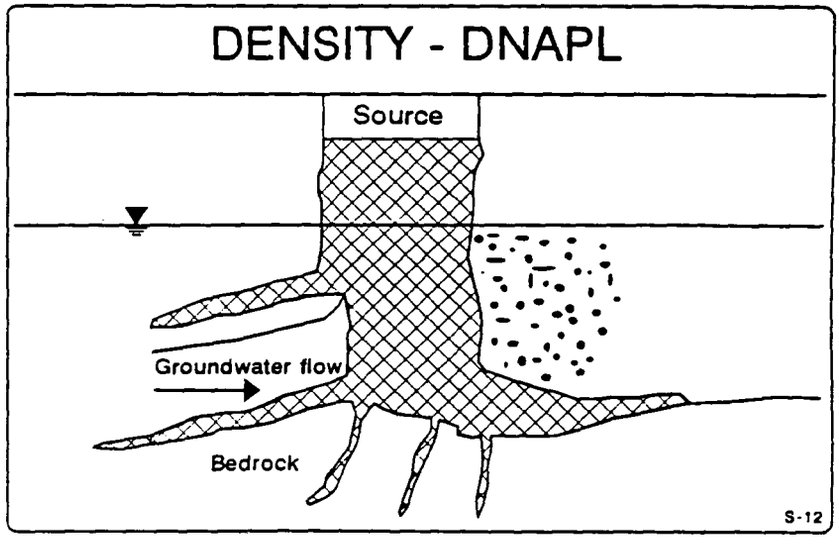


S-9

NOTES


Concentration
0
Distance from source
Advection
plus
dispersion
S-10


DENSITY - LNAPL
Source
Groundwater flow
S-11

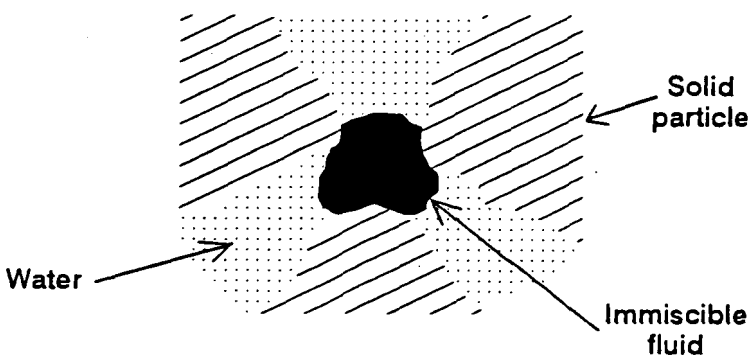

DENSITY - DNAPL
Source
Groundwater flow
Bedrock
S-12

IMMISCIBLE PHASE FLOW

- Mutually insoluble liquids
- Interferes with groundwater flow
- Liquids can become immobile at residual saturation

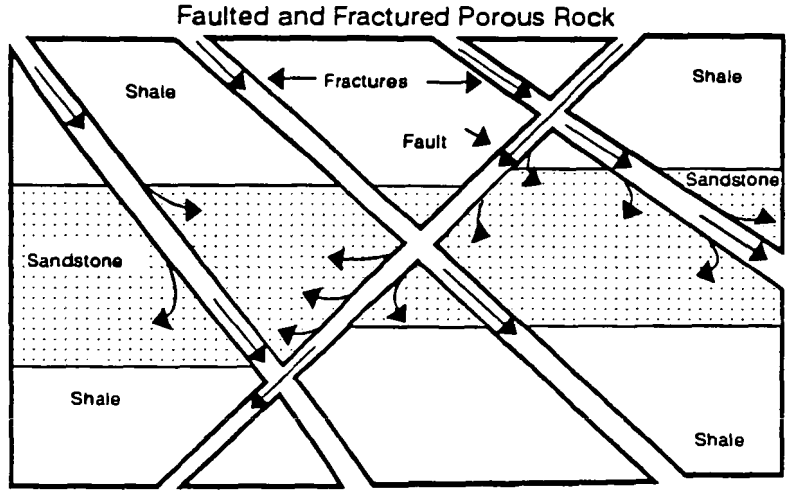
S-13

IMMISCIBLE PHASE FLOW



S-14

Faulted and Fractured Porous Rock



S-15

NOTES

CHEMICAL PROCESSES

mostly w/ clays — • Sorption

hydrate — • Hydrolysis

• Cosolvation — mixing (?)

• Ionization

loss of electron

S-16

CHEMICAL PROCESSES (cont.)

- Dissolution and precipitation
- Complexation reactions
- Redox potential

oxidation / reduction

S-17

BIOLOGICAL PROCESSES

- Microorganisms
  - Bacteria
  - Fungi
- Transformation of contaminants
  - Aerobic conditions
  - Anaerobic conditions

S-18

RETARDATION FACTOR

- Relates groundwater velocity to contaminant velocity
- Current practice: lump chemical and biological processes into retardation

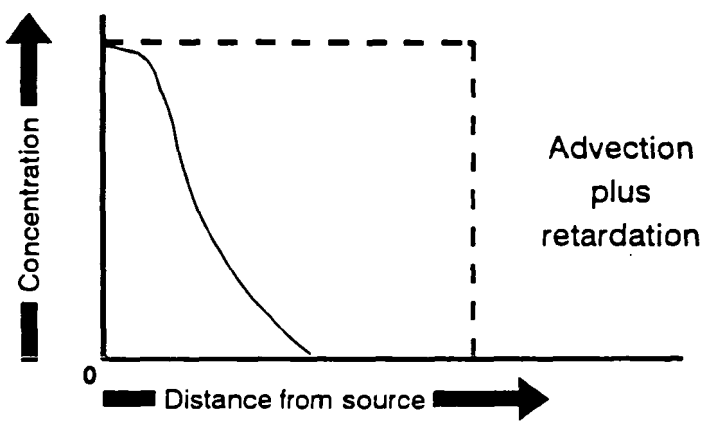
S-19

RETARDATION

$$R = 1 + \frac{\rho_b}{N_T} \times K_d$$

R = retardation factor
ρ_b = bulk density
K_d = distribution coefficient
N_T = total porosity

S-20



S-21

NOTES

Groundwater Modeling

S-22

CONCENTRATION AT DISTANCE "L"

$$C = \frac{C_o}{2}\left[\text{erfc}\left(\frac{L - vt}{2\sqrt{D_L t}}\right) + \exp\left(\frac{vL}{D_L}\right)\text{erfc}\left(\frac{L + vt}{2\sqrt{D_L t}}\right)\right]$$

D_L = longitudinal dispersion coefficient

C_o = solute concentration at source

v = average linear velocity

L = distance

t = time

erfc = complementary error function

S-23

MODELS CAN PREDICT:

- Spatial variation
- Temporal variation
- Parameter variation

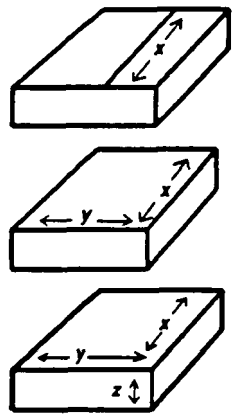
S-24

MODEL DIMENSIONS

One-dimensional

Two-dimensional

Three-dimensional



S-25

MODELING PROBLEMS

- Lack of appropriate modeling protocols and standards
- Insufficient technical support
- Inadequate education and training
- Widely used, but selection and use inconsistent

S-26

KEYS TO SUCCESSFUL USE OF MODELS

- Proper input of data and parameter estimates
- Effective communication
- Understanding the limitations of the model

S-27

NOTES

G.I.G.O.

Garbage in = Garbage out

The first axiom of computer usage

S-28

MOST COMMON EPA MODELS

Name	Relative Use
MODFLOW	29
HELP	24
RANDOM WALK	21
USGS-2D	20
USGS-MOC	19

S-29

PROBLEM 1

Cross-Section Exercise

PROBLEM 1: CROSS-SECTION EXERCISE

A. Student Performance Objectives

1. Use a topographic map to locate sites for the installation of monitoring wells at specific elevations.

2. Draw a topographic profile of a specified area.

3. Calculate a vertical exaggeration for a topographic profile.

4. Obtain geological information from monitoring well logs.

5. Use the GSA Munsell color chart and geotechnical gauge to identify rock sample colors and textures.

6. Given a geologic map, interpret elevations of geologic formations.

7. Draw a geologic cross-section using monitoring well logs and a topographic profile.

8. Interpret subsurface geology to locate aquifers of concern, identify discontinuities in geologic formations, and locate potential monitoring/remediation wells.

B. Background Information

Each group of students will have a set of six rock/sediment samples, labeled A through F, to examine. These samples represent rock/sediment samples from six of the seven different geologic formations encountered during the installation of monitoring wells at, and in the vicinity of, the Colbert Landfill site in Spokane, Washington. During the site investigation, these samples were collected from cuttings generated by mud rotary drilling. Each sample tube is also oriented with an arrow that indicates the top. **DO NOT attempt to remove the orange caps and open the tubes!**

C. Geologic Cross-Section

1. Using the GSA Munsell color chart and sample mask, match the overall color of the rocks, sand, clay, or gravel within the samples to the color chart. **Do not** determine every color if a sample is multicolored, but look for key sediment types or specific marker colors.

2. Using the geotechnical gauge, generally determine and match the grain size of the sediments with the written descriptions. For example, actual fine sand or coarse sand sizes can be found on the chart. Sediments larger than coarse sand, such as gravel and cobbles, are **NOT** shown on the geotechnical card. Using the

geotechnical gauge and the sediment characteristic diagram depicted in **Figure 1**, generally determine the degree of particle rounding and sediment sorting. Well sorted means most particles are of similar size and shape, whereas poorly sorted particles are of no particular size and vary greatly in size and shape, such as sand mixed with gravel or cobbles.

3. Using the **SAMPLES** and well log together, match these descriptions and your visual observations to the official published U.S. Geological Survey geologic description of the formations. Then identify each formation on the well logs in the space provided under the "**STRATA**" column; for example, Kiat, sample F. **START WITH WELL LOG #6 AND PROCEED TO LOG #1. EACH TUBE REPRESENTS ONLY ONE ROCK FORMATION!** Be sure to read the information written under the "REMARKS" column at the right of the log sheet for additional sample information. Your instructor will discuss the correct sample identification at the end of this portion of the exercise.

4. Using the appropriate topographic maps and graph paper provided, locate Wells 6 through 1 along the top of the graph paper from left to right along profile line A-A'. Determine the respective elevations of the wells (your instructor will demonstrate this technique).

5. Label the Y-axis of the graph paper to represent the elevation, starting from 2,100 feet at the top to 1,400 feet at the bottom. Each box on the graph represents **20 feet** in elevation.

6. Plot the location, depicting the correct surface elevation of each well on the graph. Also determine and plot the elevations of **several easily determined points** on the profile line between each of the wells in order to add more detail to the profile. This will generate a series of dots representing the elevations of the six wells and the other elevations you have determined. Make sure to select contour lines that cross the profile line. The contour interval of these particular topographic maps is 20 feet.

7. After plotting these elevations on the graph, connect them with a **smooth curve**, which will represent the shape of the topography from A-A'.

8. Using the well logs previously completed and the colored geologic map, add the existing geology and formation thickness to each well location. Each formation thickness is listed on the left side of the well log and is measured from the bottom of the next overlying formation.

9. Sketch in and interpret the geologic layers of the cross section, starting with the lowest bedrock formation. Connect all of the same geologic formations, keeping in mind that some formations have varying thicknesses and areal extent.

10. Using available groundwater information, locate the three aquifers in the cross section.

11. Using the completed cross section, locate potential sites for the installation of additional monitoring wells or remediation wells and identify formation discontinuities.

12. Compare your interpretation with the "suggested" interpretation handed out by the instructor.

D. History of Colbert Landfill

The Colbert Landfill is located 2.5 miles north of the town of Colbert near Spokane, Washington, and is owned by the Spokane County Utilities Department. This 40-acre landfill was operated from 1968 to 1986, when it was filled to capacity and closed. It received both municipal and commercial wastes from many sources. From 1975 to 1980, a local electronics manufacturing company disposed spent solvents containing methylene chloride (MC) and 1,1,1-trichloroethane (TCA) into the landfill. A local Air Force base also disposed of solvents containing acetone and methyl ethyl ketone (MEK). These solvents were trucked to the landfill in 55-gallon drums and poured down the sides of open and unlined trenches within the landfill. Approximately 300–400 gallons/month of MC and 150–200 gallons/month of TCA were disposed. In addition, an unknown volume of pesticides and tar refinery residues from other sources were dumped into these trenches.

The original site investigation was prompted by complaints from local residents who reported TCA contamination of their private wells. The population within 3 miles of the site is 1,500. In 1981, a Phase 1 investigation was conducted; a Phase 2 was completed in 1982. Groundwater samples collected from nearby private wells indicated TCA contamination at 5,600 μg/L, MC contamination at 2,500 μg/L, and acetone at a concentration of 445 μg/L. Investigation reports concluded that drinking groundwater posed the most significant risk to public health. EPA placed the site on the National Priority List (NPL) in 1983. Bottled water and a connection to the main municipal water system was supplied to residents with high TCA contamination (above the MCL), and the cost was underwritten by the potentially responsible parties (PRPs) involved.

Hydrogeological Investigation

The site lies within the drainage basin of the Little Spokane River, and residents with private wells live on all sides of the landfill. The surficial cover and subsequent lower strata in the vicinity of the site consist of glacially derived sediments of gravel and sand, below which lie layers of clay, basaltic lava flows, and granitic bedrock. Beneath the site there are three aquifers and three aquitards. The stratigraphic sequence beneath the landfill from the top (youngest) to the bottom (oldest) is:

Qfg Upper sand and gravel glacial outwash and Missoula flood deposits which together form a water table aquifer

Qglf Upper layers of glacial Lake Columbia deposits of impermeable silt and clay that serve as an aquitard; lower layers of older glaciofluvial and alluvial sand and gravel deposits that form a confined aquifer

Mvwp Impermeable but weathered Wanapum basalt flow
Mcl Impermeable and unweathered Latah Formation of silt and clay
Kiat Fractured and unfractured granitic bedrock that serves as another confined aquifer

In the upper aquifer (Qfg), which is 8–15 feet thick, groundwater flows from 4 to 13 feet per day (ft/day). The lower confined sand and gravel aquifer (Qglf) varies from a few feet thick to 150 feet thick and is hydraulically connected to the Little Spokane River. Groundwater in this aquifer flows from 2 to 12 ft/day. To the northeast of the landfill, the upper aquifer is connected to the lower aquifer. Both of these aquifers are classified as current sources of drinking water according to EPA and are used locally for potable water. The area impacted by the site includes 6,800 acres and the contamination plume extends 5 miles toward the town of Colbert. Of the contaminants present, 90 percent occur as dense, nonaqueous-phase liquids (DNAPLs) at the bottom of the upper aquifer, and natural DNAPL degradation is slow. It has been estimated that only 10 percent of the solvents have gone into solution, whereas the remainder occurs in pore spaces and as pools of pure product above impermeable layers. The TCA plume in the upper aquifer has extended 9,000 feet in 8–10 years and it moves at a rate of 2–3 ft/day. The flow rate of the contamination plume in the lower sand and gravel aquifer (Qglf) has not been calculated because of the complexity and variability of the subsurface geology. However, TCA and MC have the highest concentrations in the lower sand and gravel aquifer.

Sediment Characteristics

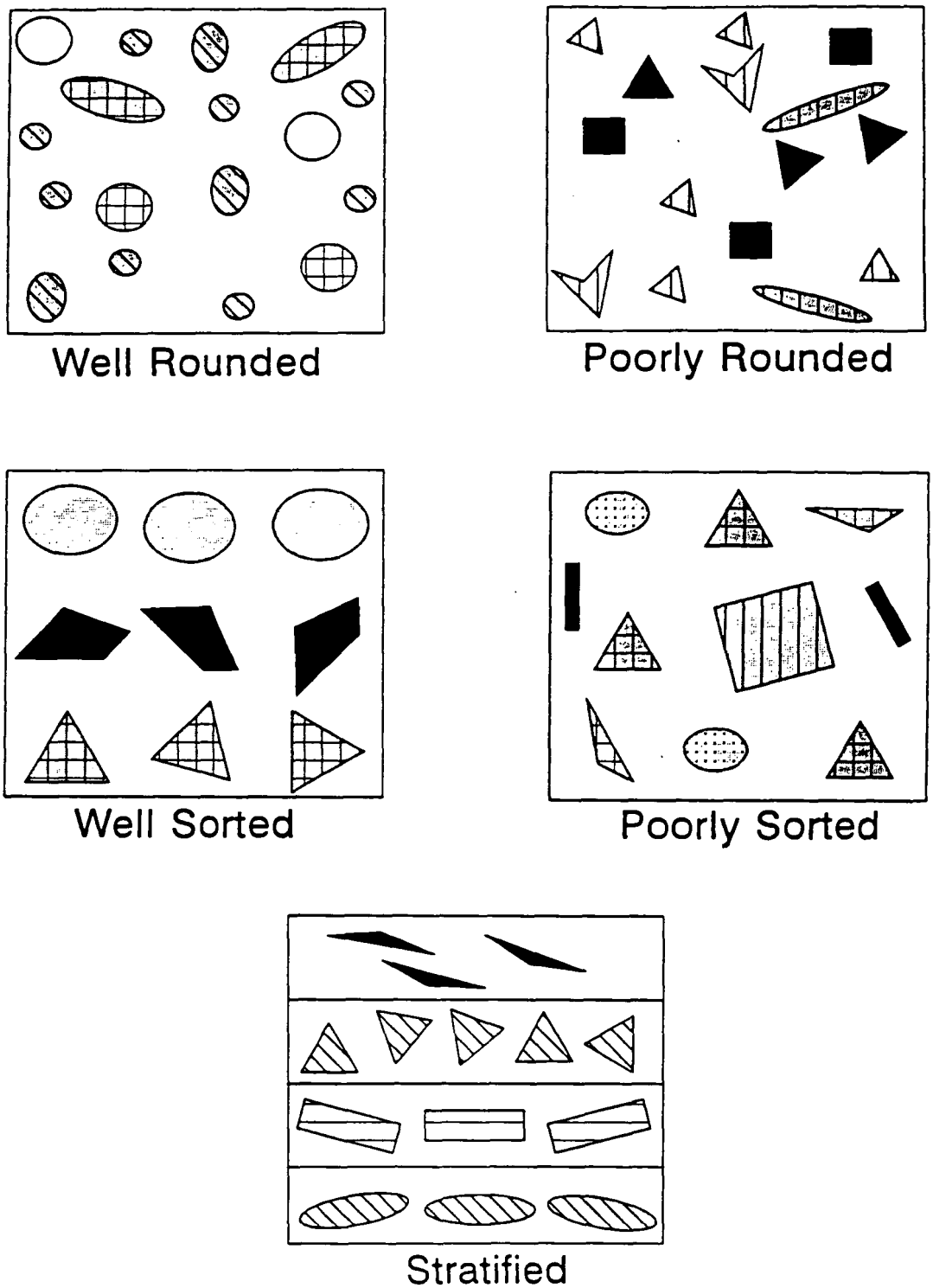
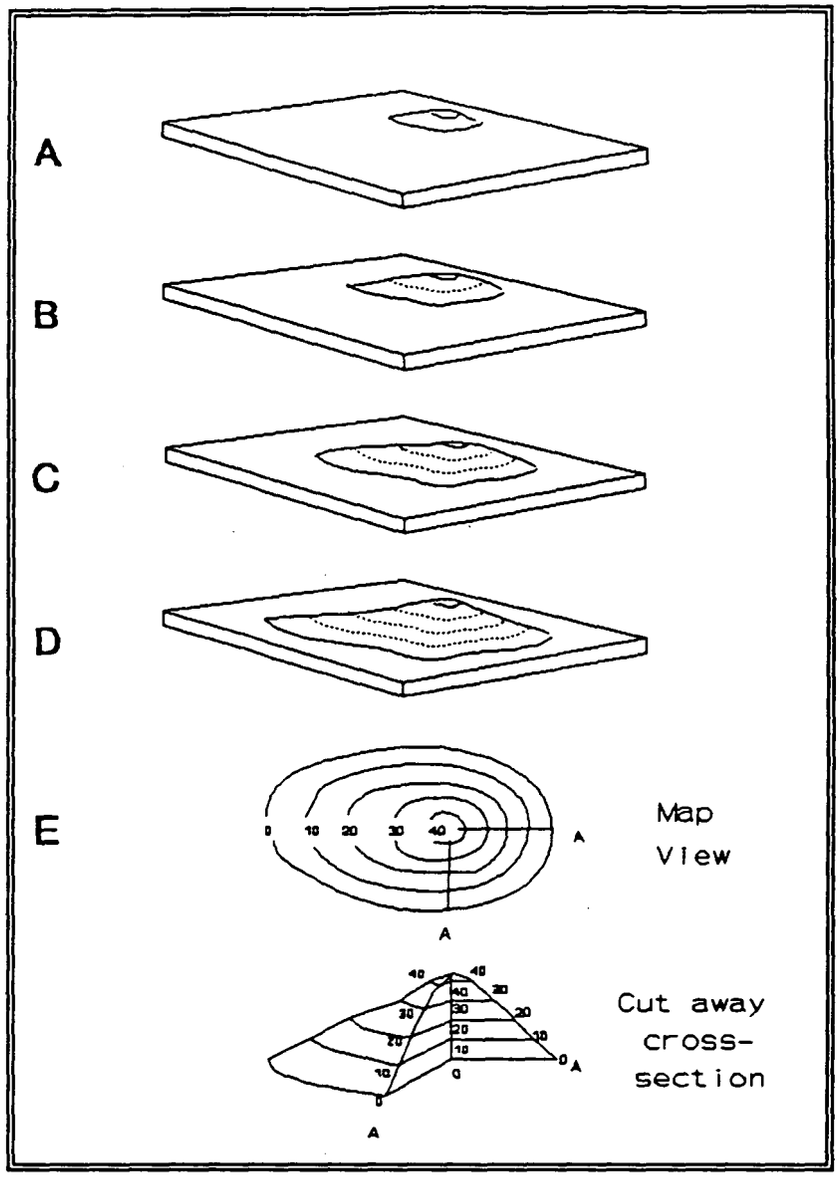


FIGURE 1



DEVELOPMENT OF CONTOUR LINES

Consider an island in a lake and the patterns made on it when the water level recedes. The shoreline represents the same elevation all around the island and is thus a contour line (see above **Figure, part A**). Suppose that the water levels of the lake drop 10 ft and that the position of the former shoreline is marked by a gravel beach (**Figure B**). Now there are two contour lines, the new lake level and the old stranded beach, each depicting accurately the shape of the new island at these two elevations. If the water level should continue to drop in increments of 10 ft, with each shoreline being marked by a beach, additional contour lines would be formed (**Figures C** and **D**). A map of the raised beaches is therefore a contour map (**Figure E**), which graphically represents the configuration of the island.

PROBLEM 2

Sediment Analysis

PROBLEM 2: SEDIMENT ANALYSIS

A. Student Performance Objectives

1. Determine the grain size distribution of unconsolidated geologic materials obtained from an aquifer.

2. Calculate a uniformity coefficient from the data obtained through sample sieving.

3. Given a formation's sieve analysis, select filter pack and screen slot size.

B. Background Information

1. Keck Field-Sieving Kit

The Keck Field-Sieving kit will be used to provide information on the grain size distribution of the unconsolidated sediments in the aquifer to be screened by the monitoring well. It will also be used to determine the correct size of filter pack material around the screen of a monitoring well, as well as determine the screen size. Sieving is only done using a dry mixture of unconsolidated sediments such as gravel, sand, silt, and clay. During the sieving, grain size ranges are retained by each sieve. The coarsest materials are retained by the top sieve, whereas the finest are collected by the bottom pan. The amount of sediment retained by each sieve is usually determined by weighing each fraction on a balance. However, with the Keck Field-Sieving Kit, this information is gathered by comparing the volume within each cylinder to the vertical percent scale along the edge of the Keck sieve holder. By initially using a sample volume that equals one full cylinder (100 percent), the percent "retained" by each screen after 5 minutes of sieving can be easily obtained. The "cumulative" percent of sand from each cylinder is calculated and plotted on special graph paper. The Y or vertical axis on the left side of the graph will represent the percent sample retained from 0 to 100 percent (the right side of the graph measures cumulative percent sediment passing), and the X or horizontal axis (along the bottom) will represent the grain size as measured in either thousandths of an inch or in U.S. standard sieve sizes (grain size in millimeters is measured along the top of the graph). Once the data points are plotted, they are connected with a smooth curve.

2. Particle Size Distributions

Because the sample is usually a mixture of sediment types, there is no single way to describe the range of particle sizes. The Wentworth Scale was developed in 1922 to classify particle size from boulders to clay. The Unified Soil Classification System was adopted by the U.S. Department of Agriculture as an extension of the Wentworth Scale to further classify fine-grained material. The particle size distribution can also be used to determine the size of the filter pack material to be used around the well screen. This material is mainly used with fine-grained sediments to make the area

around the screen more permeable, while also increasing the hydraulic diameter of the well. The grain size distribution of this material is selected such that 90 percent of it is retained by the screen slot opening. This allows the well to produce mostly sand-free water. Finally, the slope of the curve can also be used to determine the uniformity of the grain size by calculating the uniformity coefficient.

3. Uniformity Coefficient

The uniformity coefficient (UC) is calculated by dividing the 40 percent retained size of the sediment by the 90 percent retained size. For example, 40 percent of the sample was retained by 0.026 inches, while 90 percent was retained by 0.009 inches.

$$\frac{40\%\ retained}{90\%\ retained} = \frac{0.026\ in.}{0.009\ in.} = 2.9$$

The lower the value, the more uniform the particle size grading; the larger the value, the less uniform the grading. Values for UC should be less than 5.

C. Determine the Grain Size Distribution of Unconsolidated Geologic Materials Obtained from an Aquifer

1. Sieve a sample.
   a. Get a prepared Keck Field-Sieving kit from the instructors.
   b. Remove the cylinder stack from the frame by holding the frame at the **TOP** and unscrewing the knob counterclockwise.
   c. Fill the beaker with sand to the 100-ml line. This will equal one cylinder volume (100 percent).
   d. Remove the clear cap from the top cylinder and carefully pour approximately one-half of the sample from the beaker into it. Make sure the box top is beneath the cylinder to catch any spilled sand.
   e. Replace the top cap and carefully replace the cylinder stack into the frame.
   f. Slowly tighten the cap by turning the top knob clockwise.
   g. Hold the frame by **BOTH** ends and shake in a circular manner. Add an occasional vertical shake during this process.
   h. Shake for 5 minutes.
   i. **CAREFULLY** remove the cylinders from the holder, add the remaining sand sample, replace the cylinders into the frame, and continue shaking for another 5 minutes.
   j. Tap the cylinders with your fingers until the majority of the sample lies roughly flat within each cylinder.
   k. Using the vertical scale on the side of the frame, visually determine the percent sample fraction within each cylinder and record the data on the sheets provided.
   l. Carefully remove the cylinders from the frame, invert the stack, and replace the sand into the bag.
   m. Clean out each cylinder by tapping it against your hand. **DO NOT TAP**

AGAINST THE DESK OR ANY HARD SURFACE! Use the paintbrush to remove any remaining sand from the screens and gaskets.

n. Replace sieve set into box.

o. Calculate the cumulative percent of each cylinder and record the data on the data sheets.

p. Using the graph paper provided, plot your data.

2. Determine the grain size distribution of your sediment sample using your data plots and the unified soil classification scale on the bottom of the graph paper.

3. Calculate the uniformity coefficient for your sample.

Bottle #

Sediment Sieve Exercise

U.S. Sieve #	Percent Retained	Cumulative Percent

Uniformity Coefficient: UC = 40%/90%

SELECTION OF FILTER PACK AND WELL SCREEN

S-1

PURPOSE OF FILTER PACK

- Allow groundwater to flow freely into well
- Minimize or eliminate entrance of fine-grained materials

S-2

WELL SCREEN

Surrounded by:

- Filter pack coarser than the aquifer material
- Filter pack of uniform grain size
- Filter pack of higher permeability than the aquifer material

S-3

NOTES

UNIFORMITY COEFFICIENT (UC)

- Measure of the grading uniformity of sediment
- 40% retained size divided by 90% retained size
- UC of filter pack material should not exceed 2.5

S-4

FILTER PACK SELECTION

- Select by multiplying the 70% retained grain size of the aquifer materials by 4 or 6
- Use 4 if aquifer is fine grained and uniform
- Use 6 if aquifer is coarse grained and nonuniform

S-5

WELL SCREEN SELECTION

- Select screen slot opening to retain 90% of filter pack material

S-6

PROBLEM 3

Groundwater Model Demonstration

PROBLEM 4

Hydrogeological Exercises

PROBLEM 4: HYDROGEOLOGICAL EXERCISES

PART 1.

A. General Discussion

Groundwater-level data can be used to determine direction of groundwater flow by constructing groundwater contour maps and flow nets. To calculate a flow direction, at least three observation points are needed. First, relate the groundwater field levels to a common datum—map datum is usually best—and then accurately plot their position on a scale plan, as in **Figure 1**. Second, draw a pencil line between each of the observation points, and divide each line into a number of short, equal lengths in proportion to the difference in elevation at each end of the line. The third step is to join points of equal height on each of the lines to form contour lines (lines of equal head). Select a contour interval that is appropriate to the overall variation in water levels in the study area. The direction of groundwater flow is at right angles to the contour lines from points of higher head to points of lower head.

This simple procedure can be applied to a much larger number of water-level values to construct a groundwater-level contour map such as the one in the example. Locate the position of each observation point on a base map of suitable scale and write the water level against each well's position. Study these water-level values to decide which contour lines would cross the center of the map. Select one or two key contours to draw in first.

Once the contour map is complete, flow lines can be drawn by first dividing a selected contour line into equal lengths. Flow lines are drawn at right angles from this contour, at each point marked on it. The flow lines are extended until the next contour line is intercepted, and are then continued at right angles to this new contour line. Always select a contour that will enable you to draw the flow lines in a downgradient direction.

B. The Three-Point Problem

Groundwater-flow direction can be determined from water-level measurements made on three wells at a site (**Figure 1**).

1. Given:

Well Number	Head (meters)
1	26.28
2	26.20
3	26.08

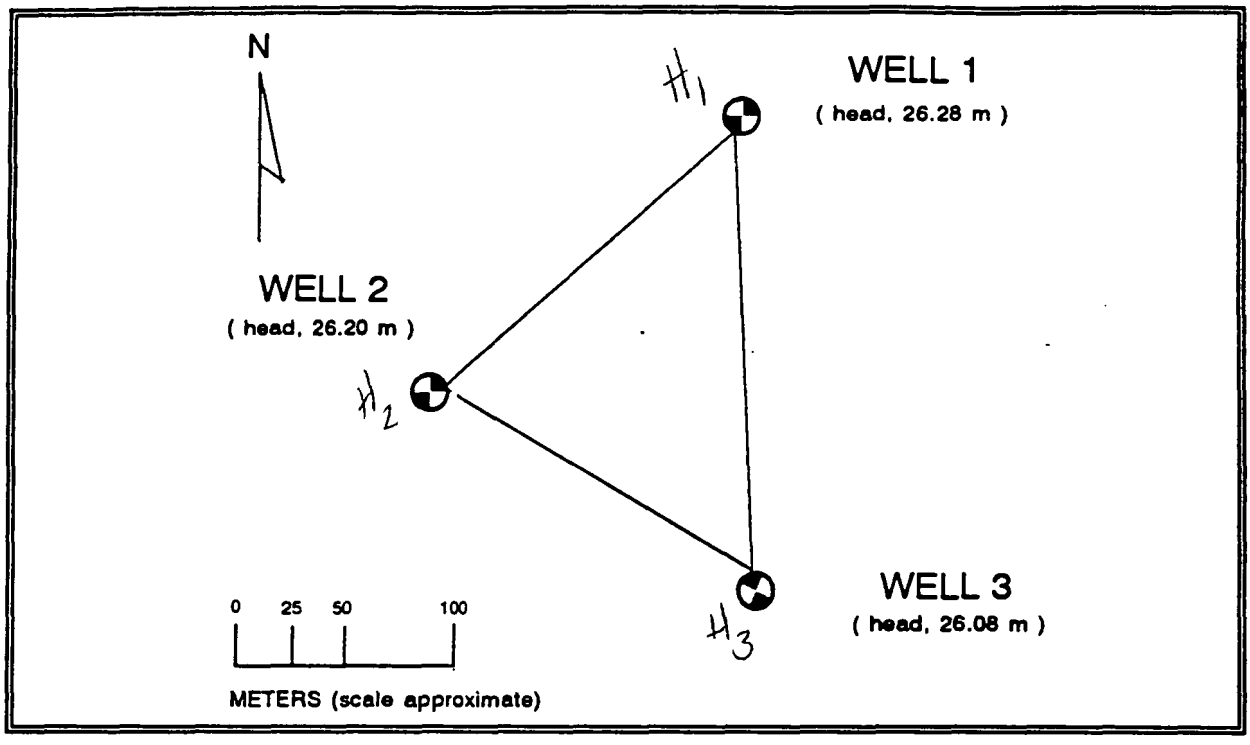


FIGURE 1

2. Procedure:

   a. Select water-level elevations (head) for the three wells depicted in **Figure 1**.

   b. Select the well with water-level elevation between the other wells (Well 2).

   c. Draw a line between Wells 1 and 3. Note that somewhere between these wells is a point, labeled A in **Figure 2**, where the water-level elevation at this point is equal to Well 2 (26.20 m).

   d. To determine the distance X from Well 1 to point A, solve the following equation (see **Figures 3, 4, and 5**):

$$\frac{H_1 - H_3}{Y} = \frac{H_1 - H_2}{X}$$

   e. Distance Y is measured directly from the map (200 m) on **Figure 3**. H_1, H_2, and H_3 represent head or water-level elevations from their respectively numbered wells.

   f. After distance X is calculated, groundwater-flow direction based on the water-level elevations can be constructed 90° to the line representing equipotential elevation of 26.20 m (**Figure 6**).

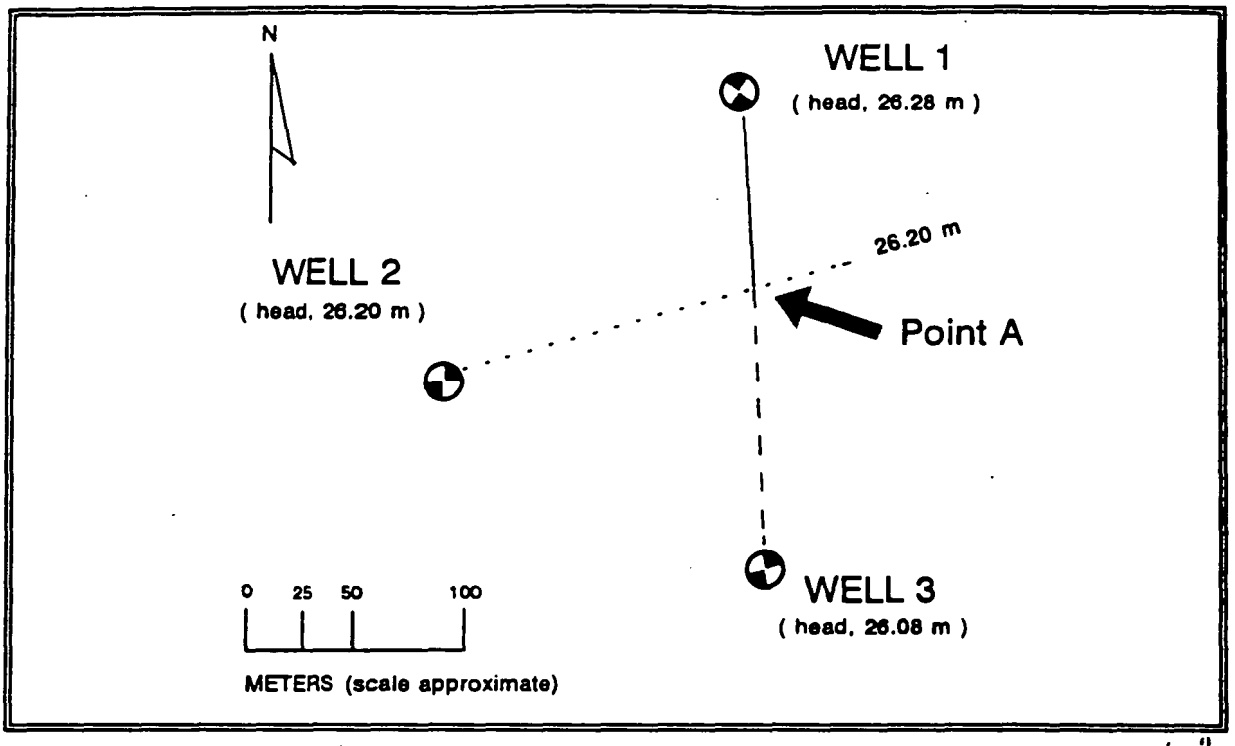


FIGURE 2

y= 1.6"

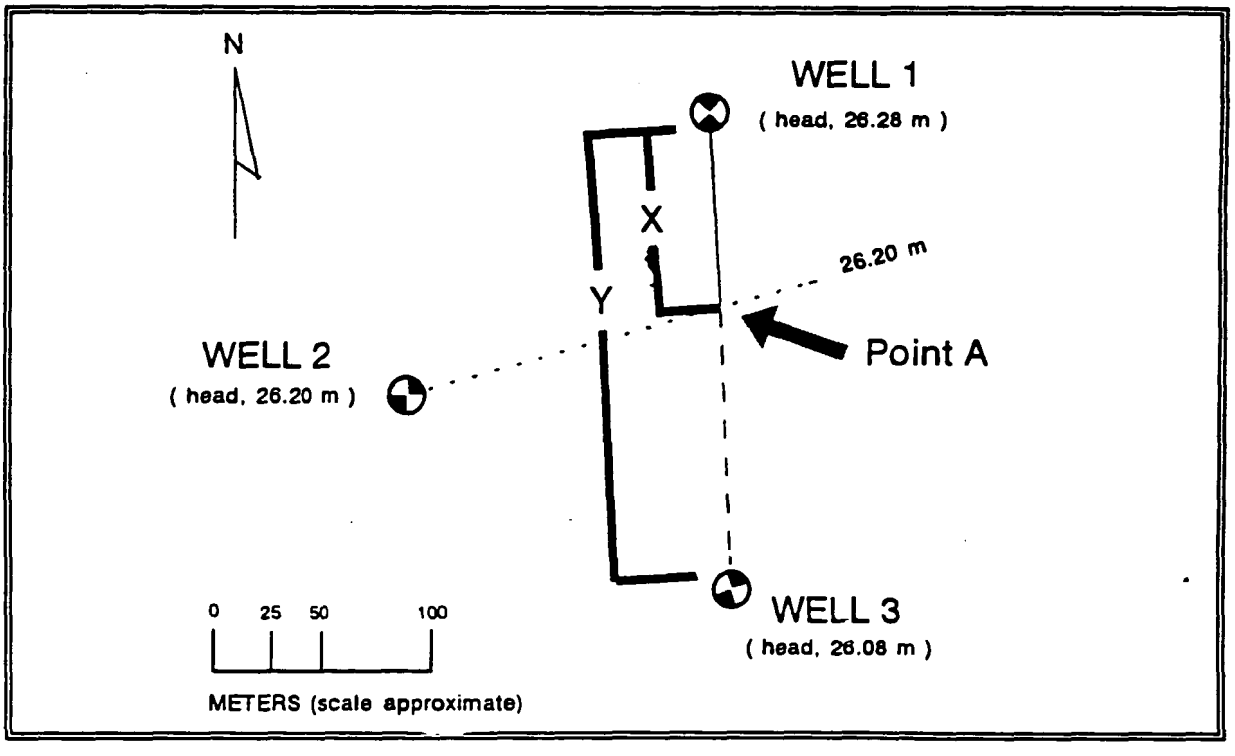


FIGURE 3

Y value – always between H_1 and H_3

$$\frac{H_1 - H_2}{X} = \frac{H_1 - H_3}{Y}$$

$$\frac{(26.28 - 26.20)}{X} = \frac{(26.28 - 26.08)}{200}$$

$$X = 80$$

FIGURE 4

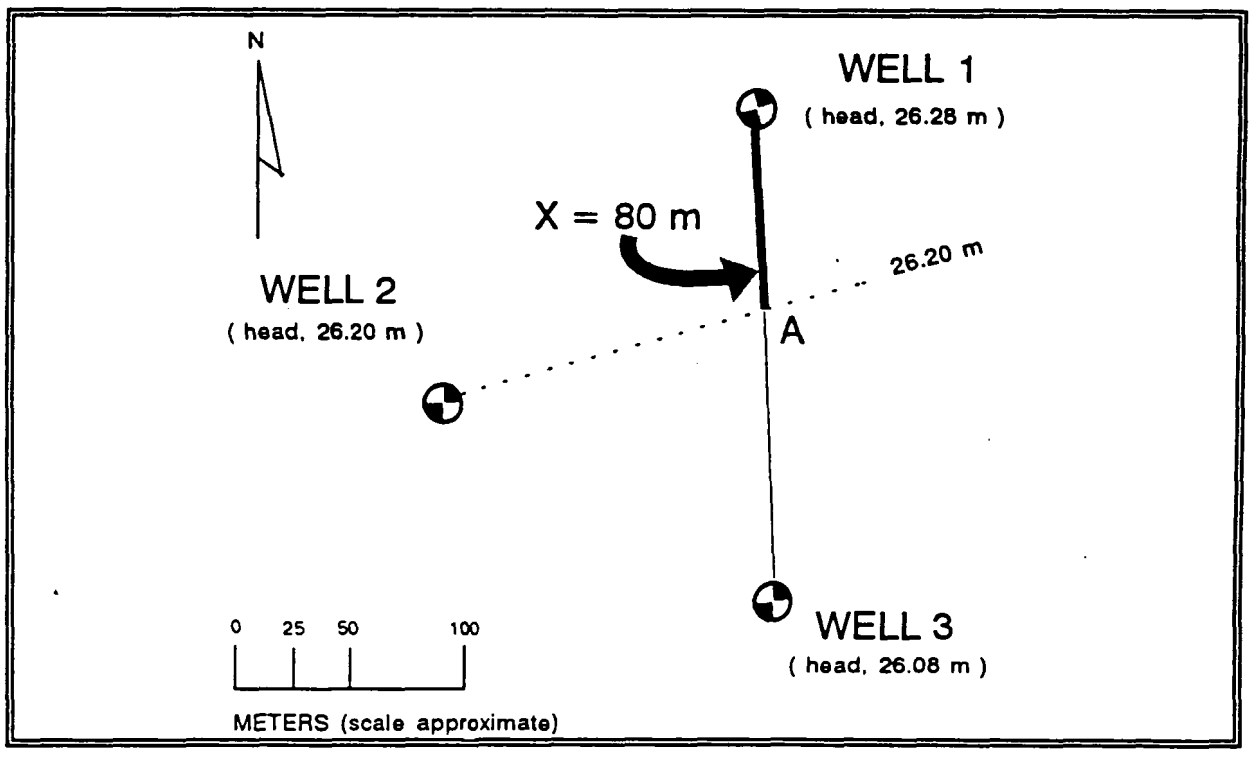


FIGURE 5

X always LAO off from H_1

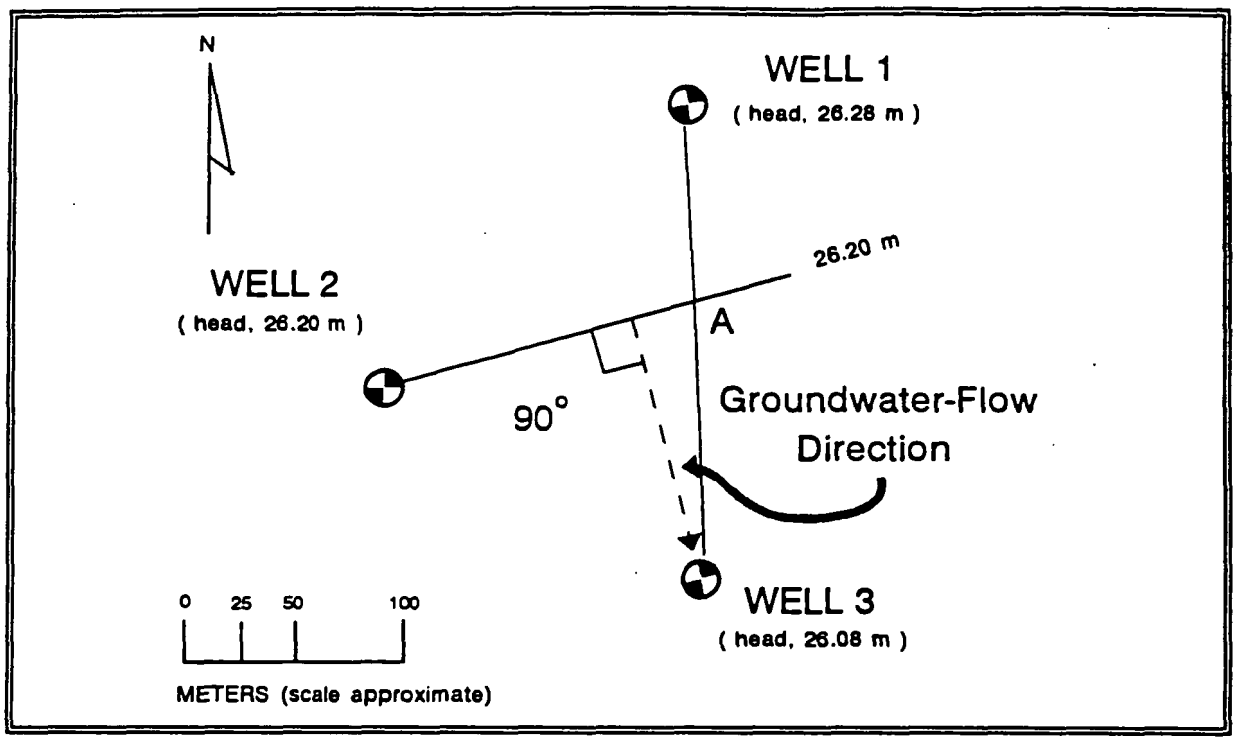


FIGURE 6

C. Colbert Landfill Three-Point Problem

1. History of Colbert Landfill

The Colbert Landfill is located 2.5 miles north of the town of Colbert near Spokane, Washington, and is owned by the Spokane County Utilities Department. This 40-acre landfill was operated from 1968 to 1986, when it was filled to capacity and closed. It received both municipal and commercial wastes from many sources. From 1975 to 1980, a local electronics manufacturing company disposed spent solvents containing methylene chloride (MC) and 1,1,1-trichloroethane (TCA) into the landfill. A local Air Force base also disposed of solvents containing acetone and methyl ethyl ketone (MEK). These solvents were trucked to the landfill in 55-gallon drums and poured down the sides of open and unlined trenches within the landfill. Approximately 300–400 gallons/month of MC and 150–200 gallons/month of TCA were disposed. In addition, an unknown volume of pesticides and tar refinery residues from other sources were dumped into these trenches.

The original site investigation was prompted by complaints from local residents who reported TCA contamination of their private wells. The population within 3 miles of the site is 1,500. In 1981, a Phase 1 investigation was conducted; a Phase 2 was completed in 1982. Groundwater samples collected from nearby private wells indicated TCA contamination at 5,600 μg/L, MC contamination at 2,500 μg/L, and acetone at a concentration of 445 μg/L. Investigation reports concluded that drinking groundwater posed the most significant risk to public health. EPA placed the site on the National

Priority List (NPL) in 1983. Bottled water and a connection to the main municipal water system was supplied to residents with high TCA contamination (above the MCL), and the cost was underwritten by the potentially responsible parties (PRPs) involved.

2. Hydrogeological Investigation

The site lies within the drainage basin of the Little Spokane River, and residents with private wells live on all sides of the landfill. The surficial cover and subsequent lower strata in the vicinity of the site consist of glacially derived sediments of gravel and sand, below which lie layers of clay, basaltic lava flows, and granitic bedrock. Beneath the site there are three aquifers and three aquitards. The stratigraphic sequence beneath the landfill from the top (youngest) to the bottom (oldest) is:

Qfg Upper sand and gravel glacial outwash and Missoula flood deposits which together form a water table aquifer

Qglf Upper layers of glacial Lake Columbia deposits of impermeable silt and clay that serve as an aquitard; lower layers of older glaciofluvial and alluvial sand and gravel deposits that form a confined aquifer

Mvwp Impermeable but weathered Wanapum basalt flow

Mcl Impermeable and unweathered Latah Formation of silt and clay

Kiat Fractured and unfractured granitic bedrock that serves as another confined aquifer

In the upper aquifer (Qfg), which is 8–15 feet thick, groundwater flows from 4 to 13 feet per day (ft/day). The lower confined sand and gravel aquifer (Qglf) varies from a few feet thick to 150 feet thick and is hydraulically connected to the Little Spokane River. Groundwater in this aquifer flows from 2 to 12 ft/day. To the northeast of the landfill, the upper aquifer is connected to the lower aquifer. Both of these aquifers are classified as current sources of drinking water according to EPA and are used locally for potable water. The area impacted by the site includes 6,800 acres and the contamination plume extends 5 miles toward the town of Colbert. Of the contaminants present, 90 percent occur as dense, nonaqueous-phase liquids (DNAPLs) at the bottom of the upper aquifer, and natural DNAPL degradation is slow. It has been estimated that only 10 percent of the solvents have gone into solution, whereas the remainder occurs in pore spaces and as pools of pure product above impermeable layers. The TCA plume in the upper aquifer has extended 9,000 feet in 8–10 years and it moves at a rate of 2–3 ft/day. The flow rate of the contamination plume in the lower sand and gravel aquifer (Qglf) has not been calculated because of the complexity and variability of the subsurface geology. However, TCA and MC have the highest concentrations in the lower sand and gravel aquifer.

3. Remedial Measures

The remediation goal for this site is to use an extraction and interception system (pump and treat) for removing groundwater contamination and to completely cap and regrade the site. A line of 8 groundwater extraction wells of variable depth, located downgradient of the site, and 10 extraction wells 100 ft deep will be used for site remediation. The wells in the lower sand and gravel aquifer will pump at a rate of 130 gallons per minute (gpm), whereas the wells in the water table aquifer will pump at a rate of 20–30 gpm.

Groundwater and soil gas monitoring is scheduled to continue for 30 years to monitor the location and movement of the groundwater contamination plume.

4. Groundwater Flow-direction Calculations

Using the data in **Table 1** from monitoring wells in the vicinity of the Colbert Landfill (see topographic map from the cross-section exercise), determine the groundwater flow direction within the shallow and deep aquifers.

Choose three wells that are relatively close together and on the same side of the Little Spokane River. Assume that north is located at the top of the page. Check your calculations.

a. Shallow groundwater flow direction:

b. Deep groundwater flow direction:

TABLE 1. CONSTRUCTION DATA ON MONITORING WELLS LOCATED IN THE VICINITY OF THE COLBERT LANDFILL, SPOKANE, WA

Table 1	A	B	C	D	E
Well Number (MW#)	Top of Casing Elevation (ft msl)	Ground Surface Elevation (ft msl)	Ground Water Elevation (ft msl)	Monitoring Well Depth (ft below ground)	Bedrock Depth (ft below ground)
1	1923.25	1920.14	1877.14s	105.10	83.40
2	1958.45	1955.50	1745.50d	263.50	269.40
3 H_1	1929.88	1926.94	1615.94d	341.20	344.30
4 H_3	1868.05	1865.85	1556.85d	340.40	343.50
5	1675.50	1672.15	variable	210.50	184.10
6 H_1	2003.70	2000.79	1958.79s	322.80	321.10
7 H_3	1948.26	1945.55	1431.21s	75.90	80.10
8 H_2	1703.20	1700.00	1695.00s	120.90	124.60
9 H_3	1906.11	1903.60	1610.75d	350.70	350.20

Y=1.9

s = shallow aquifer
d = deep aquifer

$$\frac{H_1 - H_3}{Y} = \frac{H_1 - H_2}{X}$$

$$\frac{1958.79 - 1431.21}{1.6} = \frac{1958.79 - 1695}{x}$$

$$329.79 = \frac{263.79}{x}$$

$$x = 0.8$$

PART 2.

A. Groundwater Gradient Calculation

1. Purpose

This part of the exercise uses basic principles defined in the determination of groundwater-flow directions. Groundwater gradients (slope of the top of the groundwater table) will be calculated as shown in the three-point problem.

2. Key Terms

- Head—The energy contained in a water mass produced by elevation, pressure, and/or velocity. It is a measure of the hydraulic potential due to pressure of the water column above the point of measurement and height of the measurement point above datum which is generally mean sea level. Head is usually expressed in feet or meters.
- Contour line—A line that represents the points of equal values (e.g., elevation, concentration).
- Equipotential line—A line that represents the points of equal head of groundwater in an aquifer.
- Flow lines—Lines indicating the flow direction followed by groundwater toward points of discharge. Flow lines are always perpendicular to equipotential lines. They also indicate direction of maximum potential gradient.

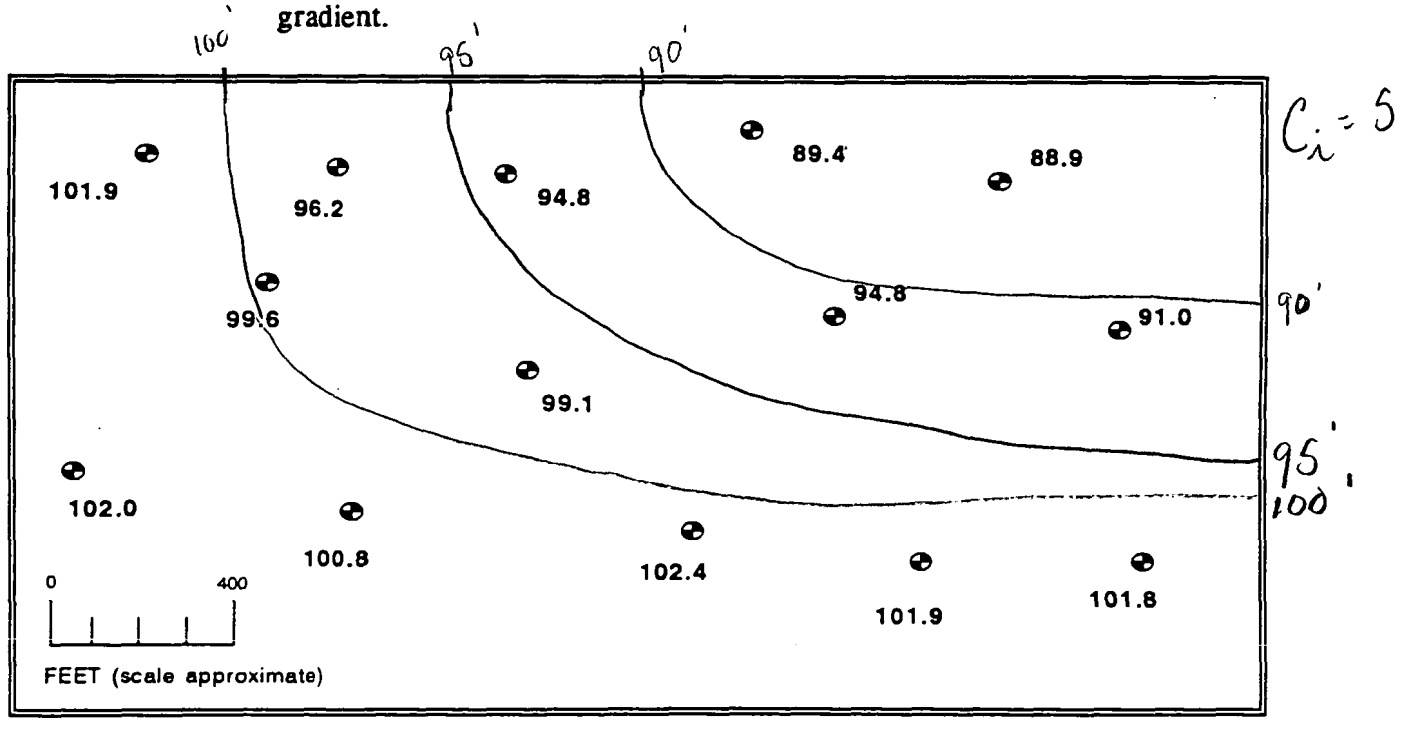


FIGURE 7. WELL LOCATIONS AND HEAD MEASUREMENTS

3. After reviewing **Figures 7-9**, perform the following:

   a. Select an appropriate contour interval that fits the water levels available and the size of the map on **Figure 10**. (Twenty-foot contour intervals should be appropriate for this problem.)

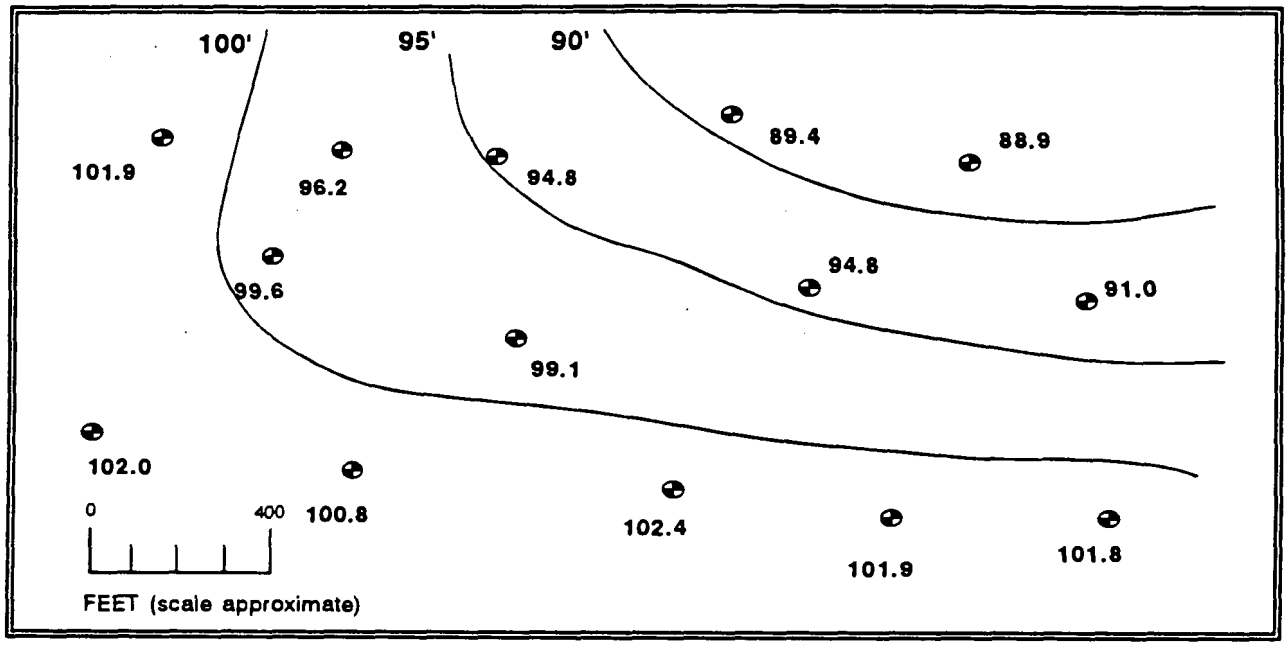


FIGURE 8. EQUIPOTENTIAL LINES WITH WELL HEAD MEASUREMENTS

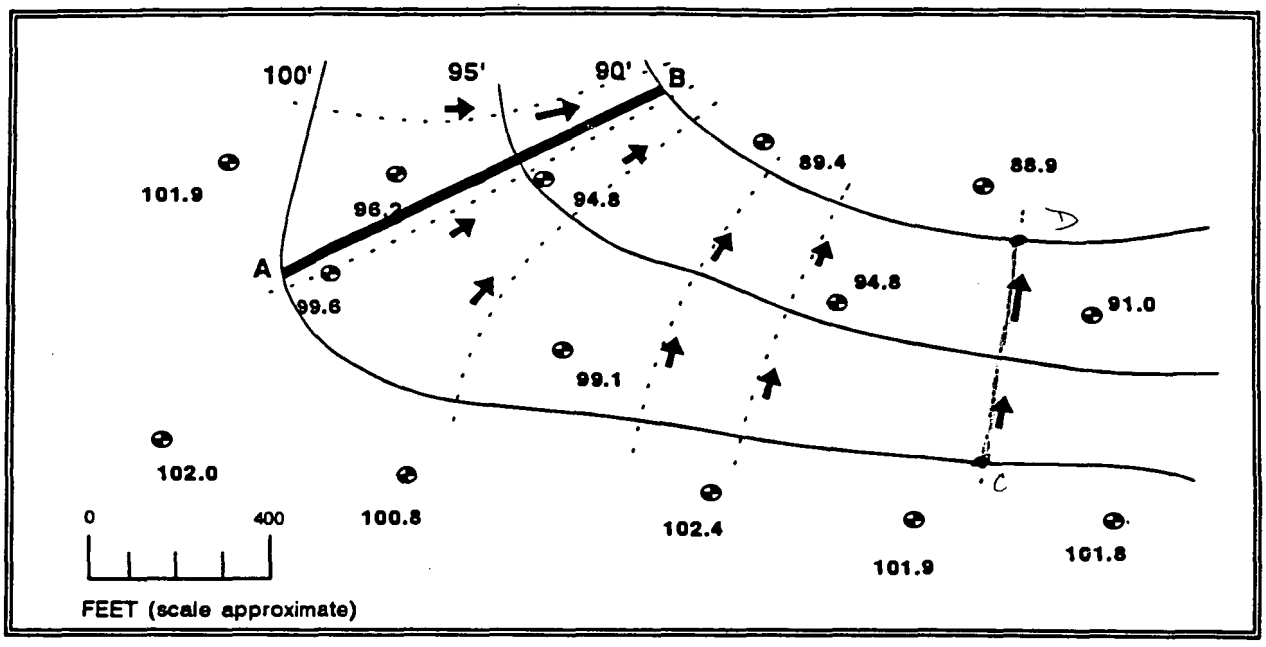


FIGURE 9. FLOW LINES ADDED TO EQUIPOTENTIAL LINES AND CALCULATION OF HYDRAULIC GRADIENT

$$\frac{\Delta H}{L} = \frac{\overset{C}{100} - \overset{D}{90}}{550'} = \frac{10}{550} = 0.018 \ \frac{\text{feet}}{\text{foot}}$$

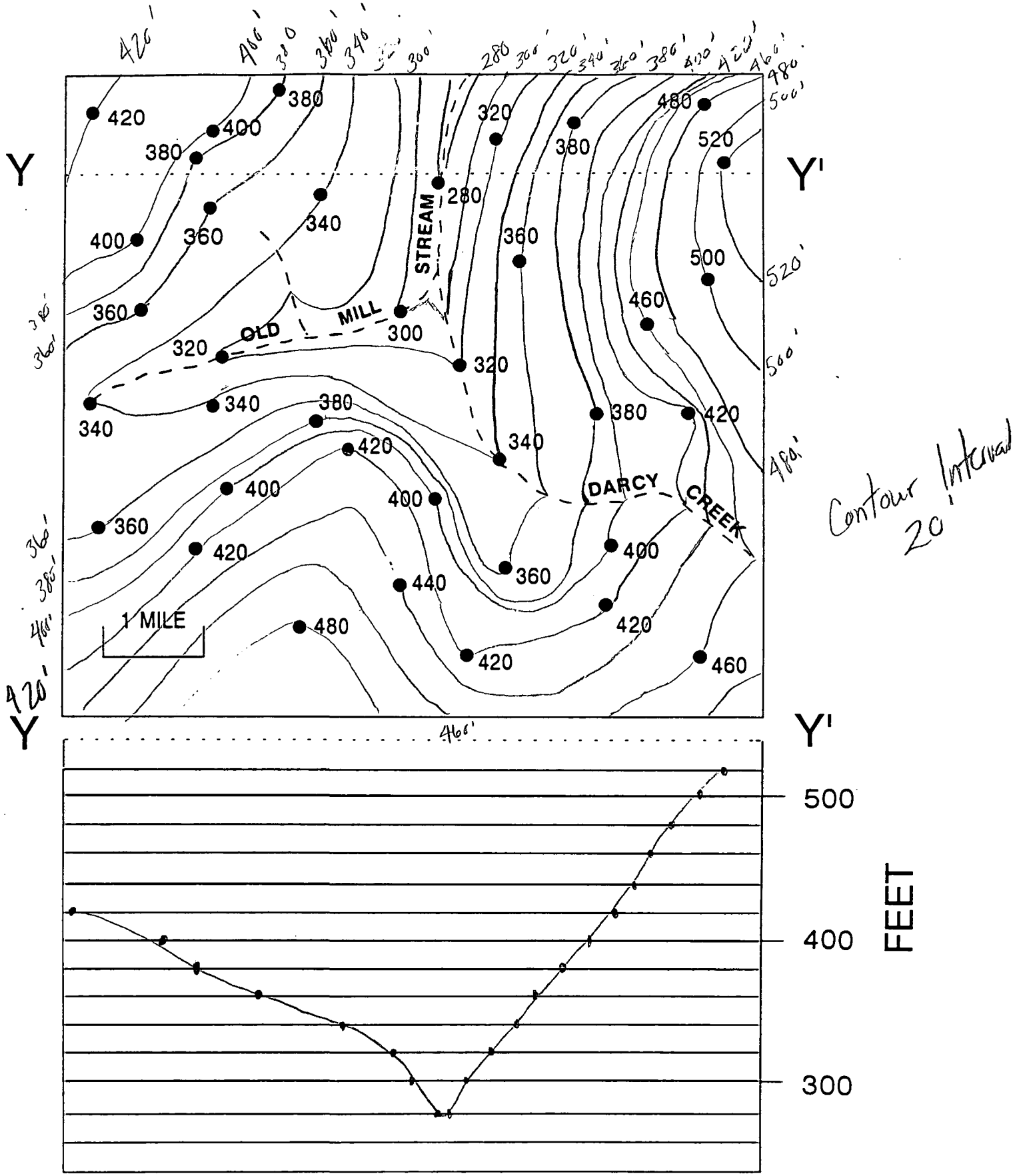

Y
Y'
STREAM
OLD MILL
DARCY CREEK
1 MILE
Contour Interval 20'
FEET
500
400
300

Figure 10

b. Draw the equipotential lines on the map (see **Figure 8**), interpolating between water-level measurements.

c. Construct flow lines perpendicular to the equipotential lines drawn in step 3 (see **Figure 9**).

d. Select a distance on your contour map between two contour lines and compute the gradient. The hydraulic gradient is calculated by measuring the scale distance between equipotential lines along a flow line that crosses the site, and dividing that value into the calculated change in head across the same distance (H_2 - H_1).

$$\frac{H_1 - H_2}{L} = \frac{\Delta H}{L} = I$$

For example, (see **Figure 9**):

Head at A = 100' (H_1)

Head at B = 90' (H_2)

Measured distance between the points is 850' (L)

Head at point A minus head at point B divided by the distance between the points equals hydraulic gradient (slope from point A to point B).

$$\frac{100\ feet - 90\ feet}{850\ feet} = \frac{10}{850} = .012\ feet/foot$$

B. Profile of the Site's Groundwater Surface

After completing the contour map, plot a profile of the sites groundwater surface at Y-Y[1] on **Figure 10**.

PART 3.

A. Bakers Quarry Flow Net Construction

1. Site History/Operation

The quarry operation began in 1905 providing local construction-grade granite. The quarry was closed in 1928 when the volumes of groundwater seeping into the pit made it economically unfeasible to continue mining (**Figure 11**). The site was abandoned and the pit filled with water. The owners of the quarry declared bankruptcy and ownership fell to the city of Tippersville in lieu of delinquent tax payments.

The quarry was used as a swimming hole and occasional dump site for local citizens until 1958, when several children drowned. The site was fenced and patrolled to prevent swimming. Uncontrolled dumping by individuals and local industry increased dramatically with the swimming ban. Dumping took place around the rim of the quarry, and the bulldozer from the town landfill was periodically used to push material into the pit. Gradually the pit was filled and several fires forced the town to terminate dumping in 1971. The surface of the site was covered with local material, primarily sand and gravel.

The site gained notoriety when an area-wide survey identified it as a potential industrial dump site. A preliminary site investigation, started on April 14, 1982, included sampling a spring located approximately 25 ft from the limits of quarrying. Priority pollutant analysis of this water sample identified ppm levels of polychlorinated biphenyls and trichloroethylene. Results from this preliminary investigation were used to justify a more extensive hydrogeologic study of the site.

2. Elements of the Hydrogeological Investigation

The first step of this investigation was to do a literature review of geologic information. A discussion with a local amateur geologist revealed a paper from a geologic investigation performed during active quarrying. Information from this study and observations at an outcrop onsite provided a geologic background for the investigation. The quarry material is a slightly gneissoid biotite-muscovite granite. Several dikes were identified in the quarry wall.

The probable high permeability and infiltration rate of the less-consolidated waste material compared to that of the granite could cause groundwater mounding in the pit area. Potential mounding, and inadequate information about groundwater flow direction, dictated a ringing of the site with monitoring wells.

Twenty-two monitoring wells were planned and installed at the site from October 1 to November 14, 1982. Eleven monitoring wells were installed in bedrock, and the unconsolidated zone was sealed with steel casing and grouted. Eleven monitoring wells were installed in the unconsolidated, heavily weathered bedrock or unconsolidated zones. For this exercise, use only the data from the 11 wells listed in **Table 2**. An explanation of these data is depicted in **Figure 12**.

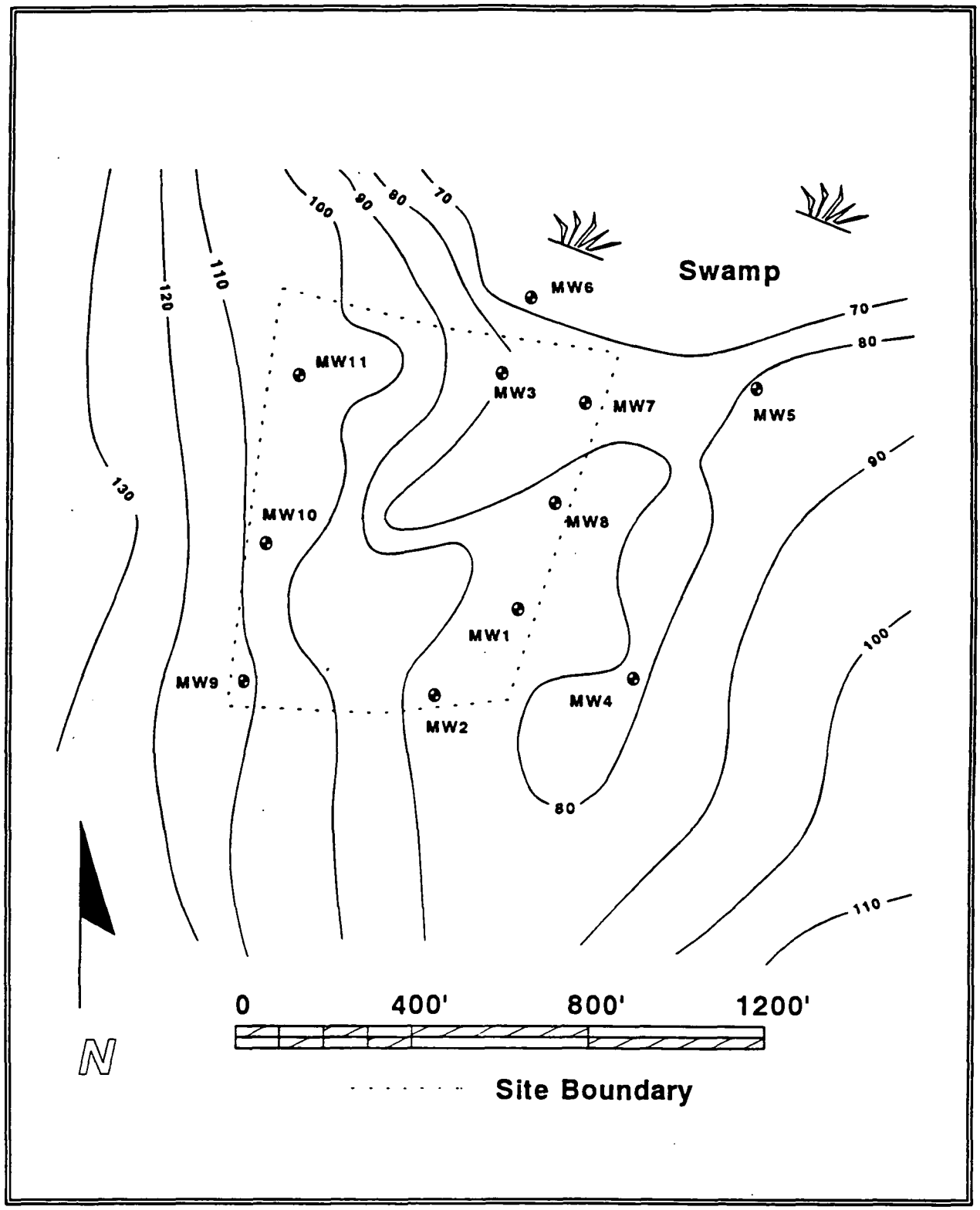
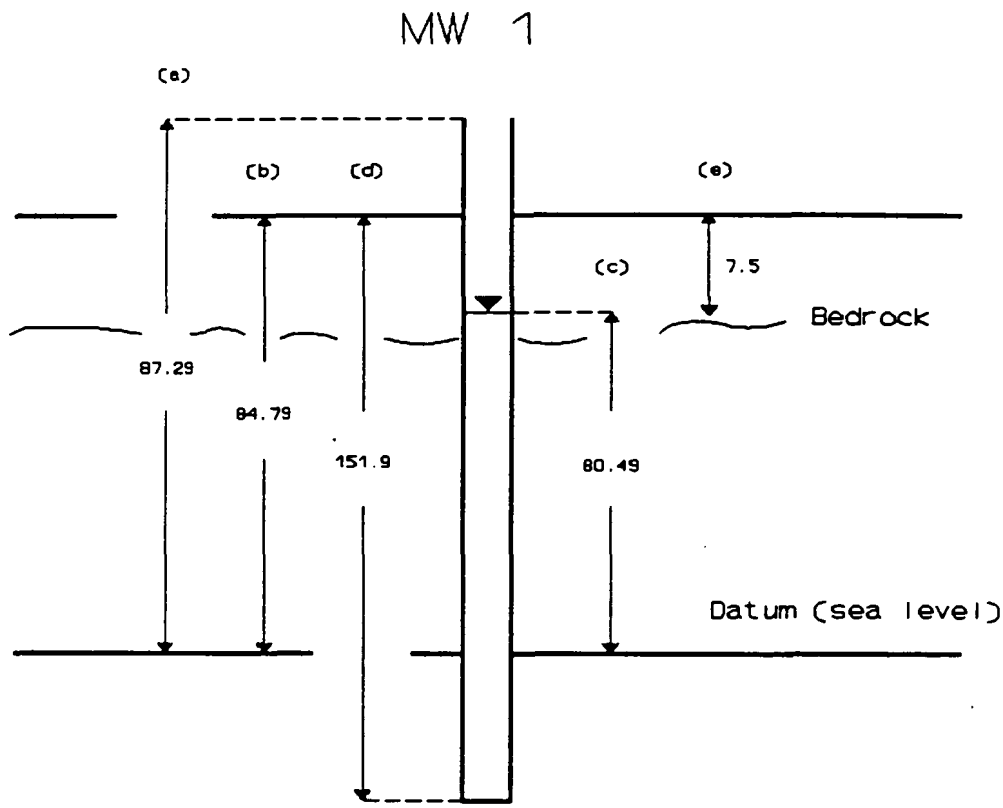


FIGURE 11. SITE MAP - BAKERS QUARRY, TIPPERSVILLE, MAINE

TABLE 2. MONITORING WELL DATA

Well Number	(a) Top of Casing Elevation (feet)*	(b) Ground Surface (GS) Elevation (feet)*	(c) Groundwater Elevation (feet)*	(d) Well Depth (feet below GS)	Bottom of Well Elevation (feet)*	(e) Bedrock Depth (feet below GS)
MW 1	87.29	84.79	80.49	151.9	-67.11	7.5
MW 2	89.94	87.99	84.69	103.05	-15.06	7.5
MW 3	88.04	85.44	75.29	103.1	-17.66	2.0
MW 4	82.50	79.80	72.40	102.3	-22.50	14.0
MW 5	82.50	80.05	73.40	102.45	-22.40	8.5
MW 6	72.50	69.50	67.50	99.6	-30.10	9.0
MW 7	80.58	78.28	74.78	99.5	-21.22	8.0
MW 8	86.03	83.53	76.93	99.2	-15.67	8.5
MW 9	114.01	111.21	92.36	99.9	11.31	10.5
MW 10	108.67	106.67	93.97	98.7	7.97	10.8
MW 11	105.07	103.37	94.97	102.1	1.27	2.5

* Datum: mean sea level



(a) Top of casing elevation (feet)

(b) Ground surface elevation (feet)

(c) Groundwater elevation (feet)

(d) Well depth below ground surface (feet)

(e) Bedrock depth (feet)

FIGURE 12. MONITORING WELL ELEVATIONS

B. Site Profile Development

1. Purpose

The development and comparison of topographic profiles across the site will help the student to understand the variability of the surface terrain usually found on most of the larger sites. The water-table profile will also be constructed.

2. Procedure

a. To construct cross-section lines, lay the edge of a piece of paper along the cross-section line selected and draw a straight line. Mark the location of the monitoring wells along the edge of the paper. (The placement of some wells may need to be projected because not all of the wells lie along a straight line.)

A - A'	MW9, MW2, and MW4 (in that order)
B - B'	MW1, MW8, and MW7
C - C'	MW11, MW3, MW7, and MW5

NOTE: Projection of wells to a cross-section line could cause distortions that might affect interpretation of the distribution of subsurface geology or soil.

b. Using the graph paper provided, transfer these well locations to the bottom of the page along the horizontal axis.

c. The vertical axis will represent elevation in feet. Mark off the elevations in 10-ft increments. Each division of the graph will represent an elevation increase of 2 ft.

d. Graph the ground surface elevation for each of the chosen monitoring wells. (This information is found in the monitoring well data, **Table 2**.)

e. Graph the groundwater elevations for these same locations.

f. Repeat this procedure for the other cross-sections lines.

g. Compare the topographic profile to the water table-profile. Are they identical? After looking at these data, are there any conclusions that can be drawn?

PART 4.

A. Student Performance Objectives

1. Perform a falling head test on geologic materials.

2. Calculate total porosity, effective porosity, and estimated hydraulic conductivity.

3. Given groundwater elevations in monitoring wells, determine the equipotential groundwater surface.

4. Given groundwater's equipotential surface, determine groundwater flow direction.

B. Perform a Falling Head Test

1. Set up burets using the stands and tube clamps.

2. Clamp the rubber tube at the bottom of the burets using the hose clamp. Fold the rubber hose to ensure a good seal before clamping (to help eliminate leaking water).

3. Position the small, round screen pieces in the bottom of the burets. Use the tamper to properly position the screens.

4. Measure 250 ml of clean water in the 500-ml plastic beaker.

5. Pour the water slowly into the buret to avoid disturbing the seated screen.

6. Measure 500 ml of gravel or sand material in the 500-ml plastic beaker.

7. Pour the gravel or sand material slowly into the water column in the buret to prevent the disturbance of the screen traps and to allow any trapped air to flow to the surface of the water in the buret.

8. Add additional measured quantities of water or gravel/sand as needed until both the water and sediment reach the zero mark on the buret. To calculate the final total volumes of water and sediment, add the volumes of additional water and gravel/sand to the initial volumes of 250 and 500 ml of water and sediment. The total volumes of water and sediment are designated W and S respectively.

9. Measure the static water level in the buret to the base of the buret stand. This is the total head of the column of water at this elevation. This measurement is designated h_0.

10. Place a plastic, 500-ml graduated beaker below the buret. (The beaker will be used to collect the water drained from the buret.) The volume of water in the beaker is designated W_D.

11. Undo the clamp and simultaneously start the timer to determine the flowrate of water through the buret. When the drained water front reaches the screen, stop the timer, clamp the buret hose, and record the elapsed time. Also record the volume of water drained during this time interval. This time is designated t.

12. Allow the water level in the buret to stabilize. Measure the length from this level to the base of the buret stand. This is the total head of the water column at this elevation after drainage has occurred. This measurement is designated h_1.

13. Subtract the measurement at h_1 from the height measurement at h_0. This length is designated L.

14. The porosity in the sediment of each buret is the volume of water necessary to fill the column of sediment in the buret to the initial static water mark at h_0 divided by the sediment volume (S). This value is total porosity and is designated N.

15. The effective porosity is estimated by dividing the volume of drained water by the sediment volume. Effective porosity is designated n.

16. Compare the initial volume of water (W) in the column before draining with the drained volume (W_D). The difference represents the volume of water retained (W_R), or the specific retention. The volume drained represents specific yield. To determine the percent effective porosity, divide the volume of drained water by the volume of total sediment volume.

17. The equation to estimate the hydraulic conductivity (K) of each buret column is derived from falling head permeameter experiments. The equations for this exercise are depicted at the bottom of **Table 3**.

TABLE 3. TOTAL HEAD WORKSHEET

Sample Number	Volume Sediment (S)	Volume Water (W)	Volume Drained Water (W_D)	Volume Retained Water (W_R)	Total Porosity Calculated (N)	Effective Porosity Calc. (n)	Length (h_0-h_1) (L)	Time (t) sec	Initial Head (h_0)	Final Head (h_1)	Ln (h_0 / h_1)	Est. Hydraulic Conductivity (K)
1												
2												
3												
4												
5												
6	500 ml	250 ml	160 ml	90 ml	0.50	0.32	45 cm	sec 24.0	70 cm	25 cm	1.03	4.44
7												

Total Porosity

$$N = \frac{W}{S}$$

Effective Porosity

$$n = \frac{W_D}{S}$$

Est. Hydraulic Cond.

$$K = \left[\frac{2.3 * L}{t}\right] * \ln \frac{h_0}{h_1}$$

$$\frac{2.3 \times 45}{24} \times 1.03$$

$$K = 4.44$$

PROBLEM 5

Aquifer Stress Tests

AQUIFER STRESS TESTS

STUDENT PERFORMANCE OBJECTIVES

At the conclusion of this unit, students will be able to:

1. List the two factors that control aquifer response during an aquifer test

2. List four aquifer test methods

3. List the purposes of the step-drawdown test

4. List the advantages and disadvantages of a slug test

5. List the advantages and disadvantages of a distance-drawdown test

6. List the advantages and disadvantages of a time-drawdown test

7. Given graph paper, graphically represent groundwater flow to show the difference between aquifer tests in unconfined and confined aquifers

8. Given aquifer test data, use the Jacob method to calculate a hydraulic conductivity for the given conditions.

NOTE: Unless otherwise stated, the conditions for performance are using all references and materials provided in the course, and the standards of performance are without error.

NOTES

AQUIFER TESTS

S-1

GROUNDWATER AND CONTAMINANT MOVEMENT

- Position and thickness of aquifers and aquitards
- Transmissivity and storage coefficient
- Hydraulic characteristics of aquitard

S-2

GROUNDWATER AND CONTAMINANT MOVEMENT (cont.)

- Position and nature of boundaries
- Location and amounts of groundwater withdrawals
- Locations, kinds, and amounts of pollutants

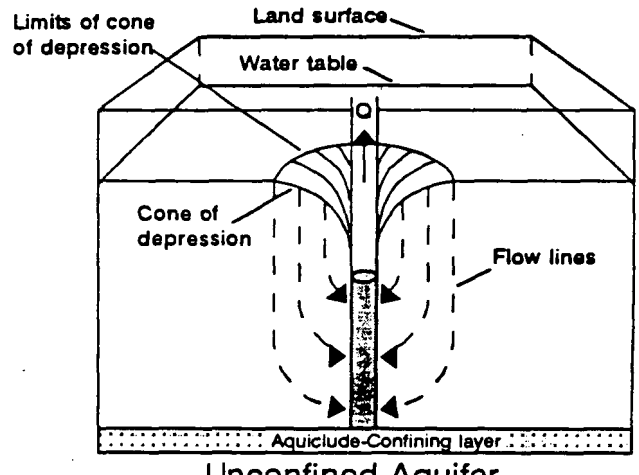
S-3

NOTES

AQUIFER RESPONSE DEPENDS ON:

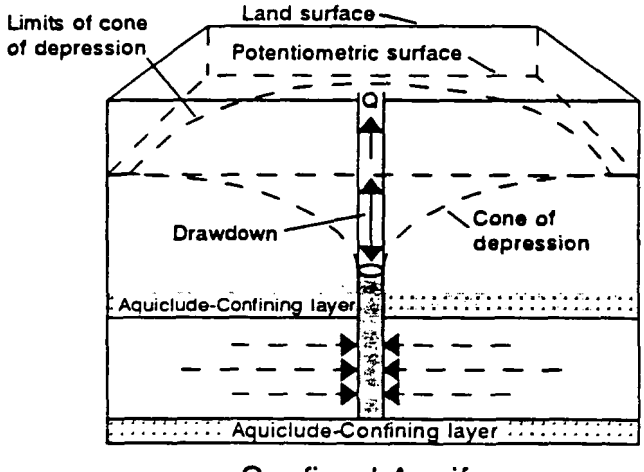
- Rate of expansion of cone of depression
  - Transmissivity of aquifer
  - Storage coefficient of aquifer
- Distance to boundaries
  - Recharge
  - Impermeable

S-4



Unconfined Aquifer

S-5



Confined Aquifer

S-6

NOTES

hydraulic conductivity only valid for small diameter

AQUIFER TEST METHODS

- Step-drawdown/well recovery tests
- Slug tests
- Distance-drawdown tests
- Time-drawdown tests

S-7

STEP DRAWDOWN
Well Recovery Tests

- Well is pumped at several successively higher rates and drawdown is recorded
- Purpose
  - Estimate transmissivity
  - Select optimum pump rate for aquifer tests
  - Identify hydraulically connected wells

S-8

STEP DRAWDOWN
Well Recovery Tests (cont.)

- Advantages
  - Short time span
  - One well

S-9

NOTES

SLUG TESTS

- Water level is abruptly raised or lowered
- Used in low-yield aquifers (<0.01 cm/s)

S-10

SLUG TESTS Advantages

- Can use small-diameter well
- No pumping - no discharge
- Inexpensive - less equipment required
- Estimates made *in situ*
- Interpretation/reporting time shortened

S-11

SLUG TESTS Disadvantages

- Very small volume of aquifer tested
- Only apply to low conductivities (0.0000001 to 0.01 cm/s)
- Transmissivity and conductivity only estimates

S-12

NOTES

SLUG TESTS
Disadvantages (cont.)

- Not applicable to large-diameter wells
- Large errors if well not properly developed
- Do not give storativity

S-13

DISTANCE-DRAWDOWN TESTS
Advantages

- Can also use time-drawdown
- Results more accurate than single well test
- Represent more of aquifer
- Can locate boundary effects

S-14

DISTANCE-DRAWDOWN TESTS
Disadvantages

- Requires multiple piezometers or monitoring wells (at least three wells)
- More expensive than single well test
- Must handle discharge water
- Requires conductivities >0.01 cm/s

S-15

NOTES

TIME-DRAWDOWN TESTS Advantages

- Only one well required
- Tests larger aquifer volume than slug test
- Less expensive than multiple-well test

S-16

TIME-DRAWDOWN TESTS Disadvantages

- Pump turbulence may interfere with water-level measurements
- Tests smaller aquifer volume than multiple-well test
- Must handle discharge water
- Requires conductivities above 0.01 cm/s

S-17

THEIS METHOD

- First formula for unsteady-state flow
  - Time factor
  - Storativity
- Derived from analogy between groundwater flow and heat flow

S-18

THEIS METHOD (cont.)

- Laborious method
  - Log-log paper
  - Curve matching
- More accurate than Jacob method

S-19

THEIS'S ASSUMPTIONS

- Aquifer is confined
- Aquifer has infinite areal extent
- Aquifer is homogeneous and isotropic
- Piezometric surface is horizontal

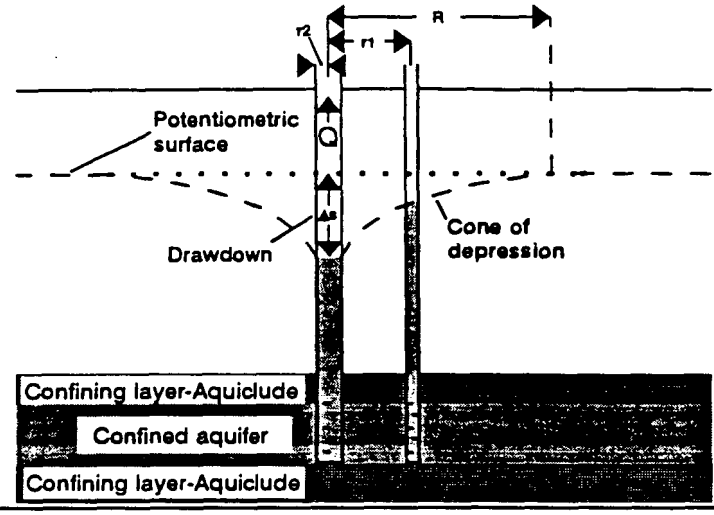
S-20

THEIS'S ASSUMPTIONS (cont.)

- Carefully controlled constant pump rate
- Well penetrates aquifer entirely
- Flow to well is in unsteady state

S-21

NOTES



S-22

THEIS EQUATION

$$T = \frac{QW(u)}{4\pi s}$$

$$S = \frac{4Ttu}{r^2}$$

T = transmissivity

Q = discharge (pumping rate)

W(u) = well function of u

s = drawdown

S = storage coefficient

t = time

r = radial distance

S-23

WELL FUNCTION - W(u)

$$W(u) = -0.577216 - \log_e u + u - \frac{u^2}{2 \times 2!} + \frac{u^3}{3 \times 3!} - \frac{u^4}{4 \times 4!} + \ldots$$

$$\text{and } u = \frac{r^2 s}{4Tt}$$

S = storage coefficient

r = distance

t = time

T = transmissivity

- W(u) is an infinite exponential series and cannot be solved directly

S-24

NOTES

JACOB METHOD

- Somewhat more convenient than Theis's method
  - Semilogarithmic paper
  - Straight line plot
  - Eliminates need to solve well function W(u)
  - No curve matching

S-25

JACOB METHOD (cont.)

- Applicable to:
  - Zone of steady-shape
  - Entire zone if steady-state

S-26

JACOB'S FORMULA

$$T = \frac{264\,Q}{\Delta s} \qquad K = \frac{T}{b}$$

T = transmissivity gallons per day per ft (gpd/ft)

Q = pump rate (gpm)

Δs = change in drawdown (ft/log cycle)

K = hydraulic conductivity in gpd/ft²

b = aquifer thickness in feet

S-27

$T = \frac{35Q}{\Delta S}$ gal ft²/day

$K = T/b$ ft/day

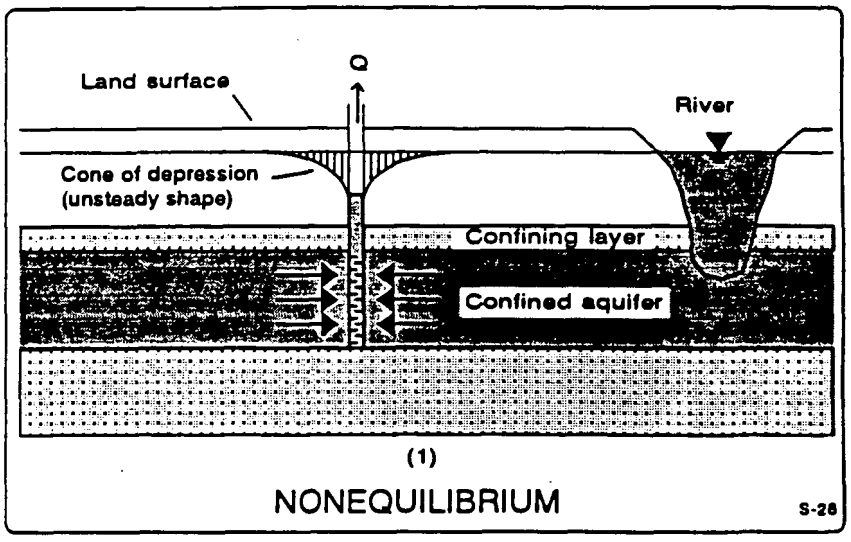
$T = \frac{264\,(109\text{ gpm})}{38\text{ ft}} = 757.26\ \frac{\text{gpd}}{\text{ft}}$

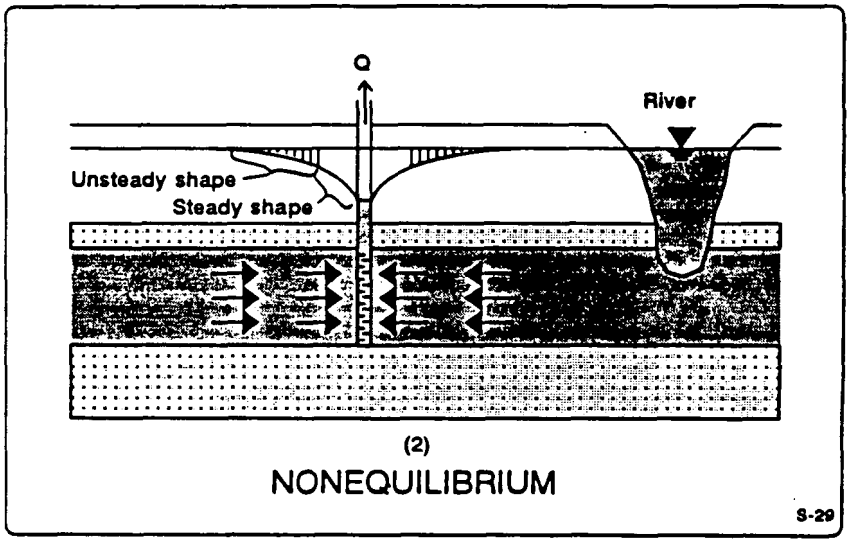
$b = 20$

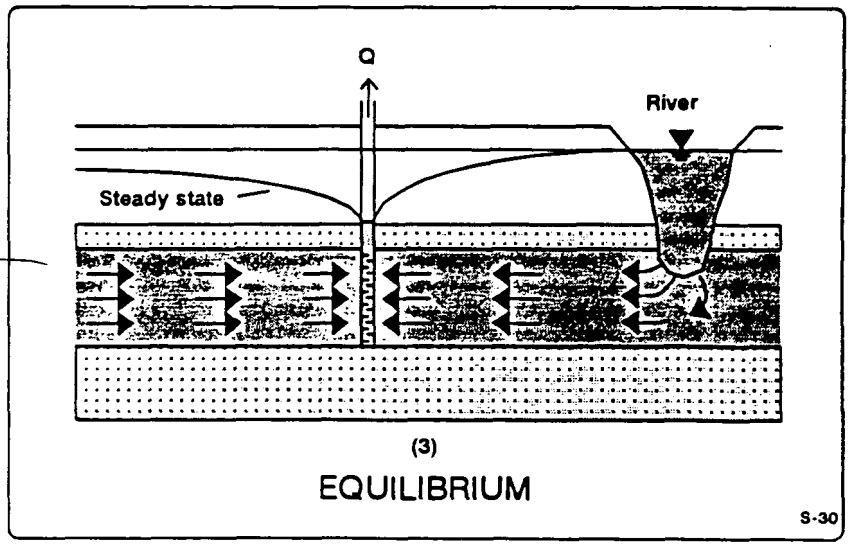
$K = \frac{757.26}{20} = 37.86$

$K = 37.86\ \frac{\text{gpd}}{\text{ft}^2}$

NOTES







Withdrawal from aquifer = recharge?

PERFORMING AN AQUIFER TEST

Jacob Time-drawdown Method

Each student will be given a sheet of four-cycle semilogarithmic graph paper. Then, follow these directions:

1. Label the long horizontal logarithmic axis (the side with the punched holes) of the graph paper t-time (minutes). Leave the first numbers (1 through 9) as is. Mark the next series of heavy lines from 10 to 100 in increments of 10 (10, 20, 30, etc.). Mark the next series from 100 to 1000 in increments of 100 (100, 200, 300, etc).

2. Label the short vertical arithmetic axis s-drawdown (feet). This will be the drawdown (s) measured from the top of the casing (provided in **Table 1**). Mark off the heavy lines by tens, starting with 0 at the top, then 10, 20, 30, 40, 50, 60, and 70 (the bottom line). Each individual mark represents 1 foot.

3. Plot the data in **Table 1** on the semilogarithmic paper with the values for drawdown on the arithmetic scale and corresponding pumping times on the logarithmic scale.

4. Draw a best-fit straight line through the data points.

5. Compute the change in drawdown over one log cycle where the data plot as a straight line.

6. Using the information given in **Table 1** (Q = 109 gpm and b = 30 feet) and Jacob's formula (provided in the manual on slide 27), calculate the value for hydraulic conductivity.

TABLE 1. PUMPING TEST DATA

Pumping Time (t) (minutes)	Drawdown (s) Measured from Top of Casing (ft)
Q = 109 gpm	b = 20 ft
0	6.1
1	6.5
2	7.5
3	8.0
4	8.6
5	9.5
6	10.5
7	11.2
8	12.0
9	13.0
10	14.0
11	15.5
12	17.0
13	18.0
14	19.3
15	20.5
18	23.5
20	25.2
22	26.7
24	28.2
26	29.5
28	30.5
32	32.0
35	34.5
40	36.6
45	38.5
50	40.5
55	42.0
60	43.5
90	50.1
120	54.8

PROBLEM 6

Groundwater Investigation Problem

PROBLEM 6: GROUNDWATER INVESTIGATION (BETTENDORF, IOWA)

A. Student Performance Objectives

1. Determine the source(s) of hydrocarbon contamination at a contaminated site.

2. Perform a Phase 1 field investigation using soil gas surveys, soil borings, and monitoring wells.

3. Present the results of the field investigation to the class.

4. Justify the conclusions of the field investigation.

B. Background Information

Task

Your environmental consulting firm has been retained by the attorney representing the Leavings to:

- Determine the source of the hydrocarbon contamination. **This is not an emergency response action.**

- Provide a brief report that includes the names of the source(s) of contamination, the total cost of investigation, and a drawing of a representative cross section through the contaminant plume.

- Justify the data that are obtained and the conclusions of the report.

Leavings Residence

On October 12, 1982, the Bettendorf, Iowa, fire department was called to the Leavings residence with complaints of gasoline vapors in the basement of the home.

On October 16, 1982, the Leavings were required to evacuate their home for an indefinite period of time until the residence could be made safe for habitation. The gasoline vapors were very strong, so electrical service to the home was turned off. Basement windows were opened to reduce the explosion potential.

Pertinent Known Facts

The contaminated site is in a residential neighborhood in Bettendorf, Iowa. It backs on commercially zoned property, which has only been partially developed to date. The residential area is about 10 years old and contains homes in the $40,000 to $70,000 range. There was apparently some cutting and filling activity at the time the area was developed.

Within 1/4 mile to the northwest and southwest, 11 reported underground storage tanks (USTs) are in use or have only recently been abandoned:

- Two tanks owned and operated by the Iowa Department of Transportation (IDOT) are located 1000 ft northwest of the site.

- Three in-place tanks initially owned by Continental Oil, and now by U-Haul, are located 700 ft southwest of the site. According to the Bettendorf Fire Department (BFD), on of the three tanks reportedly leaked.

- Three tanks owned and operated by an Amoco service station are located 1200 ft southwest of the site. BFD reports no leaks.

- Three tanks owned and operated by a Mobil Oil service station are located 1200 ft southwest of the site. BFD reports no leaks.

Neighbors that own lots 8 and 10, which adjoin the Leavings residence (Lot 9), have complained about several trees dying at the back of their property. No previous occurences of gasoline vapors have been reported at these locations.

The general geologic setting is Wisconsin loess soils mantling Kansan and Nebraskan glacial till. Valleys may expose the till surface on the side slope. Valley soils typically consist of the colluvial and alluvial silts.

Previous experience by your environmental consulting firm in this area includes a geotechnical investigation of the hotel complex located west of Utica Ridge Road and northwest of the Amoco service station. Loess soils ranged from 22 ft thick on the higher elevations of the property (western half) to 10 ft thick on the side slope. Some silt fill (5-7 ft) was noted at the east end of the hotel property. Loess soils were underlain by a gray, lean clay glacial till which apparently had groundwater perched on it. Groundwater was typically within 10-15 ft of ground surface. This investigation was performed 8 years ago and nothing in the boring logs indicated the observation of hydrocarbon vapors. However, this type of observation was not routinely reported at that time.

Other projects in the area included a maintenance yard pavement design and construction phase testing project at the IDOT facility located northwest of the Leavings residence. Loess soils were also encountered in the shallow pavement subgrade project completed 3 years ago. Consulting firm records indicated that the facility manager reported a minor gasoline spill a year before and that the spill had been cleaned up when the leaking tank was removed and replaced with a new steel tank. The second tank at the IDOT facility apparently was not replaced at that time.

Budget

The budget for implementing the field investigation is $25,000.

Interviews

- **Lot 9 (the Leavings residence):** Observations outside the residence indicate that the trees are in relatively good condition. The house was vacant. Six inches of free product that looks and smells like gasoline was observed in the open sump pit in the basement. The power to the residence was turned off, so the water level in the sump was allowed to rise. The fluid level in the sump was about 3 feet below the level of the basement floor.

- **Neighbors (Lots 8 and 10):** These property owners reported that several trees in their back yards died during the past spring. They contacted the developer of the area (who also owns the commercial property that adjoins their lots) and complained that the fill that was placed there several years ago killed some of their trees. No action was taken by the developer. Both neighbors said that when the source of the gas was located, they wanted to be notified so they could file their own lawsuits. The neighbors also noted that this past September and October were unusually wet (lots of rainfall).

- **IDOT:** The manager remembers employess of your firm testing his parking pad. He reported that one UST was replaced in 1979, whereas the other tank was installed when the facility was built in 1967. Both of the original tanks were bare metal tanks. The older tank has always contained gasoline, but the newer one contains diesel fuel. No inventory records or leak testing records are available. The manager stated that he has never had any water in his tanks. He will check with his supervisor to have the USTs precision leak tested.

- **U-Haul:** The manager said that the station used to be a Continental Oil station with three USTs. The three USTs were installed by Continental in 1970 when the station was built. Currently, only one 6000-gal UST (unleaded) remains in service for the U-Haul fleet. this tank was found to be leaking a month ago, but the manager does not know how much fuel spilled.

- **Mobil:** The manager was pleasant until he found out the purpose of the interview. He did state that he built the station in 1970 and installed three USTs at that time. He would not answer any additional questions.

- **Amoco:** The manager was not in, but an assistant provided his telephone number. In a telephone interview, the manager said he was aware of the leaking tank at the U-Haul facility and was anxious to prove the product was not from his station. He said they installed three USTs for unleaded, premium, and regular gasoline in 1972. An additional diesel UST was installed in 1978. The tanks are tested every 2 years using the Petrotite test method. The tanks have always tested tight. No inventory

control system is being used at present. He stated that if monitoring wells were needed on his property, he would be happy to cooperate.

- **Developer (Mr. M. Forester):** Mr. Forester bought the property in question in the 1960s. He developed the residential area first and some of the commercial development followed. About 40 acres remain undeveloped to date. He plans to build a shopping center on the remaining 40 acres in the future.

  Mr. Forester obtained a lot of cheap dirt and fill when the interstate cut went through about 1/2 mile west of the property in the late 1960s. He filled in a couple of good-sized valleys at that time. He has a topographic map of the area after it was filled.

  He stated that he will cooperate fully with any investigation. If any wells are needed on the property, he would like to be notified in advance. There are no buried utilities on the property except behind the residential neighborhood.

Review of Bettendorf City Hall Records

An existing topographic map and scaled land use map are available.

Ownership records indicate the land was previously owned by Mr. and Mrs. Ralph Luckless. The city hall clerk stated that she had known them prior to the sale of the farm in 1964. Zoning at that time was agricultural only. The section of the farm now in question was primarily used for grazing cattle because it was too steep for crops. The clerk remembered a couple of wooded valleys in that same field. She also remembered a muddy stream that used to run where Golden Valley Drive is now and that children used to swim in it. She also stated that one valley was between Golden Valley Drive and where all the fill is now (near U-Haul and Amoco).

The current owner of the undeveloped property is Mr. M. Forester, a developer with an Iowa City, Iowa address.

There is no record of storm or sanitary sewer lines along Utica Ridge Road south of Golden Valley Drive. Storm and sanitary sewer lines run along Spruce Hills Drive.

Iowa Geological Survey

There are no records of any wells in the section.

Adjoining section wells indicate top of bedrock at about 650 feet mean sea level (MSL). The uppermost usable aquifer is the Mississippian for elevations from 350 feet to 570 feet MSL. The materials overlying the Mississippian are Pennsylvanian shales and limestone.

Soil Conservation Survey maps

The 1974 edition indicates "Made Land" over nearly all of the area not designated as commercial zone. Made Land normally indicates areas of cut or fill.

ASSIGNMENT: PHASE 1 FIELD INVESTIGATION

TABULATION OF FEES FOR PHASE 1 FIELD INVESTIGATION GROUP ____

WORK SHEET #1	# UNITS	COST	TOTAL
Recommendation for making residence habitable	1 ea	$500 LS (lump sum)	$
Soil gas survey - mobilization fee	1 ea	$500 LS	$
Soil gas survey		$1500/ac	$
Soil boring - mobilization fee	1 ea	$500 LS	$
Soil boring with photo ionization detector - 25 feet deep max - grouted shut		$500 ea	$
Convert soil boring to PVC monitoring well (additional cost for each conversion)		$800 ea conversion	$
Convert soil boring to stainless steel monitoring well (additional cost for each conversion)		$1300 ea conversion	$
Monitoring wells - mobilization fee	1 ea	$500 LS	$
2" PVC 15 ft screen - 25 ft deep		$1200 ea	$
2" stainless steel 15 ft screen - 25 ft deep		$1700 ea	$
Well security - locking protector pipe		$300 ea	$
Field investigation engineering analysis and report	1 ea	15% $2000 min	$
TOTAL COST:			

GROUP _____

MONITORING WELLS

WELL NUMBER	1	2	3	4	5	6	7	8	9	10
GRID LOCATION										
FILL										
LOESS										
ALLUVIUM										
TILL										
NON DETECTED										
DISSOLVED PRODUCT										
FREE PRODUCT										
WATER ELEVATION										

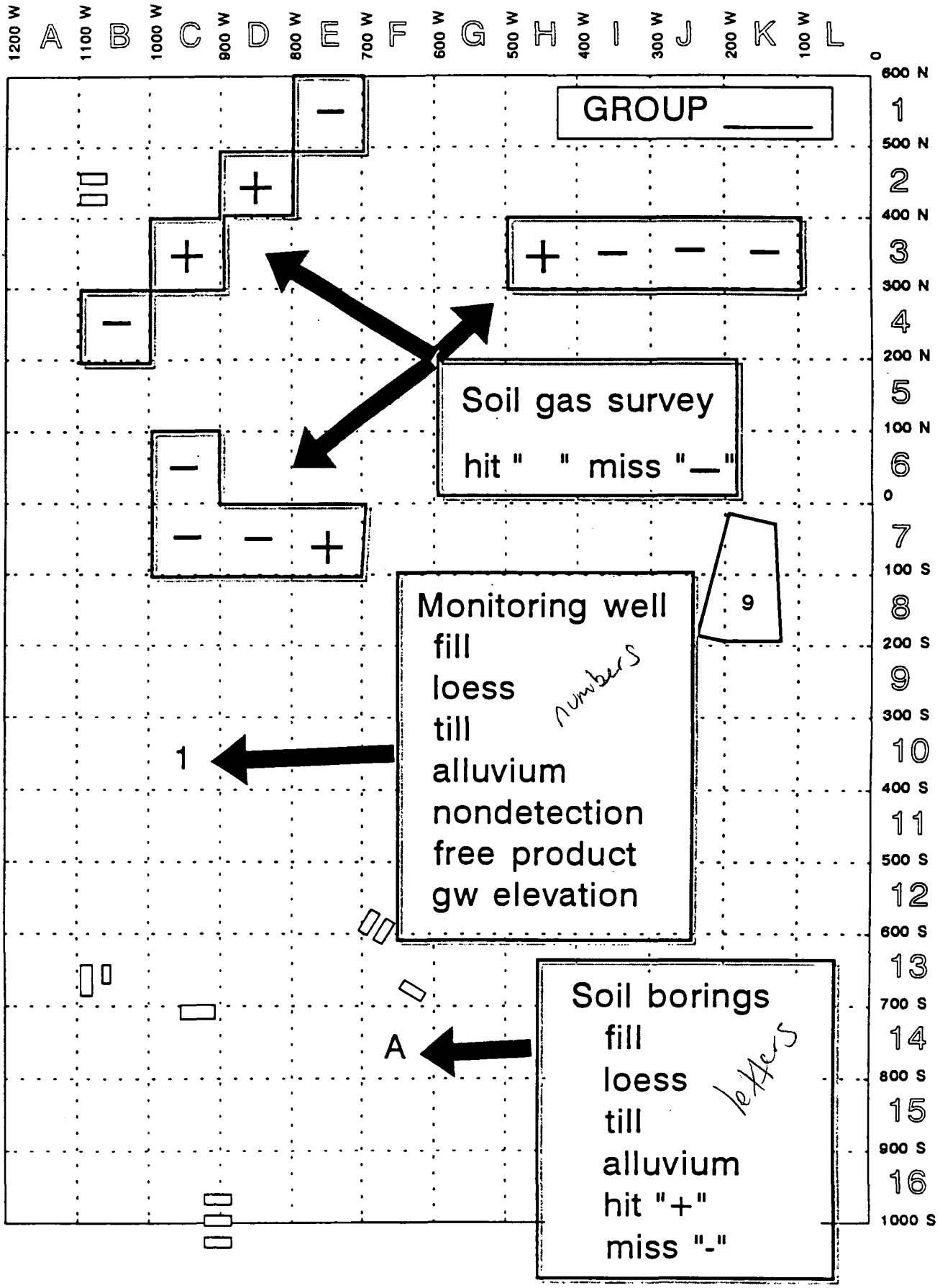
SOIL BORINGS

SOIL BORING	A	B	C	D	E	F	G	H	I	J
GRID LOCATION										
FILL										
LOESS										
ALLUVIUM										
TILL										
HIT (+)										
MISS (-)										

A

20' Loess

5' till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GROUP ____
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loess
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alluvium
nondetection
free product
gw elevation
numbers
Soil borings
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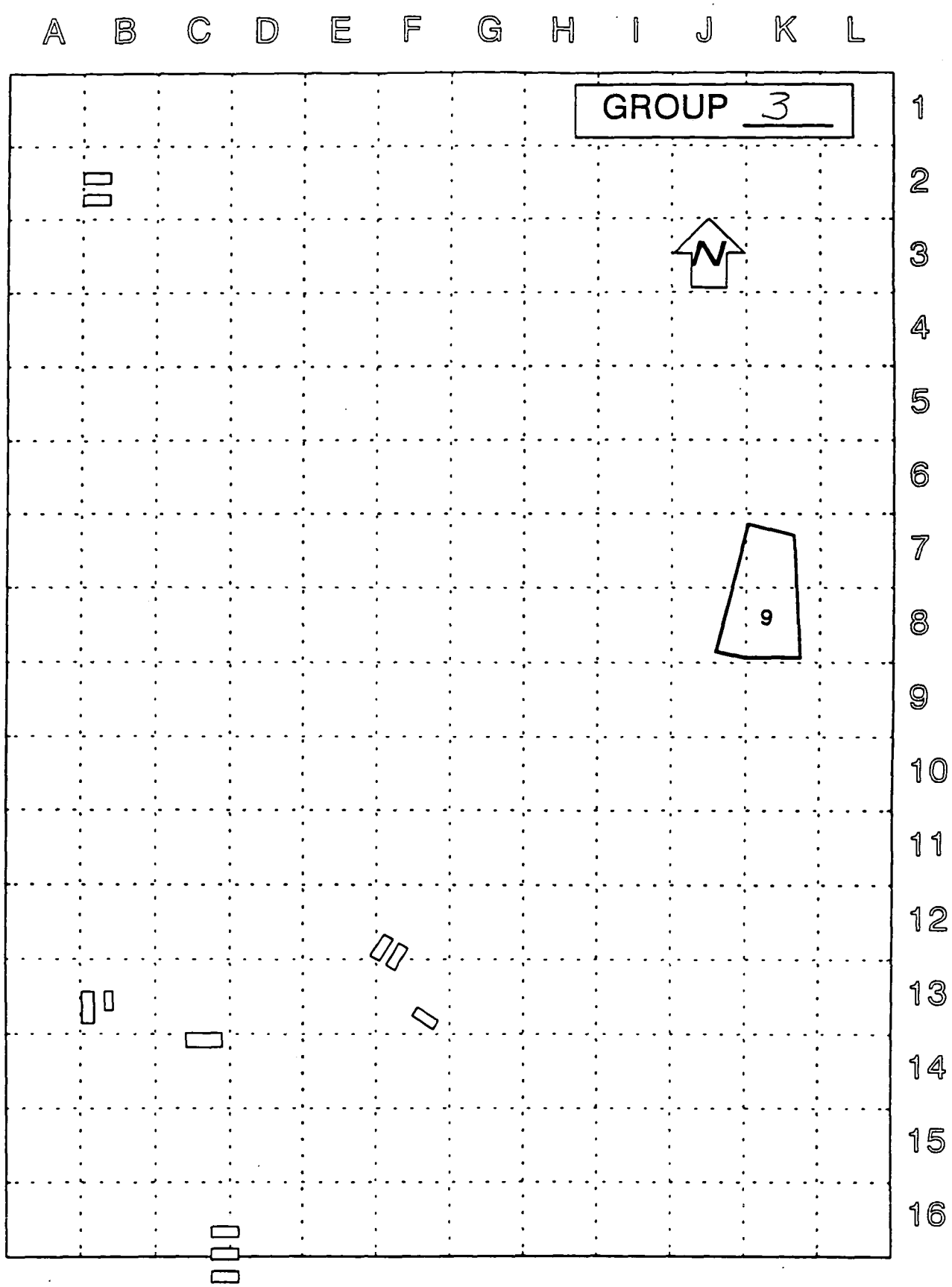
A B C D E F G H I J K L
GROUP 3
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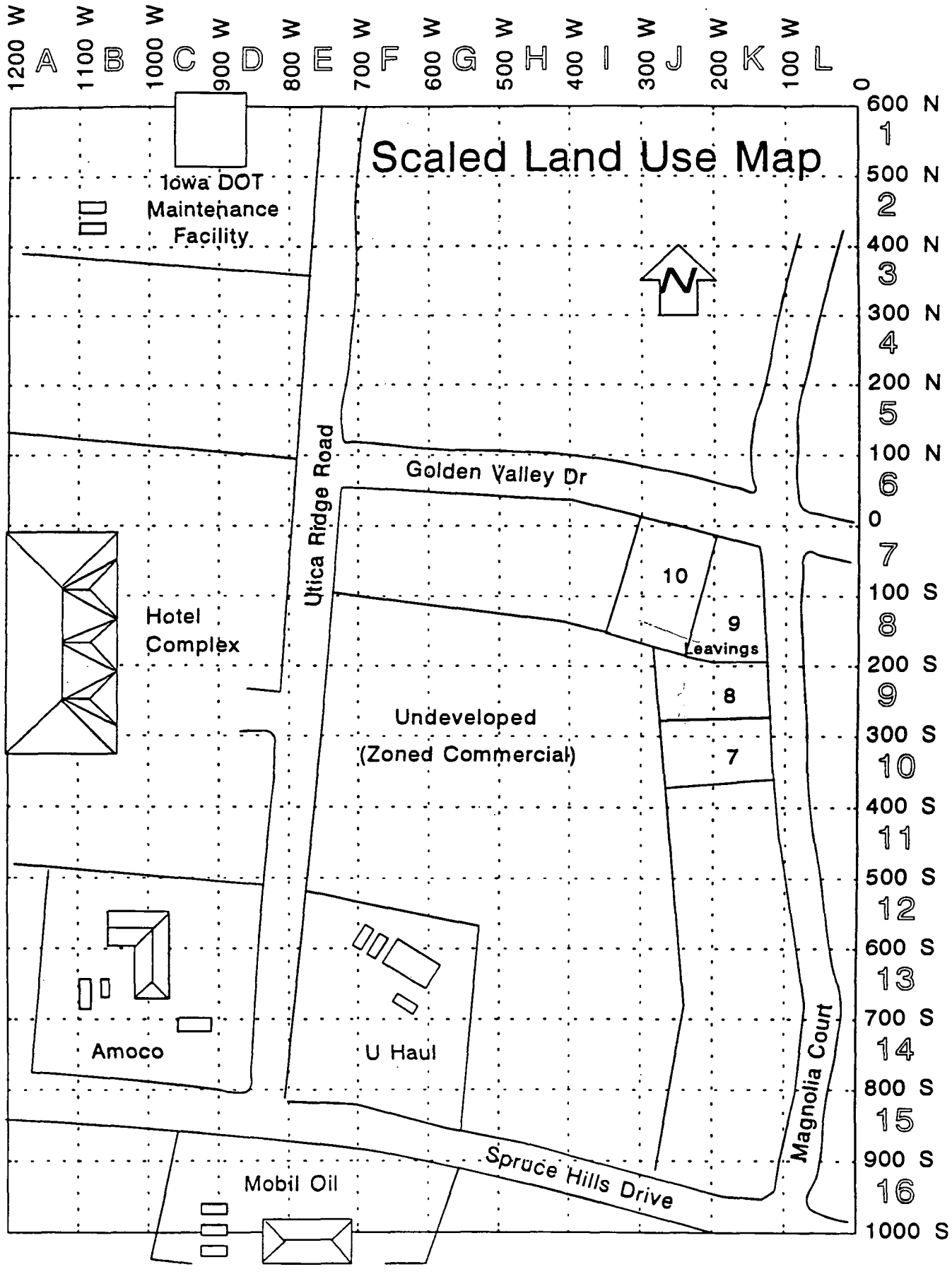
9


Scaled Land Use Map
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800 S
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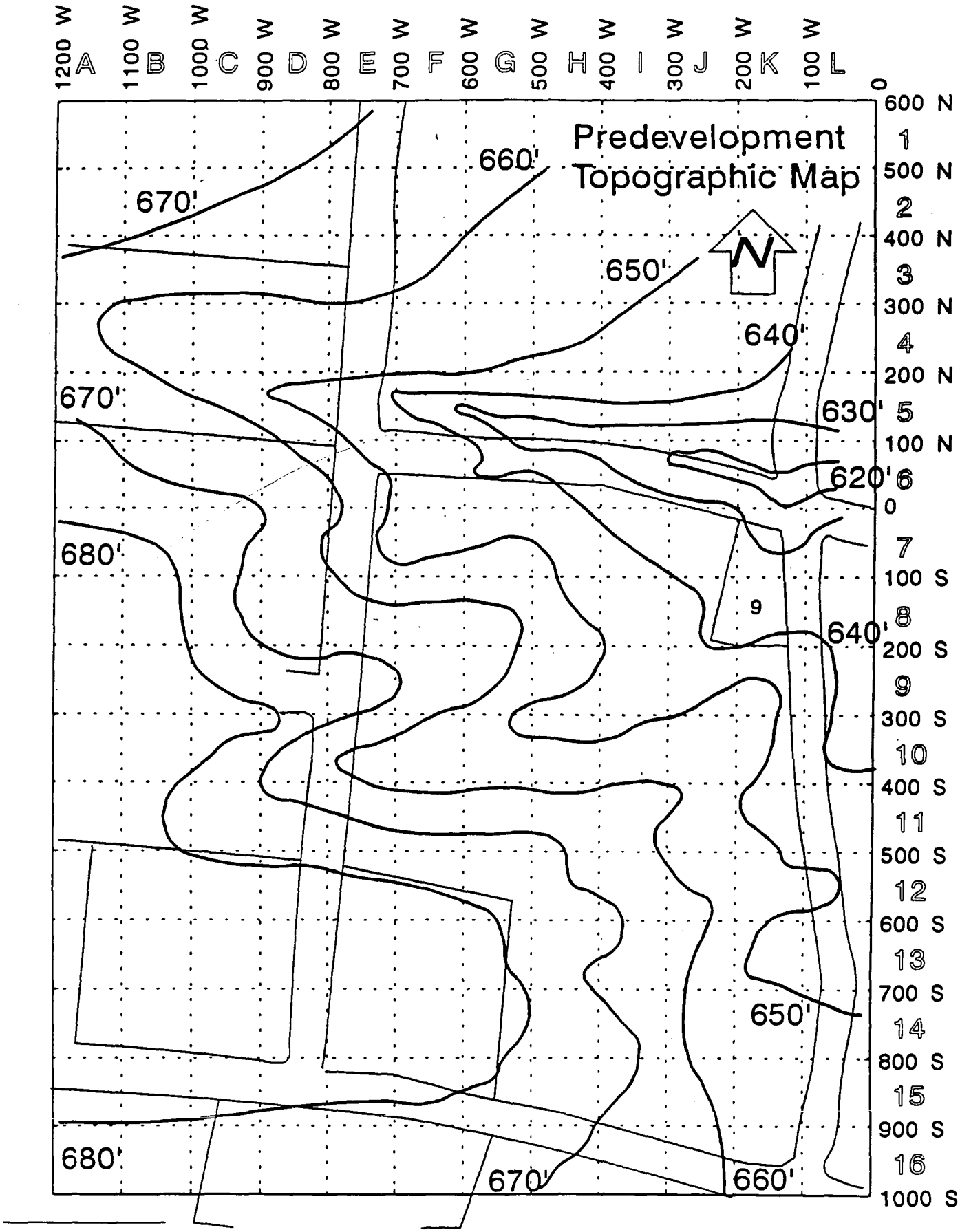
900 S
16
1000 S
Iowa DOT
Maintenance
Facility
N
Golden Valley Dr
Utica Ridge Road
Hotel
Complex
10
9
Leavings
8
7
Undeveloped
(Zoned Commercial)
Amoco
U Haul
Mobil Oil
Spruce Hills Drive
Magnolia Court


Predevelopment
Topographic Map
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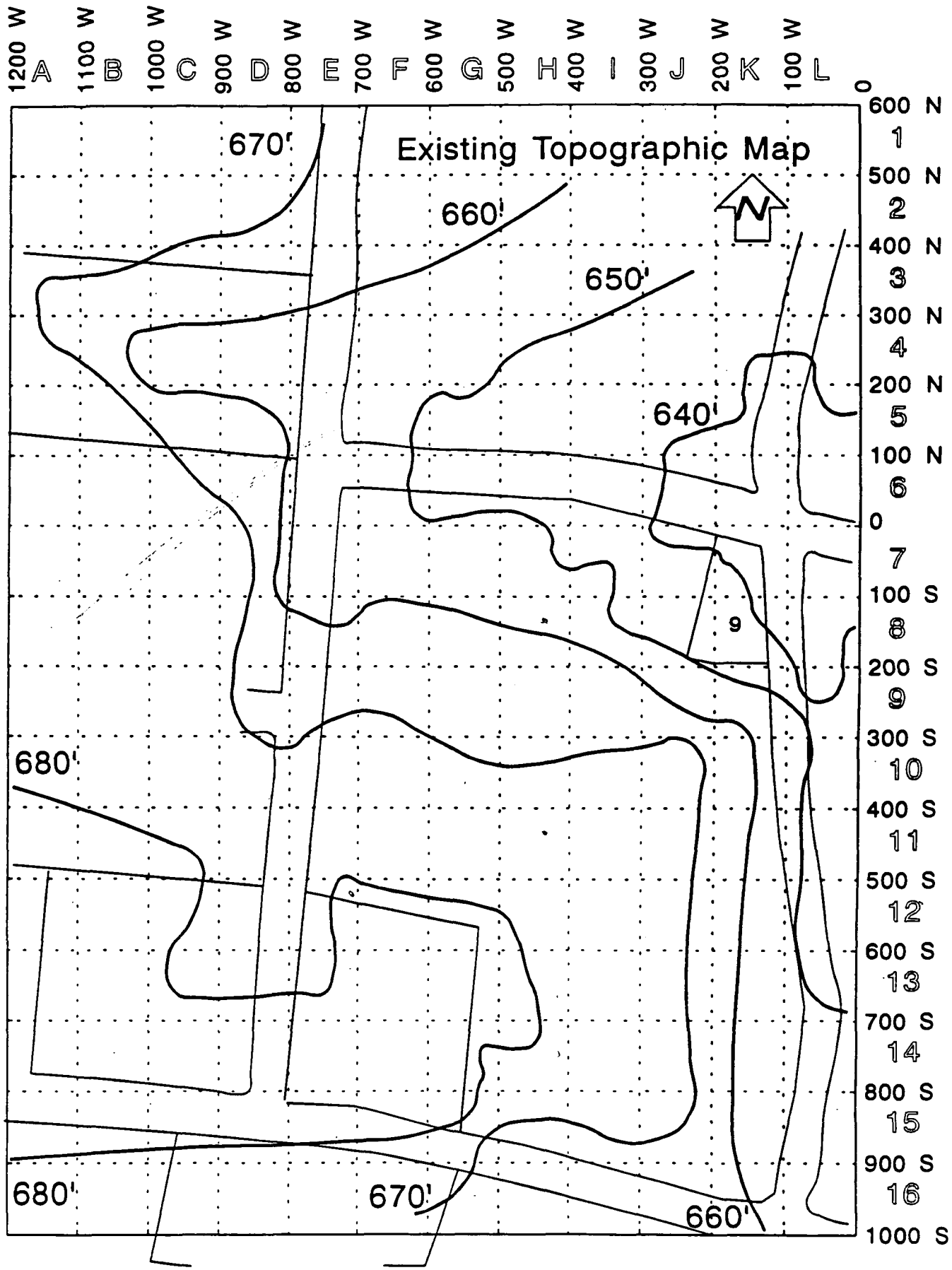
Existing Topographic Map
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APPENDIX A

Checklist for a Hydrogeological Investigation

CHECKLIST FOR A HYDROGEOLOGICAL INVESTIGATION

HAZARDOUS WASTE SITES INFORMATION LIST

When evaluating activities at sites where hazardous wastes may be causing or contributing to groundwater contamination, it is important to gather as much information as possible. The development of as much site information as possible can often provide valuable insight about site history, waste disposal practices, regional and local geology, and the potential for impacts to the environment in the site vicinity.

To make your information-gathering efforts easier, the following checklist includes some of the types of questions that could helpful to ask during a site investigation. Although these questions are oriented more toward field activities, they may also prove to be helpful to those people responsible for evaluating the adequacy of other site assessment documents.

Sources

National Water Well Association. 1991. Groundwater and Unsaturated Zone Monitoring and Sampling. 45 pp. *In:* Practical Handbook of Groundwater Monitoring.

U.S. EPA. 1986. RCRA Ground Water Monitoring Technical Enforcement Guidance Document. 208 pp.

Stropes, D.F. 1987. Unpublished Research: Technical Review of Hazardous Wastes Disposal Sites. 25 pp.

I. SITE/FACILITY HISTORY

A. Waste disposal history of the site.

1. Is this a material spill or other emergency response activity not at a Toxic Substances Storage and Disposal Facility (TSSDF)?

2. What hazardous wastes are being manufactured, stored, treated, or disposed of at the site?

3. For active manufacturing operations, what industrial processes are being used and what raw materials are used in the industrial processes?

4. Are the raw materials altered or transformed in any way during industrial processes to result in waste materials that are different from the raw materials?

5. How long has the facility been in operation?

6. Have the types of hazardous wastes manufactured, stored, treated, or disposed of at the site changed during the history of the site?

7. Have the industrial processes used at the site changed over the history of the site?

8. If the industrial processes are different, what previous industrial processes were used in the past, how long were they used, and what types of wastes were end products of the processes?

9. What environmental media (i.e., air, land, or water) have been or are being affected by the facility/site activities?

10. What is the form of the site wastes (e.g., sludge, slurry, liquid, powder, containerized, bulk storage)?

11. How much waste is generated or disposed of at the location daily?

12. What is the history of aboveground and underground storage tank use at the site?

13. What types of regulated manufacturing or pollution control units exist at the facility?

14. What governmental agencies are responsible for the regulated units?

15. Do any historical records about the site exist? If so, where are these records?

16. Has a check of any existing historical maps or aerial photos been performed to provide further insight about past site activities?

17. Is there any history of groundwater contamination as a result of the site activities?

B. Details of the site disposal activities.

1. Are the site disposal units currently in compliance with all applicable rules, regulations, and standards?

2. Are disposal areas isolated from the subsurface by the use of liners, impermeable material, etc.?

3. What type of isolating material is in use?

4. Are multiple isolation systems in use?

5. Is a leachate/contaminant collection system in use?

6. Are any monitoring wells installed adjacent to the disposal/collection system units?

7. Is there a surface water runoff control system?

8. Are any parts of the site/facility capped with an impermeable cover material?

9. What is the condition of the cap?

10. Are areas of previous waste disposal well defined?

C. What is the nature of groundwater usage from aquifers beneath the site or in adjacent areas?

1. Do any water supply wells exist in the aquifers beneath the site and adjacent areas?

2. Are water supply wells used for potable water supplies or for industrial process water?

3. Is the groundwater treated prior to use?

4. What are the pump rates of the water supply wells? Daily? Monthly? Annually?

5. What are the depths of the wells' screened intervals?

6. What other well drilling, well construction, or well completion information is available?

7. Do subsurface geologic well logs exist for the wells?

8. Are the wells upgradient, at, or downgradient of the site/facility?

9. Does pumping from these wells modify the regional groundwater table or potentiometric surface?

II. HYDROGEOLOGIC CHARACTERIZATION

A. Has the purpose of the hydrogeologic investigation been clearly and adequately defined?

1. Characterize the hydrogeologic system at the site.

2. Determine whether there has been downgradient degradation of water quality from a potential source of contamination.

3. Determine the upgradient source of contamination at a known downgradient contamination receptor (well, spring, or surface water body).

B. Has the site location and all major site features been shown on a map?

1. Has the site been located on a state map?

2. Has the site been located on a USGS 7-1/2 minute topographic quadrangle map published at a scale of 1:25,000?

3. Have coordinates for further site identification (latitude, longitude, degrees, minutes, seconds, or a site-specific grid system) been provided?

C. Has a base map of the site been prepared?

1. What is the map source?

2. Are aerial photos available?

3. Are all components of a map (north arrow, scale, map legend) shown and defined on the base map?

4. Does the map show elevations and contours?

5. Is the scale of the map adequate to delineate dimensions of onsite features adequately?

6. Does the map show features adjacent to the site that may be pertinent to the hydrogeologic investigation?

7. Are all natural physical features (e.g., topography, surface waters, surface water flow divides) shown on the map?

D. Has the subsurface geology been identified?

1. Is the geologic interpretation based on soil borings and well drilling logs?

2. Have any other reference materials been used?

3. Are aquifers present beneath the site?

4. Is the first aquifer encountered confined or unconfined?

5. Are all aquifers and confining units continuous across the site?

6. Have all geologic strata been described (e.g., thickness, rock type, unconsolidated/consolidated materials, depth)?

7. Do multiple aquifers exist at the site?

8. Have any porous vs. fractured flow media been described?

E. Do the driller's logs of the deepest borings at each well cluster show that soil material samples were collected at 5-ft intervals? If not, at what intervals were samples taken?

1. Is there a stratigraphic log of the deepest boreholes?

2. Were the borings extended to a depth below any confining beds beneath the shallowest aquifer?

3. Have enough borings of the area been done to adequately define the continuity and thickness of any confining beds?

4. Have all logs been prepared by a qualified geologist, soil scientist, or engineer using a standardized classification system?

5. Were any laboratory tests conducted on the soil and soil material samples? What types of tests were performed?

6. Were grain size distributions used to determine the size of the gravel pack or was sand filter placed in the annular space opposite the well screen?

F. Have field and/or laboratory permeability tests been performed to identify variations in aquifer and confining bed properties?

1. What type(s) of tests were performed?

2. What was the range of hydraulic conductivity values found in the aquifer? What was the arithmetic mean value?

3. What was the range of hydraulic conductivity values found in the confining bed? What was the arithmetic mean value?

4. Where are the most permeable subsurface zones located relative to the waste disposal facility?

5. Have geologic cross sections been constructed?

G. Have field and/or laboratory tests been performed to determine the specific yield, storativity, or effective porosity of the aquifer?

1. What type(s) of tests were performed?

2. What is the range of specific yield, storativity, or effective porosity values?

3. What are the average values?

H. Has the horizontal groundwater flow direction been determined?

1. Have a minimum of three piezometers been installed to determine the direction of flow in the aquifer?

2. Do any water-level readings show local variations of the water table caused by mounds or sinks?

3. Do any identified mounds or sinks result in alterations of the regional or local horizontal groundwater direction of flow?

4. Do any surface features which may have an effect on the horizontal flow exist?

5. Have all piezometer installations in the uppermost aquifer been screened at approximately the same depth below the water table?

6. Do any discernible seasonal variations in water levels exist?

7. Do any short-term variations in water levels exist? If so, what possible causes may explain these variations?

I. Has the magnitude of the horizontal hydraulic gradient been determined at various locations across the site?

1. What is the average horizontal hydraulic gradient at the site?

2. Where is the horizontal hydraulic gradient the steepest?

3. Does this location correlate to a known area of lower hydraulic conductivity in the aquifer?

4. Does this location correlate to a known area of lower aquifer thickness?

5. Where is the horizontal hydraulic gradient the lowest (flat)?

6. Does this location correlate to a known area of greater hydraulic conductivity in the aquifer?

7. Does this location correlate to a known area of greater aquifer thickness?

J. If multiple aquifers exist, have wells been installed in each aquifer to determine the vertical component of groundwater flow?

1. Have the wells in each aquifer been installed in a single borehole or in separate boreholes?

2. If a single borehole was used, what tests were conducted to ensure that no leakage between the upper and lower aquifers exist?

3. If a single borehole was used, what well installation, construction, and development techniques were used to ensure that no leakage between the upper and lower aquifers exist?

4. Based on the difference in hydraulic head between upper and lower aquifers, can the site be described as:

   a. Predominantly a recharge area?

   b. Predominantly a discharge area?

   c. Predominantly an area of horizontal flow?

5. If recharge, discharge, or horizontal flow areas exist, have these locations been shown on a hydrogeologic map (including supporting cross sections) of the site?

K. Has the magnitude of the vertical hydraulic gradients been determined at various locations across the site?

1. What is the average vertical hydraulic gradient at the site?

2. Where is the vertical hydraulic gradient the steepest?

3. Can this location be correlated to any known areas of lower hydraulic conductivity?

4. Where is the vertical hydraulic gradient the flattest?

5. Based on the vertical hydraulic gradient, what relationship exists between the shallow and deeper aquifers?

6. Is there any regional or offsite vertical hydraulic gradient information that may support or conflict with the site's vertical hydraulic gradient data?

L. Determination of seepage velocities and travel times.

1. What is the average seepage velocity of water moving from the waste facility to the downgradient site boundary?
2. What is the average travel time of water to move from the waste facility to the nearest downgradient monitoring wells?

3. What is the basis for the seepage velocity and travel time determinations?

M. Have potentiometric maps, flow nets, geologic maps, and cross sections been prepared to show the direction of groundwater flow at the site?

Horizontal Flow Components (plan view)

1. Do the contours and contour intervals between the equipotential lines adequately describe the flow regime?

   Suggested Contour Intervals:

   a. 0.1–0.5 ft if the horizontal flow component is relatively flat.

   b. 0.5–1.0 ft if the horizontal flow component is moderately steep.

   c. 1.0–5.0 ft if the horizontal flow component is extremely steep.

2. Have the equipotential lines been accurately drawn:

   a. With respect to the elevations of water levels in the wells or piezometers?

   b. With respect to nearby or onsite rivers, lakes, wells, or other boundary conditions?

   c. With respect to other naturally occurring or man-made physical features that might cause groundwater mounds or sinks in the area?

3. Do variations in the spacing of equipotential lines correspond to known areas with relative transmissivity variations?

4. Do the constructed groundwater-flow lines cross equipotential lines at right angles?

5. Can conclusions about the aquifer(s) relative homogeneity and isotropy be made based on variations in the flow lines?

Vertical Flow Components (cross sections)

1. Is the transect for the vertical flow component cross section(s) laid out along the line of a groundwater flow path as seen in the plan view?

2. Is the variation in land-surface topography accurately represented on the cross section(s)?

3. Have both vertical and horizontal scales been provided?

4. What differences exist between the vertical and horizontal scales?

5. Are all monitoring wells, piezometers, and screened intervals accurately shown?

N. What is the site water quality and geochemistry?

1. What are the upgradient groundwater quality conditions?

2. What are the downgradient groundwater quality conditions?

3. What water-quality parameters have been determined downhole?

4. What water-quality parameters have been determined at the well head?

5. Have all appropriate field water-quality determinations, equipment selections, and procedures been followed?

6. Does an adequate QA/QC procedure exist?

7. What if any, relationship exists between the site water-quality conditions and the past and/or present activities at the site?

III. DETECTION MONITORING SYSTEM

A. Are the facility upgradient and downgradient monitoring wells properly located to detect any water-quality degradation from the waste source(s)?

Horizontal Flow

1. Will groundwater from the upgradient well locations flow through or under the waste source in an unconfined aquifer?

2. Will groundwater from the upgradient well locations flow beneath the waste source and under an overlying confining bed in a confined aquifer?

3. Will groundwater from the upgradient well locations flow beneath the waste source in an unconfined aquifer separated from the waste source by an impervious liner?

4. Will groundwater from the waste source area flow toward downgradient wells?

Vertical Flow

1. Are the monitoring wells correctly screened to intercept a possible contaminant plume from the waste source based on an accurate interpretation of the vertical flow regime (recharge area, discharge area, or area of horizontal flow)?

B. Are the monitoring wells located adequately to provide sufficient groundwater flow information?

1. Are regional water levels unaffected by local groundwater mounds or sinks?

2. Are additional monitoring wells located to provide water-level information from local groundwater mounds or sinks?

3. Do upgradient and downgradient monitoring wells provide representative samples?

C. Monitoring Well Construction

1. Were precautions taken during the drilling of the borehole and installation of the well to prevent introduction of contaminants into the well?

2. Is the well casing and screen material inert to the probable major contaminants of interest?

3. What type of well casing and screen material was used?

4. Does the casing and screen material manufacturer have any available information about possible leaching of contaminants from the casing and screen material?

5. How are the well casing and screen segments connected?

6. If cement or glue has been used, what is the potential for contaminants to leach into the groundwater?

7. Were all downhole well components steam cleaned prior to installation?

8. If another cleaning technique was used, what materials were used?

D. Are there *as-built* drawings or details of each monitoring well nest or cluster showing information such as depth of well, screen intervals, type and size of screen, length of screen and riser, filter packs, seals, and protective casings?

1. Do the figures show design details of as-built wells as opposed to details of proposed wells?

2. Are the well depth(s) and diameter(s) shown?

3. Are the screened intervals and type and size of screen openings shown?

4. Is the length of the screen shown?

5. Is the length of the riser pipe and stick-up above the land surface shown?

6. Is the filter pack around the screened interval shown?

7. Does the filter pack extend at least 1 ft above and below the screened intervals?

8. What type of sealant was placed in the annulus above the filter pack?

9. What is the thickness of the seal?

10. How was the seal put in place?

11. Will a protective casing or reinforced posts be necessary to protect the monitoring well from damage?

12. Is any manufacturer's information available to verify that all materials used in the well construction do not represent potential sources of water contamination?

13. Have samples of well construction materials been kept for future analysis to verify that the materials do not represent sources of water contamination?

E. Are the screened intervals appropriate to the geologic setting and the sampling of a potential problem?

1. Is the screen set opposite a stratigraphic layer with relatively high hydraulic conductivity?

2. Is the screened interval set sufficiently below the water table so that water-level measurements can be taken and water samples can be collected during periods of low water level?

3. Is the screened interval placed in the aquifer(s) of concern?

4. If a single long screen was installed over the entire saturated thickness of the aquifer, what effect will this have on analytical data from this monitoring well?

5. Has the entire aquifer thickness been penetrated and screened?

6. Have piezometers been installed to determine vertical and horizontal flow directions?

7. Is the base of the waste disposal unit above the seasonal high water table?

8. What is the thickness of the unsaturated zone between the base of the waste disposal unit and the seasonal high water table?

F. Has a professional survey been conducted to determine the elevation and location of the measuring point at each well with reference to a common datum?

1. Is the survey at each monitoring well accurate to ± 0.01 ft?

2. Is each surveyed measuring point located at the top of the well casing?

3. What benchmark was used as a starting point for the survey?

4. Are the elevations of the measuring point at each well referenced to mean sea level and not to some local datum?

G. Has an adequate sampling and analysis program been written for the site?

1. Are the major contaminants inorganic compounds?

2. Will field filtering and preservation be done in the field?

3. Are the major contaminants organic coumpounds?

4. Where would you expect to find the contaminants in the aquifer?

   a. Floating at the top of the aquifer (LNAPLs)?

   b. Dissolved in the groundwater and flowing with it?

   c. Concentrated at the bottom of the aquifer (DNAPLs)?

5. What are the possible degradation end products of the original organic contaminants?

6. Is the sampling method adequate to prevent any loss of volatile constituents?

7. Are field measurements such as pH, Eh, specific conductance, dissolved oxygen, and temperature taken in the field?

8. Does a generic sampling and analysis protocol exist?

9. Does the sampling and analysis protocol address sample preservation, storage, transport, container identification, and chain-of-custody procedures?

10. Is the analytical laboratory certified by EPA for the analyses to be performed?

11. Did the laboratory provide input to the sampling and analysis program?

12. Is the sampling and analysis program written, clear, concise, understandable and site specific?

H. Has a QA/QC plan been written for the groundwater monitoring program?

Water-Level Measurements

1. Have worksheets containing relevant fixed data, some of which are indicated below, been prepared for use by the person taking water-level readings?

   a. Well identification number?

   b. Location of measuring point of each well?

   c. Elevation of measuring point at each well relative to mean sea level?

   d. Elevations of screened interval at each well?

   e. Type of measuring instrument to be used?

2. Do the worksheets for use by the person taking water-level readings have columns for computation of:

   a. Depth to the water table?

   b. Measuring point data to be added to or subtracted from readings of measuring instrument?

   c. Adjusted depth to water surface?

   d. Conversion of depth to water surface?

3. Do the worksheets have a space for pertinent comments?

Sample Collection

1. Are well purging procedures prior to sampling described as written procedures?

2. Is the method of purging specified?

3. Is the sample collection technique specified?

4. Is the sample storage vessel described?

5. Is the sample volume specified?

6. Is the sample identification system described?

7. Are there provisions for:

a. Trip blanks?

b. Spiked samples?

c. Duplicate, replicate, or blind samples?

8. Is the sampling frequency specified?

Sample Analysis

1. Does the laboratory have a QA/QC program for all samples?

2. Is the QA/QC program written?

3. Does the laboratory provide information on the accuracy and precision of the analytical results?

4. Did the laboratory participate in the development of the sampling and analysis plan for the groundwater monitoring program and the QA/QC plan for the nonlaboratory portion of the sampling program?

I. Is the QA/QC plan being followed during implementation of the sampling and analysis program?

1. Is the same consultant who prepared the QA/QC plan responsible for its implementation?

2. How many copies are there of the QA/QC plan?

3. Who has copies and where are they located?

4. Does the field person taking water-level measurements and collecting samples have a copy?

5. Does the field person understand the importance of following the QA/QC plan explicitly each time?

6. What safeguards and checks are there to ensure there will be no deviation from the QA/QC plan in the field and the laboratory?

J. If any more field work or data will be necessary to meet the objectives of the hydrogeologic investigation, what types of additional field installations and data will be needed?

APPENDIX B

Sampling Protocols

GENERALIZED GROUNDWATER SAMPLING PROTOCOL

Step	Goal	Recommendations
Hydrologic measurements	Establish nonpumping water level	Measure the water level to ±0.3 cm (±0.01 ft)
Well purging	Remove or isolate stagnant H_2O, which would otherwise bias representative sample	Pump water until well purging parameters (e.g., pH, T, Ω^{-1}, Eh) stabilize to ±10% over at least two successive well volumes pumped
Sample collection	Collect samples at land surface or in well bore with minimal disturbance of sample chemistry	Pumping rates should be limited to ~100 mL/min for volatile organics and gas-sensitive parameters
Filtration/preservation	Filtration permits determination of soluble constituents and is a form of preservation; it should be done in the field as soon as possible after sample collection	For trace metals, inorganic anions/cations, and alkalinity. Do not filter TOC, TOX, or other volatile organic compound samples; filter other organic compound samples only when required
Field determinations	Field analyses of samples will effectively avoid bias in determining parameters/constituents that do not store well (e.g., gases, alkalinity, and pH)	Samples for determining gases, alkalinity, and pH should be analyzed in the field if at all possible
Field blanks/standards	These blanks and standards will permit the correction of analytical results for changes that may occur after sample collection. Preserve, store, and transport with other samples.	At least one blank and one standard for each sensitive parameter should be made up in the field on each day of sampling. Spiked samples are also recommended for good QA/QC.
Sample storage, transportation, and chain of custody (COC)	Refrigerate and protect samples to minimize their chemical alteration prior to analysis. Document movement of samples from collector to laboratory.	Observe maximum sample holding or storage periods recommended by EPA. Documentation of actual holding periods should be carefully performed. Establish COC forms, which must accompany all samples during shipment.

Adapted from: U.S. EPA. 1985. Practical Guide for Ground-Water Sampling. EPA/600/2-85/104. Robert S. Kerr Environmental Research Laboratory, Ada, OK.

APPENDIX C

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APPENDIX D

Sources of Information

SOURCES OF INFORMATION

SOURCES OF U.S. ENVIRONMENTAL PROTECTION AGENCY DOCUMENTS

Center for Environmental Research Information (CERI) (no charge for documents)

Center for Environmental Research Information (CERI)
ORD Publications
26 West Martin Luther King Drive
Cincinnati, OH 45268
513 569-7562
FTS 8-684-7562

Public Information Center (PIC) (no charge for public domain documents)

Public Information Center (PIC)
U.S. Environmental Protection Agency
PM-211B
401 M Street, S.W.
Washington, DC 20460
202 382-2080
FTS 8-382-2080

Superfund Docket and Information Center (SDIC)

U.S. Environmental Protection Agency
Superfund Docket and Information Center (SDIC)
OS-245
401 M Street, S.W.
Washington, DC 20460
202 260-6940
FTS 8-382-6940

National Technical Information Services (NTIS) (cost varies)

National Technical Information Services (NTIS)
U.S. Department of Commerce
5285 Port Royal Road
Springfield, VA 22161
703 487-4650
1-800-553-NTIS(6847)

Superintendent of Documents

Government Printing Office
202 783-3238

SOURCES OF MODELS AND MODEL INFORMATION

Superfund Exposure Assessment Manual

EPA/540/1-88/001, April 1988
Chapter 3 "Contaminant Fate Analysis" - 35 models

National Ground Water Association

National Ground Water Association
6375 Riverside Dr.
Dublin, OH 43017
614 761-1711

International Groundwater Modeling Center (IGWMC)

Paul K. M. van der Heijde, Director IGWMC
Institute for Ground-Water Research and Education
Colorado School of Mines
Golden, CO 80401-1887
303 273-3103
303 273-3278 (fax)

Groundwater Flow Model

Groundwater Education of Michigan (GEM) Regional Center
Institute for Water Sciences
1024 Trimpe Hall
Western Michigan University
Kalamazoo, MI 49008
616 387-4986
Cost (as of 3/95): $275.00 (including shipping)

UST Video: Groundwater Cleanup

Industrial Training Systems Corp.
20 West Stow Road
Marlton, NJ 08053
609 983-7300
Cost: $595.00

GEOPHYSICS ADVISOR EXPERT SYSTEM VERSION 2.0

Gary R. Olhoeft, Jeff Lucius, Cathy Sanders
U.S. Geological Survey
Box 25046 DFC - Mail Stop 964
Denver, CO 80225
303 236-1413/1200

U.S. Geological Survey preliminary computer program for Geophysics Advisor Expert System. Distributed on 3.5" disk and written in True BASIC 2.01 to run under Microsoft MS-DOS 2.0 or later on IBM-PC or true compatible computers with 640k or greater memory available to the program. No source code is available.

This expert system program was created for the U.S. Environmental Protection Agency, Environmental Monitoring Systems Laboratory, Las Vegas, Nevada. The expert system is designed to assist and educate non-geophysicists in the use of geophysics at hazardous waste sites. It is not meant to replace the expert advice of competent geophysicists.

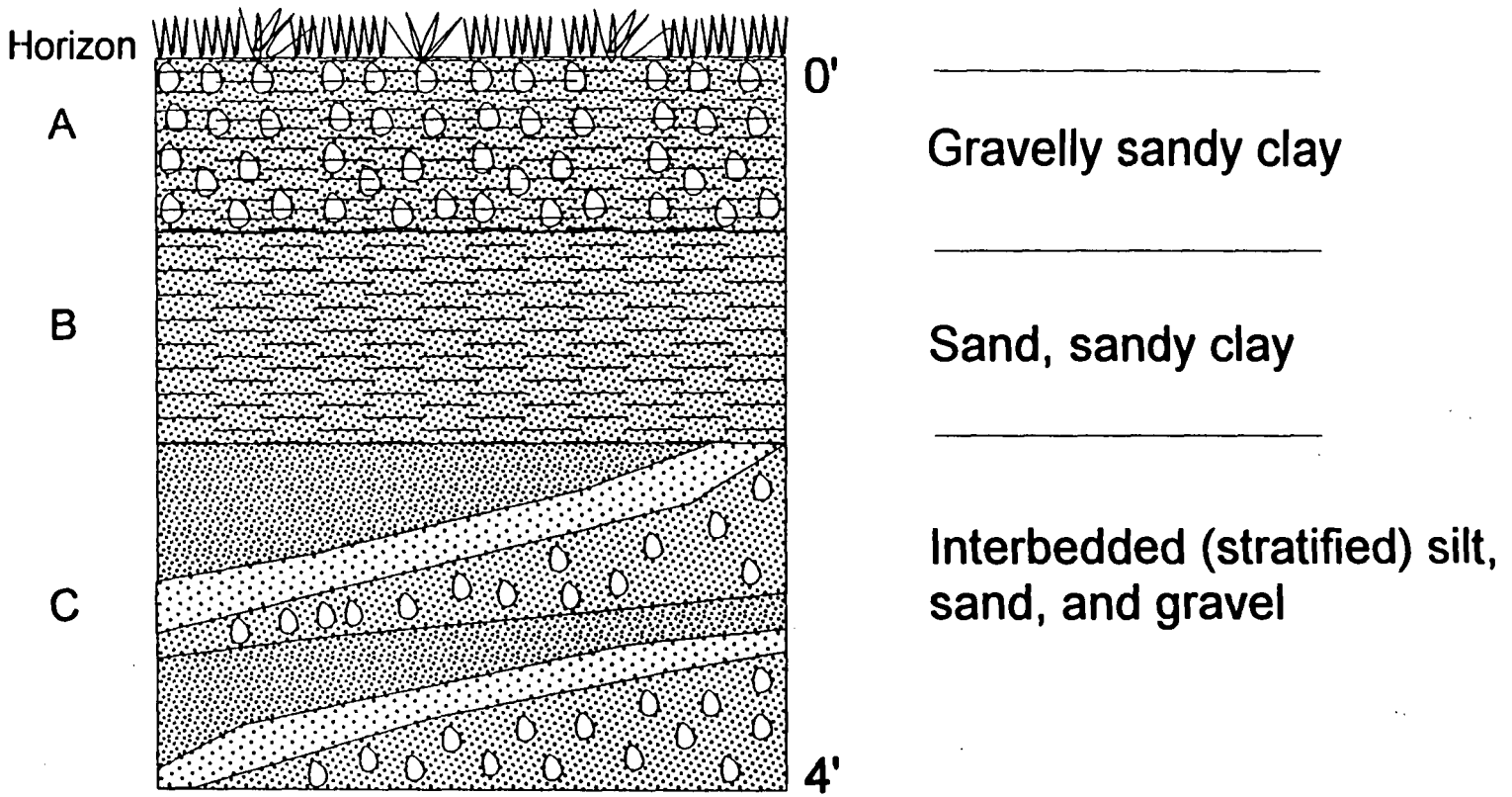
COMPREHENSIVE LISTING OF AERIAL PHOTOGRAPHY

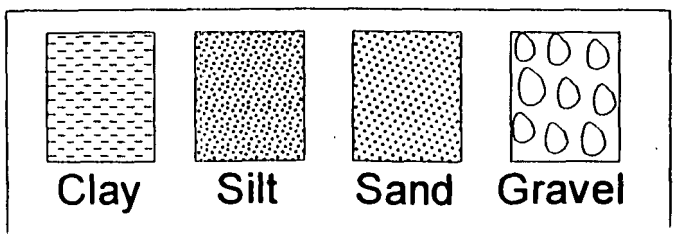
U.S. Department of Agriculture, ASCS
Aerial Photography Field Office
2222 West 2300 South
P.O. Box 30010
Salt Lake City, UT 84130-0010
801 524-5856

APPENDIX E

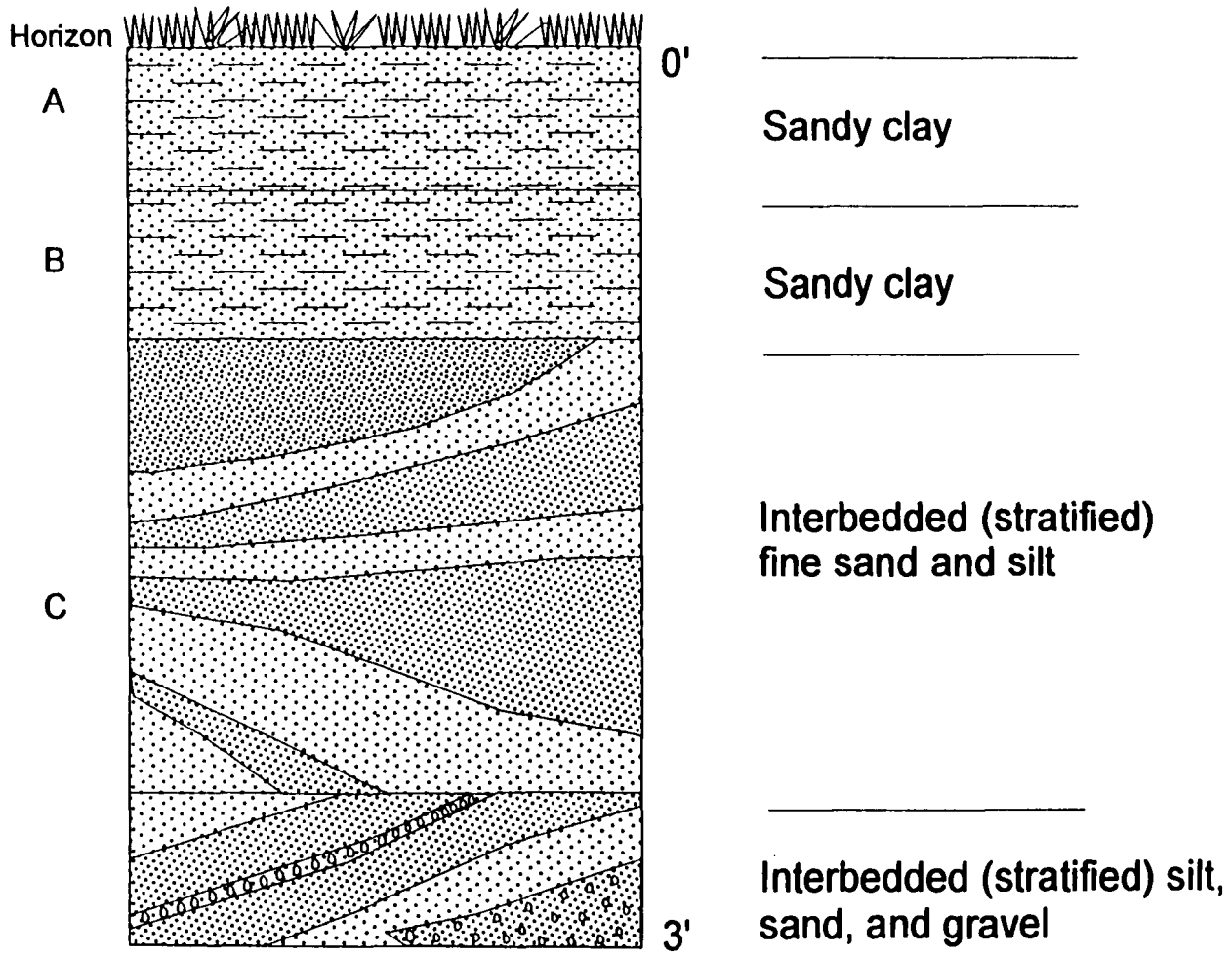
Soil Profiles

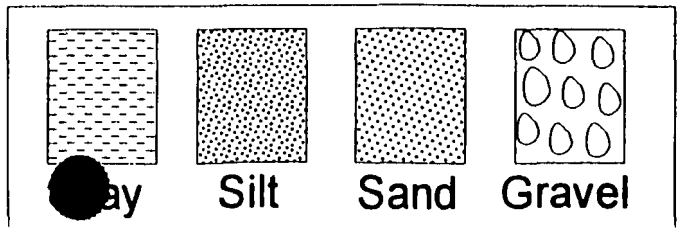
SOIL PROFILE DEVELOPED ON ALLUVIAL FAN DEPOSITS



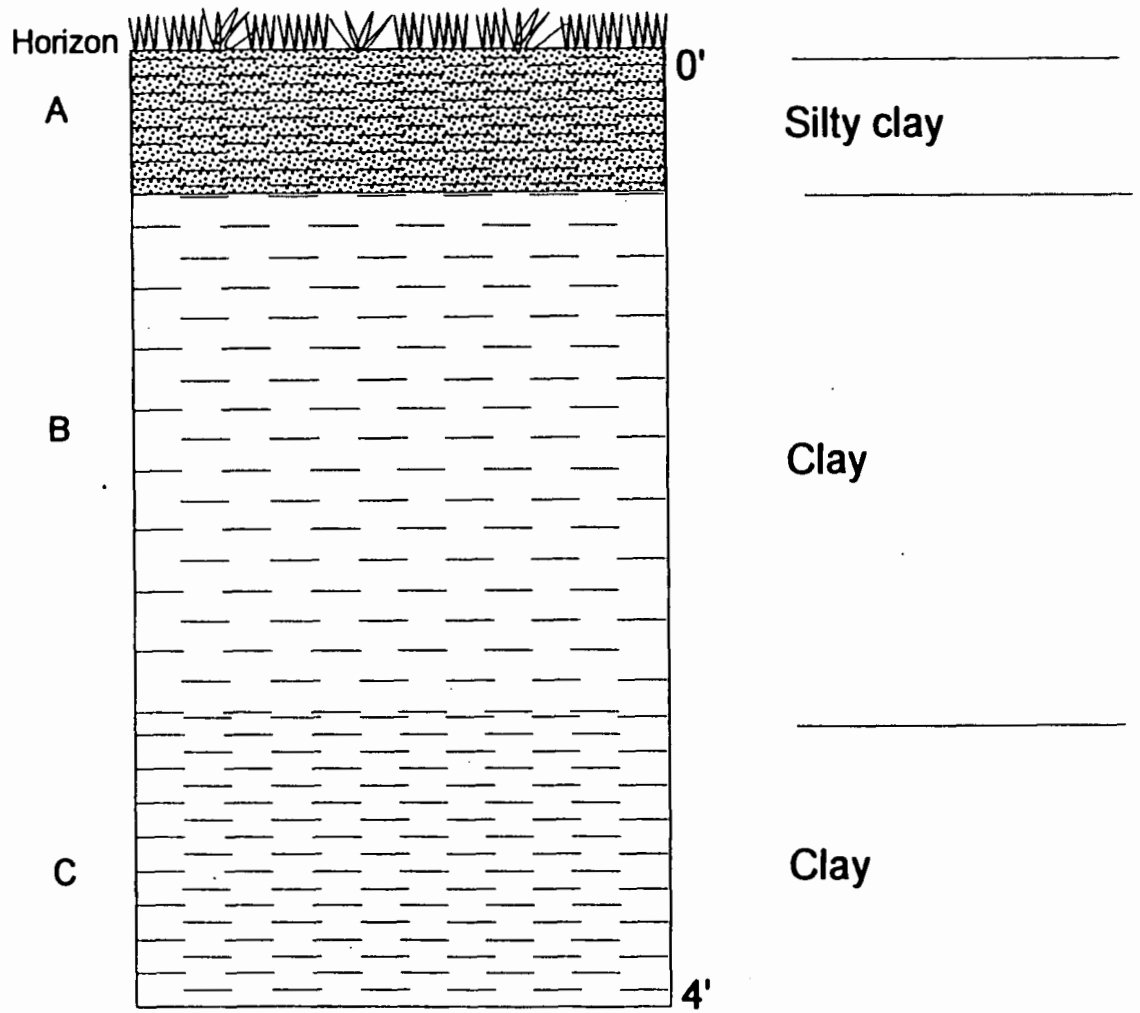


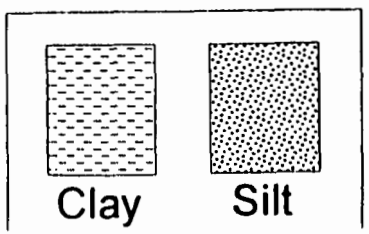
SOIL PROFILE DEVELOPED ON VALLEY FILL DEPOSITS



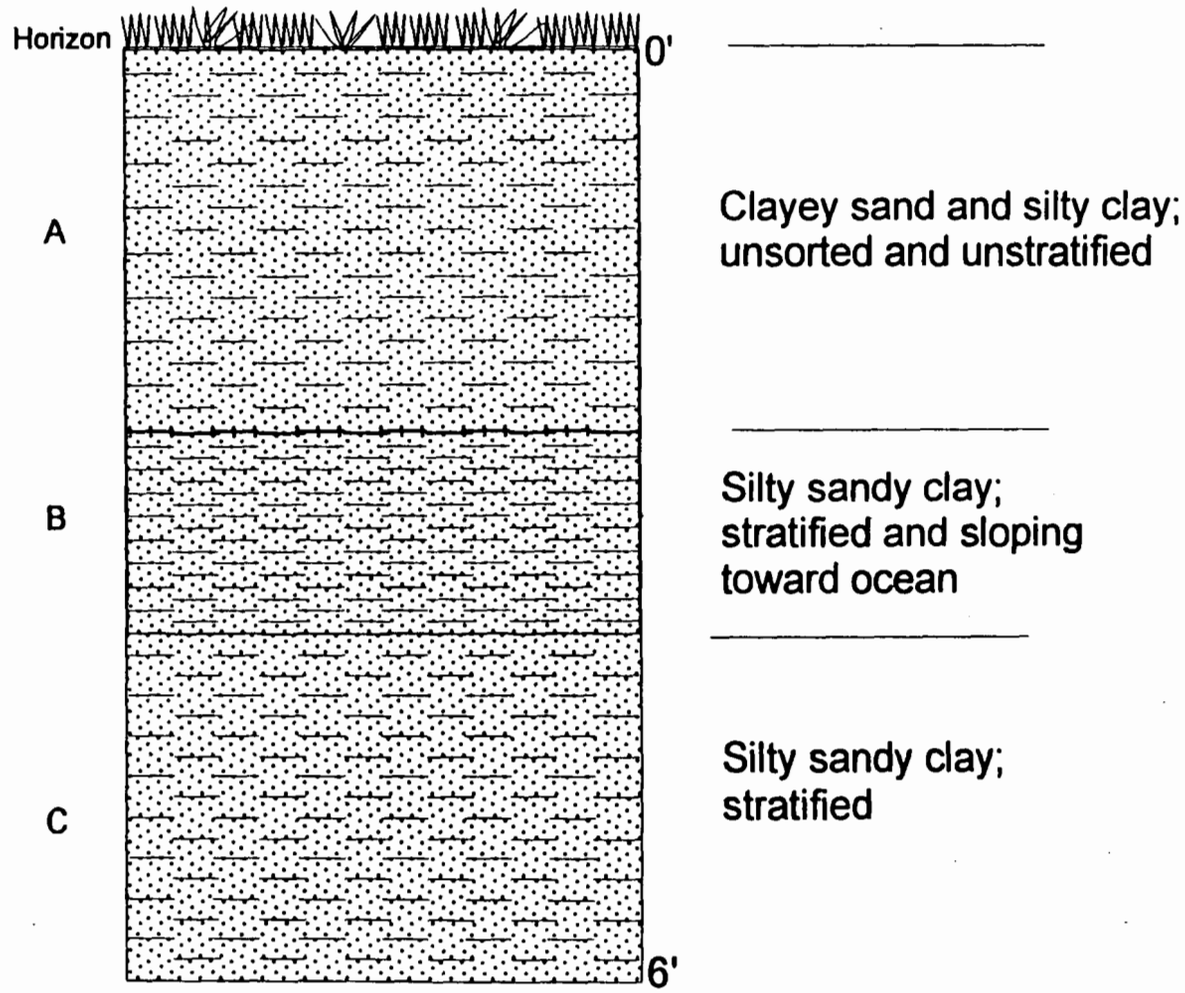


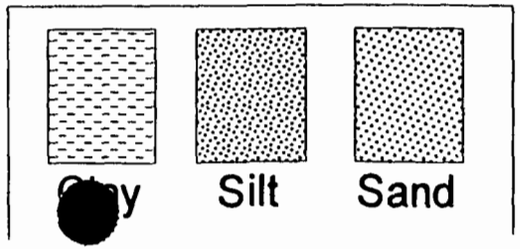
SOIL PROFILE DEVELOPED ON ALLUVIUM



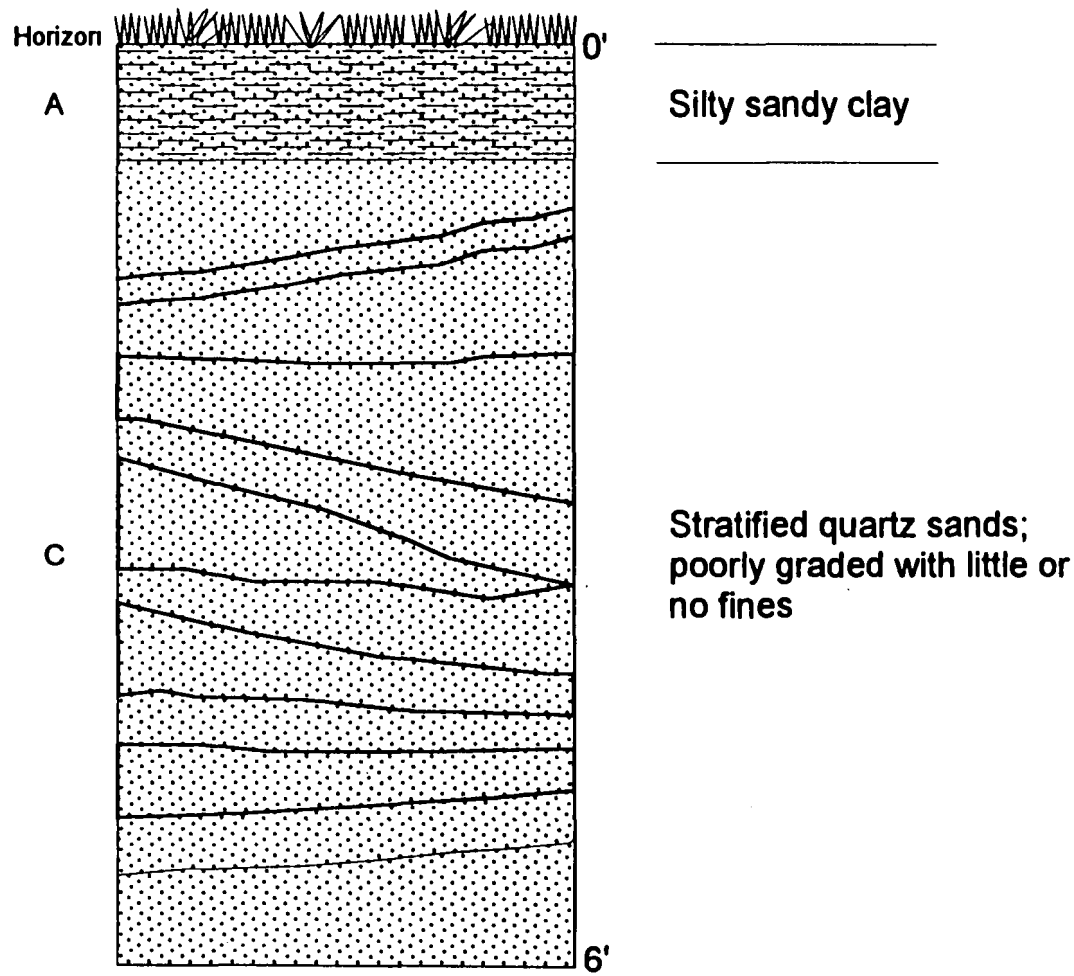


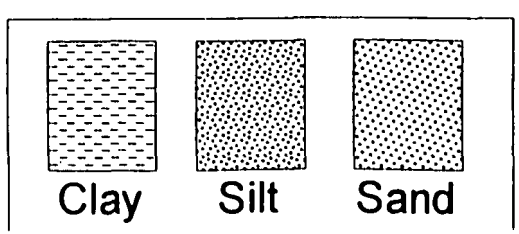
SOIL PROFILE DEVELOPED ON COASTAL PLAIN DEPOSITS IN A HUMID CLIMATE





SOIL PROFILE DEVELOPED ON COASTAL BEACH DEPOSITS IN A HUMID CLIMATE





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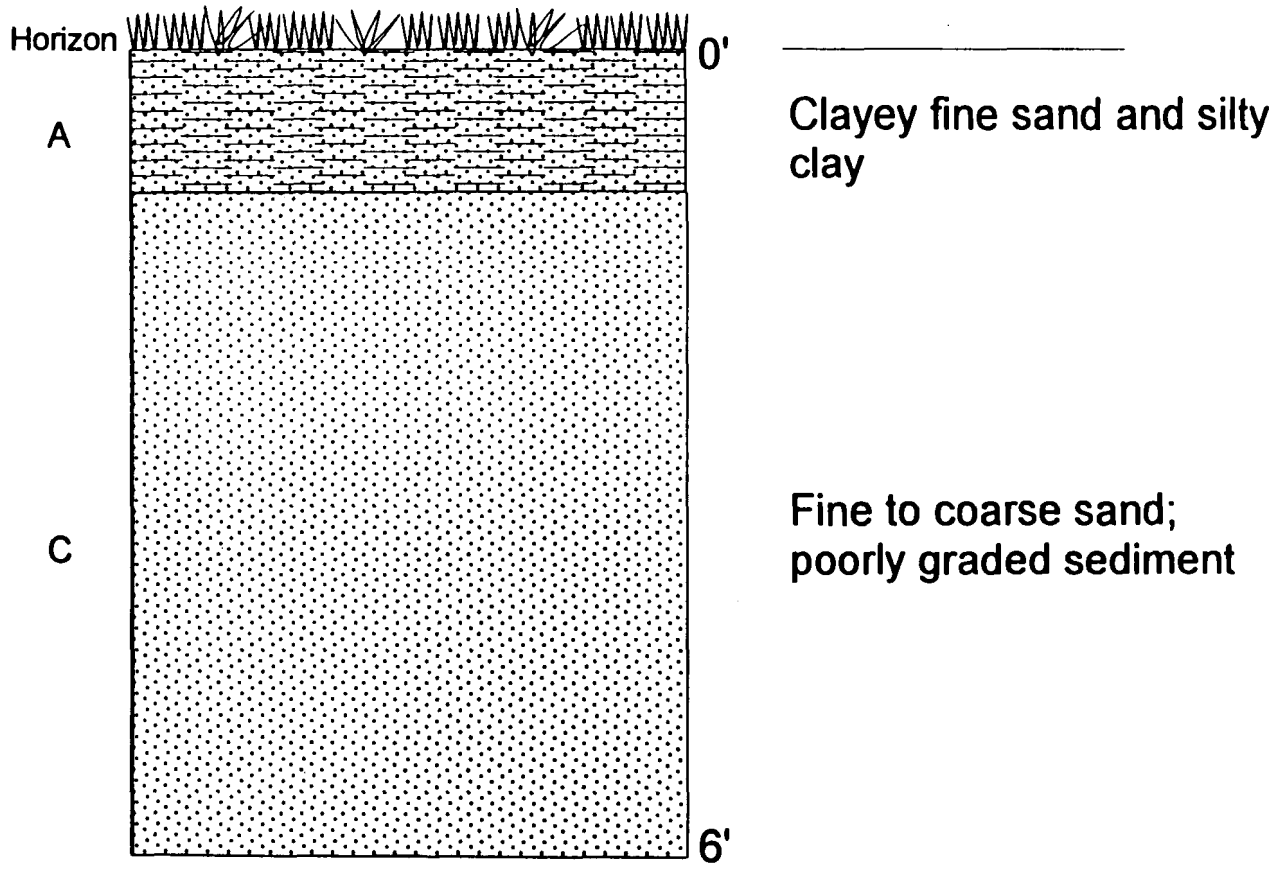

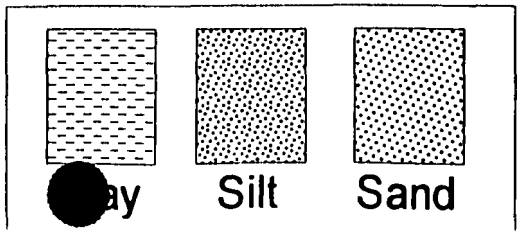

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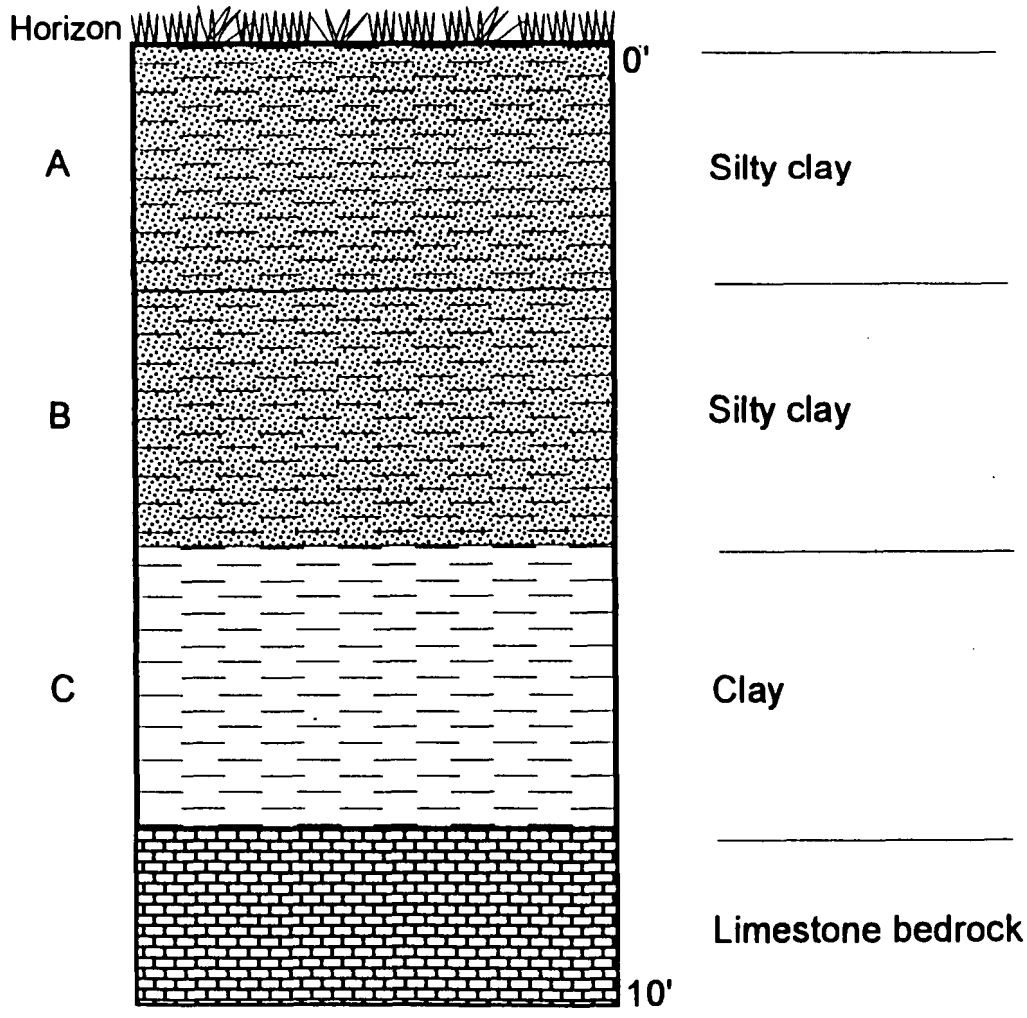

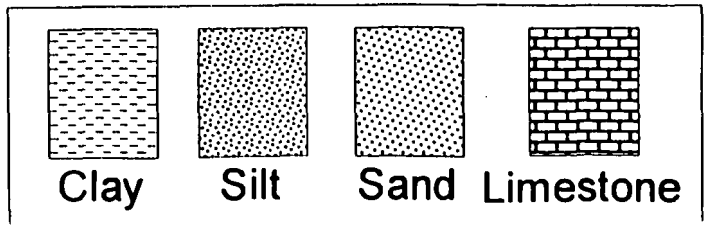

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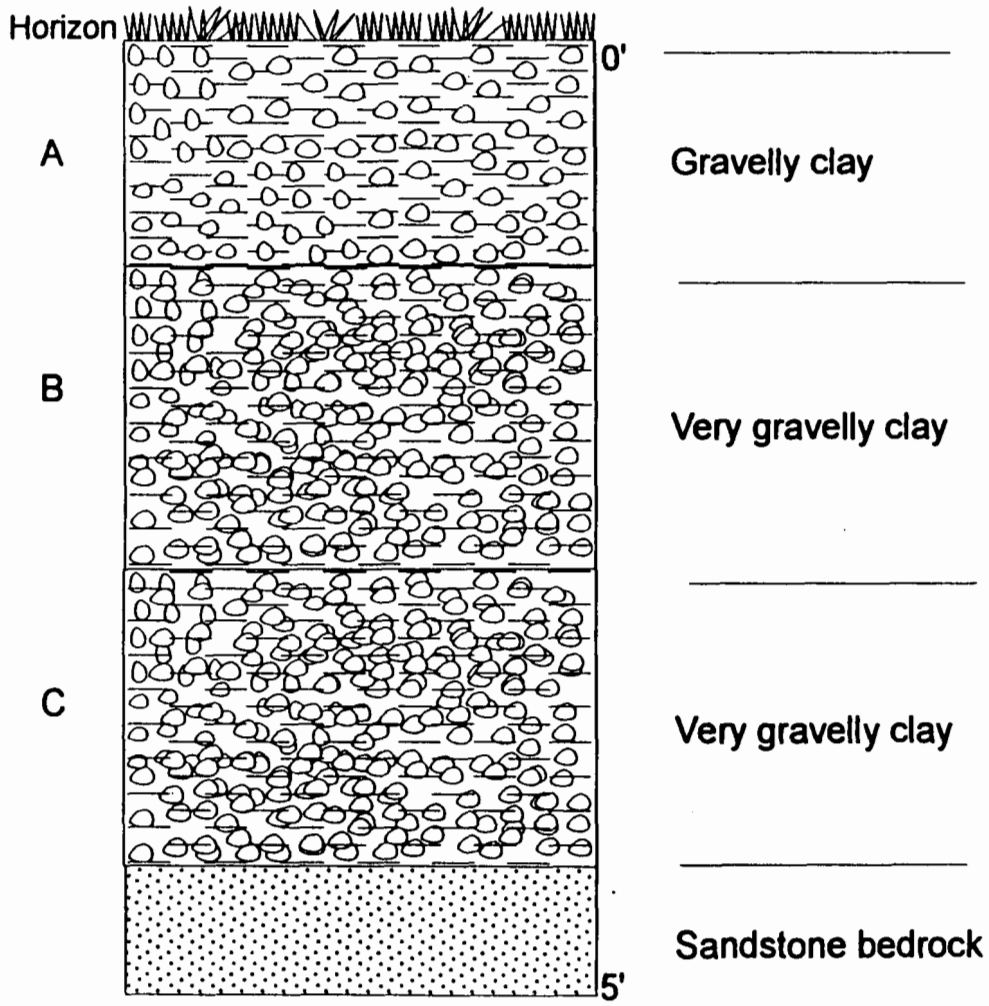

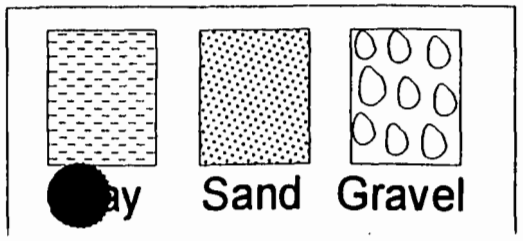

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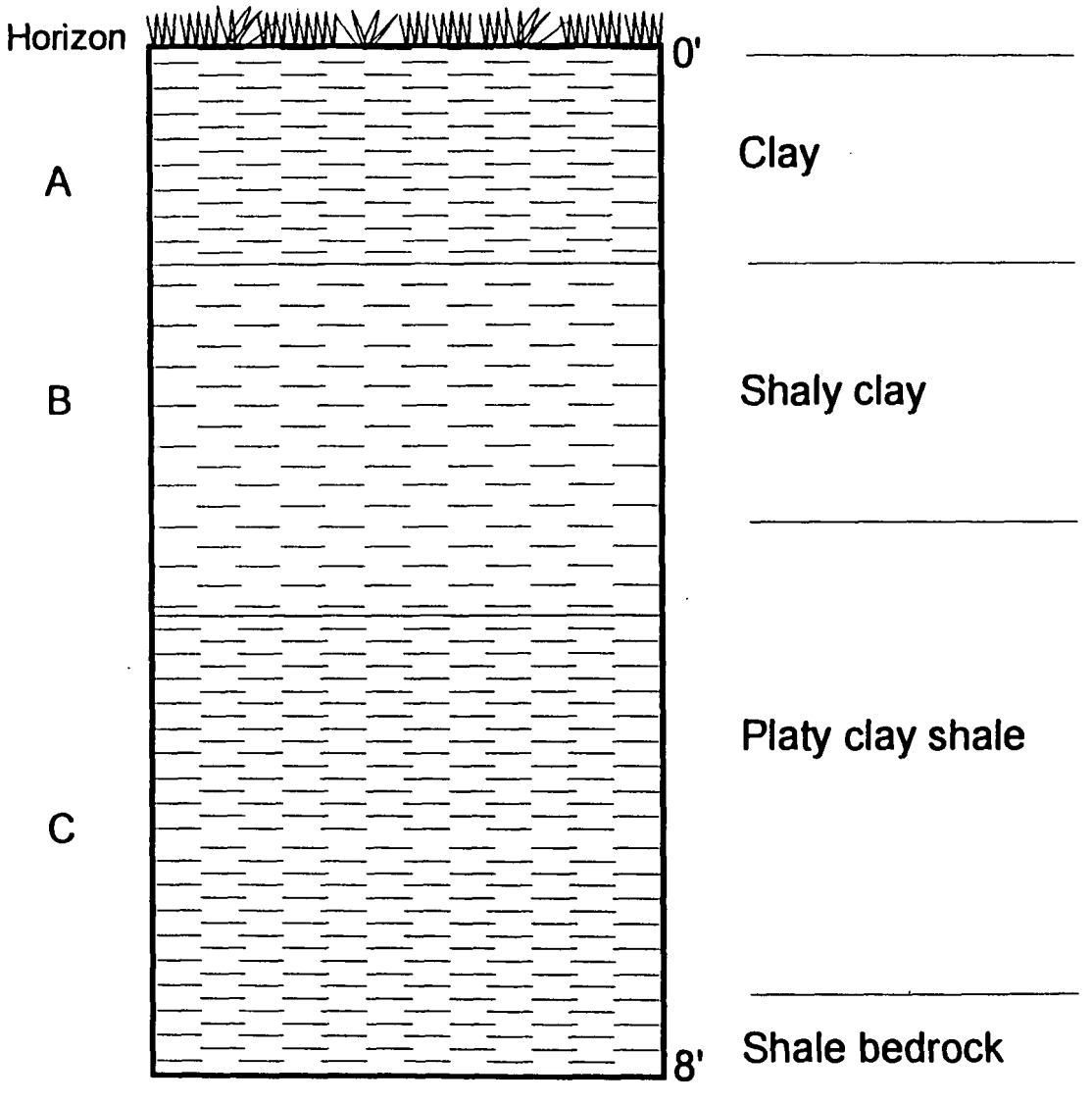

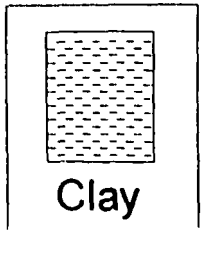

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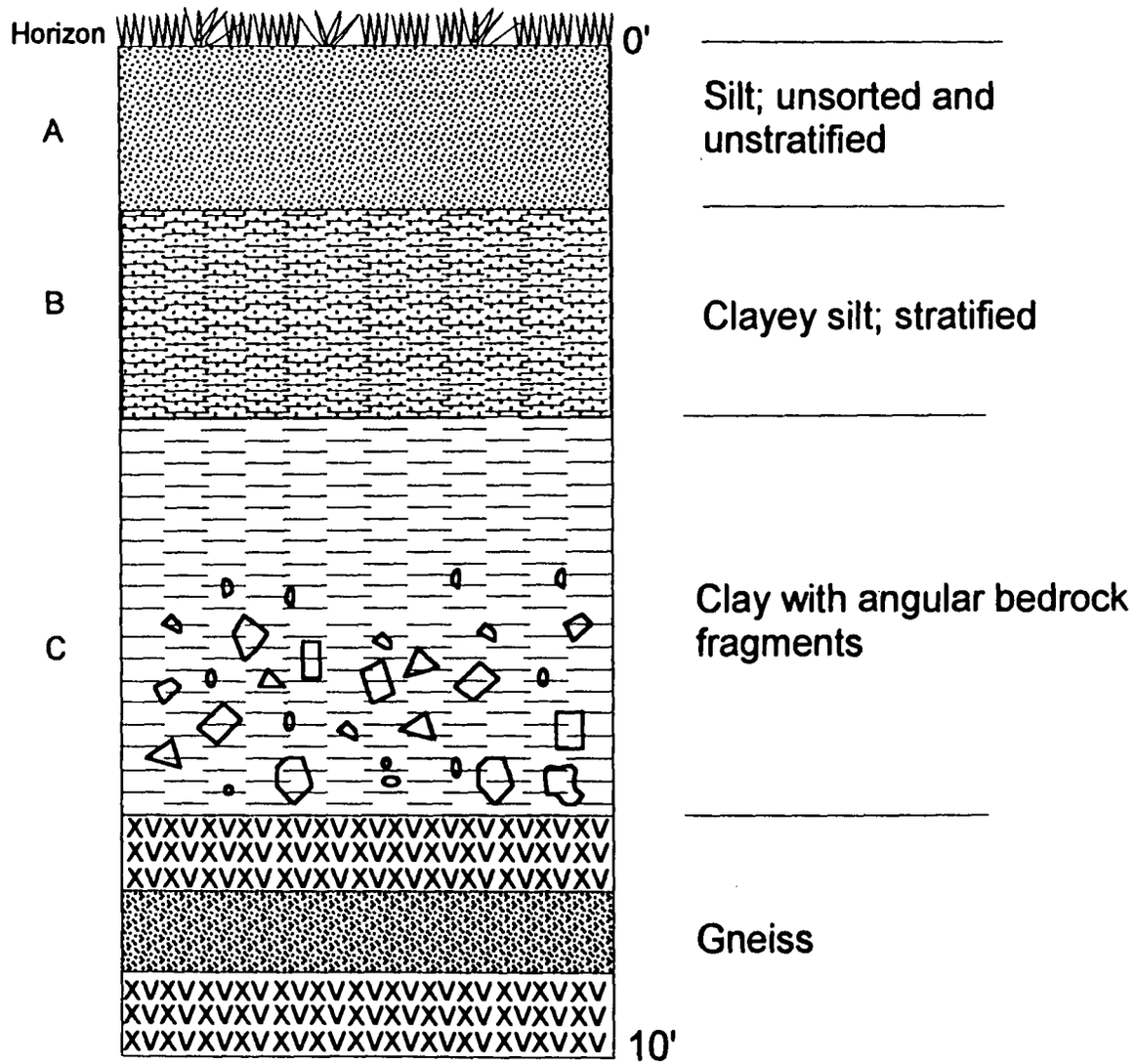

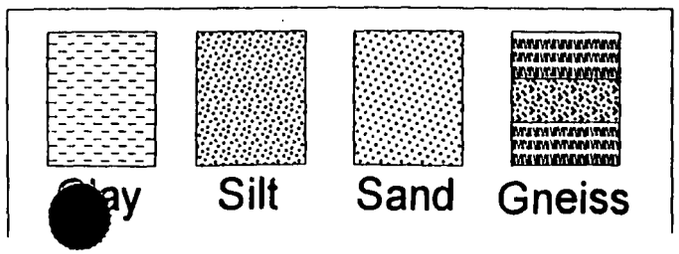

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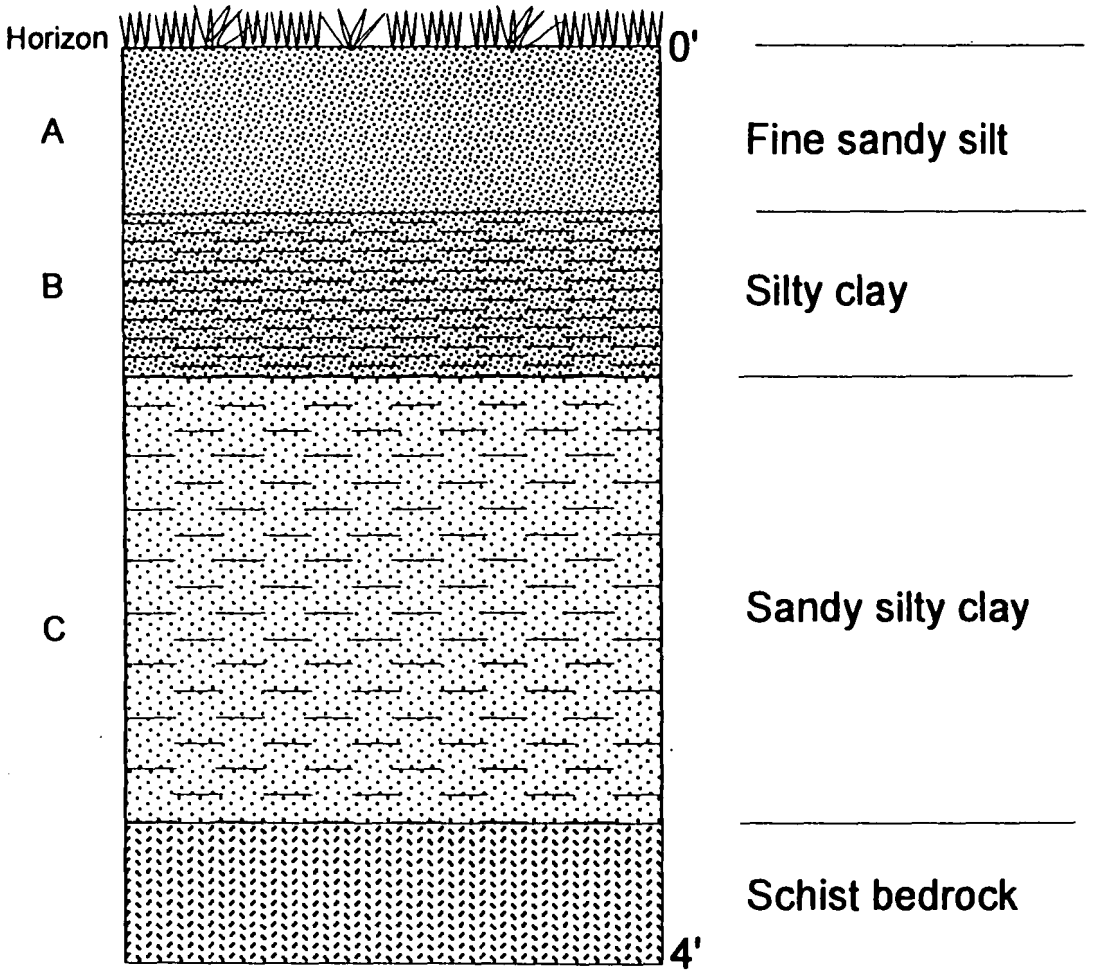

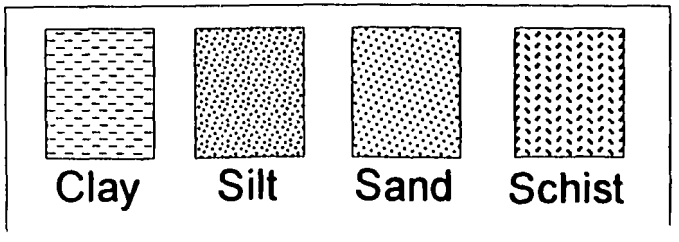

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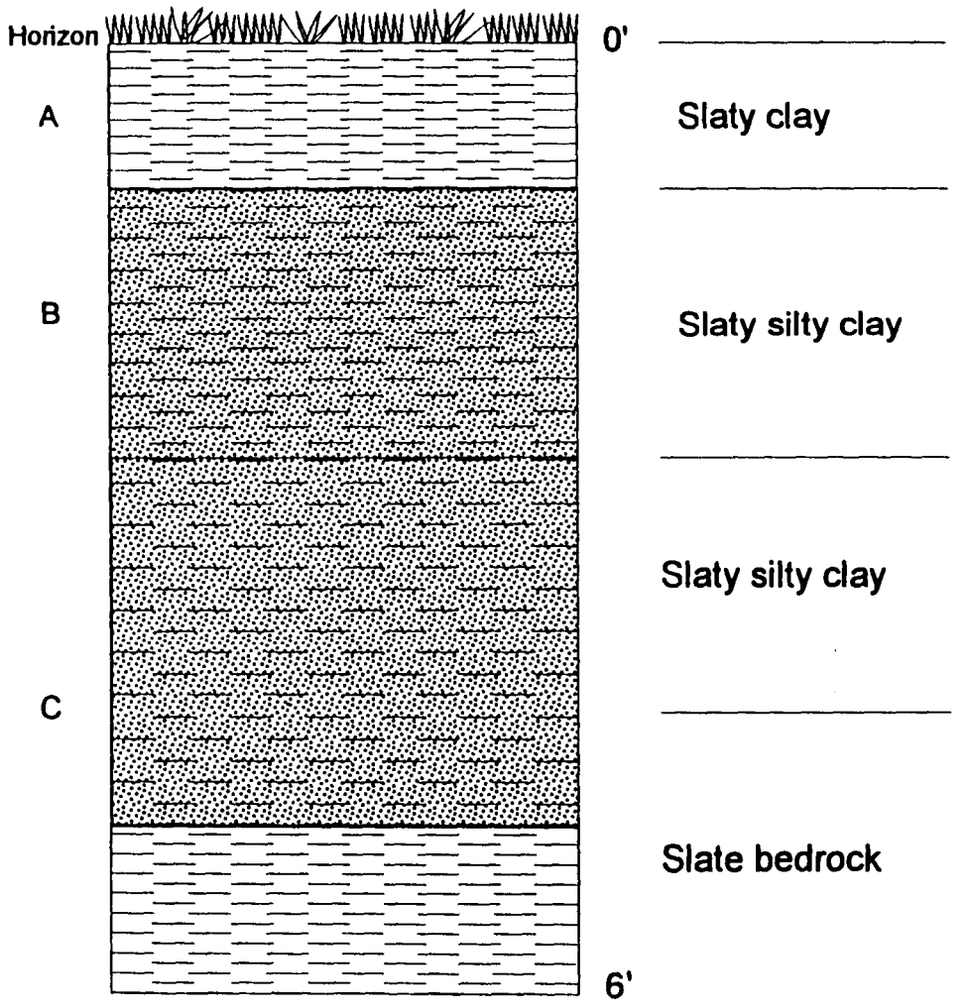

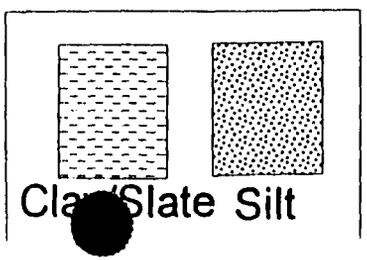

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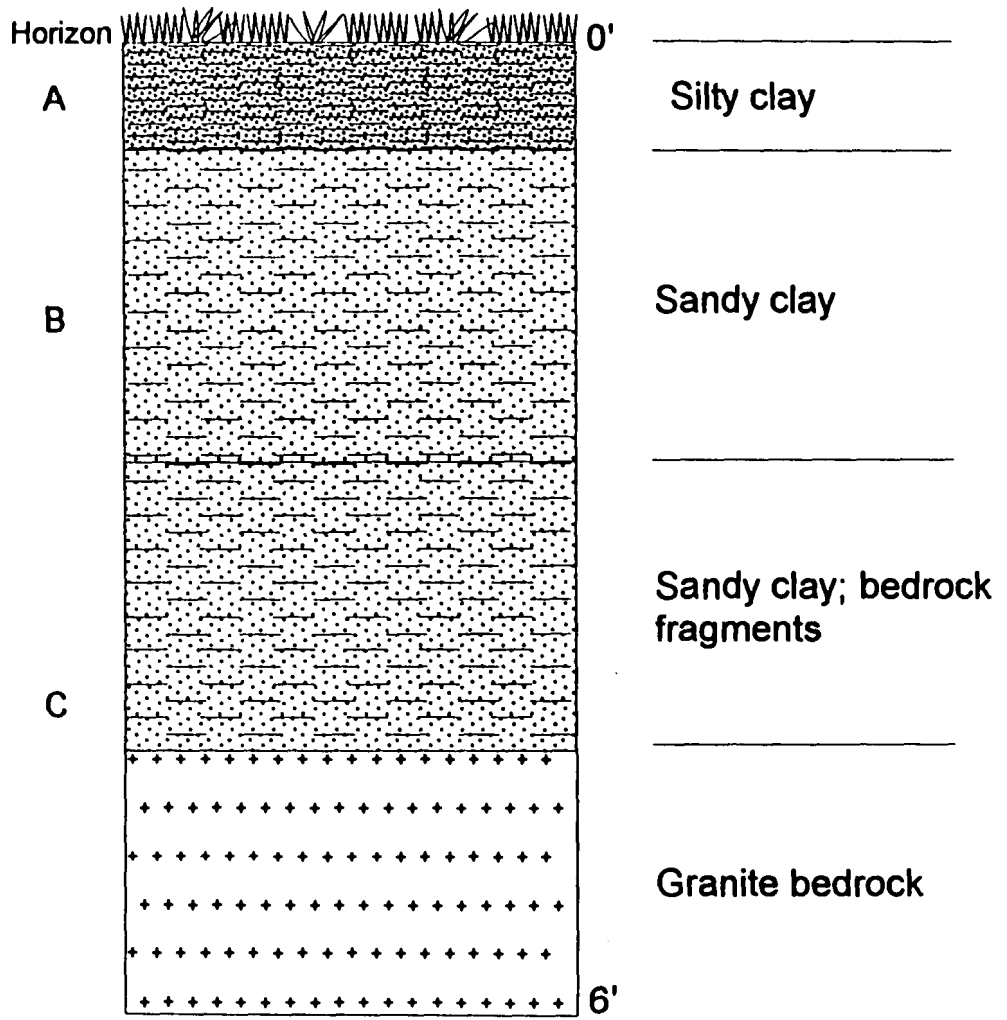

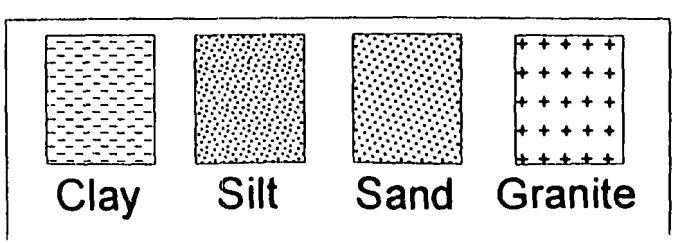

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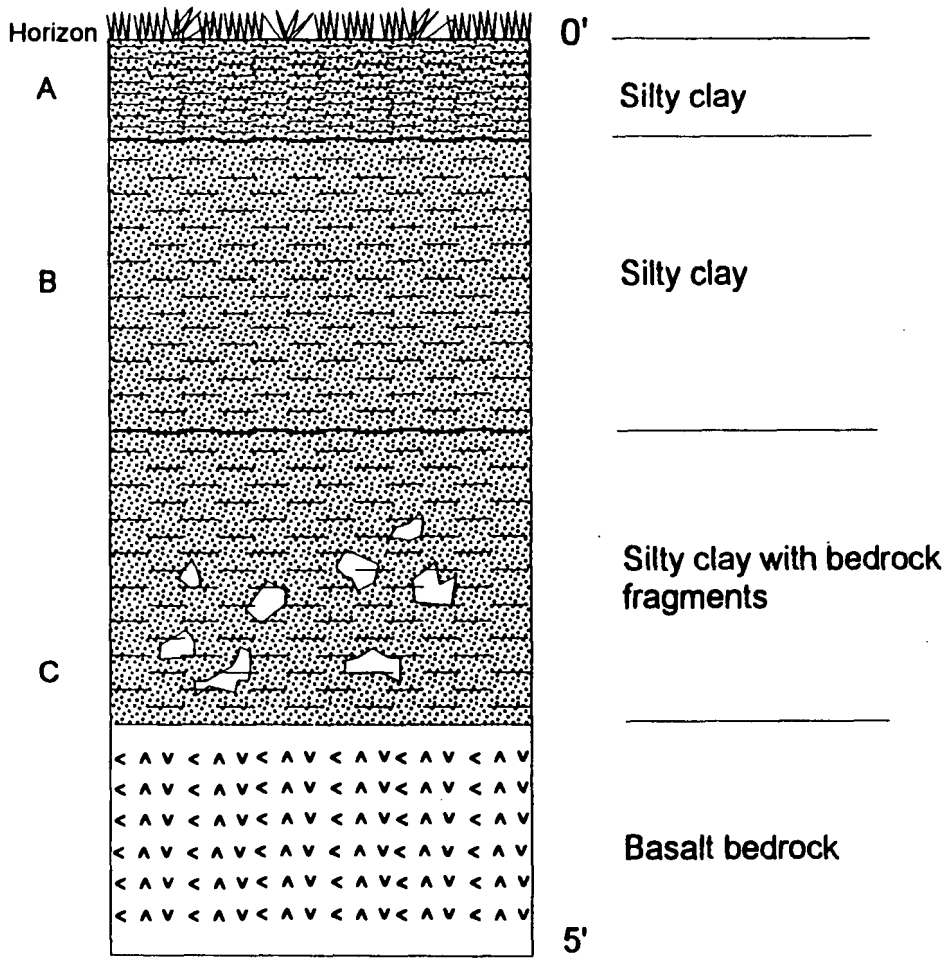

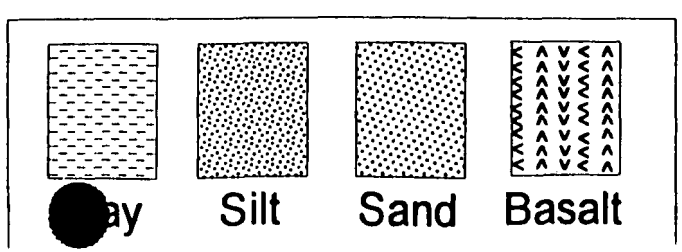

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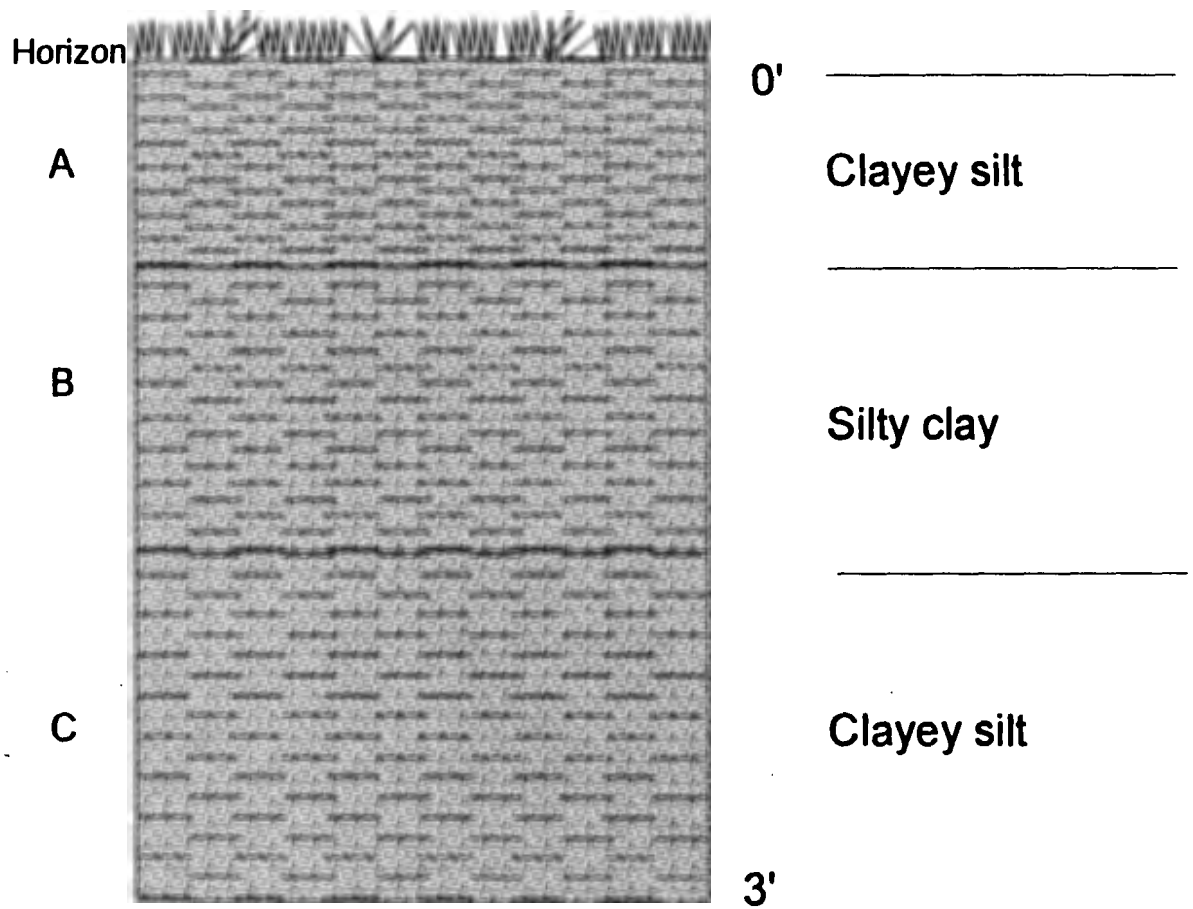

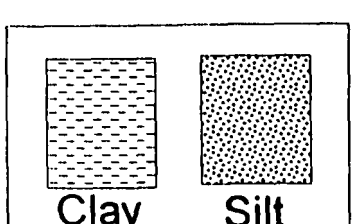

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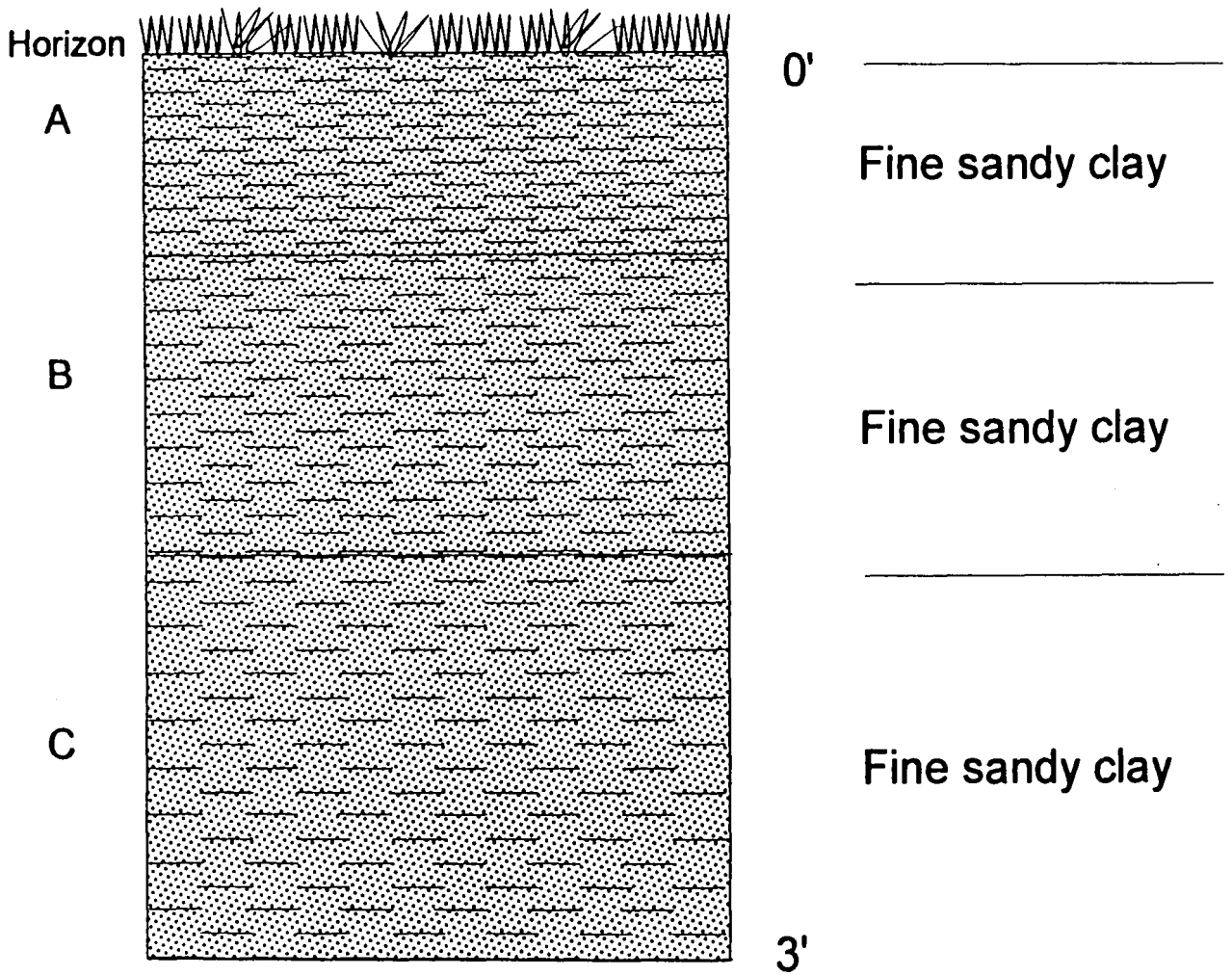

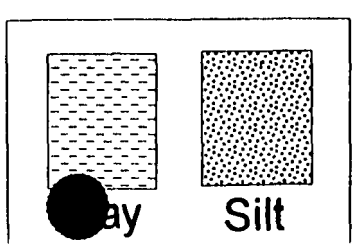

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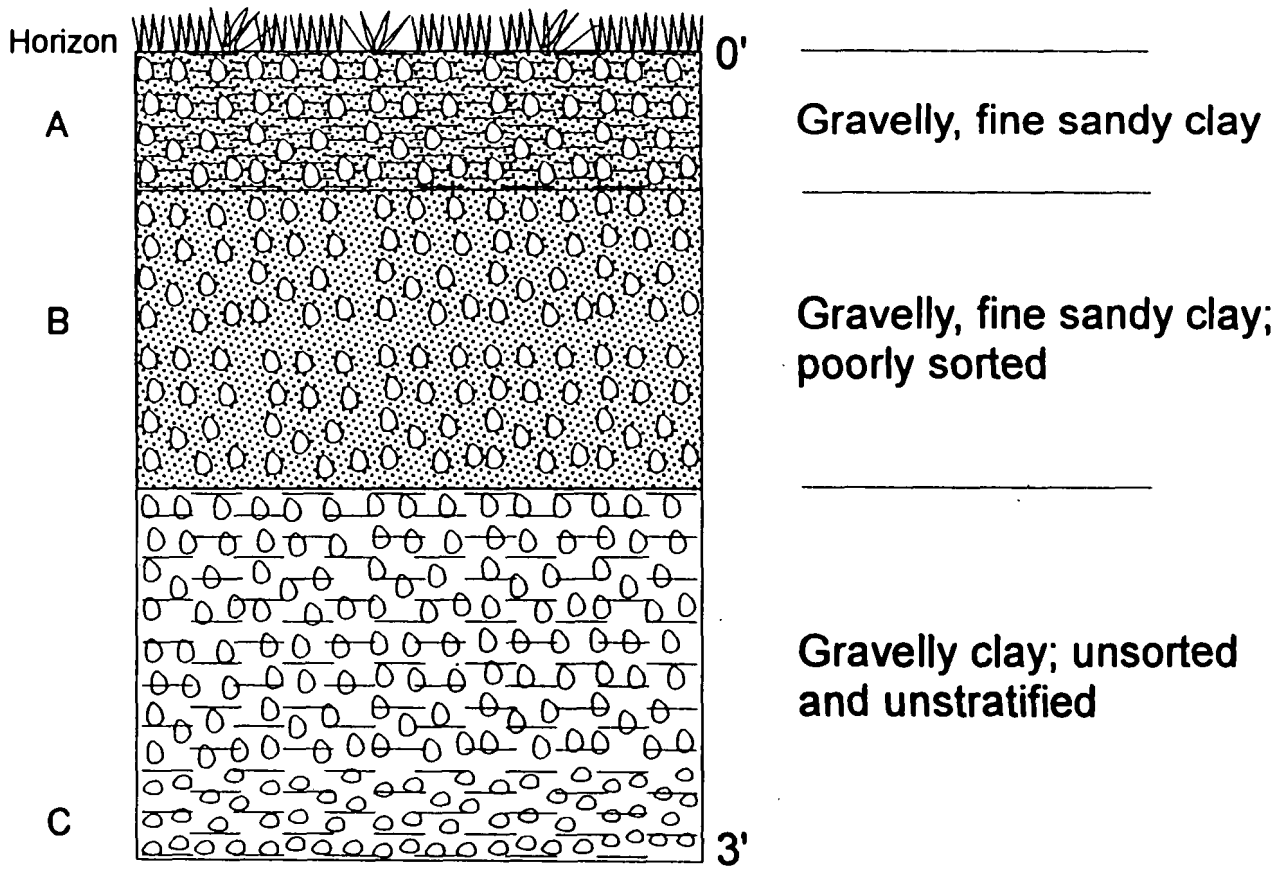

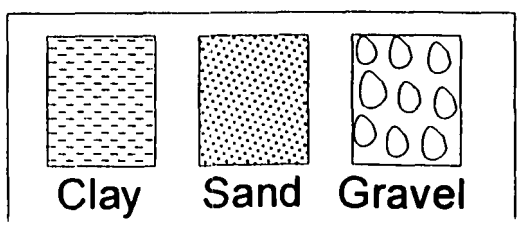

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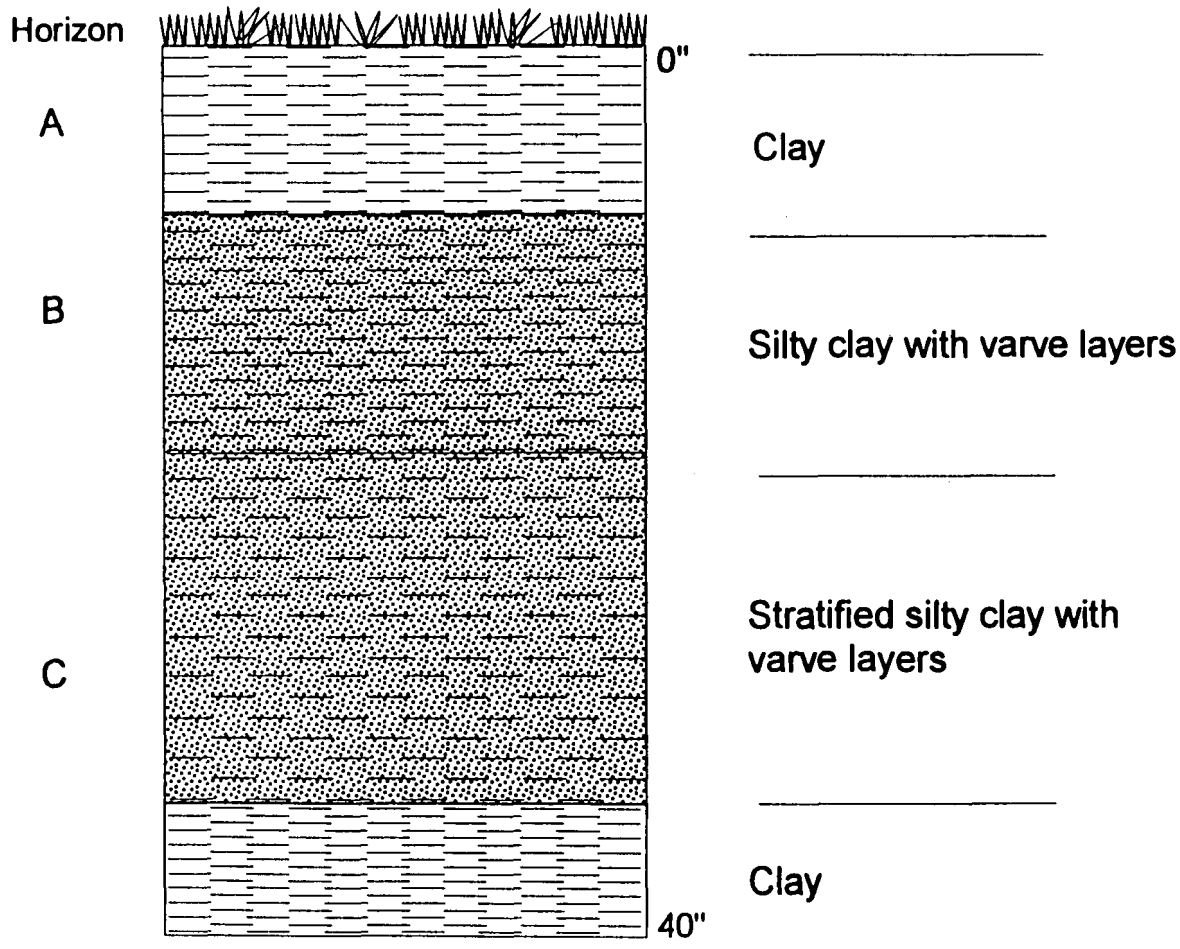

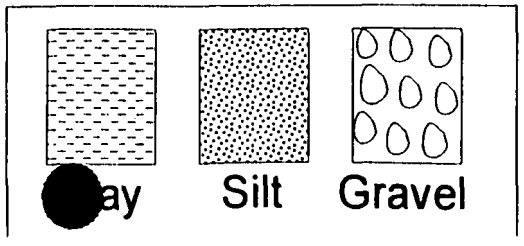

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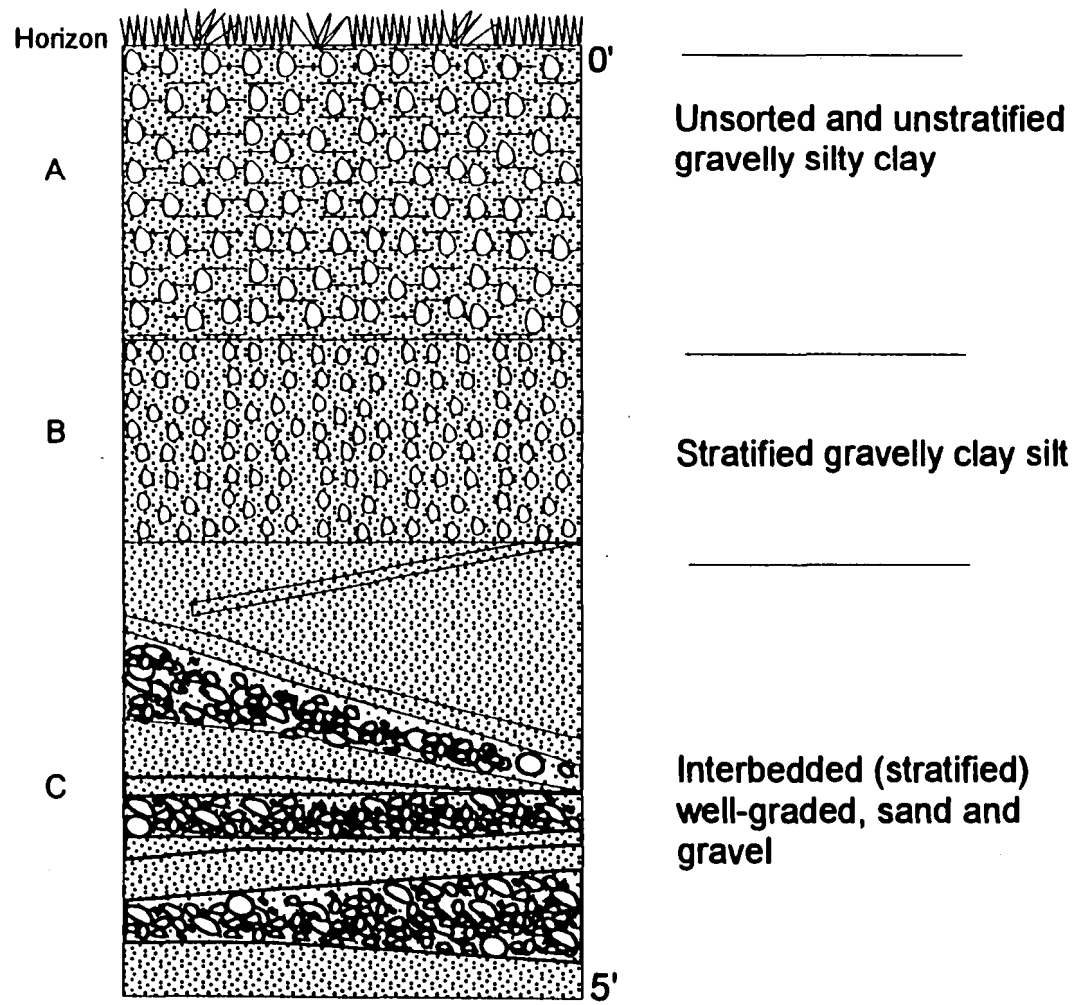

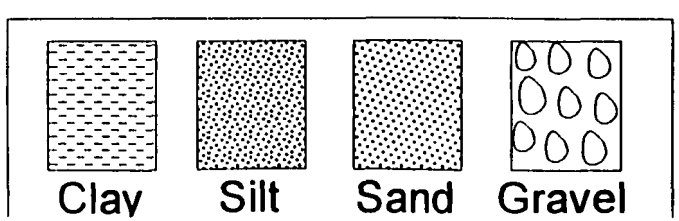

VOL. I

United States
Environmental Protection
Agency

Office of Research and
Development
Washington DC 20460

EPA 625/6-90/016a
September 1990

EPA

Handbook

Ground Water

Volume I: Ground Water and Contamination

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U.S. Environmental Protection Agency
Office of Research and Development

Center for Environmental Research Information
Cincinnati, OH 45268

NOTICE

This document has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

This document is not intended to be a guidance or support document for specific regulatory program. Guidance documents are available from EPA and must be consulted to address specific regulatory issues.

Contents

		Page
Chapter 1.	Basic Geology	1
Chaper 2.	Classification of Ground-Water Regions	18
Chapter 3.	Ground Water-Surface Water Relationship	50
Chapter 4.	Basic Hydrogeology	74
Chapter 5.	Ground-Water Contamination	94
Chapter 6.	Ground-Water Investigations	114
Chapter 7.	Ground-Water Restoration	128

Acknowledgments

Many individuals contributed to the preparation and review of this handbook. The document was prepared by Eastern Research Group, Inc., for EPA's Center for Environmental Research Information, Cincinnati, OH. Contract administration was provided by the Center for Environmental Research Information.

Volume I, Ground Water and Contamination, will be followed by Volume II, Methodology. Although extensively revised, Volume I was obtained from previous publlications, "Handbook: Ground Water" (EPA/625/6-87/016) and "Protection of Public Water Supplies from Ground-Water Contamination" (EPA/625/4-85/016).

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Preface

The subsurface environment of ground water is characterized by a complex interplay of physical, geochemical and biological forces that govern the release, transport and fate of a variety of chemical substances. There are literally as many varied hydrogeologic settings as there are types and numbers of contaminant sources. In situations where ground-water investigations are most necessary, there are frequently many variables of land and ground-water use and contaminant source characteristics which cannot be fully characterized.

The impact of natural ground-water recharge and discharge processes on distributions of chemical constituents is understood for only a few types of chemical species. Also, these processes may be modified by both natural phenomena and man's activities so as to further complicate apparent spatial or temporal trends in water quality. Since so many climatic, demographic and hydrogeologic factors may vary from place to place, or even small areas within specific sites, there can be no single "standard" approach for assessing and protecting the quality of ground water that will be applicable in all cases.

Despite these uncertainties, investigations are under way and they are used as a basis for making decisions about the need for, and usefulness of, alternative corrective and preventive actions. Decision makers, therefore, need some assurance that elements of uncertainty are minimized and that hydrogeologic investigations provide reliable results.

A purpose of this document is to discuss measures that can be taken to ensure that uncertainties do not undermine our ability to make reliable predictions about the response of contamination to various corrective or preventive measures.

EPA conducts considerable research in ground water to support its regulatory needs. In recent years, scientific knowledge about ground-water systems has been increasing rapidly. Researchers in the Office of Research and Development have made improvements in technology for assessing the subsurface, in adapting techniques from other disciplines to successfully identify specific contaminants in ground water, in assessing the behavior of certain chemicals in some geologic materials and in advancing the state-of-the-art of remedial technologies.

An important part of EPA's ground-water research program is to transmit research information to decision makers, field managers and the scientific community. This publication has been developed to assist that effort and, additionally, to help satisfy an immediate Agency need to promote the transfer of technology that is applicable to ground-water contamination control and prevention.

The need exists for a resource document that brings together available technical information in a form convenient for ground-water personnel within EPA and state and local governments on whom EPA ultimately depends for proper ground-water management. The information contained in this handbook is intended to meet that need. It is applicable to many programs that deal with the ground-water resource. However, it is not intended as a guidance or support document for a specific regulatory program.

GUIDANCE DOCUMENTS ARE AVAILABLE FROM EPA AND MUST BE CONSULTED TO ADDRESS SPECIFIC REGULATORY ISSUES.

Chapter 1

BASIC GEOLOGY

Introduction

Geology, the study of the earth, includes the investigation of earth materials, the processes that act on these materials, the products that are formed, the history of the earth, and the origin and development of life forms. There are several subfields of geology. Physical geology deals with all aspects of the earth and includes most earth science specialities. Historical geology is the study of the origin of the earth, continents and ocean basins, and life forms, while economic geology is an applied approach involved in the search and exploitation of mineral resources, such as metallic ores, fuels, and water. Structural geology deals with the various structures of the earth and the forces that produce them. Geophysics is the examination of the physical properties of the earth and includes the study of earthquakes and methods to evaluate the subsurface.

From the perspective of ground water, all of the subfields of geology are used, some more than others. Probably the most difficult concept to comprehend by individuals with little or no geological training is the complexity of the subsurface, which is hidden from view and, at least presently, cannot be adequately sampled. In geologic or hydrogeologic studies, it is best to always keep in mind a fundamental principle of geology, that is, the present is the key to the past. This means that the processes that are occurring today are the same processes that occurred throughout the geologic past—only the magnitude has changed from one time to the next.

Consider, for example, the channel and flood plain of a modern day river or stream. The watercourse constantly meanders from one side of the flood plain to another, eroding the banks and carrying the sediments farther downstream. The channel changes in size and position, giving rise to deposits of differing grain size and, perhaps, composition. The changes may be abrupt or gradual, both vertically and horizontally, as is evident from an examination of the walls of a gravel pit or the bluffs along a river. Because of the dynamic nature of streams and deltas, one will find a geologic situation that is perplexing, not only to the individual involved in a ground-water investigation, but to the geologist as well. Each change in grain size will cause a difference in permeability and ground-water velocity, while changes in mineral composition can lead to variances in water quality. At the other end of the depositional spectrum are deposits collected in lakes, seas, and the oceans, which are likely to be much more widespread and uniform in thickness, grain size, and composition.

As one walks from the sandy beach of a lake into the water, the sediments become finer and more widely distributed as the action of waves and currents sort the material brought into the lake by streams. Farther from shore, the bottom of the lake may consist of mud, which is a mixture of silt, clay, and organic matter. In some situations the earthy mud grades laterally into a lime ooze or mud. In geologic time these sediments become lithified or changed into rock...the sand to sandstone, the mud to shale, and the limy mud to limestone. It is important to note, however, that the sand, mud, and lime were all deposited at the same time, although with lithification each sediment type produced a different sedimentary rock.

Minerals

The earth, some 7,926 miles in diameter at the equator, consists of a core, mantle, and crust, which have been defined by the analysis of seismic or earthquake waves. Only a thin layer of the crust has been examined by humans. It consists of a variety of rocks, each of which is made up of one or more minerals.

Most minerals contain two or more elements, but of all of the elements known, only eight account for nearly 98 percent of the rocks and minerals:

Oxygen 46%
Silicon 27.72%
Aluminum 8.13%
Iron 5%
Calcium 3.63%
Sodium 2.83%
Potassium 2.59%
Magnesium 2.09%

Without detailed study, it is usually difficult to distinguish one mineral from another, except for a few common varieties, such as quartz, pyrite, mica, and some gemstones. On the other hand, it is important to have at least a general understanding of mineralogy because it is the mineral make-up of rocks that, to a large extent, controls the type of water that a rock will contain under natural conditions and the way it will react to contaminants or naturally occurring substances.

The most common rock-forming minerals are relatively few and deserve at least a mention. They can be divided into three broad groups: (1) the carbonates, sulfates, and oxides, (2) the rock-forming silicate minerals, and (3) the common ore minerals.

Carbonates, Sulfates, and Oxides

Calcite, a calcium carbonate ($CaCO_3$), is the major mineral in limestone. It is quite soluble, which accounts for its usual presence in water. The most common mineral is quartz. It is silicon dioxide (SiO_2), hard, and resistant to both chemical and mechanical weathering. In sedimentary rocks it generally occurs as sand-size grains (sandstone) or even finer, such as silt or clay size, and it may also appear as a cement. Because of the low solubility of silicon, silica generally appears in concentrations less than 25 mg/L in water. Limonite is actually a group name for the hydrated ferric oxide minerals ($Fe_2O_3H_2O$), which occur so commonly in many types of rocks. Limonite is generally rusty or blackish with a dull, earthy luster and a yellow-brown streak. It is a common weathering product of other iron minerals. Because limonite and other iron-bearing minerals are nearly universal, dissolved iron is a very common constituent in water and causes staining of clothing and plumbing fixtures. Gypsum, a hydrated calcium sulfate ($CaSO_4 \cdot 2H_2O$), occurs as a sedimentary evaporite deposit and as crystals in shale and some clay deposits. Quite soluble, it is the major source of sulfate in ground water.

Rock-Forming Silicates

The most common rock-forming silicate minerals include the feldspars, micas, pyroxenes, amphiboles, and olivine. Except in certain igneous and metamorphic rocks, these minerals are quite small and commonly require a microscope for identification. The feldspars are alumino-silicates of potassium or sodium and calcium. Most of the minerals in this group are white, gray, or pink. Upon weathering they turn to clay and release the remaining chemical elements to water. The micas, called muscovite and biotite, are platy alumino-silicate minerals that are common and easily recognized in igneous, metamorphic, and sedimentary rocks. The pyroxenes, a group of silicates of calcium, magnesium, and iron, as well as the amphiboles, which are complex hydrated silicates of calcium, magnesium, iron, and aluminum, are common in most igneous and metamorphic rocks. They appear as small, dark crystals of accessory minerals. Olivine, a magnesium-iron silicate, is generally green or yellow and is common in certain igneous and metamorphic rocks. None of the rock-forming silicate minerals have a major impact on water quality in most situations.

Next to organic matter, clay minerals are the most chemically active materials in soil and unconsolidated materials. Both consolidated rocks and unconsolidated sediments that have a high clay mineral content tend to have low permeabilities and, consequently, water movement through them is very slow. The two broad groups of clay minerals commonly recognized are the silicate clays and the hydrous oxide clays. Silicate clays form from the weathering of primary silicate minerals, such as feldspars and olivine. They have a sheet-like lattice structure and a strong adsorptive capacity. Silicate clays are classified according to different stacking arrangements of the lattice layers and their tendency to expand in water. The stacking type strongly affects certain properties of clays, including (1) surface area, (2) the tendency to swell during hydration, and (3) cation exchange capacity (CEC), which is a quantitative measure of the ability of a mineral surface to adsorb ions.

Table1-1 summarizes some properties of silicate clay minerals, which are listed from the most reactive (montmorillonite and vermiculite) to least reactive (kaolinite). The montmorillonite group is most sensitive to swelling and has a high CEC. The structure in kaolinite results in both a low surface area and CEC. Illite and chlorite have intermediate surface areas, CEC, and sensitivities to swelling. Clay minerals in sedimentary rocks are usually mixtures of different groups. In addition, mixed-layer clay minerals can form and these have properties and compositions that are intermediate between two well-defined clay types (e.g., chlorite-illite, illite-montmorillonite). Hydrous oxide clays, which are less well understood than silicate clays, are oxides of iron, magnesium, and aluminum that are associated with water molecules. Compared to silicate clays, CEC is lower in hydrous oxide clays.

Property	Type of Clay[a]				
	Montmorillonite (Smetite)	Verriculite	Illite	Chlorite	Kaolinite
Lattice type[c]	2:1	2:1	2:1	2:2	1:1
Expanding?	Yes	Slightly	No	No	No
Specific surface (m^2/g)	700-800	700-800	65-120	25-40	7-30
External surface	High	High	Medium	Medium	Low
Internal surface	Very High	High	Medium	Medium	None
Swelling capacity	High	Med-High	Medium	Low	Low
Cation exchange capacity (meq/100g)	80-150	100-150+	10-40	10-40	3-15
Other similar Clays	Beidellite Nontrorite Saporite Bentonite[d]				Halloysite Anauxite Dickit

[a] Clays are arranged from most reactive (montmorillonite) to least reactive (kaolinite).

[b] The term smectite is now used to refer to the montmorillonite group of clays (Soil Science Society of America, 1987)

[c] Tetrahedral:octahedral layers.

[d] Bentonite is a clay formed from weathering of volcanic ash and is made up mostly of montmorillonite and beidellite.

[e] Upper range occurs with smaller particle size.

Sources: Adapted from Grim (1968), Brady (1974), and Ahlrichs (1972).

Table 1-1. Important Characteristics of Silicate Clay Minerals

Ores

The three most common ore minerals are galena, sphalerite, and pyrite. Galena, a lead sulfide (PbS), is heavy, brittle, and breaks into cubes. Sphalerite is a zinc sulfide (ZnS) mineral that is brownish, yellowish, or black. It ordinarily occurs with galena and is a major ore of zinc. The iron sulfide pyrite (FeS), which is also called fool's gold, is common in nearly all types of rocks. It is the weathering of this mineral that leads to acid-mine drainage, which is common in many coal fields and metal sulfide mining regions.

Rocks, Their Origin and Properties

Three types of rock comprise the crust of the earth. Igneous rocks solidified from molten material either within the earth (intrusive) or on or near the surface (extrusive). Metamorphic rocks were originally igneous or sedimentary rocks that were modified by temperature, pressure, and chemically active fluids. Sedimentary rocks are the result of the weathering of preexisting rocks, erosion, and deposition. Geologists have developed elaborate systems of nomenclature and classification of rocks, but these are of little value in hydrogeologic studies and, therefore, only the most basic descriptions will be presented.

Igneous Rocks

Igneous rocks are classified on the basis of their composition and grain size. Most consist of feldspar and a variety of dark minerals; several others also contain quartz. If the parent molten material cools slowly deep below the surface, minerals will have an opportunity to grow and the rock will be coarse grained. Magma that cools rapidly, such as that derived from volcanic activity, is so fine grained that individual minerals generally cannot be seen even with a hand lens. In some cases the molten material began to cool slowly, allowing some minerals to grow, and then the rate increased dramatically so that the

remainder formed a fine groundmass. This texture, consisting of large crystals in a fine-grained matrix, is called porphyritic.

Intrusive igneous rocks can only be seen where they have been exposed by erosion. They are concordant if they more or less parallel the bedding of the enclosing rocks and discordant if they cut across the bedding. The largest discordant igneous masses are called batholiths and they occur in the eroded centers of many ancient mountains. Their dimensions are in the range of tens of miles. Batholiths usually consist largely of granite, which is surrounded by metamorphic rocks.

Discordant igneous rocks also include dikes that range in width from a few inches to thousands of feet. Many are several miles long. Sills are concordant bodies that have invaded sedimentary rocks along bedding planes. They are relatively thin. Both sills and dikes tend to cool quite rapidly and, as a result, are fine grained.

Extrusive rocks include lava flows or other types associated with volcanic activity, such as the glassy rock, pumice, and the consolidated ash called tuff. These are fine grained or even glassy.

With some exceptions, igneous rocks are dense and have very little porosity or permeability. Most, however, are fractured to some degree and can store and transmit a modest amount of water. Some lava flows are notable exceptions because they contain large diameter tubes or a permeable zone at the top of the flow where gas bubbles migrated to the surface before the rock solidified. These rocks are called scoria.

Metamorphic Rocks

Metamorphism is a process that changes preexisting rocks into new forms because of increases in temperature, pressure, and chemically active fluids. Metamorphism may affect igneous, sedimentary, or other metamorphic rocks. The changes brought about include the formation of new minerals, increase in grain size, and modification of rock structure or texture, all of which depend on the original rock's composition and the intensity of the metamorphism.

Some of the most obvious changes are in texture, which serves as a means of classifying metamorphic rocks into two broad groups, the foliated and non-foliated rocks. Foliated metamorphic rocks typify regions that have undergone severe deformation, such as mountain ranges. Shale, which consists mainly of silt and clay, is transformed into slate by the change of clay to mica. Mica, being a platy mineral, grows with its long axis perpendicular to the principal direction of stress, forming a preferred orientation. This orientation, such as the development of cleavage in slate, may differ greatly from the original bedding.

With increasing degrees of metamorphism, the grains of mica grow to a larger size so that the rock has a distinct foliation, which is characteristic of the metamorphic rock, schist. At even higher grades of metamorphism, the mica may be transformed to a much coarser-grained feldspar, producing the strongly banded texture of gneiss.

Non-foliated rocks include the hornfels and another group formed from rocks that consist mainly of a single mineral. The hornfels occur around an intrusive body and were changed by "baking" during intrusion. The second group includes marble and quartzite, as well as several other forms. Marble is metamorphosed limestone and quartzite is metamorphosed quartz sandstone.

There are many different types of metamorphic rocks, but from a hydrogeologic viewpoint they normally neither store nor transmit much water and are of only minor importance as aquifers. Their primary permeability is notably small, if it exists at all, and fluids are forced to migrate through secondary openings, such as faults, joints, or other types of fractures.

Sedimentary Rocks

Sedimentary rocks are deposited, either in a body of water or on the land, by running water, by wind, and by glaciers. Each depositional agent leaves a characteristic stamp on the material it deposits. The sediments carried by these agents were first derived by the weathering and erosion of preexisting rocks. The most common sedimentary rocks are shale, siltstone, sandstone, limestone, and glacial till. The change from a loose, unconsolidated sediment to a rock is the process of lithification. Although sedimentary rocks appear to be the dominant type, in reality they make up but a small percentage of the earth. They do, however, form a thin crust over much of the earth's surface, are the type most readily evident, and serve as the primary source of ground water.

The major characteristics of sedimentary rocks are sorting, rounding, and stratification. A sediment is well sorted if the grains are nearly all the same size. Wind is the most effective agent of sorting and this is followed by water. Glacial till is unsorted and consists of a wide mixture of material that ranges from large boulders to clay.

While being transported, sedimentary material loses

its sharp, angular configuration as it develops some degree of rounding. The amount of rounding depends on the original shape, composition, transporting medium, and the distance traveled.

Sorting and rounding are important features of both consolidated and unconsolidated material because they have a major control on permeability and porosity. The greater the degree of sorting and rounding the higher will be the water-transmitting and storage properties. This is why a deposit of sand, in contrast to glacial till, can be such a productive aquifer.

Most sedimentary rocks are deposited in a sequence of layers or strata. Each layer or stratum is separated by a bedding plane, which probably reflects variations in sediment supply or some type of short-term erosion. Commonly bedding planes represent changes in grain size. Stratification provides many clues in our attempt to unravel geologic history. The correlation of strata between wells or outcrops is called stratigraphy.

Sedimentary rocks are classified on the basis of texture (grain size and shape) and composition. Clastic rocks consist of particles of broken or worn material and include shale, siltstone, sandstone, and conglomerate. These rocks are lithified by compaction, in the case of shale, and by cementation. The most common cements are clay, calcite, quartz, and limonite. The last three, carried by ground water, precipitate in the unconsolidated material under specific geochemical conditions.

The organic or chemical sedimentary rocks consist of strata formed from or by organisms and by chemical precipitates from sea water or other solutions. Most have a crystalline texture. Some consist of well preserved organic remains, such as reef deposits and coal seams. Chemical sediments include, in addition to some limestones, the evaporites, such as halite (sodium chloride), gypsum, and anhydrite. Anhydrite is an anhydrous calcium sulfate.

Geologists also have developed an elaborate classification of sedimentary rocks, which is of little importance to the purpose of this introduction. In fact, most sedimentary rocks are mixtures of clastic debris, organic material, and chemical precipitates. One should keep in mind not the various classifications, but rather the texture, composition, and other features that can be used to understand the origin and history of the rock.

The term texture has different meanings in geology and soil science. In soil science it is simply the relative proportions of clay-, silt-, and sand-sized particles in soil or unconsolidated material. The term fabric applies to the total of all physical features of a rock or soil that can be observed. Soil fabric analysis involves the study of distinctive physical features resulting from soil-forming processes, which also strongly influence the location and rate of water movement in soil.

A variety of scales are available for the classification of materials based on particle-size distribution. In geology, the Wentworth-Udden scale is most widely used: boulder (>256 mm), cobble (64-256 mm), pebble (4-64 mm), granule or gravel (2-4 mm), sand (1/16-2 mm), silt (1/256-1/16 mm), and clay (<1/256 mm). The U.S. Department of Agriculture (USDA) soil textural classification system is most widely used by soil scientists, and engineers usually use the Unified soil classification system. The hydrologic properties of soils are strongly related to particle-size distribution.

Weathering

Generally speaking, a rock is stable only in the environment in which it was formed. Once removed from that environment, it begins to change, rapidly in a few cases, but more often slowly, by weathering. The two major processes of weathering are mechanical and chemical, but they usually proceed in concert.

Mechanical Weathering

Mechanical weathering is the physical breakdown of rocks and minerals. Some is the result of fracturing due to the volumetric increase when water in a crack turns to ice, some is the result of abrasion during transport by water, ice, or wind, and a large part is the result of gravity causing rocks to fall and shatter. Mechanical weathering alone only reduces the size of the rock; its chemical composition is not changed. The weathered material formed ranges in size from boulders to silt.

Chemical Weathering

Chemical weathering, on the other hand, is an actual change in composition as minerals are modified from one type to another. Many, if not most of the changes are accompanied by a volumetric increase or decrease, which in itself further promotes additional chemical weathering. The rate depends on temperature, surface area, and available water.

The major reactions involved in chemical weathering are oxidation, hydrolysis, and carbonation. Oxidation is a reaction with oxygen to form an oxide, hydrolysis is reaction with water, and carbonation is a reaction with CO_2 to form a carbonate. In these reactions the total volume increases and, since chemical weathering is most effective on grain surfaces, disintegration of a rock occurs.

Quartz, whether vein deposits or individual grains, undergoes practically no chemical weathering; the end product is quartz sand. Some of the feldspars weather to clay and release calcium, sodium, silica, and many other elements that are transported in water. The iron-bearing minerals provide, in addition to iron and magnesium, weathering products that are similar to the feldspars.

Basic Soil Concepts

Although the term soil is often loosely used to refer to any unconsolidated material, soil scientists distinguish it from other unconsolidated geologic materials by observable features, such as accumulation of organic matter, formation of soil structure, and leaching, that result from soil-forming processes.

The soil at a particular location is the result of the interaction of five factors: (I) parent material, (2) topography, (3) climate, (4) biota, and (5) time. The interaction of these factors results in the formation of a soil profile, the description of which forms the basis for classifying a soil. Specific soil-forming processes that influence soil profile development include (I) organic matter accumulation; (2) weathering of minerals to clays; (3) the depletion of clay and other sesquioxide minerals from upper horizons (eluviation), with subsequent enrichment in lower horizon (illuviation); (4) leaching or accumulation of soluble salts; (5) the formation of soil structure by the aggregation of soil particles into larger units called peds; and (6) the formation of slowly permeable layers called fragipans.

Perhaps the most distinctive features of a soil profile are its major horizons. The O horizon, if present, is a layer of partially decomposed organic material. The A horizon, which lies at the surface or near surface, is a mineral horizon characterized by maximum accumulation of organic matter; it usually has a distinctly darker color than lower horizons. The B horizon, the zone of most active weathering, is commonly enriched in clays, and has a well-defined soil structure. The C horizon is unconsolidated material that has experienced little or no weathering. The R horizon is solid rock.

Soil physical properties, such as texture, structure, and pore size distribution, are the major determinants in water movement in soil. Depending on the specific soil, water movement may be enhanced or retarded compared to unweathered geologic materials. Organic matter enhances water-holding capacity and infiltration. The formation of soil structure also enhances permeability, particularly in clayey soils. On the other hand, the formation of restrictive layers, such as fragipans, may substantially reduce infiltration compared to unweathered materials. Micromorphological and general fabric analysis of soil is used infrequently in the study of ground-water contamination, more because of unfamiliarity with the methods than their lack of value.

Minerals in the soil are the chemical signature of the bedrock from which they originated. Rainfall and temperature are two significant factors that dictate the rate and extent to which mineral solids in the soil react with water. Organic matter and clay content are major parameters of importance in studying the transport and fate of contaminants in soil.

Erosion and Deposition

Once a rock begins to weather, the by-products await erosion or transportation, which must be followed by deposition. The major agents involved in this part of the rock cycle are running water, wind, and glacial ice.

Waterborne Deposits

Mass wasting is the downslope movement of large amounts of detrital material by gravity. Through this process, sediments are made available to streams that carry them away to a temporary or permanent site of deposition. During transportation some sorting occurs and the finer silt and clay are carried farther downstream. The streams, constantly filling, eroding, and widening their channels, leave materials in their valleys that indicate much of the history of the region. Stream valley deposits, called alluvium, are shown on geologic maps by the symbol Qal, meaning Quaternary age alluvium. Alluvial deposits are distinct, but highly variable in grain size, composition, and thickness. Where they consist of glacially derived sand and gravel, called outwash, they form some of the most productive water-bearing units in the world.

Sediments, either clastic or chemical/organic, transported to past and present seas and ocean basins spread out to form, after lithification, extensive formations of sandstone, siltstone, shale, and limestone. In the geologic past, these marine deposits covered vast areas and when uplifted they formed the land surface, where they again began to weather in anticipation of the next trip to the ocean.

The major features of marine sedimentary rocks are their widespread occurrence and rather uniform thickness and composition, although extreme changes exist in many places. If not disturbed by some type of earth movement, they are stratified and horizontal. Furthermore, each lithologic type is unique relative to adjacent units. The bedding planes or contacts that

divide them represent distinct differences in texture or composition. From a hydrologic perspective, differences in texture from one rock type to another produce boundaries that strongly influence ground-water flow. Consequently, ground water tends to flow parallel to these boundaries, that is, within a particular geologic formation, rather than across them.

Windborne Deposits

Wind-laid or eolian deposits are relatively rare in the geologic record. The massively cross-bedded sandstone of the Navajo Sandstone in Utah's Zion National Park and surrounding areas is a classic example in the United States. Other deposits are more or less local and represent dunes formed along beaches of large water bodies or streams. Their major characteristic is the high degree of sorting. Dunes, being relatively free of silt and clay, are very permeable and porous, unless the openings have been filled by cement. They allow rapid infiltration of water and can form major water-bearing units, if the topographic and geologic conditions are such that the water does not rapidly drain.

Another wind-deposited sediment is loess, which consists largely of silt. It lacks bedding but is typified by vertical jointing. The silt is transported by wind from deserts, flood plains, and glacial deposits. Loess weathers to a fertile soil and is very porous. It is common along the major rivers in the glaciated parts of the United States and in China, parts of Europe, and adjacent to deserts and deposits of glacial outwash.

Glacial Deposits

Glaciers erode, transport, and deposit sediments that range from clay to huge boulders. They subdue the land surface over which they flow and bury former river systems. The areas covered by glaciers during the last Ice Age in the United States are described in Chapter 2, but the deposits extend far beyond the former margins of the ice. The two major types of glaciers include valley or mountain glaciers and the far more extensive continental glaciers. The deposits they leave are similar and differ, for the most part, only in scale.

As a glacier slowly passes over the land surface, it incorporates material from the underlying rocks into the ice mass, only to deposit that material elsewhere when the ice melts. During this process, it modifies the land surface, both through erosion and deposition. The debris associated with glacial activity is collectively termed glacial drift. Unstratified drift, usually deposited directly by the ice, is glacial till, a heterogeneous mixture of boulders, gravel, sand, silt, and clay. Glacial debris reworked by streams and in lakes is stratified drift. Although stratified drift may range widely in grain size, the sorting far surpasses that of glacial till. Glacial lake clays are particularly well sorted.

Glacial geologists usually map not on the basis of texture, but rather the type of landform that was developed, such as moraines, outwash, drumlins, and so on. The various kinds of moraines and associated landforms are composed largely of unstratified drift with incorporated layers of sand and gravel. Stratified drift is found along existing or former stream valleys or lakes that were either in the glacier or extended downgradient from it. Meltwater stream deposits are mixtures of sand and gravel. In places, some have coalesced to develop extensive outwash plains.

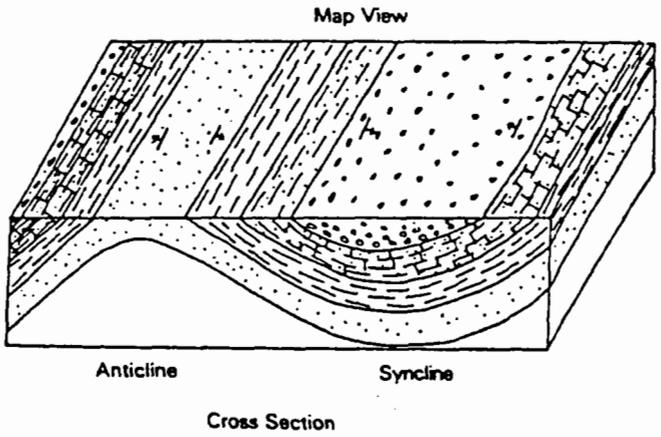
Glaciers advanced and retreated many times, reworking, overriding, and incorporating sediments from previous advances into the ice, subsequently redepositing them elsewhere. There was a constant inversion of topography as buried ice melted causing adjacent, waterlogged till to slump into the low areas. During advances, the ice might have overridden older outwash layers so that upon melting these sand and gravel deposits were covered by a younger layer of till. Regardless of the cause, the final effect is one of complexity of origin, history, and stratigraphy. When working with glacial till deposits, it is nearly always impossible to predict the lateral extent or thickness of a particular lithology in the subsurface. Surficial stratified drift is more uniform than till in thickness, extent, and texture.

Geologic Structure

A general law of geology is that in any sequence of sedimentary rocks that has not been disturbed by folding or faulting, the youngest unit is on the top. A second general law is that sedimentary rocks are deposited in a horizontal or nearly horizontal position. The fact that rocks are found overturned, displaced vertically or laterally, and squeezed into open or tight folds, clearly indicates that the crust of the earth is a dynamic system. There is a constant battle between the forces of destruction (erosion) and construction (earth movements).

Folding

Rocks, folded by compressional forces, are common in and adjacent to former or existing mountain ranges. The folds range from a few inches to 50 miles or so across. Anticlines are rocks folded upward into an arch. Their counterpart, synclines, are folded downward like a valley (fig. 1-1). A monocline is a flecture in which the rocks are horizontal, or nearly so, on either side of the flecture.



The arrow indicates the direction of dip. In an anticline, the rocks dip away from the crest and in a syncline they dip toward the center.

Figure 1-1. Dip and Strike Symbols Commonly Shown on Geologic Maps

Although many rocks have been folded into various structures, this does not mean that these same structures form similar topographic features. As the folding takes place over eons, the forces of erosion attempt to maintain a low profile. As uplift continues, erosion removes weathering products from the rising mass, carrying them to other places of deposition. The final topography is related to the erodibility of the rocks, with resistant strata, such as sandstone, forming ridges, and the less resistant material, such as shale, forming valleys (fig. 1-2). Consequently, the geologic structure of an area may bear little resemblance to its topography.

The structure of an area can be determined from field studies or a geologic map, if one exists. Various types of folds and their dimensions appear as unusual patterns on geologic maps. An anticline, for example, will be depicted as a series of rock units in which the oldest is in the middle, while a syncline is represented by the youngest rock in the center. More or less equidimensional anticlines and synclines are termed domes and basins, respectively.

The inclination of the top of a fold is the plunge. Folds may be symmetrical, asymmetrical, overturned, or recumbent. The inclination of the rocks is indicated by dip and strike symbols. The strike is perpendicular to the dip and the degree of dip is commonly shown by a number. The dip may range from less than a degree to vertical.

Unconformities

An unconformity is a break in the geologic record. It is caused by a cessation in deposition that is followed by erosion and subsequent deposition. The geologic record is lost by the period of erosion because the rocks that contained the record were removed.

If a sequence of strata is horizontal but the contact between two rock groups in the sequence represents an erosional surface, that surface is said to be a disconformity (fig. 1-3). Where a sequence of strata has been tilted and eroded and then younger, horizontal rocks are deposited over them, the contact is an angular unconformity. A nonconformity occurs where eroded

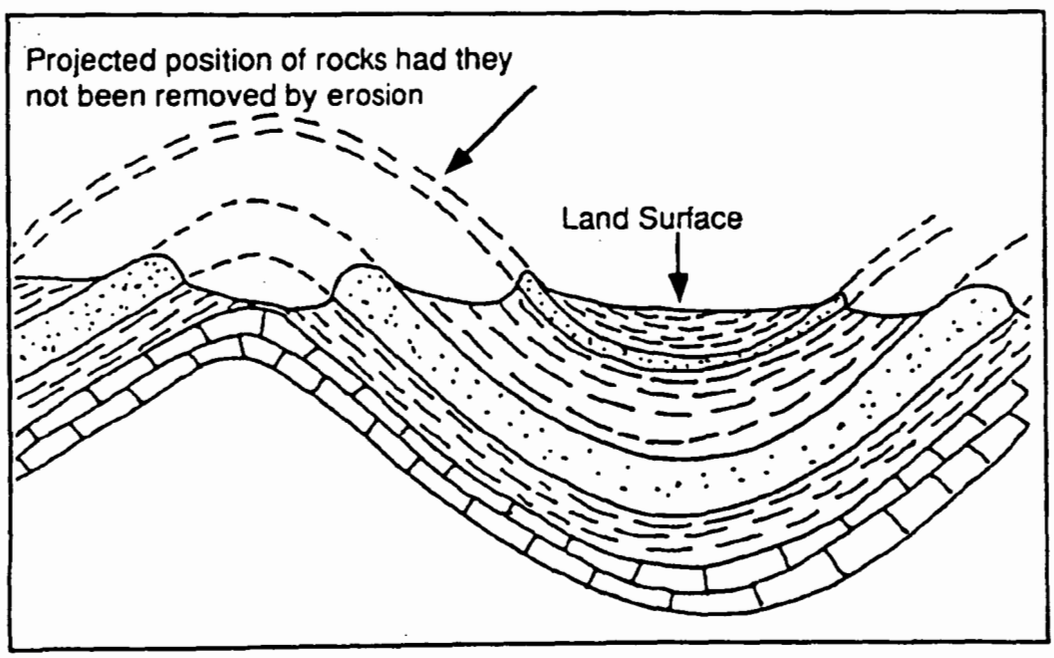


Figure 1-2. Geologic Structure May Influence Surface Topography

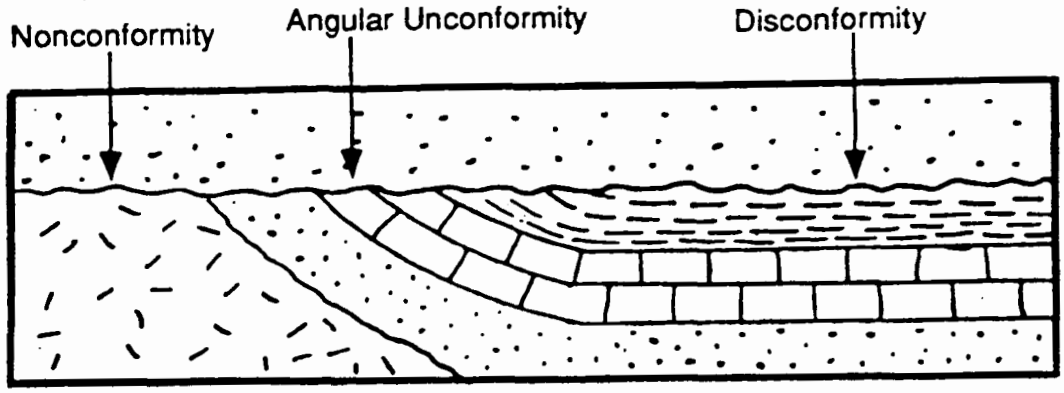


Figure 1-3. An Unconformity Represents a Break In the Geologic Record

igneous or metamorphic rocks are overlain by sedimentary rocks.

Fractures

Fractures in rocks are either joints or faults. A joint is a fracture along which no movement has taken place; a fault implies movement. Movement along faults is as little as a few inches to tens of miles. Probably all consolidated rocks and a good share of the unconsolidated deposits contain joints. Joints may exert a major control on water movement and chemical quality. Characteristically joints are open and serve as major conduits or pipes. Water can move through them quickly, perhaps carrying contaminants, and, being open, the filtration effect is lost. It is a good possibility that the outbreak of many waterborne diseases that can be tied to ground-water supplies is the result of the transmission of infectious agents through fractures to wells and springs.

Faults are most common in the deformed rocks of mountain ranges, suggesting either lengthening or shortening of the crust. Movement along a fault may be horizontal, vertical, or a combination. The most common types of faults are called normal, reverse, and lateral (fig. 1-4). A normal fault, which indicates stretching of the crust, is one in which the upper or hanging wall has moved down relative to the lower or foot wall. The Red Sea, Dead Sea, and the large lake basins in the east African highlands, among many others, lie in grabens, which are blocks bounded by normal faults (fig. 1-4). A reverse or thrust fault implies compression and shortening of the crust. It is distinguished by the fact that the hanging wall has moved up relative to the foot wall. A lateral fault is one in which the movement has been largely horizontal. The San Andreas Fault, extending some 600 miles from San Francisco Bay to the Gulf of California, is the most notable lateral fault in the United States. It was movement along this fault the produced the 1906 San Francisco earthquake.

Geologic Time

Geologic time deals with the relation between the emplacement or disturbance of rocks and time. In order to provide some standard classification, the geologic time scale was developed (table 1-2). It is based on a sequence of rocks that were deposited during a particular time interval. Commonly the divisions are based on some type of unconformity. In considering geologic time, three types of units are defined. These are rock units, time-rock units, and time units.

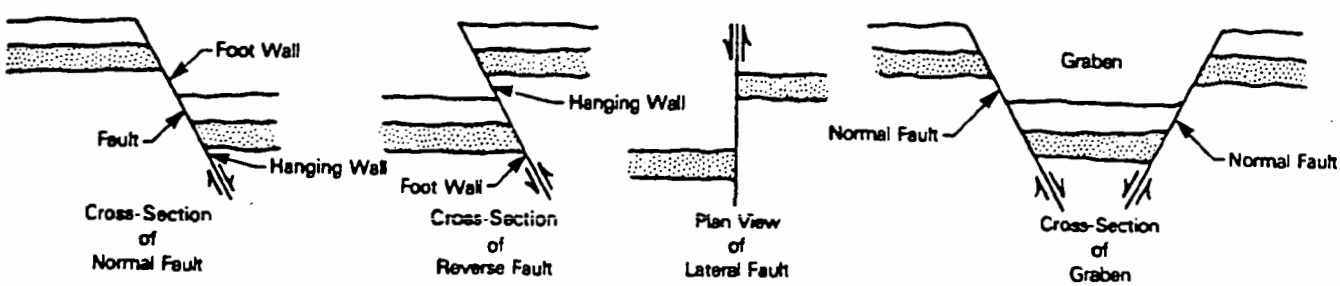


Figure 1-4. Cross Sections of Normal, Reverse and Lateral Faults

Era	Period	Epoch	Millions of Years Ago
Cenozoic	Quaternary	Recent	
		Pleistocene	0-2
	Tertiary	Pliocene	2-13
		Miocene	13-25
		Oligocene	25-36
		Eocene	36-58
		Paleocene	58-63
Mesozoic	Cretaceous		63-135
	Jurassic		135-181
	Triassic		181-230
Paleozoic	Permian		230-280
	Pennsylvanian		280-310
	Mississippian		310-345
	Devonian		345-405
	Silurian		405-425
	Ordovician		425-500
	Cambrian		500-600
Precambrian	Lasted at least 2.5 billion years		

Table 1-2. Geologic Time Scale

Rock Units

A rock unit refers to some particular lithology. These may be further divided into geologic formations, which are of sufficient size and uniformity to be mapped in the field. The Pierre Shale, for example, is a widespread and, in places, thick geologic formation that extends over much of the Northern Great Plains. Formations can also be divided into smaller units called members. Formations have a geographic name that may be coupled with a term that describes the major rock type. Two or more formations comprise a group.

Time and Time-Rock Units

Time-rock units refer to the rock that was deposited during a certain period of time. These units are divided into system, series, and stage. Time units refer to the time during which a sequence of rocks were deposited. The time-rock term, system, has the equivalent time term, period. That is, during the Cretaceous Period rocks of the Cretaceous System were deposited and they consist of many groups and formations. Time units are named in such a way that the eras reflect the complexity of life forms that existed, such as the Mesozoic or "middle life." System or period nomenclature largely is based on the geographic location in which the rocks were first described, such as Jurassic, which relates to the Jura Mountains of Europe.

The terms used by geologists to describe rocks relative to geologic time are useful to the ground-water investigator in that they allow one to better perceive a regional geologic situation. The terms alone have no significance as far as water-bearing properties are concerned.

Geologic Maps And Cross Sections

Geologists use a number of techniques to graphically represent surface and subsurface conditions. These include surficial geologic maps, columnar sections, cross-sections of the subsurface, maps that show the configuration of the surface of a geologic unit, such as the bedrock beneath glacial deposits, maps that indicate the thickness or grain size of a particular unit, a variety of contour maps, and a whole host of others.

A surficial geologic map depicts the geographic extent of formations and their structure. Columnar sections describe the vertical distribution of rock units, their lithology, and thickness. Geologic cross sections attempt to illustrate the subsurface distribution of rock units between points of control, such as outcrops or well bores. An isopach map shows the geographic range in thickness of a unit. These maps and cross-sections are based largely or entirely on well logs, which are descriptions of earth material penetrated during the drilling of a well or test hole.

Whatever the type of graphical representation, it must be remembered that maps of the subsurface and cross-sections represent only interpretations, most of which are based on scanty data. In reality, they are merely graphical renditions that are presumably based on scientific thought, a knowledge of depositional characteristics of rock units, and a data base that provides some control. They are not exact because the features they attempt to show are complex, nearly always hidden from view, and difficult to sample.

All things considered, graphical representations are exceedingly useful, if not essential, to subsurface studies. On the other hand, a particular drawing that is prepared for one purpose may not be suitable for another purpose even though the same units are involved. This is largely due to scale and generalizations.

A geologic map of a glaciated area is shown in fig. 1-5. The upland area is mantled by glacial till (Qgm) and the surficial material covering the relatively flat flood plain has been mapped as alluvium (Qal). Beneath the alluvial cover are other deposits of glacial origin that consist of glacial till, outwash, and glacial lake deposits. A water well drillers log of a boring in the valley states "this well is just like all of the others in the valley" and that the upper 70 feet of the valley fill consists of a "mixture of clay, sand, silt, and boulders." This is underlain by 30 feet of "water sand," which is the aquifer. The aquifer overlies "slate, jingle rock, and coal." The terminology may be quaint, but it is nonetheless a vocabulary that must be interpreted.

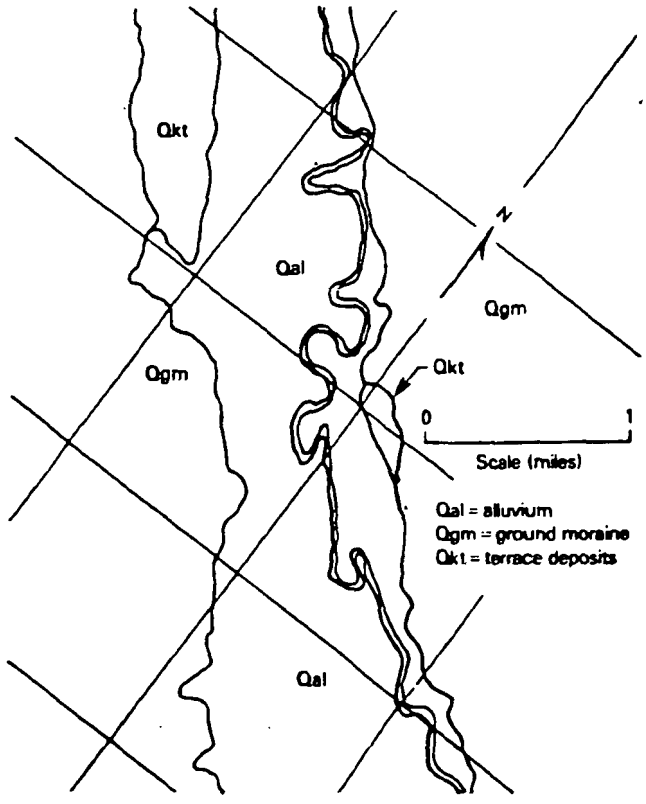


Figure 1-5. Generalized Geologic Map of a Glaciated Area Along the Souris River Valley in Central North Dakota

Examination of the local geology, as evidenced by strata that crop out along the hill sides, indicates that the bedrock or older material that underlies the glacial drift consists of shale, sandstone cemented by calcite, and lignite, which is an immature coal. These are the geologic terms, at least in this area, for "slate, jingle rock, and coal," respectively.

For generalized purposes, it is possible to use the drillers log to construct a cross section across or along the stream valley (fig. 1-6). In this case, one would assume for the sake of simplicity, the existence of an

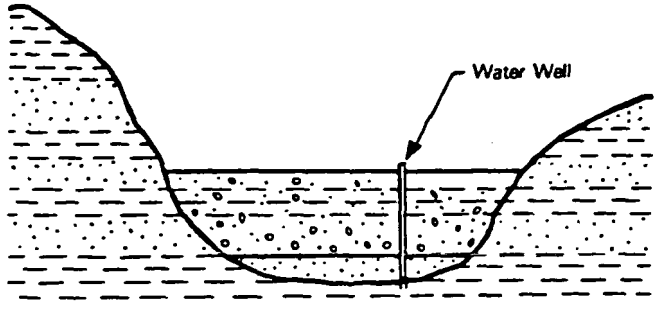


Figure 1-6. Generalized Geologic Cross Section of the Souris River Valley Based on Driller's Log

aquifer that is rather uniform in composition and thickness. A second generation cross section, shown in fig. 1-7, is based on several bore-hole logs described by a geologist who collected samples as the holes were being drilled. Notice in this figure that the subsurface appears to be much more complex, consisting of several isolated permeable units that are incorporated within the fine-grained glacial deposits that fill the valley. In addition, the aquifer does not consist of a uniform thickness of sand, but rather a unit that ranges from 30 to 105 feet in thickness and from sand to a mixture of sand and gravel. The water-bearing characteristics of each of these units are all different. This cross section too is quite generalized, which becomes evident as one examines an actual log of one of the bore holes (table 1-3).

In addition to showing more accurately the composition of the subsurface, well logs also can provide some interesting clues concerning the relative permeabilities of the water-bearing units. Referring to Table 1-4, a generalized log of well 1 describing the depth interval ranging from 62 to 92 feet, contains the remark "losing water" and in well 5, at a depths of 80 to 120 feet, is the notation, "3 bags of bentonite." In the first case "losing water" means that the material being penetrated by the drill bit from 62 to 92 feet was more permeable than the annulus of the cutting-filled bore hole. Some of the water used for drilling, which is pumped down the hole through the drill pipe to remove the cuttings, found it easier to move out into the formation than to flow back up the hole. The remark is a good indication of a permeability that is higher than that present in those sections where water was not being lost.

In the case of well 5, the material extending from 80 to 120 feet was so permeable that much of the drilling fluid was moving into the formation and there was no return of the cuttings. To regain circulation, bentonite, or to use the field term, "mud," was added to the drilling fluid to seal the permeable zone. Even though the geologist described the aquifer materials from both zones similarly, the section in well 5 is more permeable than the one in well 1, which in turn is more permeable than the other coarse-grained units penetrated where there was no fluid loss.

The three most important points to be remembered here are, first, graphical representations of the surface or subsurface geology are merely guesses of what might actually exist, and even these depend to some extent on the original intended usage. Secondly, the subsurface is far more complex than is usually anticipated, particularly in regard to unconsolidated deposits. Finally, evaluating the original data, such as well logs, might lead to a better appraisal of the

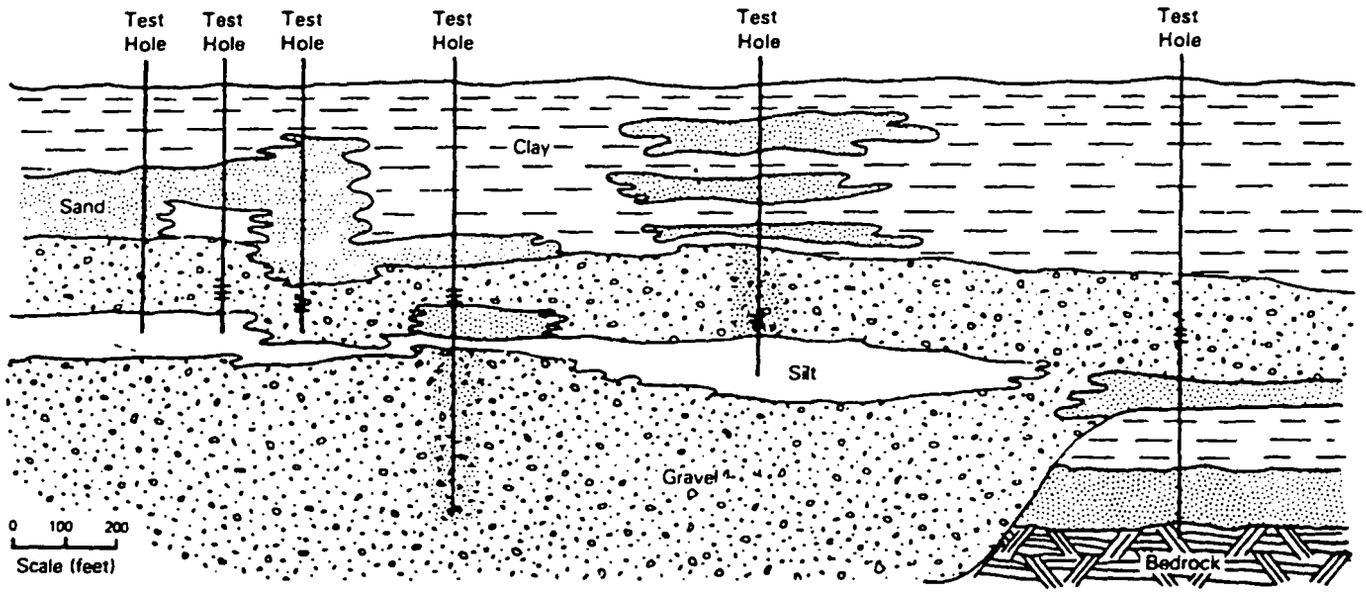


Figure 1-7. Geologic Cross Section of the Souris River Valley Based on Detailed Logs of Test Holes

Sample Description and Drilling Condition	Depth (ft)
Topsoil, silty clay, black	0-1
Clay, silty, yellow brown, poorly consolidated	1-5
Clay, silty, yellow gray, soft, moderately compacted	5-10
Clay, silty, as above, silty layers, soft	10-15
Silty, clayey, gray, soft, uniform drilling	15-20
Clay, silty, some fine to medium sand, gray	20-30
Clay, gray to black, soft, very tight	30-40
Clay, as above, gravelly near top	40-50
Clay, as above, no gravel	50-60
Clay, as above, very silty in spots, gray	60-70
Clay and silt, very easy drilling	70-80
Clay, as above to gravel, fine to coarse, sandy, thin clay layers, taking lots of water	80-90
Gravel, as above, some clay near top, very rough drilling, mixed three bags of mud, lots of lignite chips	90-100
Gravel, as above, cobbles and boulders	100-120
Gravel, as above, to sand, fine to coarse, lots of lignite, much easier drilling	120-130
Clay, gravelly and rocky, rough drilling, poor sample return	130-140
Sandy clay, gravelly and rocky, rough drilling, poor sample return (till)	140-150
Sandy clay, as above, poor sample return	150-160
Clay, sandy, gray, soft, plastic, noncalcareous	160-170
Clay, sandy, as above, tight, uniform drilling	170-180
Clay, as above, much less sand, gray, soft, tight, plastic	180-190
Clay, as above, no sand, good sample return	190-200
Clay, as above	200-210

Table 1-3. Geologist's Log of a Test Hole, Souris River Valley, North Dakota

subsurface, an appraisal that far surpasses the use of generalized lithologic logs alone.

Ground Water In Igneous And Metamorphic Rocks

Nearly all of the porosity and permeability of igneous and metamorphic rocks are the result of secondary openings, such as fractures, faults, and the dissolution of certain minerals. A few notable exceptions include large lava tunnels present in some flows, interflow or coarse sedimentary layers between individual lava flows, and deposits of selected pyroclastic materials.

Because the openings in igneous and metamorphic rocks are, volumetrically speaking, quite small, rocks of this type are poor suppliers of ground water. Moreover, the supplies that are available commonly drain rapidly after a period of recharge by infiltration of precipitation. In addition they are subject to contamination from the surface where these rocks crop out.

The width, spacing, and depth of fractures ranges widely, as do their origin. Fracture widths vary from about .0008 inches the surface to .003 inches at a depth of 200 feet, while spacing increased from 5 to 10 feet near the surface to 15 to 35 feet at depth in the Front Range of the Rocky Mountains (Snow, 1968). In the same area porosity decreased from below 300 feet or so, but there are many recorded exceptions. Exfoliation fractures in the crystalline rocks of the Piedmont near Atlanta, GA range from 1 to 8 inches in width (Cressler and others, 1983).

Material	Depth (ft)
Test Hole 1	
Fill	0-3
Silt, olive-gray	3-14
Sand, fine-medium	14-21
Silt, sandy, gray	21-25
Clay, gray	25-29
Sand, fine-coarse	29-47
Clay, gray	47-62
Gravel, fine to coarse, losing water	62-92
Silt, sandy, gray	92-100
Observation well depth 80 feet	
Test Hole 2	
Fill	0-2
Clay, silty and sandy, gray	2-17
Clay, very sandy, gray	17-19
Sand, fine-medium	19-60
Sand, fine-coarse with gravel	60-80
Gravel, coarse, 2 bags bentonite and bran	80-100
Observation well depth 88 feet	
Test Hole 3	
Silt, yellow	0-5
Clay, silty, black	5-15
Sand, fine to coarse	15-29
Clay, silty, gray	29-65
Sand, medium-coarse, some gravel	65-69
Gravel, sandy, taking water	69-88
Sand, fine to medium, abundant chips of lignite	88-170
Observation well depth 84 feet	
Test Hole 4	
Fill	0-5
Silt, brown	5-12
Sand, fine-medium	12-28
Clay, silty and sandy, gray	28-37
Sand, fine	37-49
Clay, dark gray	49-55
Sand, fine	55-61
Clay, sandy, gray	61-66
Sand, fine-coarse, some gravel	66-103
Silt, gray	103-120
Observation well depth 96 feet	
Test Hole 5	
Clay, silty, brown	0-10
Silt, clayey, gray	10-80
Gravel, fine-coarse, sandy, taking lots of water 3 bags bentonite	80-120
Sand, fine to coarse, gravelly	120-130
Clay, gravelly and rocky (till)	130-150
Sand, fine, Fort Union Group	150-180
Observation well depth 100 feet	

Table 1-4. Generalized Geologic Logs of Five Test Holes, Souris River Valley, North Dakota

The difficulty of evaluating water and contaminant movement in fractured rocks is that the actual direction of movement may not be in the direction of decreasing head, but rather in some different though related direction. The problem is further compounded by the difficulty in locating the fractures. Because of these characteristics, evaluation of water availability, direction of movement, and velocity is exceedingly difficult. As a general rule in the eastern part of the United States, well yields, and therefore fractures, permeability, and porosity, are greater in valleys and broad ravines than on flat uplands, which in turn is higher than on hill slopes and hill crests.

Unless some special circumstance exists, water obtained from igneous and metamorphic rocks is nearly always of excellent chemical quality. Dissolved solids are commonly less than 100 mg/L. Water from metamorphosed carbonate rocks may have moderate to high concentrations of hardness.

Ground Water In Sedimentary Rocks

Usable supplies of ground water can be obtained from all types of sedimentary rocks, but the fine-grained strata, such as shale and siltstone, may only provide a few gallons per day and even this can be highly mineralized. Even though fine-grained rocks may have relatively high porosities, the primary permeability is very low. On the other hand, shale is likely to contain a great number of joints that are both closely spaced and extend to depths of several tens of feet. Therefore, rather than being impermeable, they can be quite transmissive. This is of considerable importance in waste disposal schemes because insufficient attention might be paid during engineering design to the potential for flow through fractures. In addition, the leachate that is formed as water infiltrates through waste might be small in quantity but highly mineralized. Because of the low bulk permeability, it would be difficult to remove the contaminated water or even to properly locate monitoring wells.

From another perspective, fine-grained sedimentary rocks, owing to their high porosity, can store huge quantities of water. Some of this water can be released to adjacent aquifers when a head difference is developed by pumping. No doubt fine-grained confining units provide, on a regional scale, a great deal of water to aquifer systems. The porosity, however, decreases with depth because of compaction brought about by the weight of overlying sediments.

The porosity of sandstones range from less than 1 percent to a maximum of about 30 percent. This is a function of sorting, grain shape, and cementation. Cementation can be variable both in space and time and on outcrops can differ greatly from that in the subsurface.

Fractures also play an important role in the movement of fluids through sandstones and transmissivities may be as much as two orders of magnitude greater in a fractured rock than in an unfractured part of the same geologic formation.

Sandstone units that were deposited in a marine or near marine environment can be very wide spread, covering tens of thousands of square miles, such as the St. Peter Sandstone of Cambrian age. Those representing ancient alluvial channel fills, deltas, and related environments of deposition are more likely to be discontinuous and erratic in thickness. Individual units are exceedingly difficult to trace in the subsurface. Regional ground-water flow and storage may be strongly influenced by the geologic structure.

Carbonate rocks are formed in many different environments and the original porosity and permeability are modified rapidly after burial. Some special carbonate rocks, such as coquina and some breccias, may remain very porous and permeable, but these are the exception.

It is the presence of fractures and other secondary openings that develop high yielding carbonate aquifers. One important aspect is the change from calcite to dolomite ($CaMg[CO_3{}_2]$), which results in a volumetric reduction of 13 percent and the creation of considerable pore space. Of particular importance and also concern in many of the carbonate regions of the world, is the dissolution of carbonates along fractures and bedding planes by circulating ground water. This is the manner in which caves and sinkholes are formed. As dissolution progresses upward in a cave, the overlying rocks may collapse to form a sinkhole that contains water if the cavity extends below the water table. Regions in which there has been extensive dissolution of carbonates leading to the formation of caves, underground rivers, and sinkholes, are called karst. Notable examples include parts of Missouri, Indiana, and Kentucky.

Karst areas are particularly troublesome, even though they can provide large quantities of water to wells and springs. They are easily contaminated, and it is commonly difficult to trace the contaminant because the water can flow very rapidly, and there is no filtering action to degrade the waste. Not uncommonly a well owner may be unaware that he is consuming unsafe water. An individual in Kentucky became concerned because his well yield had declined. The well, which drew water from a relatively shallow cave below the water table, was cased with a pipe, on the end of which was a screen. When the screen was pulled, it was found to be completely coated with fibrous material. The owner was disconcerted to learn that the fibrous covering was derived from toilet paper.

Ground Water In Unconsolidated Sediments

Unconsolidated sediments accumulate in many different environments, all of which leave their trademark on the characteristics of the deposit. Some are thick and areally extensive, as the alluvial fill in the Basin and Range Province, others are exceedingly long and narrow, such as the alluvial deposits along streams and rivers, and others may cover only a few hundred square feet, like some glacial forms. In addition to serving as major aquifers, unconsolidated sediments are also important as sources of raw materials for construction.

Although closely related to sorting, the porosities of unconsolidated materials range from less than 1 to more than 90 percent, the latter representing uncompacted mud. Permeabilities also range widely. Cementing of some type and degree is probably universal, but not obvious, with silt and clay being the predominant form.

Most unconsolidated sediments owe their emplacement to running water and, consequently, some sorting is expected. On the other hand, water as an agent of transportation will vary in both volume and velocity, which is climate dependent, and this will leave an imprint on the sediments. It is to be expected that stream related material, which most unconsolidated material is, will be variable in extent, thickness, and grain size. Other than this, one can draw no general guidelines; therefore, it is essential to develop some knowledge of the resulting stratigraphy that is characteristic of the most common environments of deposition. The water-bearing properties of glacial drift, of course, are exceedingly variable, but stratified drift is more uniform and better sorted than glacial till

Relation Between Geology, Climate, and Ground-Water Quality

The availability of ground-water supplies and their chemical quality are closely related to precipitation. As a general rule, the least mineralized water, both in streams and underground, occurs in areas of the greatest amount of rainfall. Inland, precipitation decreases, water supplies diminish, and the quality deteriorates. The mineral composition of water-bearing rocks exerts a strong influence on ground-water quality and thus, the solubility of the rocks may override the role of precipitation.

Where precipitation exceeds 40 inches per year, shallow ground water usually contains less than 500 mg/L and commonly less than 250 mg/L of dissolved solids. Where precipitation ranges between 20 and 40 inches, dissolved solids may range between 400 and 1,000 mg/L, and in drier regions they commonly exceed 1,000 mg/L.

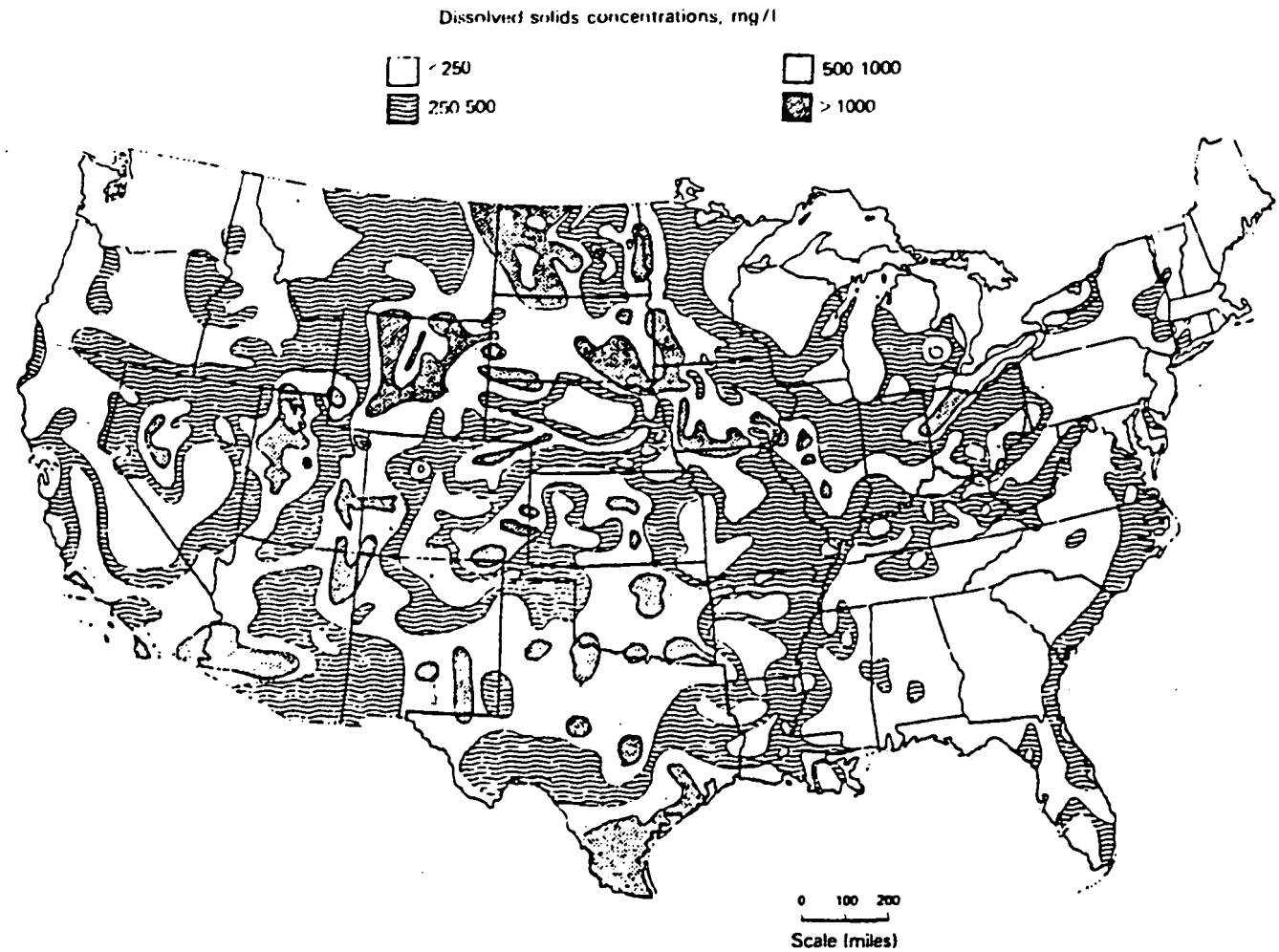


Figure 1-8. Dissolved Solids Concentrations in Ground Water Used for Drinking in the United States (from Pettyjohn and others, 1979)

The dissolved solids concentration of ground water increases toward the interior of the continent. The increase is closely related to precipitation and the solubility of the aquifer framework. The least mineralized ground water is found in a broad belt that extends southward from the New England states, along the Atlantic Coast to Florida, and then continues to parallel much of the Gulf Coast. Similarly, along the Pacific Coast from Washington to central California, the mineral content is also very low. Throughout this belt, dissolved solids concentrations generally are less than 250 mg/L and commonly less than 100 mg/L (fig. 1-8).

The Appalachian region consists of a sequence of strata that range from nearly flat-lying to complexly folded and faulted. Likewise, ground-water quality in this region also is highly variable, being generally harder and containing more dissolved minerals than does water along the coastal belt. Much of the difference in quality, however, is related to the abundance of carbonate aquifers, which provide waters rich in calcium and magnesium.

Westward from the Appalachian Mountains to about the position of the 20-inch precipitation line (eastern North Dakota to Texas), dissolved solids in ground water progressively increase. They are generally less than 1,000 mg/L and are most commonly in the 250 to 750 mg/L range. The water is moderately to very hard, and in some areas concentrations of sulfate and chloride are excessive.

From the 20-inch precipitation line westward to the northern Rocky Mountains, dissolved solids are in the 500 to 1,500 mg/L range. Much of the water from glacial drift and bedrock formations is very hard and

contains significant concentrations of calcium sulfate. Other bedrock formations may contain soft sodium bicarbonate, sodium sulfate, or sodium chloride water.

Throughout much of the Rocky Mountains, ground-water quality is variable, although the dissolved solids concentrations commonly range between 250 and 750 mg/L. Stretching southward from Washington to southern California, Arizona, and New Mexico is a vast desert region. Here the difference in quality is wide and dissolved solids generally exceed 750 mg/L. In the central parts of some desert basins the ground water is highly mineralized, but along the mountain flanks the mineral content may be quite low.

Extremely hard water is found over much of the Interior Lowlands, Great Plains, Colorado Plateau, and Great Basin. Isolated areas of high hardness are present in northwestern New York, eastern North Carolina, the southern tip of Florida, northern Ohio, and parts of southern California. In general, the hardness is of the carbonate type.

On a regional level, chloride does not appear to be a significant problem, although it is troublesome locally due largely to industrial activities, the intrusion of seawater caused by overpumping coastal aquifers, or interaquifer leakage related to pressure declines brought about by withdrawals.

In many locations, sulfate levels exceed the federal recommended limit of 250 mg/L; regionally sulfate may be a problem only in the Great Plains, eastern Colorado Plateau, Ohio, and Indiana. Iron problems are ubiquitous because concentrations exceeding only .3 mg/L will cause staining of clothing and fixtures. Fluoride is abnormally high in several areas, particularly parts of western Texas, Iowa, Illinois, Indiana, Ohio, New Mexico, Wyoming, Utah, Nevada, Kansas, New Hampshire, Arizona, Colorado, North and South Dakota, and Louisiana.

A water-quality problem of growing concern, particularly in irrigated regions, is nitrate, which is derived from fertilizers, sewage, and through natural causes. When consumed by infants less than six months old for a period of time, high nitrate concentrations can cause a disease known as "blue babies." This occurs because the child's blood cannot carry sufficient oxygen; the disease is easily overcome by using low nitrate water for formula preparation. Despite the fact that nitrate concentrations in ground water appear to have been increasing in many areas during the last 30 years or so, there have been no reported incidences of "blue babies" for more than 20 years, at least in the states that comprise the Great Plains.

Conclusions

In detail, the study of geology is complex, but the principles outlined above should be sufficient for a general understanding of the topic, particularly as it relates to ground water. If interested in a more definitive treatment, the reader should examine the references at the end of the chapter.

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Chapter 2

CLASSIFICATION OF GROUND-WATER REGIONS

To describe concisely ground-water conditions in the United States, it is necessary to divide the country into regions in which these conditions are generally similar. Because the presence and availability of ground water depends primarily on geologic conditions, ground-water regions also are areas in which the composition, arrangement, and structure of rock units are similar (Heath, 1982).

To divide the country into ground-water regions, it is necessary to develop a classification that identifies features of ground-water systems that affect the occurrence and availability of ground water. The five features pertinent to such a classification are: (I) the components of the system and their arrangement, (2) the nature of the water-bearing openings of the dominant aquifer or aquifers with respect to whether they are of primary or secondary origin, (3) the mineral composition of the rock matrix of the dominant aquifers with respect to whether it is soluble or insoluble, (4) the water storage and transmission characteristics of the dominant aquifer or aquifers, and (5) the nature and location of recharge and discharge areas.

The first two of these features are primary criteria used in all delineations of ground-water regions. The remaining three are secondary criteria that are useful in subdividing what might otherwise be large and unwieldy regions into areas that are more homogeneous and, therefore, more convenient for descriptive purposes. Table 2-1 lists each of the five features together with explanatory information. The fact that most of the features are more or less interrelated is readily apparent from the comments in the column headed "Significance of Feature."

Ground-Water Regions of the United States

On the basis of the criteria listed above the United States, exclusive of Alaska and Hawaii, can be divided into 11 ground-water regions.

Figure 2-1 shows the boundaries of these 11 regions. A special area, region 12, which consists of those segments of the valleys of perennial streams that are underlain by sand and gravel thick enough to be hydrologically significant (thicknesses generally more than about 26 feet), is shown in Figure 2-2.

The nature and extent of the dominant aquifers and their relations to other units of the ground-water system are the primary criteria used in delineating the regions. Consequently, the boundaries of the regions generally coincide with major geologic boundaries and at most places do not coincide with drainage divides. Although this lack of coincidence emphasizes that the physical characteristics of ground-water systems and stream systems are controlled by different factors, it does not mean that the two systems are not related. Ground-water systems and stream systems are intimately related, as shown in the following discussions of each of the ground-water regions.

1. Western Mountain Ranges

(Mountains with thin soils over fractured rocks, alternating with narrow alluvial and, in part, glaciated valleys)

The Western Mountain Ranges, shown in Figure 2-3, encompass three areas totaling 278,000 mi^2. The largest area extends in an arc from the Sierra Nevada in California, north through the Coast Ranges and Cascade Mountains in Oregon and Washington, and east and south through the Rocky Mountains in Idaho and Montana into the Bighorn Mountains in Wyoming and the Wasatch and Uinta Mountains in Utah. The second area includes the southern Rocky Mountains, which extend from the Laramie Range in southeastern Wyoming through central Colorado into the Sangre de Cristo Range in northern New Mexico. The smallest area includes the part of the Black Hills of South Dakota in which Precambrian rocks are exposed.

As would be expected in such a large region, both the origin of the mountains and the rocks that form them are complex. Most of the mountain ranges are underlain by

Feature	Aspect	Range in Conditions	Significance of Feature
Component of the system	Unconfined aquifer	Thin, discontinuous, hydrologically insignificant. Minior aquifer, serves primarily as a storage reservoir and recharge condult for underlying aquifer. The dominant aquifer.	Affect response of the system to pumpage and other stresses. Affect recharge and discharge conditions. Determine susceptibility to pollution.
	Confining beds	Not present, or hydrologically insignificant. Thin, markedly discontinuous, or very leaky. Thick, extensive, and impermeable. Complexly interbedded with aquifers or productive zones.	
	Confined aquifers	Not present, or hydrologically insignificant. Thin or not highly productive. Multiple thin aquifers interbedded with nonproductive zones. The dominant aquifer—thick and productive.	
	Presence and arrangements of components	A single, unconfined aquifer. Two interconnected aquifers of essentially equal hydrologic importance. A three-unit system consisting of an unconfined aquifer, a confining bed, and confined aquifer. A complexly interbedded sequence of aquifers and confining beds.	
Water-bearing openings of dominant aquifer	Primary openings	Pores in unconsolidated deposits. Pores in semiconsolidated rocks. Pores, tubes, and cooling fractures in volcanic (extrusive-igneous) rocks.	Control water-storage and transmission characteristics. Affect disperson and dilution of wastes.
	Secondary openings	Fractures and faults in crystalline and consolidated sedimentary rocks. Solution-enlarged openings in limestones and other soluble rocks.	
Composition of rock matrix of dominant aquifer	Insoluble	Essentially insoluble. Both relatively insoluble and soluble constituents.	Affects water-storage and transmission characteristics. Has major influence on water quality.
	Soluble	Relatively soluble.	
Storage and transmission characteristics of dominant aquifer	Porosity	Large, as in well-sorted, unconsolidated deposits. Moderate, as in poorly-sorted unconsolidated deposits and semiconsolidated rocks. Small, as in fractured crystalline and consolidated sedimentary rocks.	Control response to pumpage and other stresses. Determine yield of wells. Affect long-term yield of system. Affect rate at which pollutants move.
	Transmissivity	Large, as in cavernous limestones, some lava flows, and clean gravels. Moderate, as in well-sorted, coarse-grained sands, and semiconsolidated limestones. Small, as in poorly-sorted, fine-grained deposits and most fractured rocks. Very small, as in confining beds.	
Recharge and discharge conditions of dominant aquifer	Recharge	In upland areas between streams, particularly in humid regions. Through channels of losing streams. Largely or entirely by leakage across confining beds from adjacent aquifers.	Affect response to stress and long-term yields. Determine susceptibility to pollution. Affect water quality.
	Discharge	Through springs or by seepage to stream channels, estuaries, or the ocean. By evaporation on flood plains and in basin "sinks." By seepage across confining beds into adjacent aquifers.	

Table 2-1. Features of Ground-Water Systems Useful in the Delineation of Ground-Water Regions

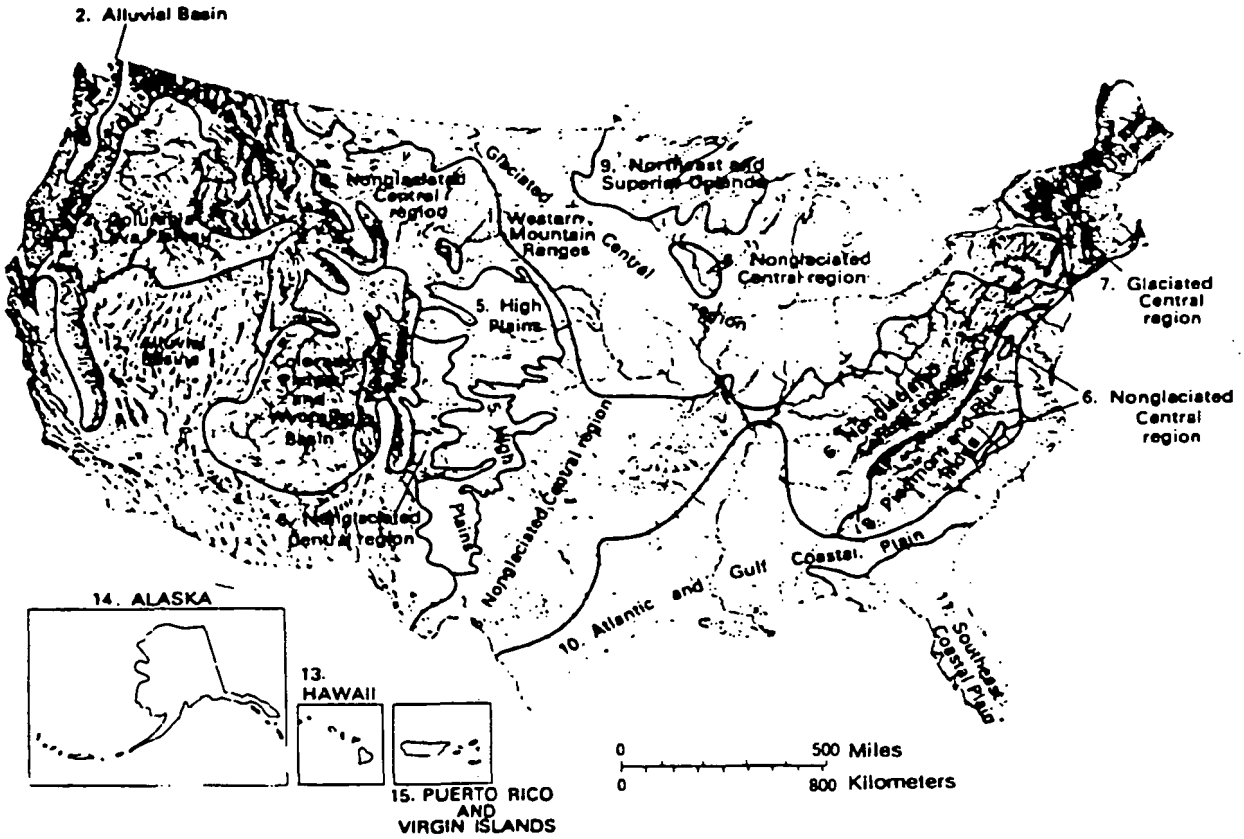


Figure 2-1. Ground-Water Regions Used in This Report [The Alluvial Valleys Region (region 12) is shown on figure 2-2]

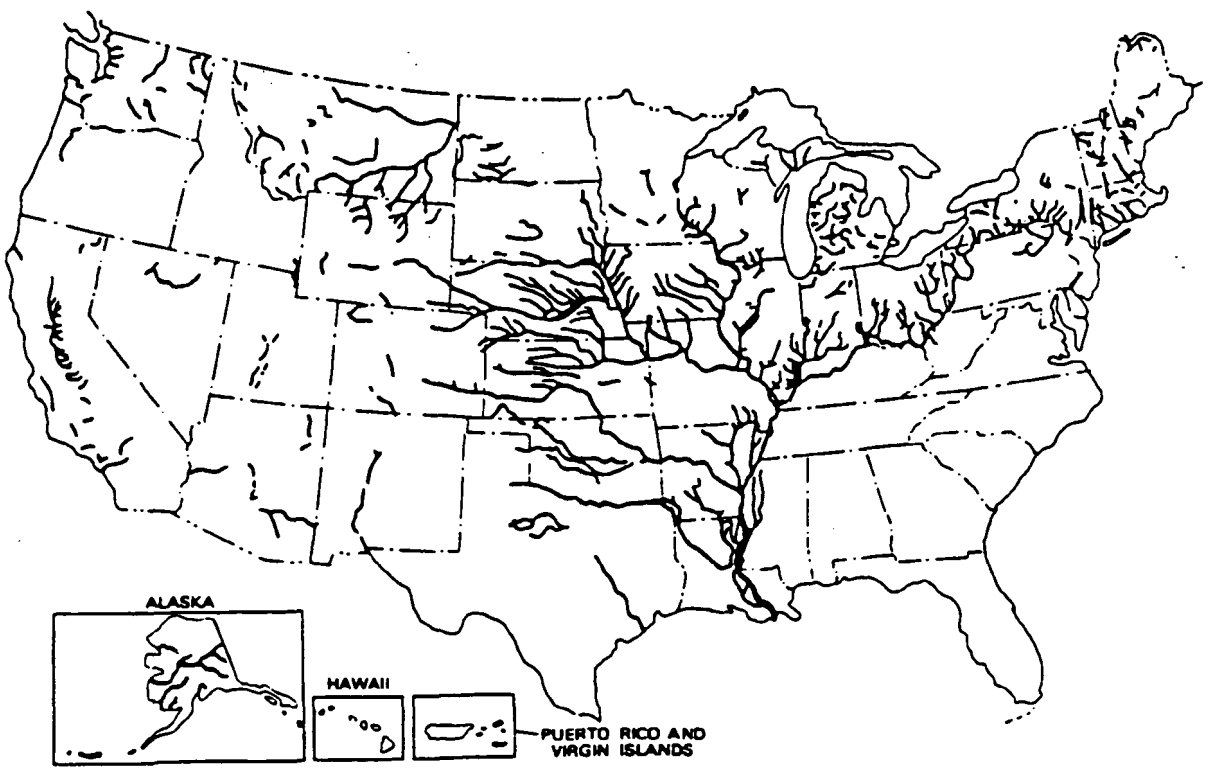


Figure 2-2. Alluvial Valleys Ground-Water Region

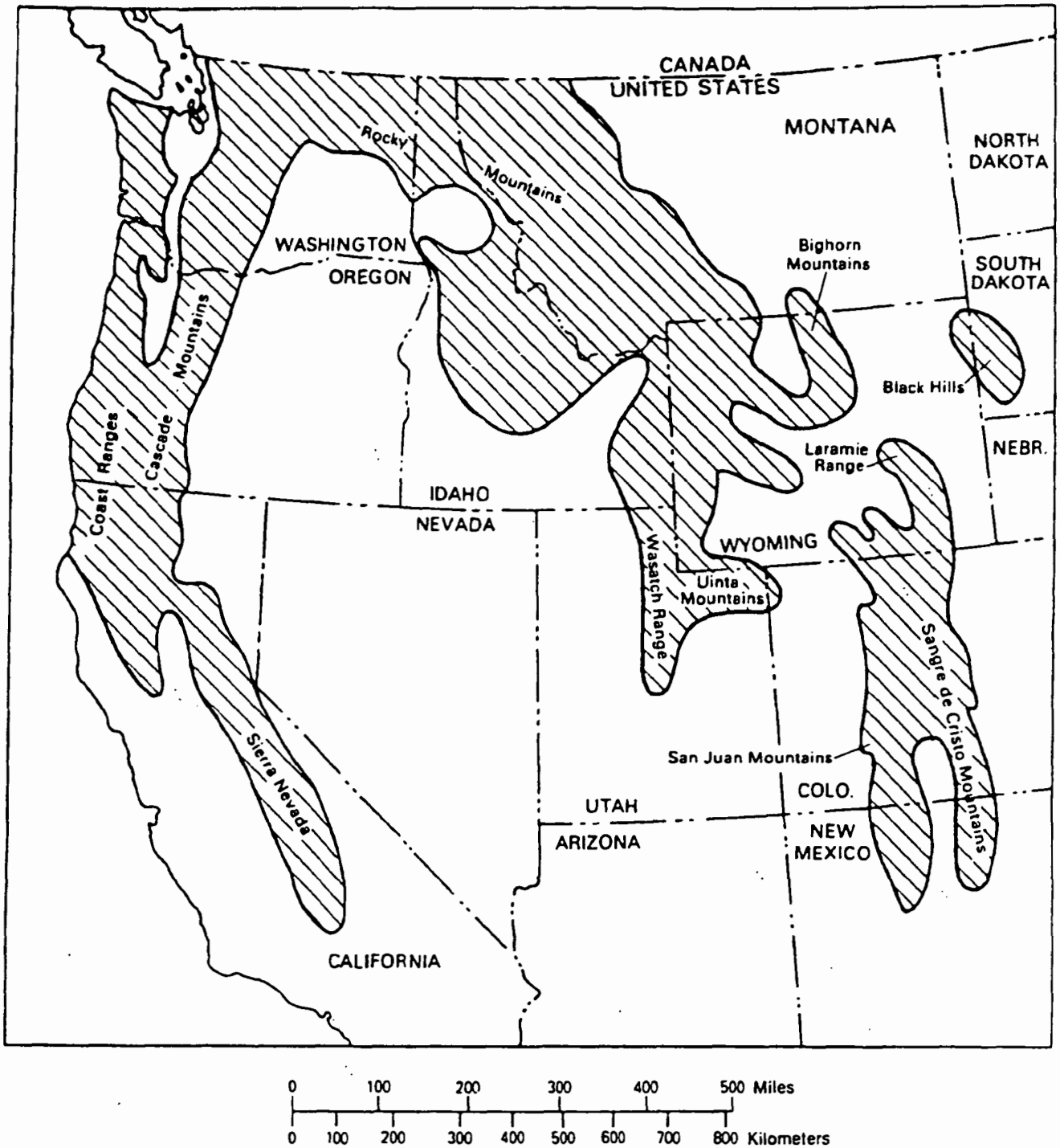


Figure 2-3. Western Mountain Ranges Region

granitic and metamorphic rocks flanked by consolidated sedimentary rocks of Paleozoic to Cenozoic age. The other ranges, including the San Juan Mountains in southwestern Colorado and the Cascade Mountains in Washington and Oregon, are underlain by lavas and other igneous rocks.

The summits and slopes of most of the mountains consist of bedrock exposures or of bedrock covered by a layer of boulders and other rock fragments produced by frost action and other weathering processes acting on the bedrock. This layer is generally only a few feet thick on the upper slopes but forms a relatively thick apron along the base of the mountains. The narrow valleys are underlain by relatively thin, coarse, bouldery alluvium washed from the higher slopes. The large synclinal valleys and those that occupy downfaulted structural troughs are underlain by moderately thick deposits of coarse-grained alluvium transported by streams from the adjacent mountains, as shown in Figure 2-4.

The Western Mountain Ranges and the mountain ranges in adjacent regions are the principal sources of water supplies developed at lower altitudes in the western half of the conterminous United States. As McGuinness (1963) noted, the mountains of the West are moist "islands" in a sea of desert or semidesert that covers the western half of the Nation. The heaviest precipitation falls on the western slopes; thus, these slopes are the major source of runoff and are also the most densely vegetated. Much of the precipitation falls as snow during the winter.

The Western Mountain Ranges are sparsely populated and have relatively small water needs. The region is an exporter of water to adjacent "have-not" areas. Numerous surface reservoirs have been constructed in the region. Many such impoundments have been developed on streams that drain the western flank of the Sierra Nevada in California and the Rocky Mountains in Colorado.

Melting snow and rainfall at the higher altitudes in the region provide abundant water for ground-water recharge. However, the thin soils and bedrock fractures in areas underlain by crystalline rocks fill quickly, and the remaining water runs off overland to streams. Because of their small storage capacity, the underground openings provide limited base runoff to the streams, which at the higher altitudes flow only during rains or snowmelt periods. Thus, at the higher altitudes in this

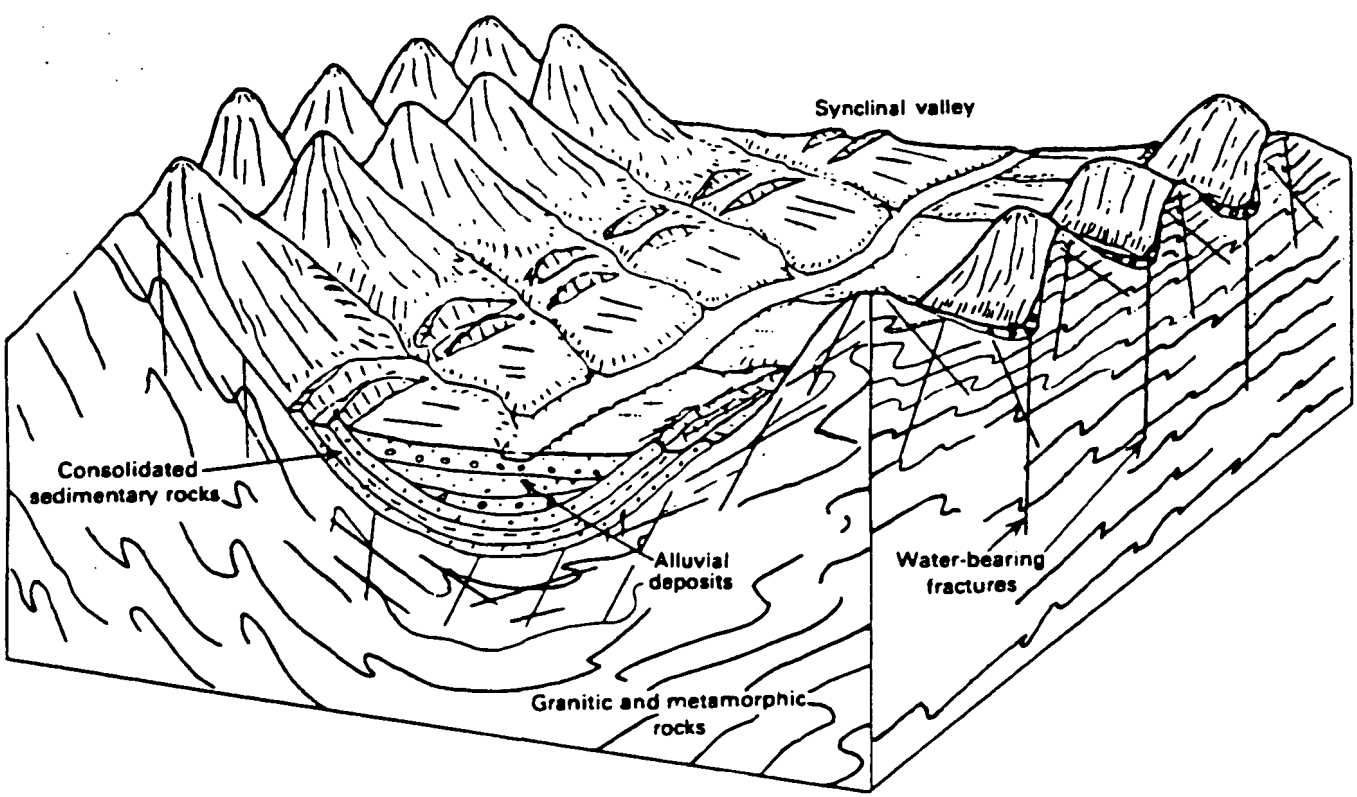


Figure 2-4. Topographic and Geologic Features in the Southern Rocky Mountains Part of the Western Mountain Ranges Region

region underlain by crystalline rocks, relatively little opportunity exists for development of ground-water supplies. The best opportunities exist in the valleys that contain at least moderate thicknesses of saturated alluvium or in areas underlain by permeable sedimentary or volcanic rocks. Ground-water supplies in the valleys are obtained both from wells drawing from the alluvium and from wells drawing from the underlying rocks. The yields of wells in crystalline bedrock and from small, thin deposits of alluvium are generally adequate only for domestic and stock needs. Large yields can be obtained from the alluvial deposits that overlie the major lowlands and from wells completed in permeable sedimentary or volcanic rocks.

2. Alluvial Basins
(Thick alluvial deposits in basins and valleys bordered by mountains and locally of glacial origin)

The Alluvial Basins region occupies a discontinuous area of 396,000 mi^2 extending from the Puget Sound-Williamette Valley area of Washington and Oregon to west Texas. The region consists of an irregular alternation of basins or valleys and mountain ranges. In the Alluvial Basins region, basins and valleys are the dominant feature. The principal exception is the Coast Ranges of southern California which topographically more closely resemble the Western Mountain Ranges.

Most of the Nevada and all of the Utah parts of this region are an area of internal drainage referred to as the Great Basin. No surface or subsurface flow leaves this part of the region and all water reaching it from adjacent areas and from precipitation is evaporated or transpired.

The basins and valleys range from about 280 ft below sea level in Death Valley in California to 6,550 ft above sea level in the San Luis Valley in Colorado. The basins range in size from a few hundred feet in width and a mile or two in length to, for the Central Valley of California, as much as 50 mi in width and 400 mi in length. The crests of the mountains are commonly 3,300 to 4,900 ft above the adjacent valley floors.

The surrounding mountains, and the bedrock beneath the basins, consist of granite and metamorphic rocks of Precambrian to Tertiary age and consolidated sedimentary rocks of Paleozoic to Cenozoic age. The rocks are broken along fractures and faults that may serve as water-bearing openings. However, the openings in the granitic and metamorphic rocks in the mountainous areas have a relatively small capacity to store and to transmit ground water.

The dominant element in the hydrology of the region is the thick (several hundred to several thousand feet) layer of generally unconsolidated alluvial material that partially fills the basins. Figures 2-5, 2-6, and 2-7 illustrate this dominant element. Generally, the coarsest material occurs adjacent to the mountains; the material gets progressively finer toward the centers of the basins. However, as Figure 2-6 shows, in most alluvial fans there are layers of sand and gravel that extend into the central parts of the basins. In time, the fans formed by adjacent streams coalesced to form a continuous and thick deposit of alluvium that slopes gently from the

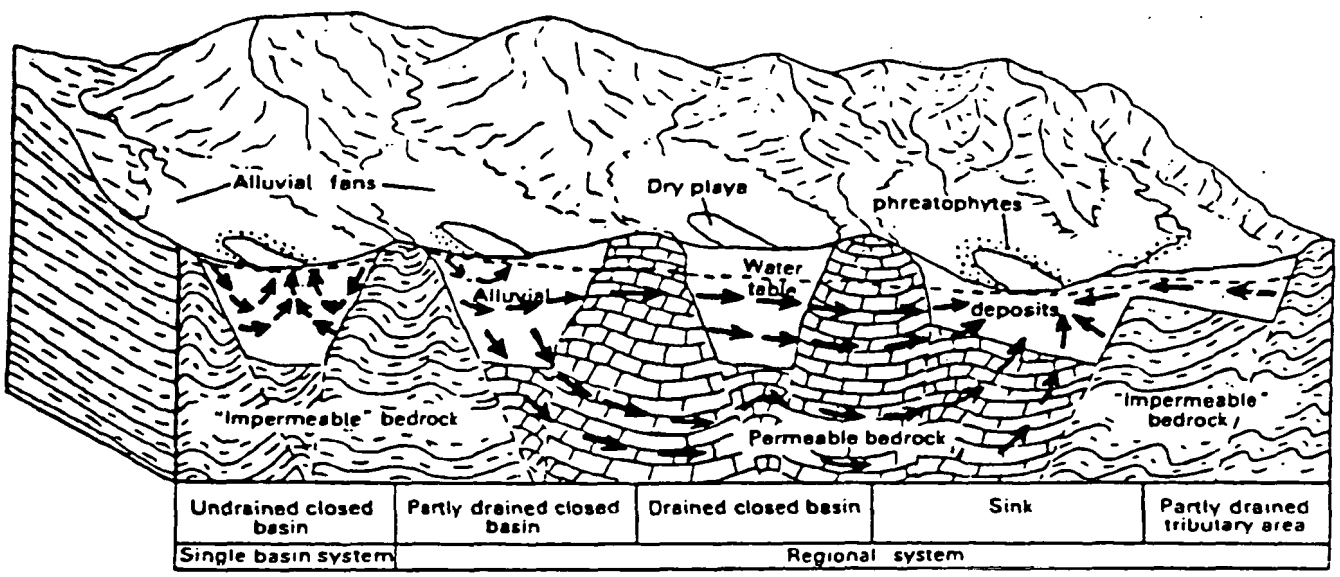


Figure 2-5. Common Ground-Water Flow Systems in the Alluvial Basins Region (From U.S. Geological Survey Professional Paper 813-G)

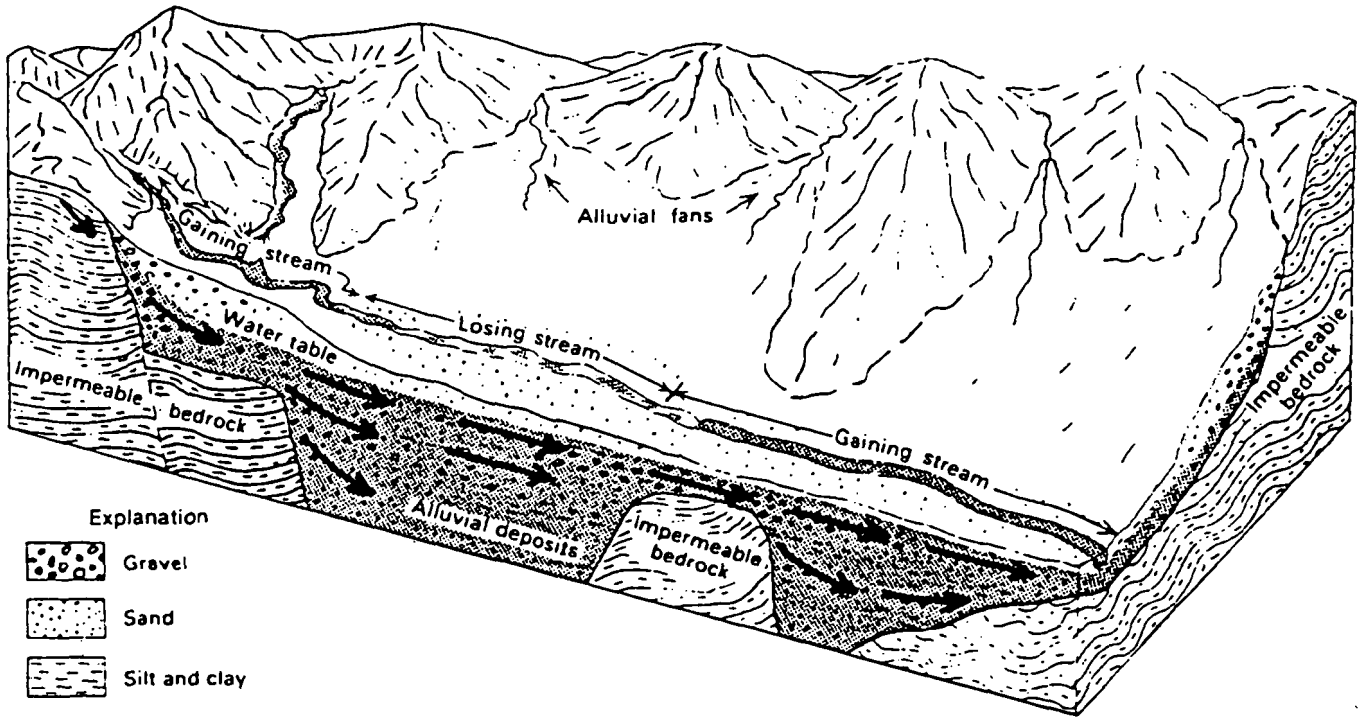


Figure 2-6. Common Relationships between Ground Water and Surface Water in the Alluvial Basins Region (Modified from U.S. Geological Survey Professional Paper 813-G)

mountains toward the center of the basins. These alluvial-fan deposits are overlain by or grade into fine-grained flood plain, lake, or playa deposits in the central part of most basins. The fine-grained deposits are especially suited to large-scale cultivation.

The Puget Sound and Williamette Valley areas differ geologically from the remainder of the region. The Puget Sound area is underlain by thick and very permeable deposits of gravel and sand laid down by glacial meltwater. The gravel and sand are interbedded with clay in parts of the area. The Williamette Valley is mostly underlain by interbedded sand, silt, and clay deposited on floodplains by the Williamette River and other streams.

The Alluvial Basins region is the driest area in the United States, with large parts of it being classified as semiarid and arid. Annual precipitation in the valleys in Nevada and Arizona ranges from about 4 to 6 in. However, in the mountainous areas throughout the region, in the northern part of the Central Valley of California, and in the Washington-Oregon area, annual precipitation ranges from about 16 in to more than 31 in. The region also receives runoff from streams that originate in the mountains of the Western Mountain Ranges region.

Because of the very thin cover of unconsolidated material on the mountains, precipitation runs off rapidly down the valleys and out onto the fans, where it infiltrates. The water moves through the sand and gravel layers toward the centers of the basins. The centers of many basins consist of flat-floored, vegetation-free areas onto which ground water may discharge and on which overland runoff may collect during intense storms. The water that collects in these areas (playas), evaporates relatively quickly, leaving both a thin deposit of clay and other sediment and a crust of the soluble salts that were dissolved in the water, as Figure 2-5 illustrates.

Studies in the region have shown that the hydrology of the alluvial basins is more complex than that described in the preceding paragraph, which applies only to what has been described as "undrained closed basins." As Figure 2-5 shows, water may move through permeable bedrock from one basin to another, arriving, ultimately, at a large playa referred to as a "sink." Water discharges from sinks not by "sinking" into the ground, but by evaporating. In those parts of the region drained by perennial watercourses ground water discharges to the streams from the alluvial deposits. However, before entering the streams, water may move down some valleys through the alluvial deposits for tens of miles. A reversal of this situation occurs along the lower Colorado River and at the upstream end of the valleys of some of the other perennial streams; in these areas, water moves from the streams into the alluvium to supply the needs of the adjacent vegetated zones.

Figure 2-7. Areas Underlain by Sand and Gravel in the Alluvial Basins Region

Ground water is the major source of water in the Alluvial Basins region. Because of the dry climate, agriculture requires intensive irrigation. Most of the ground water is obtained from the sand and gravel deposits in the valley alluvium. These deposits are interbedded with finer grained layers of silt and clay that are also saturated with water. When hydraulic heads in the sand and gravel layers are lowered by withdrawals, the water in the silt and clay begins to move slowly into the sand and gravel. The movement, which in some areas takes decades to become significant, is accompanied by compaction of the silt and clay and subsidence of the land surface. Subsidence is most severe in parts of the Central Valley, where it exceeds 30 ft in one area, and in southern Arizona, where subsidence of more than 13 ft has been observed.

3. Columbia Lava Plateau

(Thick sequence of lava flows irregularly interbedded with thin unconsolidated deposits and overlain by thin soils)

As Figure 2-8 shows, the Columbia Lava Plateau occupies an area of 141,000 mi^2 in northeastern California, eastern Washington and Oregon, southern Idaho, and northern Nevada. As its name implies, it is basically a plateau, standing generally between 1,640 and 5,900 ft above sea level, that is underlain by a great thickness of lava flows irregularly interbedded with silt, sand, and other unconsolidated deposits.

The great sequence of lava flows, which ranges in thickness from less than 160 ft adjacent to the bordering mountain ranges to more than 3,300 ft in south-central Washington and southern Idaho, is the principal water-bearing unit in the region. As Figure 2-9 shows, the water-bearing lava is underlain by granite, metamorphic rocks, older lava flows, and sedimentary rocks, none of which are very permeable. Individual lava flows in the water-bearing zone range in thickness from several feet to more than 160 ft and average about 50 ft. The volcanic rocks yield water mainly from permeable zones that occur at or near the contacts between some flow layers. Parts of some flows are separated by soil zones and, at places, by sand, silt, and clay. These sedimentary layers, where they occur between lava flows, are commonly referred to as "interflow sediments." Gravel, sand, silt, and clay cover the volcanic rocks and the older exposed bedrock in parts of the area.

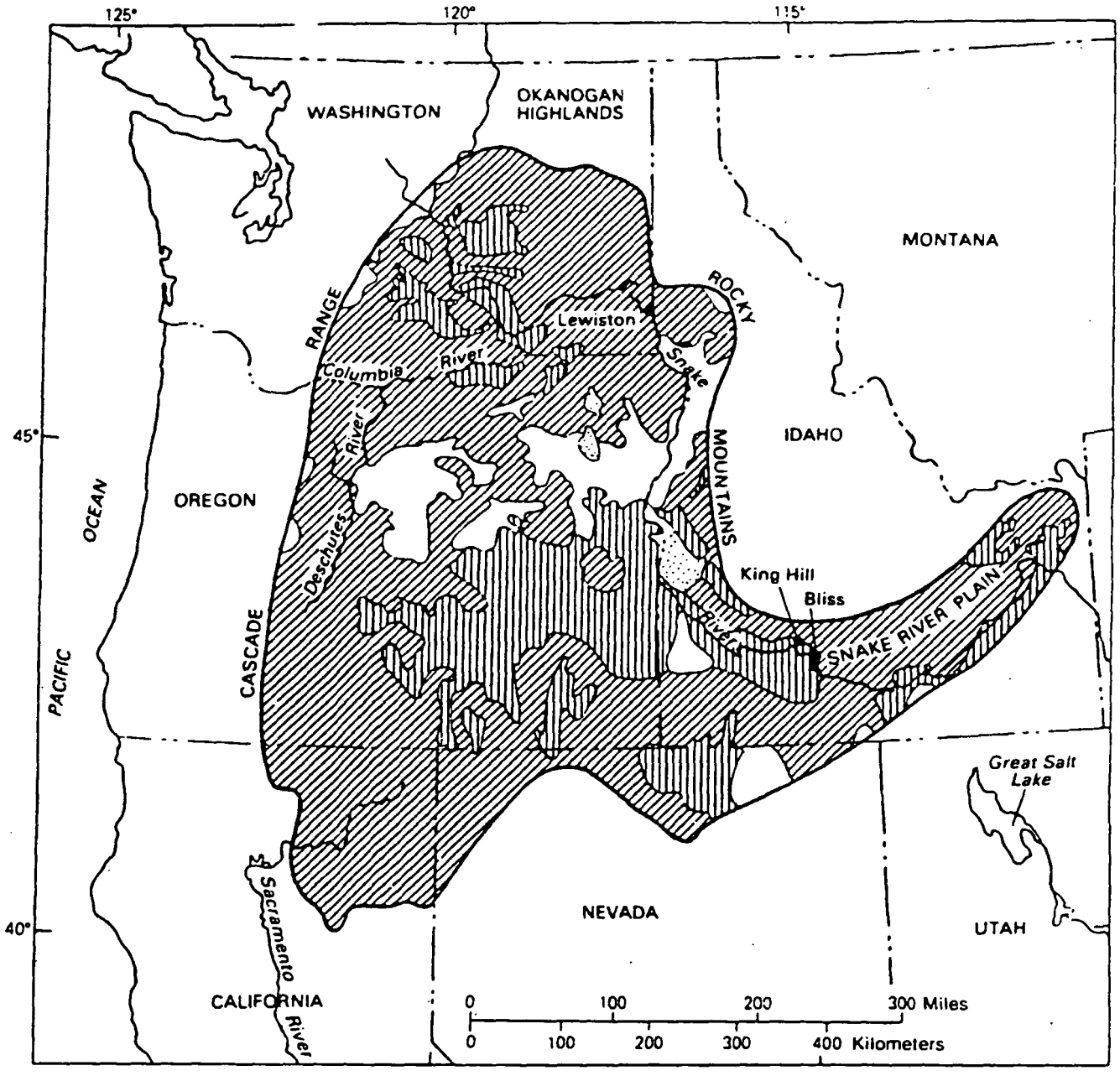
From the standpoint of the hydraulic characteristics of the volcanic rocks, it is useful to divide the Columbia Lava Plateau region into two parts: (I) the area in southeastern Washington, northeastern Oregon, and the Lewiston area of Idaho, part of which is underlain by volcanic rocks of the Columbia River Group; and (2) the remainder of the area shown on Figure 2-8, which also includes the Snake River Plain. The basalt underlying the Snake River Plain is referred to as the Snake River Basalt; that underlying southeastern Oregon and the remainder of this area has been divided into several units, to which names of local origin are applied (Hampton, 1964).

The Columbia River Group is of Miocene to Pliocene (?) age and consists of relatively thick flows that have been deformed into a series of broad folds and offset locally along normal faults. Movement of ground water occurs primarily through the interflow zones near the top of flows and, to a much smaller extent, through fault zones and through joints developed in the dense central and lower parts of the flows. The axes of sharp folds and the offset of the interflow zones along faults form subsurface dams that affect the movement of ground water. Water reaching the interflow zones tends to move down the dip of the flows from fold axes and to collect updip behind faults that are transverse to the direction of movement (Newcomb, 1962). As a result, the basalt in parts of the area is divided into a series of barrier-controlled reservoirs, which are only poorly connected hydraulically to adjacent reservoirs.

The water-bearing basalt underlying California, Nevada, southeastern Oregon, and southern Idaho is of Pliocene to Holocene age and consists of small, relatively thin flows that have been affected to a much smaller extent by folding and faulting than has the Columbia River Group. The thin flows contain extensive, highly permeable interflow zones that are relatively effectively interconnected through a dense network of cooling fractures. Structural barriers to ground-water movement are of minor importance. This is demonstrated by conditions in the 17,000 mi^2 area of the Snake River Plain east of Bliss, Idaho.

The interflow zones form a complex sequence of relatively horizontal aquifers that are separated vertically by the dense central and lower parts of the lava flows and by interlayered clay and silt. Hydrologists estimate that the interflow zones, which range in thickness from about 3 ft to about 26 ft, account for about 10 percent of the basalt. MacNish and Barker (1976) have estimated that the hydraulic conductivity along the flow-contact zones may be a billion times higher than the hydraulic conductivity across the dense zones. The lateral extent of individual aquifers is highly variable.

The large differences in hydraulic conductivity between the aquifers and the intervening "confining zones" result in significant differences in hydraulic heads between different aquifers. These differences reflect the head losses that occur as water moves vertically through the



Explanation

- Chiefly sedimentary rocks
- Chiefly volcanic rocks
- Sedimentary and volcanic rocks
- Major aquifers thin or absent

Figure 2-8. Generalized Distribution and Types of Major Aquifers of the Columbia Lava Plateau Region (Modified from U.S. Geological Survey Professional Paper 813-S)

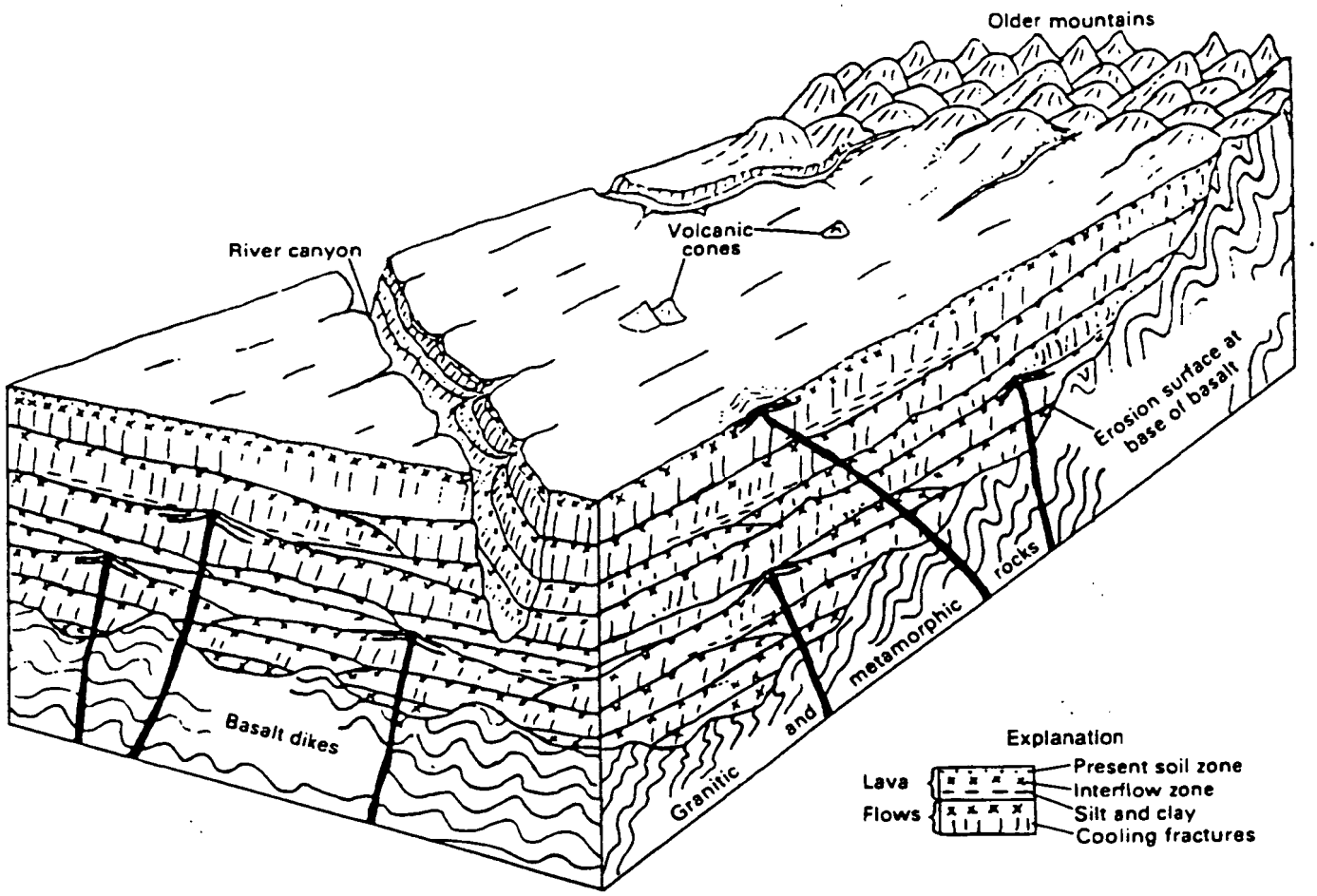


Figure 2-9. Topographic and Geologic Features of the Columbia Lava Plateau Region

system. As a result, heads decrease with increasing depth in recharge areas and increase with increasing depth near the streams that serve as major lines of ground-water discharge. As Figure 2-10 shows, the difference in heads between different aquifers can result in the movement of large volumes of water between aquifers through the openhole (uncased) sections of wells.

Much of the Columbia Lava Plateau region is in the "rain shadow" east of the Cascades and, as a result, receives only 8 to 47 in of precipitation annually. The areas that receive the least precipitation include the plateau area immediately east of the Cascades and the Snake River Plain. Recharge to the ground-water system depends on several factors, including the amount and seasonal distribution of precipitation and the permeability of the surficial materials. Most precipitation occurs in the winter and thus coincides with the cooler, nongrowing season when conditions are most favorable for recharge. The Columbia-North Pacific Technical Staff (1970) estimates that recharge may amount to 24 in in areas underlain by highly permeable young lavas that receive abundant precipitation. Considerable recharge also occurs by infiltration of water from streams that flow onto the plateau from the adjoining mountains. These sources of natural recharge are supplemented in agricultural areas by the infiltration of irrigation water.

Discharge from the ground-water system occurs as seepage to streams, as spring flow, and by evapotranspiration in areas where the water table is at or near the land surface. The famous Thousand Springs and other springs along the Snake River canyon in southern Idaho are, in fact, among the most spectacular displays of ground-water discharge in the world.

The large withdrawal of water in the Columbia Lava Plateau for irrigation, industrial, and other uses has resulted in declines in ground-water levels of as much as 100 to 200 ft in several areas. In most of these areas, the declines have been slowed or stopped through

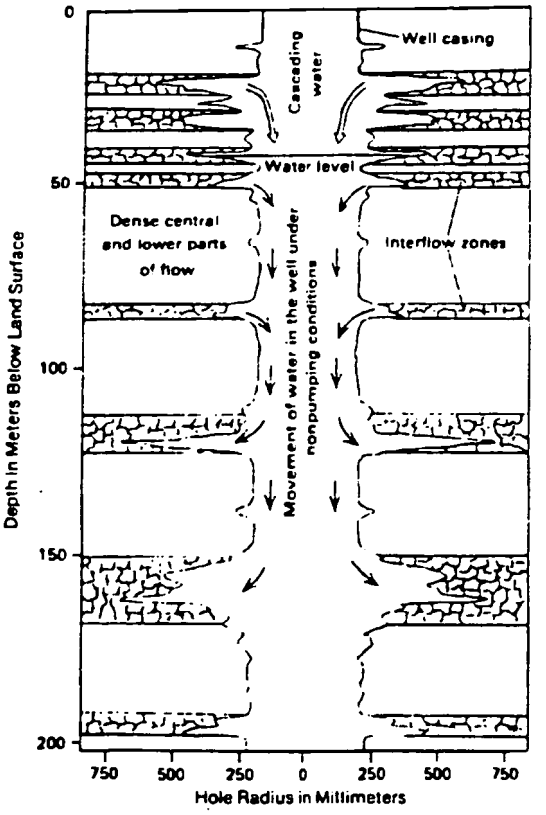


Figure 2-10. Well in a Recharge Area in the Columbia River Group (Modified from Luzier and Burt, 1974)

regulatory restrictions or other changes that have reduced withdrawals. Declines are still occurring, at rates as much as a few feet per year, in a few areas.

4. Colorado Plateau and Wyoming Basin
(Thin soils over consolidated sedimentary rocks)

The Colorado Plateau and Wyoming Basin region occupies an area of 160,000 mi^2 in Arizona, Colorado, New Mexico, Utah, and Wyoming. It is a region of canyons and cliffs of thin, patchy, rocky soils, and of sparse vegetation adapted to the arid and semiarid climate. The large-scale structure of the region is that of a broad plateau standing at an altitude of 8,200 to 11,500 ft and underlain by horizontal to gently dipping layers of consolidated sedimentary rocks. As Figure 2-11 shows, the plateau structure has been modified by an irregular alternation of basins and domes, in some of which major faults have caused significant offset of the rock layers. The region is bordered on the east, north, and west by mountain ranges that tend to obscure its plateau structure. It also contains rather widely scattered extinct volcanoes and lava fields.

The rocks that underlie the region consist principally of sandstone, shale, and limestone of Paleozoic to Cenozoic age. In parts of the region these rock units include significant amounts of gypsum (calcium sulfate). In the Paradox Basin in western Colorado the rock units include thick deposits of sodium- and potassium-bearing

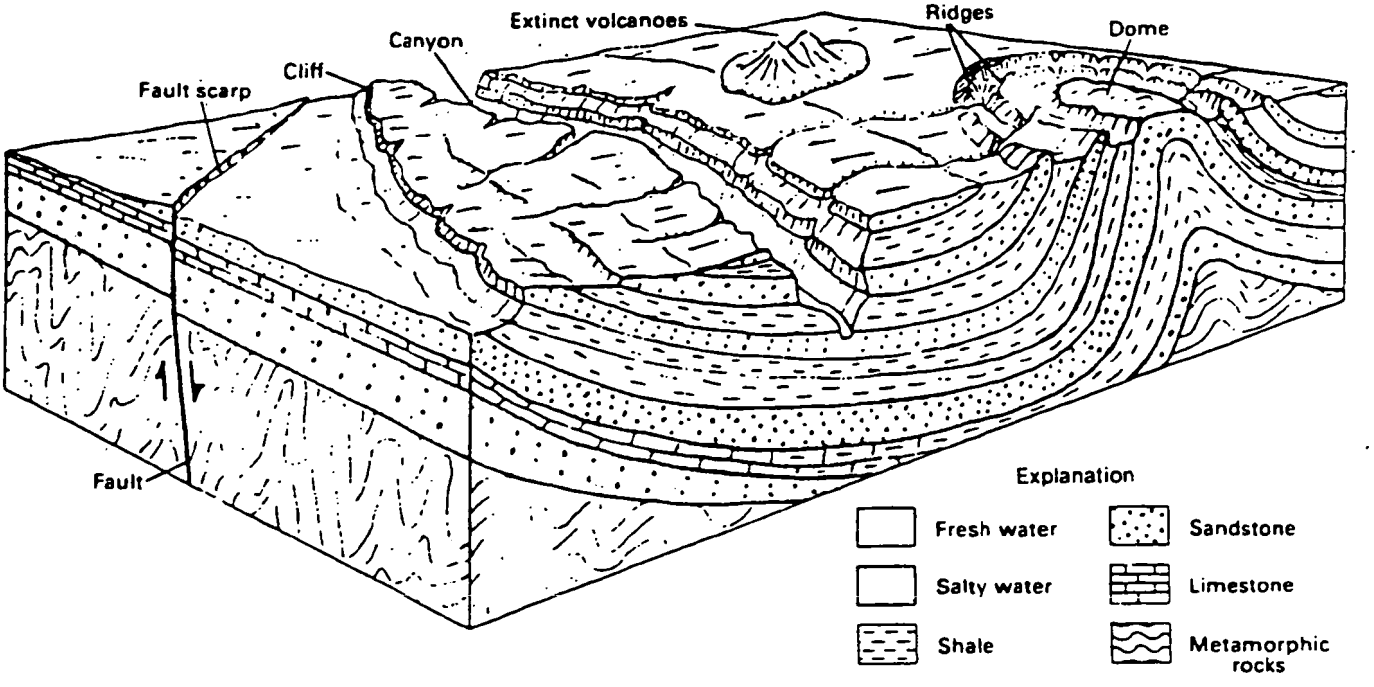


Figure 2-11. Topographic and Geologic Features of the Colorado Plateau and Wyoming Basin Region

minerals, principally halite (sodium chloride). The sandstones and shales are most prevalent and most extensive. The sandstones are the principal sources of ground water and contain water in fractures developed both along bedding planes and across the beds and in interconnected pores. The most productive sandstones are those that are only partially cemented and retain significant primary porosity.

Unconsolidated deposits are of relatively minor importance in this region. Thin deposits of alluvium capable of yielding small to moderate supplies of ground water occur along parts of the valleys of major streams, especially adjacent to the mountain ranges in the northern and eastern parts of the region. In most of the remainder of the region there are large expanses of exposed bedrock, and the soils, where present, are thin and rocky.

Recharge of the sandstone aquifers occurs where they are exposed above the cliffs and in the ridges. Average precipitations ranges from about 6 in in the lower areas to about 39 in in the higher mountains. The heaviest rainfall occurs in the summer in isolated, intense thunderstorms during which some recharge occurs where intermittent streams flow across sandstone outcrops. However, most recharge occurs in the winter during snowmelt periods. Water moves down the dip of the beds away from the recharge areas to discharge along the channels of major streams through seeps and springs and along the walls of canyons cut by the streams.

The quantity of water available for recharge is small, but so are the porosity and the transmissivity of most of the sandstone aquifers. The water in the sandstone aquifers is unconfined in the recharge areas and is confined downdip. Because most of the sandstones are consolidated, the storage coefficient in the confined parts of the aquifers is very small. Even small rates of withdrawal cause extensive cones of depression around pumping wells.

The Colorado Plateau and Wyoming Basin is a dry, sparsely populated region in which most water supplies are obtained from the perennial streams. Less than 5 percent of the water needs are supplied by ground water, and the development of even small ground-water supplies requires the application of considerable knowledge of the occurrence of both rock units and their structure, and of the chemical quality of the water. Also, because of the large surface relief and the dip of the aquifers, wells even for domestic or small livestock supplies must penetrate to depths of a few hundred feet in much of the area. Thus, the development of ground-water supplies is far more expensive than in most other parts of the country. These negative aspects notwithstanding, ground water in the region can support a substantial increase over the present withdrawals.

As in most other areas of the country underlain by consolidated sedimentary rock, mineralized (saline) water—that is, water containing more than 1,000 mg/L of dissolved solids—is widespread. Most of the shales and siltstones contain mineralized water throughout the region and below altitudes of about 6,500 ft. Freshwater—water containing less than 1,000 mg/L of dissolved solids—occurs only in the most permeable sandstones and limestones. Much of the mineralized water is due to the solution of gypsum and halite. Although the aquifers that contain mineralized water are commonly overlain by aquifers containing freshwater, this situation is reversed in a few places where aquifers containing mineralized water are underlain by more permeable aquifers containing freshwater.

5. High Plains
(Thick alluvial deposits over fractured sedimentary rock)

The High Plains region occupies an area of 174,000 mi^2 extending from South Dakota to Texas. The plains are a remnant of a great alluvial plain built in Miocene time by streams that flowed east from the Rocky Mountains. Erosion has removed a large part of the once extensive plain, including all of the part adjacent to the mountains, except in a small area in southeastern Wyoming.

The original depositional surface of the alluvial plain is still almost unmodified in large areas, especially in Texas and New Mexico, and forms a flat, imperceptibly eastward-sloping tableland that ranges in altitude from about 6,500 ft near the Rocky Mountains to about 1,600 ft along its eastern edge. The surface of the southern High Plains contains numerous shallow circular depressions, called playas, that intermittently contain water following heavy rains. As Figure 2-12 shows, other significant topographic features include sand dunes, which are especially prevalent in central and northern Nebraska, and wide, downcut valleys of streams that flow eastward across the area from the Rocky Mountains.

The High Plains region is underlain by one of the most productive and most extensively developed aquifers in the United States. The alluvial materials derived from the Rocky Mountains, which are referred to as the Ogallala Formation, are the dominant geologic unit of the High Plains aquifer. The Ogallala ranges in thickness from a few tens of feet to more than 650 ft and consists of poorly sorted and generally unconsolidated clay, silt, sand and gravel.

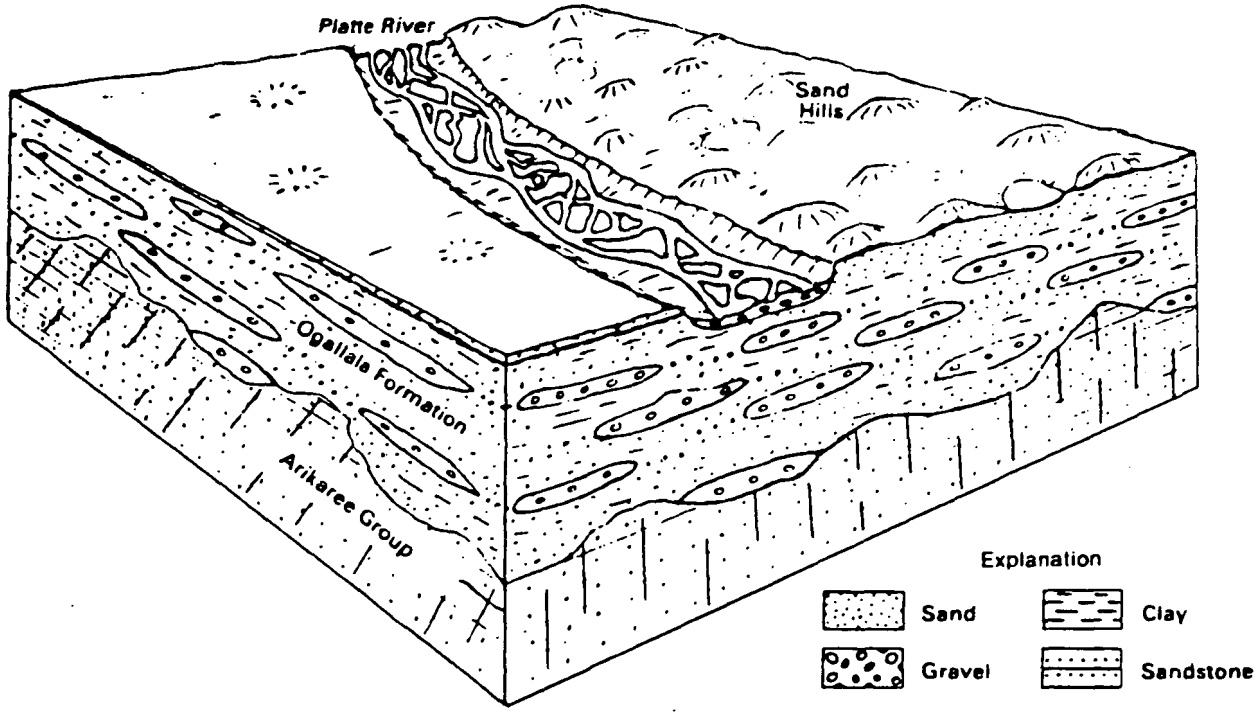


Figure 2-12. Topographic and Geologic Features of the High Plains Region

Younger alluvial materials of Quaternary age overlie the Ogallala Formation of late Tertiary age in most parts of the High Plains. Where these deposits are saturated, they form a part of the High Plains aquifer; in parts of south-central Nebraska and central Kansas, where the Ogallala is absent, they comprise the entire aquifer. The Quarternary deposits are composed largely of material derived from the Ogallala and consist of gravel, sand, silt, and clay. An extensive area of dune sand occurs in the Sand Hills area north of the Platte River in Nebraska.

Other, older geologic units that are hydrologically connected to the Ogallala include the Arikaree Group of Miocene age and a small part of the underlying Brule Formation. The Arikaree Group is predominantly a massive, very fine to fine-grained sandstone that locally contains beds of volcanic ash, silty sand, and sandy clay. The maximum thickness of the Arikaree is about 1000 ft, in western Nebraska. The Brule Formation of Oligocene age underlies the Arikaree. In most of the area in which it occurs, the Brule forms the base of the High Plains aquifer. However, in the southeastern corner of Wyoming and the adjacent parts of Colorado and Nebraska, the Brule contains fractured sandstones hydraulically interconnected to the overlying Arikaree Group; in this area the Brule is considered to be a part of the High Plains aquifer.

In the remainder of the region, the High Plains aquifer is underlain by several formations, ranging in age from Cretaceous to Permian and composed principally of shale, limestone, and sandstone. The oldest of these underlies parts of northeastern Texas, western Oklahoma, and central Kansas and contains layers of relatively soluble minerals including gypsum, anhydrite, and halite (common salt), which are dissolved by circulating ground water.

Prior to the erosion that removed most of the western part of the Ogallala, the High Plains aquifer was recharged by the streams that flowed onto the plain from the mountains to the west as well as by local precipitation. The only source of recharge now is local precipitation, which ranges from about 16 in along the western boundary of the region to about 24 in along the eastern boundary. Precipitation and ground-water recharge on the High Plains vary in an east-west direction, but recharge to the High Plains aquifer also varies in a north-south direction. The average annual rate of recharge has been determined to range from about 0.2 in in Texas and New Mexico to about 4 in in the Sand Hills in Nebraska This large difference is explained by differences in evaporation and transpiration and by differences in the permeability of surficial materials.

In some parts of the High Plains, especially in the southern part, the near-surface layers of the Ogallala have been cemented with lime (calcium carbonate) to form a material of relatively low permeability called caliche. Precipitation on areas underlain by caliche soaks slowly into the ground. Much of this precipitation collects in playas that are underlain by silt and clay, with the result that most of the water evaporates. It is only during years of excessive precipitation that significant recharge occurs and this, as noted above, averages only about0.2 in per year in the southern part of the High Plains. In the Sand Hills area about 20 percent of the precipitation (or about4 in annually) reaches the water table as recharge.

Figure 2-13 shows that the water table of the High Plains aquifer has a general slope toward the east. Gutentag and Weeks (1980) estimate that, on the basis of the average hydraulic gradient and aquifer characteristics, that water moves through the aquifer at a rate of about 1 ft per day.

Natural discharge from the aquifer occurs to streams, to springs and seeps along the eastern boundary of the plains, and by evaporation and transpiration in areas where the water table is within a few feet of the land surface. However, at present the largest discharge is probably through wells. The widespread occurrence of permeable layers of sand and gravel, which permit the construction of large-yield wells almost any place in the region, has led to the development of an extensive agricultural economy largely dependent on irrigation. Most of this water is derived from ground-water storage, resulting in a long-term continuing decline in ground-water levels in parts of the region of as much as 3 ft per year.

The depletion of ground-water storage in the High Plains is a matter of increasing concern in the region. However, from the standpoint of the region as a whole, the depletion does not yet represent a large part of the storage that is available for use. Weeks and Gutentag (1981) estimate, on the basis of a specific yield of 15 percent of the total volume of saturated material, that the available (usable) storage in 1980 was about 3.3 billion acre-ft. Luckey, Gutentag, and Weeks (1981) estimate that this is only about 5 percent less than the storage that was available at the start of withdrawals. However, in areas where intense irrigation has long been practiced, depletion of storage is severe.

6. Nonglaciated Central Region

(Thin regolith over fractured sedimentary rocks)

As Figure 2-14 shows, the Nonglaciated Central region is an area of about 671,000 mi^2 extending from the Appalachian Mountains on the east to the Rocky Mountains on the west. The part of the region in eastern Colorado and northeastern New Mexico is separated from the remainder of the region by the High Plains region. The Nonglaciated Central region also includes the Triassic Basins in Virginia and North Carolina and the "driftless" area in Wisconsin, Minnesota, Iowa, and Illinois where glacial deposits, if present, are thin and of no hydrologic importance.

The region is geologically complex. Most of it is underlain by consolidated sedimentary rocks that range in age from Paleozoic to Tertiary and consist largely of sandstone, shale, limestone, dolomite, and conglomerate. A small area in Texas and western Oklahoma is underlain by gypsum. Figure 2-15 shows that throughout most of the region the rock layers are horizontal or gently dipping. Principal exceptions are the Valley and Ridge section, the Wichita and Arbuckle Mountains in Oklahoma, and the Ouachita Mountains in Oklahoma and Arkansas, in all of which the rocks have been folded and extensively faulted. As Figure 2-16 shows, around the Black Hills and along the eastern side of the Rocky Mountains the rock layers have been bent up sharply toward the mountains and truncated by erosion. The Triassic Basins in Virginia and North Carolina are underlain by moderate to gently dipping beds of shale and sandstone that have been extensively faulted and invaded by narrow bodies of igneous rock.

The land surface in most of the region is underlain by regolith formed by chemical and mechanical breakdown of the bedrock. In the western part of the Great Plains the residual soils are overlain by or intermixed with wind-laid deposits. In areas underlain by relatively pure limestone, the regolith consists mostly of clay and is generally only a few feet thick. Where the limestones contain chert and in the areas underlain by shale and sandstone, the regolith is thicker, up to 100 ft or more in some areas. The chert and sand form moderately permeable soils, whereas the soils developed on shale are finer grained and less permeable.

As Figure 2-15 shows, the principal water-bearing openings in the bedrock are fractures, which generally occur in three sets. The first set, and the one that is probably of greatest importance from the standpoint of groundwater as well yields, consists of fractures developed along bedding planes. The two remaining sets are essentially vertical and thus cross the bedding planes at a steep angle. The primary difference between the sets of vertical fractures is in the orientation of the fractures in each set. The vertical fractures facilitate movement of water across the rock layers and thus serve as the principal hydraulic connection between the bedding-plane fractures.

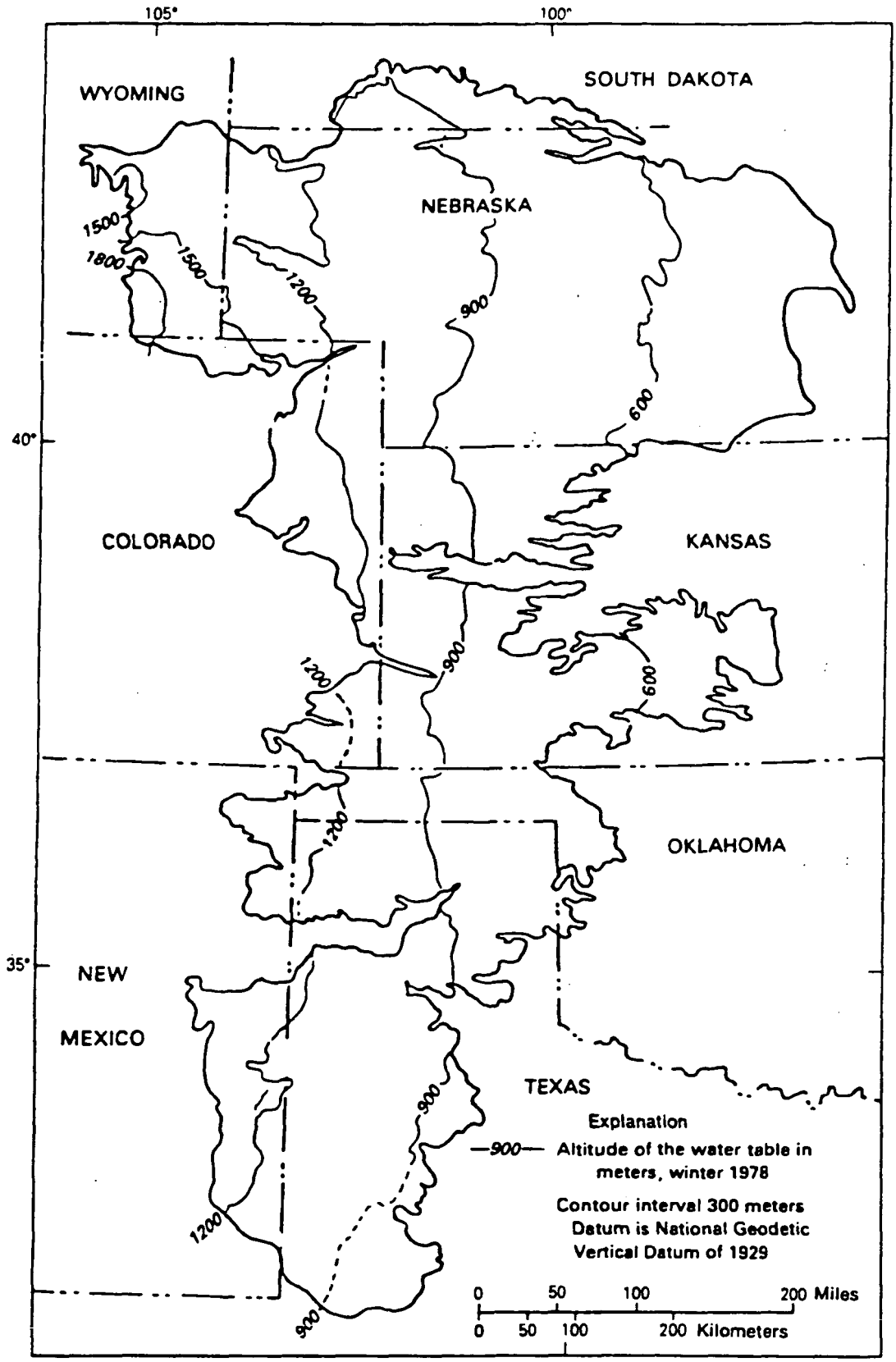


Figure 2-13. Altitude of the Water Table of the High Plains Aquifer

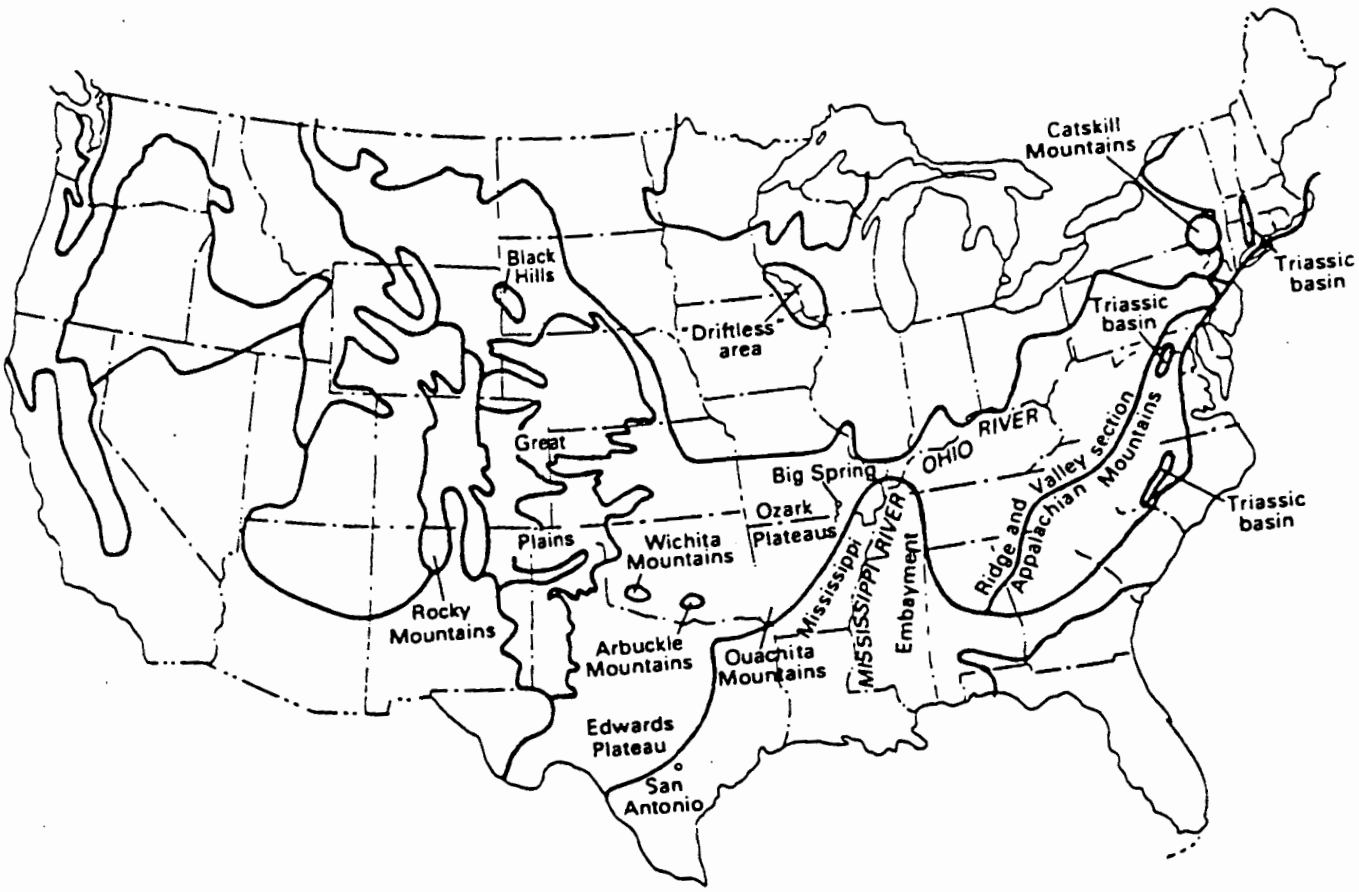


Figure 2-14. Location of Geographic Features Mentioned in the Discussions or Regions Covering the Central and Eastern Parts of the United States

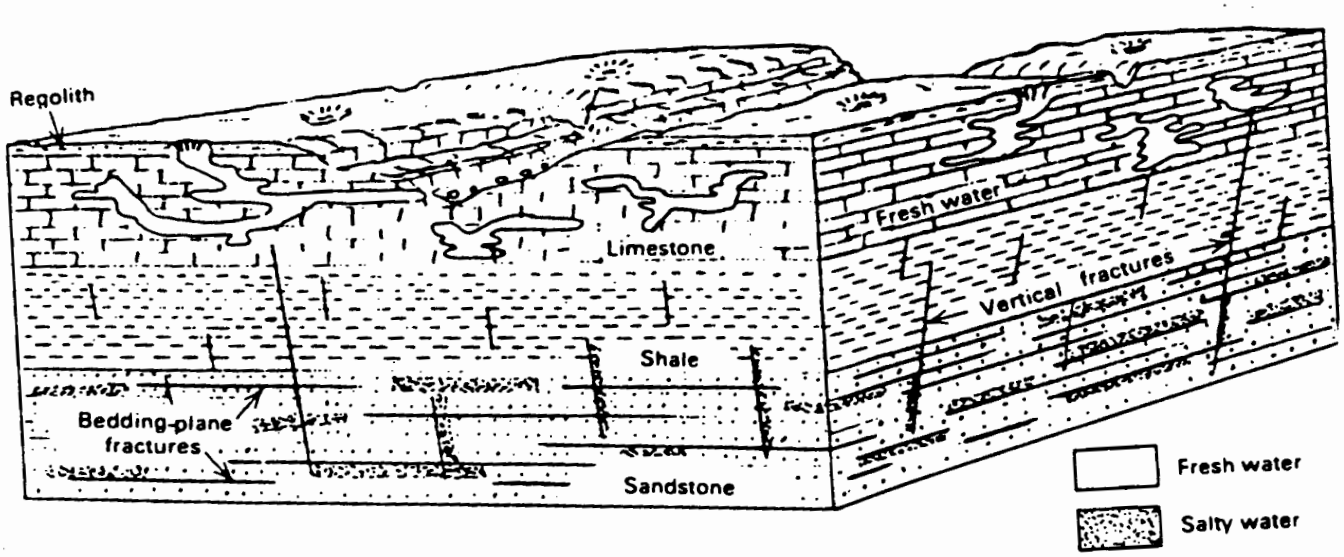


Figure 2-15. Topographic and Geologic Features of the Nonglaciated Central Region

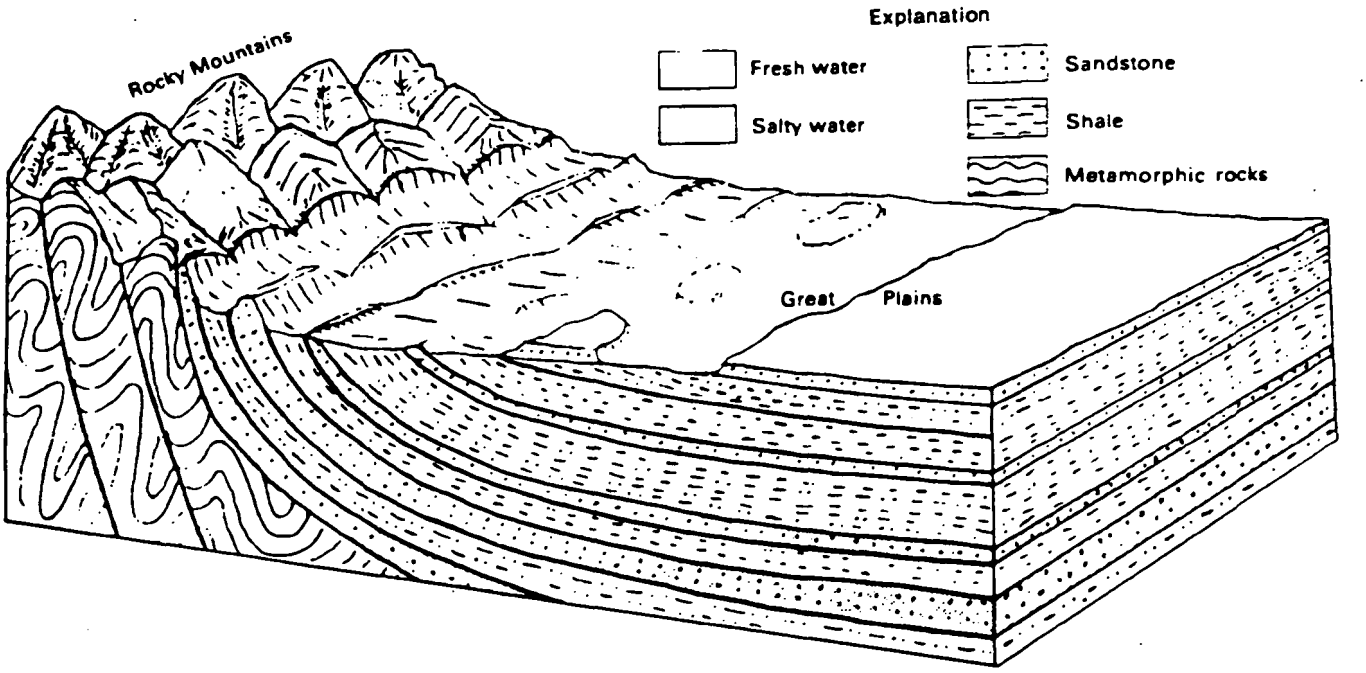


Figure 2-16. Topographic and Geologic Features Along the Western Boundary of the Nonglaciated Central Region

In the parts of the region in which the bedrock has been folded or bent, fractures range from horizontal to vertical. Fractures parallel to the land surface, where present, are probably less numerous and of more limited extent than in areas of flat-lying rocks.

The openings developed along most fractures are less than a0.04 in wide. The principal exception occurs in limestones and dolomites. Water moving through these rocks gradually enlarges the fractures to form, in time, extensive cavernous openings or cave systems. Many large springs emerge from these openings.

Recharge of the ground-water system in this region occurs primarily in the outcrop areas of the bedrock aquifers in the uplands between streams. Precipitation in the region ranges from about 16 in per year in the western part to more than 47 in in the eastern part. This wide difference in precipitation is reflected in recharge rates, which range from about 0.2 in per year in west Texas and New Mexico to as much as 20 in per year in Pennsylvania and eastern Tennessee.

Discharge from the ground-water system is by springs and seepage into streams and by evaporation and transpiration.

The yield of wells depends on: (I) the number and size of fractures that are penetrated and the extent to which they have been enlarged by solution, (2) the rate of recharge, and (3) the storage capacity of the bedrock and regolith. Yields of wells in most of the region are small, in the range of about 2.5 to about 250 gallons per minute, making the Nonglaciated Central region one of the least favorable ground-water regions in the country. Even in parts of the areas underlain by cavernous limestone, yields are moderately low because of both the absence of a thick regolith and the large water-transmitting capacity of the cavernous openings, which quickly discharge the water that reaches them during periods of recharge.

The exceptions to the small well yields are the cavernous limestones of the Edwards Plateau, the Ozark Plateaus, and the Ridge and Valley section. Figure 2-14 shows the location of these areas. The Edwards Plateau in Texas is bounded on the south by the Balcones Fault Zone, in which limestone and dolomite up to 500 ft in thickness has been extensively faulted, which facilitates the development of solution openings. This zone forms one of the most productive aquifers in the country. Wells of the City of San Antonio are located in this zone; individually, they have yields of more than 16,000 gallons per minute.

As Figures 2-15 and 2-16 show, another feature that makes much of this region unfavorable for ground-water development is the occurrence of salty water at relatively shallow depths. In most of the Nonglaciated Central region, except the Ozark Plateaus, the Ouachita and Arbuckle Mountains, and the Ridge and Valley section, the water in the bedrock contains more than

1,000 mg/L of dissolved solids at depths less than 500 ft.

7. Glaciated Central Region
(Glacial deposits over fractured sedimentary rocks)

Figure 2-14 shows the Glaciated Central region, which occupies an area of 500,000 mi^2 extending from the Triassic Basin in Connecticut and Massachusetts and the Catskill Mountains in New York on the east to the northern part of the Great Plains in Montana on the west. Figure 2-17 shows that the Glaciated Central region is underlain by relatively flat-lying consolidated sedimentary rocks that range in age from Paleozoic to Tertiary. The bedrock is overlain by glacial deposits that, in most of the area, consist chiefly of till, an unsorted mixture of rock particles deposited directly by the ice sheets. The till is interbedded with and overlain by sand and gravel deposited by meltwater streams, by silt and clay deposited in glacial lakes, and, in large parts of the North-Central States, by loess, a well-sorted silt believed to have been deposited primarily by the wind.

On the Catskill Mountains and other uplands in the eastern part of the region, the glacial deposits are typically only a few to several feet thick. In much of the central and western parts of the region, the glacial deposits exceed 330 ft in thickness. The principal exception is the "driftless" area in Wisconsin, Minnesota, Iowa, and Illinois where the bedrock is overlain by thin soils. This area, both geologically and hydrologically, resembles the Nonglaciated Central region and is, therefore, included as part of that region.

The glacial deposits are thickest in valleys in the bedrock surface. In most of the region westward from Ohio to the Dakotas, the thickness of the glacial deposits exceeds the relief on the preglacial surface, with the result that the locations of valleys and stream channels in the preglacial surface are no longer discernible from the land surface. Figure 2-17 shows that the glacial deposits in buried valleys include, in addition to till and lacustrine silts and clays, substantial thicknesses of highly permeable sand and gravel.

Ground water occurs both in the glacial deposits and in the bedrock. Water occurs in the glacial deposits in pores between the rock particles and in the bedrock primarily along fractures.

Large parts of the region are underlain by limestones and dolomites in which fractures have been enlarged by solution. On the whole, caves and other large solution openings are much less numerous and hydrologically much less important in the Glaciated Central region.

The glacial deposits are recharged by precipitation on the interstream areas and serve both as a source of water to shallow wells and as a reservoir for recharge to the underlying bedrock. Precipitation ranges from about 16 in per year in the western part of the region to about

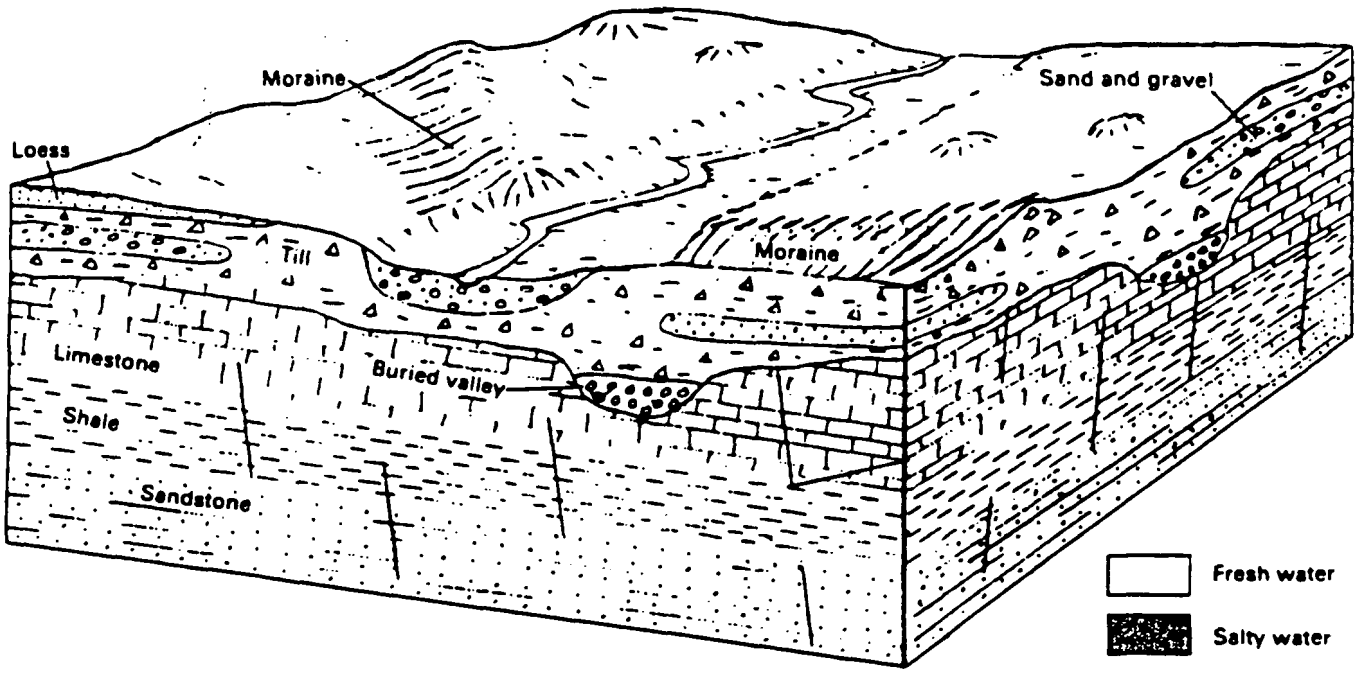


Figure 2-17. Topographic and Geologic Features of the Glaciated Central Region

39 in n the east. On sloping hillsides underlain by clay-rich till, the annual rate of recharge, even in the humid eastern part of the region, probably does not exceed 2 in. In contrast, relatively flat areas underlain by sand and gravel may receive as much as 12 in of recharge annually in the eastern part of the region.

Ground water in small to moderate amounts can be obtained any place in the region, both from the glacial deposits and from the bedrock. Large to very large amounts of ground water are obtained from the sand and gravel deposits and from some of the limestones, dolomites, and sandstones. The shales are the least productive bedrock formations in the region.

Because of the widespread occurrence of limestone and dolomite, water from both the glacial deposits and the bedrock contains as much as several hundred milligrams per liter of dissolved minerals and is moderately hard. Concentrations of iron in excess of 0.3 mg/L are a problem in water from some of the sandstone aquifers in Wisconsin and Illinois and locally in glacial deposits throughout the region. Sulfate in excess of 250 mg/L is a problem in water both from the glacial deposits and from the bedrock in parts of New York, Ohio, Indiana, and Michigan.

As is the case in the Nonglaciated Central region mineralized water occurs at relatively shallow depth in bedrock in large parts of this region. The thickness of the freshwater zone in the bedrock depends on the vertical hydraulic conductivity of both the bedrock and the glacial deposits and on the effectiveness of the hydraulic connection between them. Both the freshwater and the underlying saline water move toward the valleys of perennial streams to discharge. As a result, the depth to saline water is less under valleys than under uplands. At depths of 1,600 to 3,300 ft in much of the region, the mineral content of the water approaches that of seawater (about 35,000 mg/L). At greater depths, the mineral content may reach concentrations several times that of seawater.

8. Piedmont Blue Ridge Region
(Thick regolith over fractured crystalline and metamorphosed sedimentary rocks)

The Piedmont and Blue Ridge region is an area of about 95,000 mi^2 extending from Alabama on the south to Pennsylvania on the north. The Piedmont part of the region consists of low, rounded hills and long, rolling, northeast-southwest trending ridges. The Blue Ridge is mountainous and includes the highest peaks east of the Mississippi.

The Piedmont and Blue Ridge region is underlain by bedrock of Precambrian and Paleozoic age consisting of igneous, and metamorphosed igneous, and sedimentary rocks. The land surface in the Piedmont and Blue Ridge is underlain by clay-rich, unconsolidated material derived from in situ weathering of the underlying bedrock. This material, which averages about 33 to 65 ft in thickness and may be as much as 330 ft thick on some ridges, is referred to as saprolite. In many valleys, especially those of larger streams, flood plains are underlain by thin, moderately well-sorted alluvium deposited by the streams. While the distinction between saprolite and alluvium is not important, the term regolith is used to refer to the layer of unconsolidated deposits.

As Figure 2-18 shows the regolith contains water in pore spaces between rock particles. The bedrock, on the other hand, does not have any significant intergranular porosity. It contains water, instead, in sheetlike openings formed along fractures. The hydraulic conductivities of the regolith and the bedrock are similar and range from about 0.003 to 3 ft per day. The major difference in their water-bearing characteristics is their porosities, the porosity of regolith being about 20 to 30 percent and the porosity of the bedrock about 0.01 to 2 percent. Small supplies of water adequate for domestic needs can be obtained from the regolith through large-diameters bored or dug wells. However, most wells, especially those where moderate supplies of water are needed, are relatively small in diameter and are cased through the regolith and finished with open holes in the bedrock. Although, the hydraulic conductivity of the bedrock is similar to that of the regolith, bedrock wells generally have much larger yields than regolith wells because, being deeper, they have a much larger available drawdown.

All ground-water systems function both as reservoirs that store water and as pipelines that transmit water from recharge areas to discharge areas. The yield of bedrock wells in the Piedmont and Blue Ridge region depends on the number and size of fractures penetrated by the open hole and on the replenishment of the fractures by seepage into them from the overlying regolith. Thus, the ground-water system in this region can be viewed, from the standpoint of ground-water development, as a terrain in which the reservoir and pipeline functions are effectively separated. Because of its larger porosity, the regolith functions as a reservoir that slowly feeds water downward into the fractures in the bedrock. The fractures serve as an intricate interconnected network of pipelines that transmit water either to springs or streams or to wells.

Recharge of the ground-water system occurs on the areas above the flood plains of streams, and natural discharge occurs as seepage springs that are common

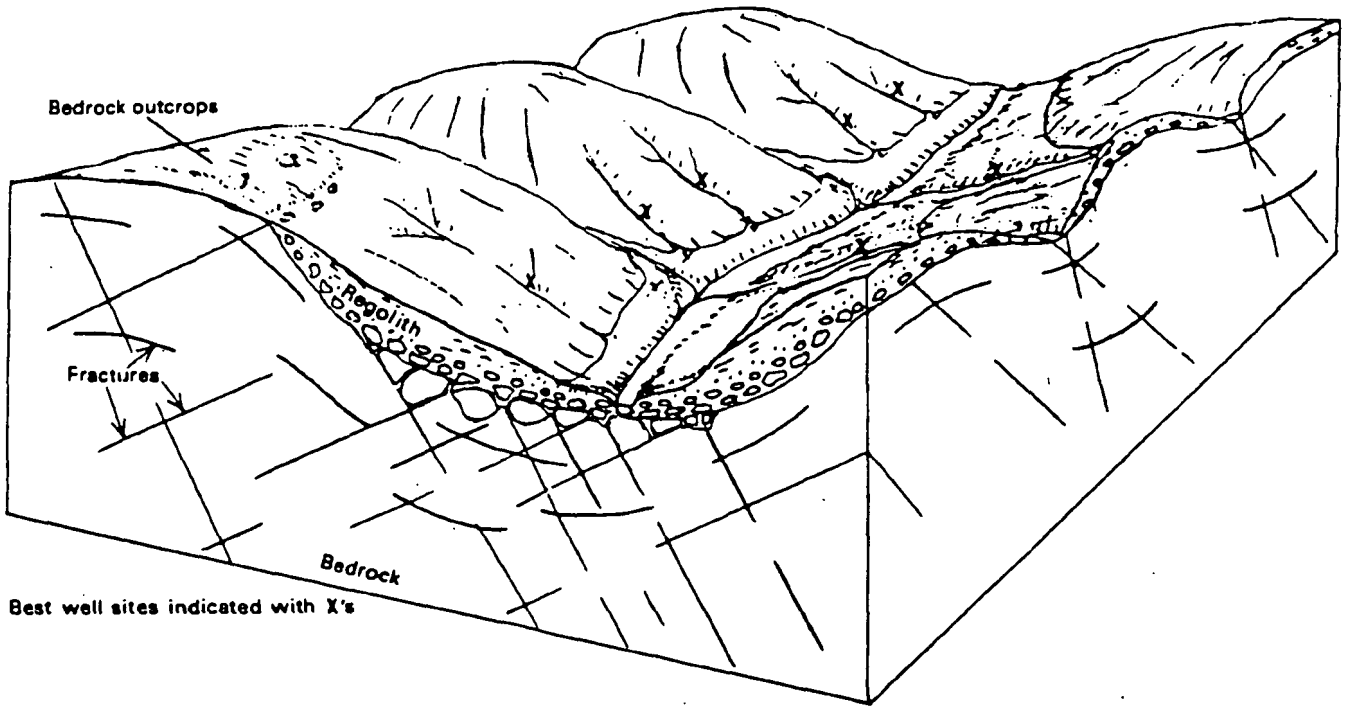


Figure 2-18. Topographic and Geologic Features of the Piedmont and Blue Ridge Region

near the bases of slopes and as seepage into streams. With respect to recharge conditions, it is important to note that forested areas, which include most of the Blue Ridge and much of the Piedmont, have thick and very permeable soils overlain by a thick layer of forest litter. In these areas, even on steep slopes, most of the precipitation seeps into the soil zone, and most of this moves laterally through the soil and a thin, temporary, saturated zone to surface depressions or streams to discharge. The remainder seeps into the regolith below the soil zone, and much of this ultimately seeps into the underlying bedrock.

The Piedmont and Blue Ridge region has long been known as an area generally unfavorable for ground-water development. This reputation seems to have resulted both from the small reported yields of the numerous domestic wells in use in the region that were, generally, sited as a matter of convenience and from a failure to apply existing technology to the careful selection of well sites where moderate yields are needed. As water needs in the region increase and as reservoir sites on streams become increasingly more difficult to obtain, it will be necessary to make intensive use of ground water.

9. Northeast and Superior Uplands
(Glacial deposits over fractured crystalline rocks)

The Northeast and Superior Uplands region is made up of two separate areas totaling about 160,000 mi^2. The Northeast Upland encompasses the Adirondack Mountains, the Lake Champlain valley, and nearly all of New England. The Superior Upland encompasses most of the northern parts of Minnesota and Wisconsin adjacent to the western end of Lake Superior.

Bedrock in the region ranges in age from Precambrian to Paleozoic, and as Figure 2-19 shows, consists mostly of intrusive igneous rocks and metamorphosed sedimentary rocks. Most have been intensively folded and cut by numerous faults.

As Figures 2-19 and 2-20 show, the bedrock is overlain by unconsolidated glacial deposits including till and gravel, sand, silt, and clay. The thickness of the glacial deposits ranges from a few feet on the higher mountains, which also have large expanses of barren rock, to more than300 ft in some valleys. The most extensive glacial deposit is till. In most of the valleys and other low areas, the till is covered by glacial outwash consisting of

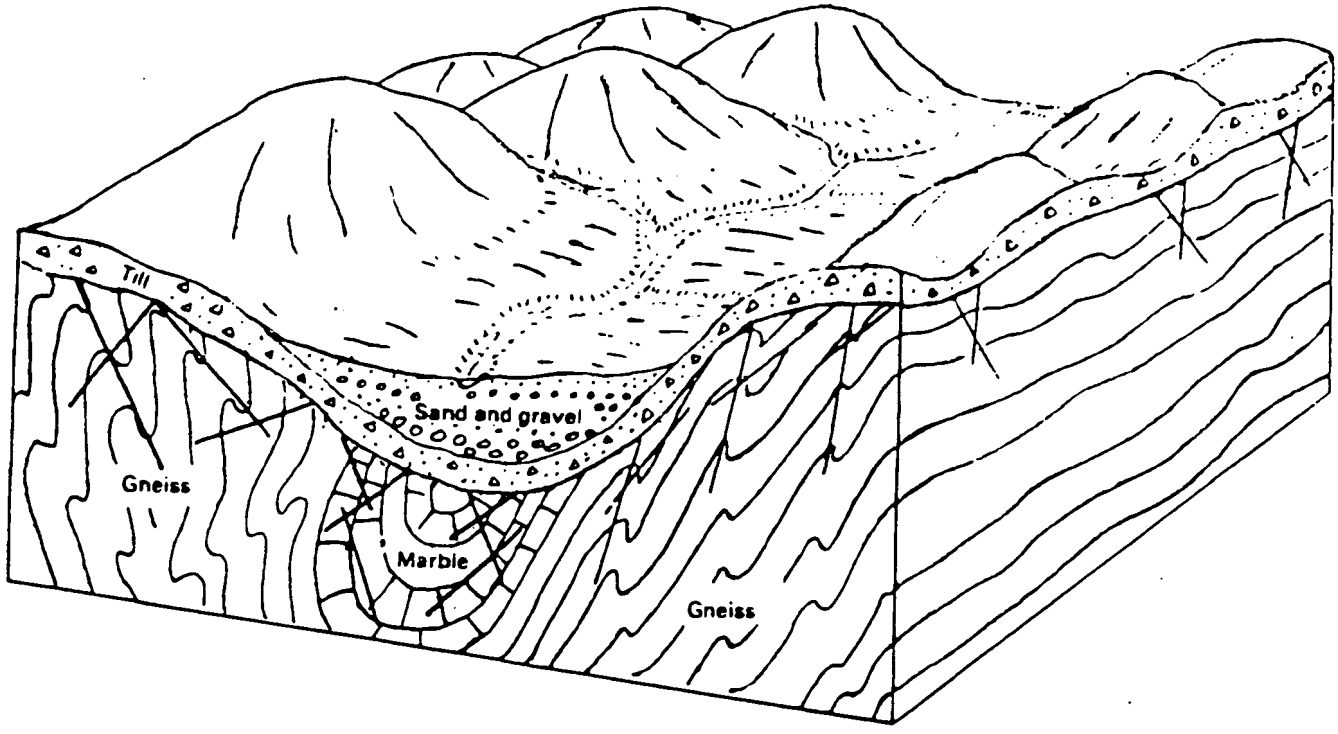


Figure 2-19. Topographic and Geologic Features of the Northeast and Superior Uplands Region.

interlayered sand and gravel, ranging in thickness from a few feet to more than 65 ft.

Ground-water supplies are obtained in the region from both the glacial deposits and the underlying bedrock. The largest yields come from the sand and gravel deposits, which in parts of the valleys of large streams are as much as 200 ft thick. Water occurs in the bedrock in fractures similar in origin, occurrence, and hydraulic characteristics to those in the Piedmont and Blue Ridge region.

Recharge from precipitation generally begins in the fall after plant growth stops. It continues intermittently over the winter during thaws and culminates during the period between the spring thaw and the start of the growing season. Precipitation on the Northeast Upland, about 47 in per year, is twice that on the Superior Upland, with the result that recharge is largest in the Northeast. The glacial deposits in the region serve as a storage reservoir for the fractures in the underlying bedrock.

Water supplies in the Northeast and Superior Uplands region are obtained from open-hole drilled wells in bedrock, from drilled and screened or openend wells in sand and gravel, and from large-diameter bored or dug wells in till. The development of water supplies from bedrock, especially in the Superior Upland, is more uncertain than from the fractured rocks in the Piedmont and Blue Ridge region because the ice sheets that advanced across the region removed the upper, more fractured part of the rock and also tended to obscure many of the fracture-caused depressions in the rock surface with the layer of glacial till.

Most of the rocks that underlie the Northeast and Superior Uplands are relatively insoluble, and consequently, the ground water in both the glacial deposits and the bedrock generally contains less than 500 mg/L of dissolved solids. Two of the most significant water-quality problems confronting the region, especially the Northeast Upland section, are acid precipitation and pollution caused by salts used to de-ice highways. Much of the precipitation falling on the Northeast in 1982 had a pH in the range of 4 to 6 units. Because of the low buffering capacity of the soils derived from rocks underlying the area, there is relatively little opportunity for the pH to be increased. One of the results of this is the gradual elimination of living organisms from many lakes and streams. The effect on ground-water quality, which will develop much more slowly, has not yet been

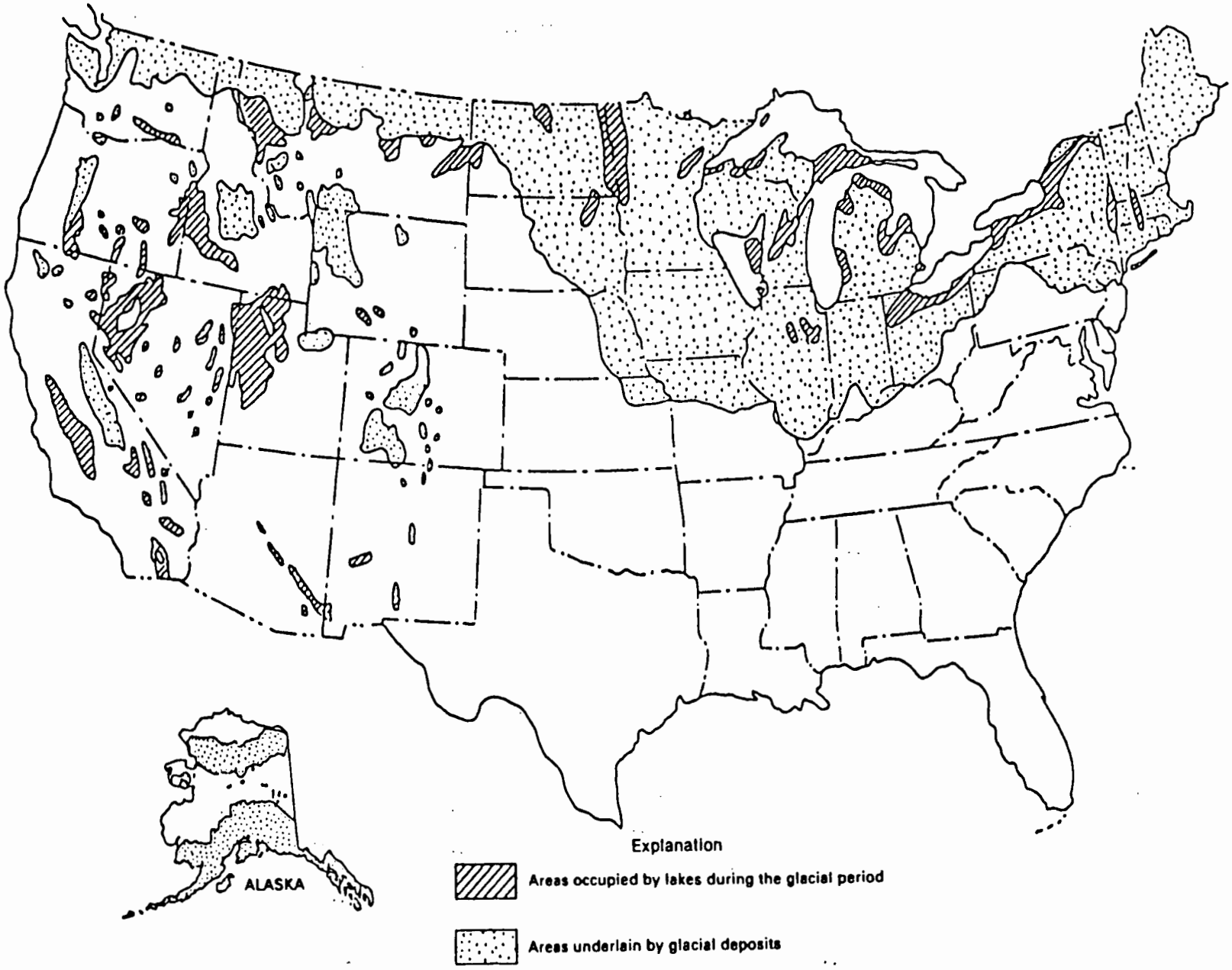


Figure 2-20. Glacial Features of the United States (Adapted from U.S. Geological Survey, 1970, p. 76)

determined. The second problem—that of de-icing salts—affects ground-water quality adjacent to streets and roads maintained for winter travel.

10. Atlantic and Gulf Coastal Plain
(Complexly interbedded sand, silt and clay)

The Atlantic and Gulf Coastal Plain region is an area of about 326,000 mi^2 extending from Cape Cod, Massachusetts, to the Rio Grande in Texas. This region does not include Florida and parts of the adjacent states.

The topography of the region ranges from extensive, flat, coastal swamps and marshes, 3 to 6 ft above sea level, to rolling uplands, 300 to 800 ft above sea level, along the inner margin of the region.

The region is underlain by unconsolidated sediments that consist principally of sand, silt, and clay. These sediments, which range in age from Jurassic to the present, range in thickness from less than a foot near the inner edge of the region to more than 39,000 ft in southern Louisiana. The sediments are complexly interbedded to the extent that most of the named geologic units into which they have been divided contain layers of the different types of sediment that underlie the region. These named geologic units dip toward the coast or toward the axis of the Mississippi embayment, with the result that those that crop out at the surface form a series of bands roughly parallel to the coast or to the axis of the embayment, as shown in Figure 2-21.

Although sand, silt, and clay are the principal types of material underlying the Atlantic and Gulf Coastal Plain, there are also small amounts of gravel interbedded with the sand, a few beds composed of mollusk shells, and small amounts of limestone present in the region. The most important limestone is the semi-consolidated Castle Hayne Limestone of Eocene age, which underlies an area of about 10,000 mi^2 in eastern North Carolina, is more than 650 ft thick in much of the area, and is the most productive aquifer in North Carolina. A soft, clayey limestone (the chalk of the Selma Group) of Late Cretaceous age underlies parts of eastern Mississippi and western Alabama, but instead of being an aquifer, it is an important confining bed.

From the standpoint of well yields and ground-water use, the Atlantic and Gulf Coastal Plain is one of the most important regions in the country. Recharge to the ground-water system occurs in the interstream areas, both where sand layers crop out and by percolation downward across the interbedded clay and silt layers. Discharge from the system occurs by seepage to streams, estuaries, and the ocean.

Wells that yield moderate to large quantities of water can be constructed almost anywhere in the region. Because most of the aquifers consist of unconsolidated sand, wells require screens; where the sand is fine-grained and well sorted, the common practice is to

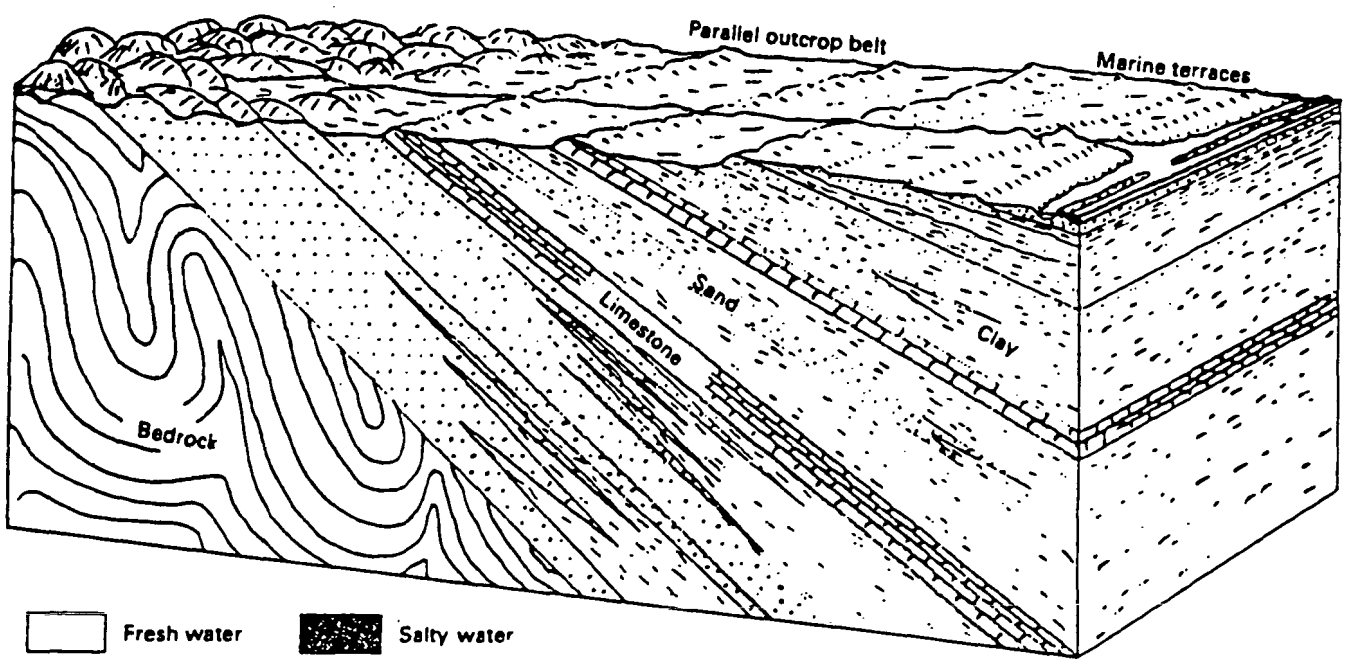


Figure 2-21. Topographic and Geologic Features of the Gulf Coastal Plain

surround the screens with a coarse sand or gravel envelope.

Withdrawals near the outcrop areas of aquifers are rather quickly balanced by increases in recharge and (or) reductions in natural discharge. Withdrawals at significant distances downdip do not appreciably affect conditions in the outcrop area and thus must be partly or largely supplied from water in storage in the aquifers and confining beds.

If withdrawals are continued for long periods in areas underlain by thick sequences of unconsolidated deposits, the lowered ground-water levels in the aquifer may result in drainage of water from layers of silt and clay. The depletion of storage in fine-grained beds results in subsidence of the land surface. Subsidence in parts of the Houston area totaled about 30 ft as of 1978. Subsidence near pumping centers in the Atlantic Coastal Plain has not yet been confirmed but is believed to be occurring at a slower rate than along the Texas Gulf Coast.

Depletion of storage in the aquifers underlying large areas of the Atlantic and Gulf Coastal Plain is reflected in long-term declines in ground-water levels. These declines suggest that withdrawals in these areas are exceeding the long-term yield of the aquifers.

Another problem that affects ground-water development in the region concerns the presence of saline water in the deeper parts of most aquifers. In some of the deeper aquifers, the interface between freshwater and saltwater is inshore, but in parts of the region, including parts of Long Island, New Jersey, and Mississippi, the interface in the most intensively developed aquifers is a significant distance offshore. Pumping near the interfaces has resulted in local problems of saltwater encroachment.

Another significant feature of the ground-water system in this region is the presence of "geopressured" zones at depths of 5,900 to 20,000 ft in Texas and Louisiana, which contain water at a temperature of 80°C to more than 273°C. Water in these zones contains significant concentrations of natural gas, and the water in some zones is under pressures sufficient to support a column of water more than 13,000 ft above land surface. Because the elevated temperature, natural gas, and high pressure are all potential energy sources, these zones are under intensive investigation.

11. Southeast Coastal Plain
(Thick layers of sand and clay over semiconsolidated carbonate rocks)

Figure 2-22 shows the Southeast Coastal Plain, an area of about 82,000 mi^2 in Alabama, Florida, Georgia, and South Carolina. It is a relatively flat, low-lying area. Much of the area, including the Everglades in southern Florida, is a nearly flat plain less than 30 ft above sea level.

The land surface of the Southeast Coastal Plain is underlain by unconsolidated deposits of Pleistocene age consisting of sand, gravel, clay, and shell beds and, in southeastern Florida, by semi-consolidated limestone. In most of the region, the surficial deposits rest on formations, primarily of middle to late Miocene age, composed of interbedded clay, sand, and limestone. The formations of middle to late Miocene age or surficial deposits overlie semi-consolidated limestones and dolomites that are as much as 5,000 ft thick.

The Tertiary limestone that underlies the Southeast Coastal Plain constitutes one of the most productive aquifers in the United States and is the feature that justifies treatment of the region separately from the remainder of the Atlantic and Gulf Coastal Plain. The aquifer, which is known as the Floridan aquifer, underlies all of Florida and southeast Georgia and small areas in Alabama and South Carolina. The Floridan aquifer consists of layers several feet thick composed largely of loose aggregations of shells and fragments of marine organisms interbedded with much thinner layers of cement and cherty limestone. The Floridan, one of the most productive aquifers in the world, is the principal source of ground-water supplies in the Southeast Coastal Plain region.

In southern Florida, south of Lake Okeechobee, and in a belt about 18 mi wide northward along the east coast of Florida to the vicinity of St. Augustine, the water in the Floridan aquifer contains more than 100 mg/L of chloride. In this area, most water supplies are obtained from surficial aquifers. The most notable of these aquifers underlies the southeastern part of Florida and, in the Miami area, consists of 100 to 330 ft of cavernous limestone and sand and is referred to as the Biscayne aquifer. The Biscayne is an unconfined aquifer, which is recharged by local precipitation and by infiltration of water from canals that drain water from impoundments developed in the Everglades. It is the principal source of water for municipal, industrial, and irrigation uses and can yield as much as 1,300 gal per min to small-diameter wells less than 80 ft deep finished with open holes only 3 to 6 ft long.

The surficial aquifers in the remainder of the region are composed primarily of sand, except in the coastal zones of Florida where the sand is interbedded with shells and thin limestones. These surficial aquifers serve as sources of small ground-water supplies

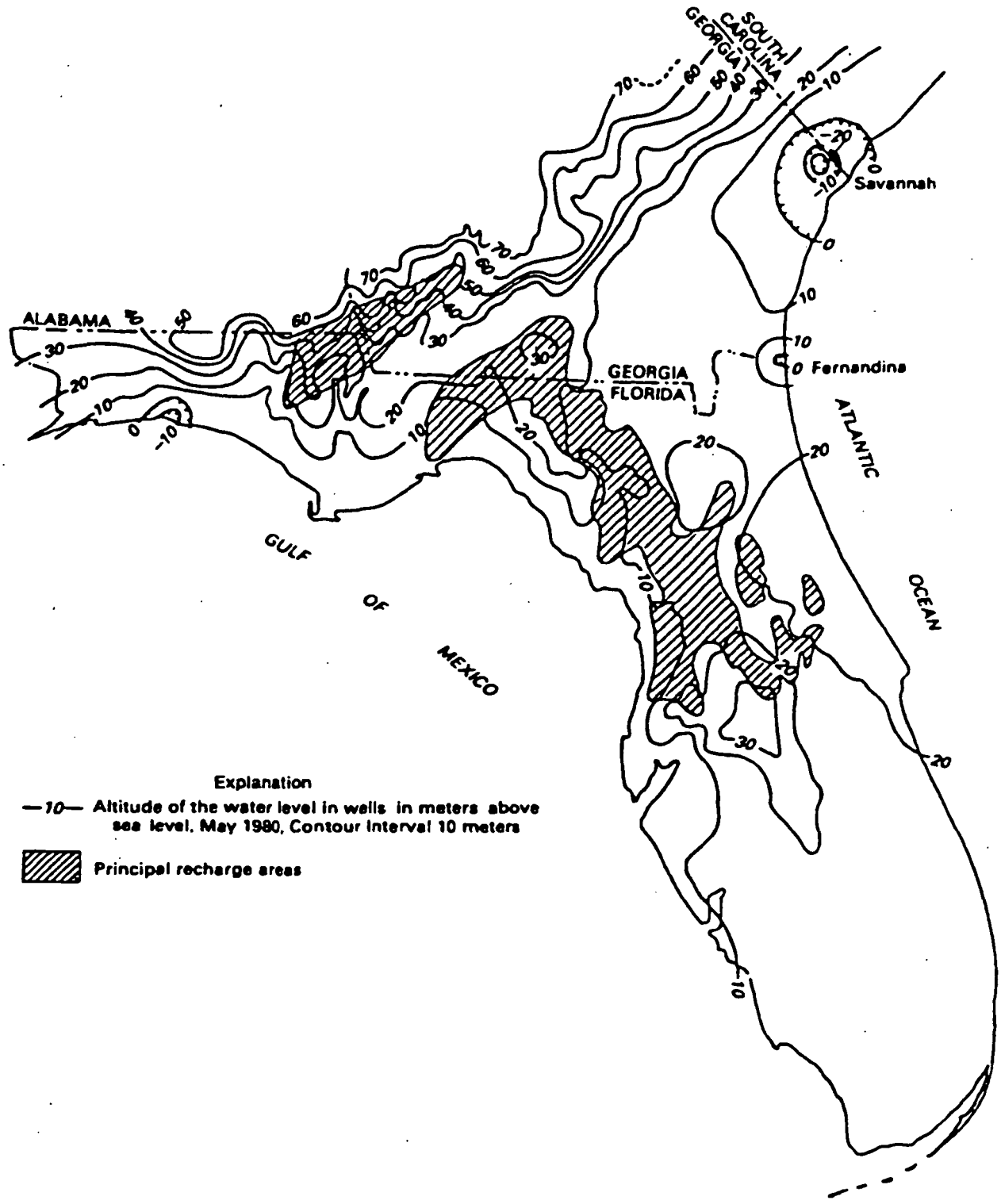


Figure 2-22. Potentiometric Surface for the Floridan Aquifer (Adapted from Johnston, Healy, and Hayes, 1981)

throughout the region and are the primary sources of ground water where the water in the Floridan aquifer contains more than about 250 mg/L of chloride.

The Floridan aquifer is the principal source of ground water in the region. Ground water in the upper part of the aquifer is unconfined in the principal recharge areas in Georgia and in west-central Florida, which are shown in Figure 2-22. In the remainder of the region, water in the aquifer is confined by clay in the Hawthorn Formation and in other beds that overlie the aquifer.

Recharge occurs where the potentiometric surface of the Floridan aquifer is lower than the water table in the overlying surficial aquifer. As Figure 2-22 shows, the principal recharge areas include a broad area along the west side of Florida extending from the central part of the peninsula to south-central Georgia and an area extending from west-central Florida through southeast Alabama into southwest Georgia. In these areas, recharge rates are estimated to exceed 5 in. per yr. Recharge occurs by infiltration of precipitation directly into the limestone, where it is exposed at the land surface, and by seepage through the permeable soils that partly mantle the limestone in the outcrop areas. Considerable recharge also occurs in the higher parts of the recharge areas through permeable openings in the confining beds, where these beds have been breached by the collapse of caverns in the limestone during the process of sinkhole formation. Figure 2-23 illustrates this sinkhole formation. Thus, the land surface in most of Florida north of Lake Okeechobee is marked by thousands of closed depressions ranging in diameter from a few feet to several miles. The larger depressions are occupied by lakes generally referred to as sinkhole lakes.

Discharge from the Floridan aquifer occurs through springs and by seepage to streams. Considerable discharge also occurs by diffuse seepage across the overlying confining beds in areas where the potentiometric surface of the aquifer stands at a higher altitude than the water table. In most of these areas wells open to the aquifer will flow at the land surface.

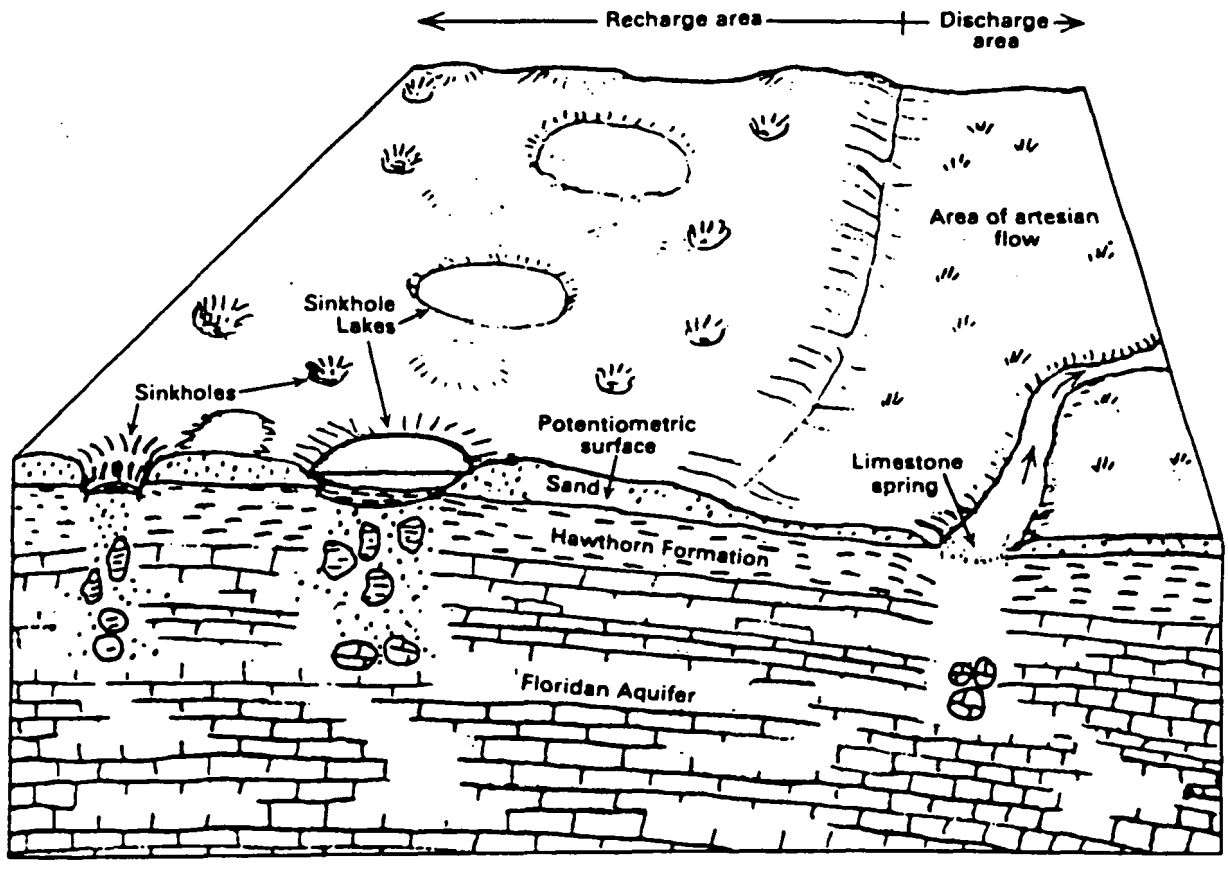


Figure 2-23. Topographic and Geologic Features of the Southeast Coastal Plain Region

The most spectacular discharge from the Floridan aquifer is through sinkholes exposed along streams and offshore.

Water supplies are obtained from the Floridan aquifer by installing casing through the overlying formations and drilling an open hole in the limestones and dolomites comprising the aquifer. Total withdrawals from the aquifer are estimated to have been about 3.5 billion gallons per day in 1978. Large withdrawals also occur from the other aquifers in the region.

12. Alluvial Valleys
(Thick sand and gravel deposits beneath floodplains and terraces of streams)

In the preceding discussions of ground-water regions, streams and other bodies of surface water were mentioned as places of ground-water discharge. In most areas ground-water systems and surface streams form a water system so intimately interconnected that a change in one causes a change in other. For example, withdrawals from ground-water systems reduce discharge to streams and thereby reduce streamflow. The movement of water from streams into ground-water systems in response to withdrawals is not a significant feature in most areas because ground-water withdrawals are dispersed over the uplands between streams rather than concentrated near them. An exception to this occurs where stream channels and floodplains are underlain by highly permeable deposits of sand and gravel. The large yields of these deposits, as well as the variability and availability of streamflow, encourage the development of these sand and gravel deposits as sources of ground water, and thus, encourage the concentration of withdrawals near streams. From the standpoint of ground-water hydrology, three criteria are used to differentiate alluvial valleys from other valleys. These criteria are as follows:

1. The alluvial valleys contain sand and gravel deposits thick enough to supply water to wells at moderate to large rates. [Commonly, the water-transmitting capacity of the sand and gravel is at least 10 times larger than that of the adjacent (enclosing) rocks.]
2. The sand and gravel deposits are in hydraulic contact with a perennial stream that serves as a source of recharge and whose flow normally far exceeds that demand from any typical well field.
3. The sand and gravel deposit occurs in a clearly defined band ("channel") that normally does not extend beyond the floodplain and adjacent terraces. In other words, the width of the deposit is small or very small compared with its length.

According to these criteria, the valleys of streams that were not affected by glacial meltwater are not considered alluvial valleys. The floodplains in these valleys are commonly underlain only by thin deposits of fine-grained alluvium. These criteria also eliminate the "buried" valleys of the glaciated area. Although the water-transmitting capacity of the sand and gravel in buried valleys may be large, the yield to wells in most of them is small because of the limited opportunity for recharge through the surrounding, less-permeable materials.

The alluvial valleys are commonly underlain, in addition to sand and gravel, by deposits of silt and clay. In many of the glaciated valleys in New York and New England the land surface is underlain by a layer of sand and gravel that ranges in thickness from 3 to 6 ft to more than 30 ft. The bottom of this deposit ranges, from one part of a valley to another, from a position above the water table to several feet below the bottom of streams. This surficial deposit of sand and gravel is commonly underlain by interbedded silt and clay which is, in turn, underlain by a discontinuous "basal" layer of sand and gravel.

The sequence of deposits in the alluvial valleys depends, of course, on the history of deposition in the valleys. Figure 2-24 shows that the sand and gravel in the valleys of major streams, such as those of the Mississippi, Missouri, and Ohio, are commonly overlain by deposits of clay and other fine-grained alluvium deposited during floods since the end of the glacial period.

Under natural conditions the alluvial deposits are recharged by precipitation on the valleys, by ground water moving from the adjacent and underlying aquifers, by overbank flooding of the streams, and, in some glacial valleys, by infiltration from tributary streams. Water in the alluvial deposits discharges to the streams in the valleys.

The layers of sand and gravel in the alluvial valleys are among the most productive aquifers in the country. They have been extensively developed as sources of water for municipalities, industries, and irrigation. Some of the gravel layers have hydraulic conductivities nearly as large as those of cavernous limestone. The large yields of the sand and gravel depend not only on their large water-transmitting capacity but also on their hydraulic connection to the streams flowing in the valleys. Large withdrawals from the deposits result in a reduction in ground-water discharge to the streams and, if large enough, cause infiltration of water from the streams into the deposits.

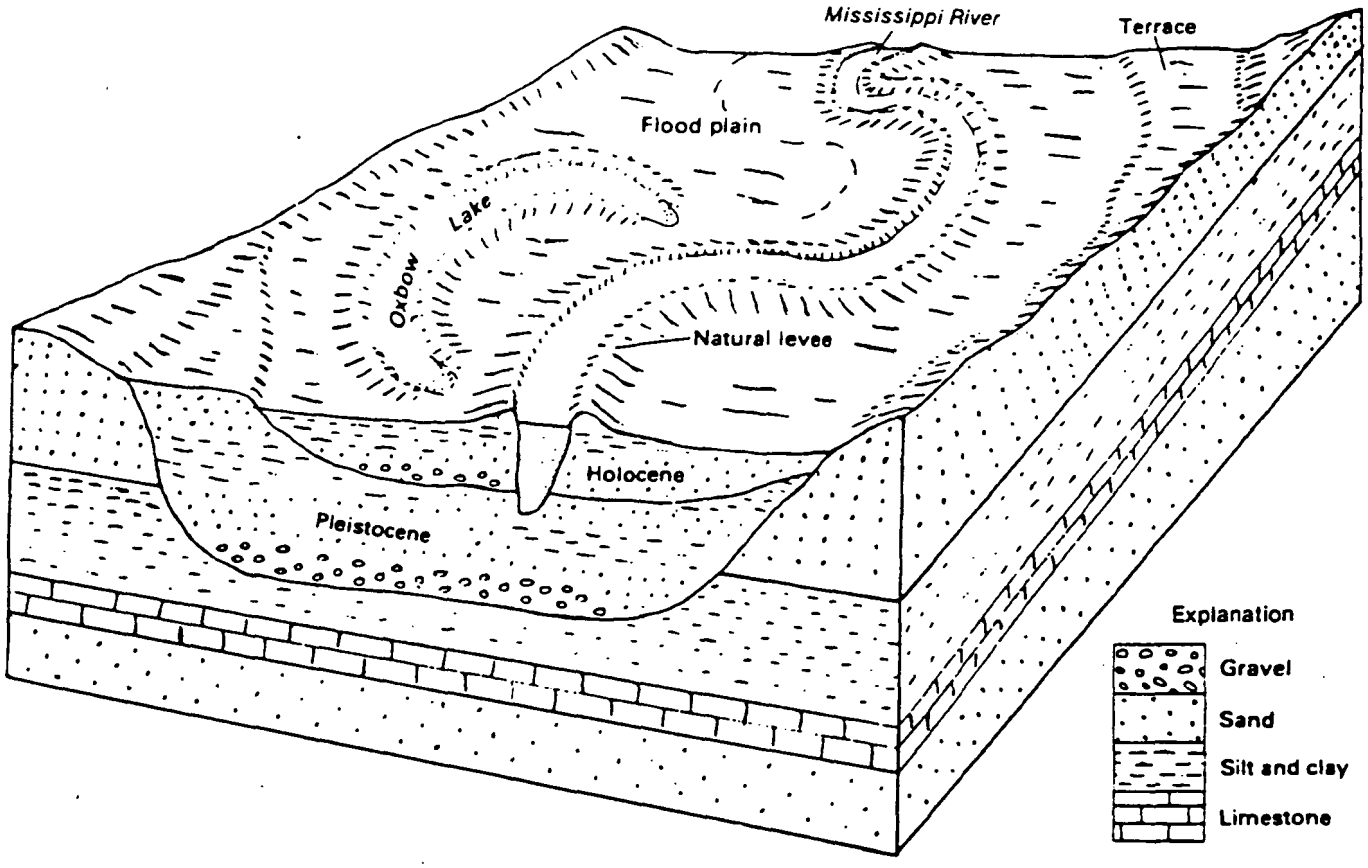


Figure 2-24. Topographic and Geologic Features of a Section of the Aluvial Valley of the Mississippi River

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A large number of publications were consulted, for both general and specific information, in the preparation of this paper. Specific reference to these publications generally is omitted in the text, both to avoid interruption of the discussions and to save space. Publications that served as primary references are listed below, under the categories of general references and references to regional discussions. General references include publications that were used both for background information on the classification of ground-water systems and for general information on the regions. References to the regional discussions include publications that served as a source of additional information on the individual regions.

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Chapter 3

GROUND WATER-SURFACE WATER RELATIONSHIP

Introduction

The interrelations between ground water and surface water are of great importance in both regional and local hydrologic investigations and a wide variety of information can be obtained by analyzing streamflow data. Most commonly the surface water investigator deals with stream hydrographs, channel characteristics, geomorphology, or flood routing. Although the hydrogeologist may evaluate induced infiltration into a streamside aquifer, he is generally more interested in aquifer characteristics, such as hydraulic conductivity, thickness, boundaries, and well yields. Many hydrologists tend to ignore the fact that, at least in humid areas, ground-water runoff accounts for a significant part of a stream's total flow.

Evaluation of the ground-water component of runoff can provide important and useful information regarding regional recharge rates, aquifer characteristics, and ground-water quality, and can indicate areas of high potential yield to wells. The purpose of this chapter is to describe a number of techniques that can be used to evaluate runoff to obtain a better understanding and evaluation of ground-water resources. In particular, the following will be examined:

1. Ground-water runoff
2. Surface runoff
3. Regional ground-water recharge rates
4. Determination of areas of relatively high permeability or water-yielding characteristics
5. Determination of the background concentration of ground-water quality
6. Estimation of evapotranspiration
7. Determination of the percentage of precipitation that is evaportranspired, becomes ground-water runoff, or becomes surface-water runoff.

The approaches taken, admittedly some highly subjective, are based on: (I) short-term runoff events, (2) long-term hydrographs, and (3) dry-weather flow measurements. In the first approach a single event, such as a flood wave of a few hours or few days duration, can be analyzed, while the latter two approaches are based on annual stream hydrographs, flow-duration curves, or seepage runs. Short-term events may provide a considerable amount of information for a local area, while long-term events are most useful for regional studies. Streamflow may consist of several components including ground-water runoff, surface runoff, effluent, and precipitation that falls directly into the channel.

The volume of water that is added by precipitation directly into the channel is relatively small compared to the stream's total flow. The contribution by waste effluent may or may not be significant, since it depends on the activities that are occurring in the basin. In permeable basins in humid regions, ground-water runoff may account for 70 or 80 percent of the stream's annual discharge. The remainder is surface runoff, which originates as precipitation or snow melt that flows directly into the stream channel. This chapter is concerned largely with ground-water runoff and surface runoff and the separation of these two components.

In order to fully appreciate the origin and significance of ground-water runoff, it is first necessary to examine the regional ground-water flow system. Figure 3-1 illustrates a typical flow pattern. Particularly in humid and semi-arid regions, the water table generally conforms with the surface topography. Consequently, the hydraulic gradient or water table slopes away from divides and topographically high areas toward adjacent low areas, such as streams and rivers. Topographic highs and lows, therefore, serve as recharge and discharge areas, respectively.

Ground-water flow systems may be local, intermediate, or regional. As these terms imply, ground-water flow paths may be short, amounting to a few yards at one

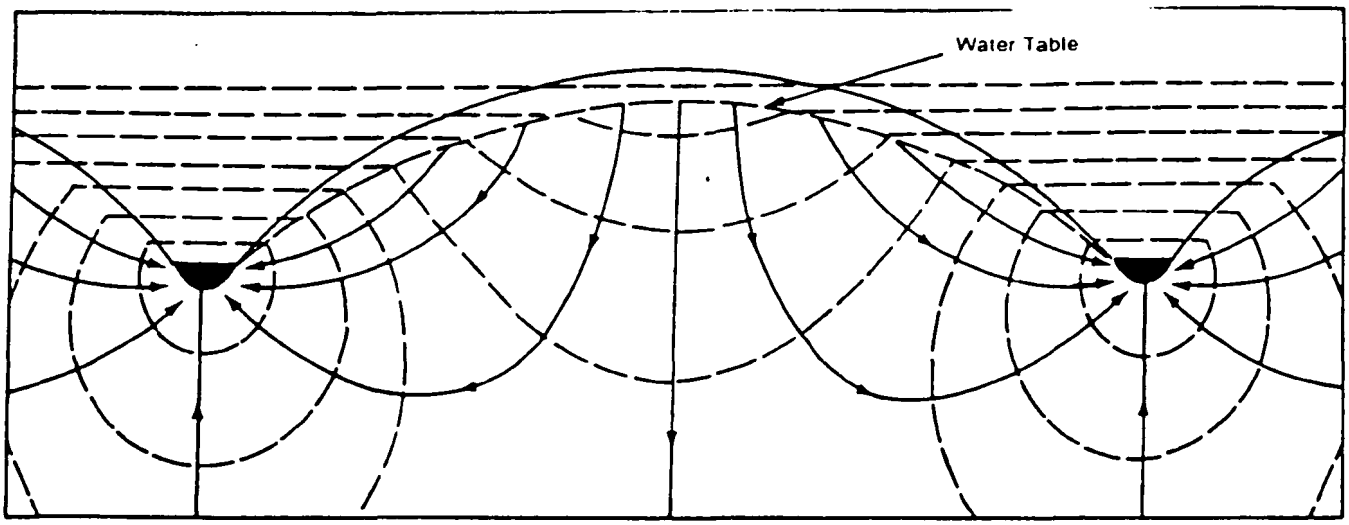


Figure 3-1. Approximate Flow Pattern in Uniformly Permeable Material between the Sources Distributed over the Air-Water Interface and the Valley Sinks (After Hubbert, 1940)

extreme to many miles in the regional case. Individual flow lines are, of course, influenced by the stratigraphy and, in particular, are controlled by hydraulic conductivity.

As water infiltrates a recharge area, the mineral content is relatively low. The quality changes, however, along the flow path and dissolved solids, as well as several other constituents, generally increase with increasing distances traveled in the subsurface. It is for this reason that even nearby streams may be typified by different chemical quality. A stream, seep, or spring in a local discharge area may be less mineralized than that issuing from a regional discharge zone because of the increase in mineralization that takes place along longer flow paths. It must be remembered, however, that other conditions, such as soil type, solubility of the enclosing rocks, surface drainage characteristics, and waste disposal practices, may have a profound effect on water quality at any particular site.

Even streams in close proximity may differ considerably in discharge even though the size of the drainage area and climatic conditions are similar. Figure 3-2 gives the superimposed hydrographs of White River in southwestern South Dakota and the Middle Loup River in northwestern Nebraska, which are good examples. White River has a low discharge throughout most of the year, but from May to September, flash floods are common. The wide extreme in discharge is characteristic of a flashy stream.

The flow of Middle Loup River is nearly constant, although from late spring to early fall higher flows may occur. These peaks, however, differ considerably from those found in White River because the increase in discharge takes place over a longer interval, the stage does not range widely, and the recession occurs more slowly. The differences in hydrographs of these two nearby rivers is puzzling, until the geology and topography of their respective basins are examined.

White River flows through the Badlands of South Dakota, an area of abrupt changes in relief, steep slopes, little vegetative cover, and rocks that consist largely of silt and clay, both of which may contain an abundance of bentonite. When wet, bentonite, a swelling clay, increases greatly in volume. As a result of these features, rainfall in the White River basin tends to quickly run off and there is little opportunity for infiltration and ground-water recharge to occur. Thus, intense rainstorms cause flash floods, such as those that occurred in June, August, and September.

The Middle Loup basin is carved into the undulating grassland topography of the Sandhills of Nebraska, where surficial materials consist of wind-blown sand. Since the low relief, grass-covered surface promotes infiltration, precipitation is readily absorbed by the underlying sand. As a result, there is very little surface runoff and a great amount of infiltration and ground-water recharge. The ground water slowly migrates to the river channel, thus providing a high sustained flow. In a comparison of the hydrographs of these two rivers, it is evident that the geologic framework of the basin

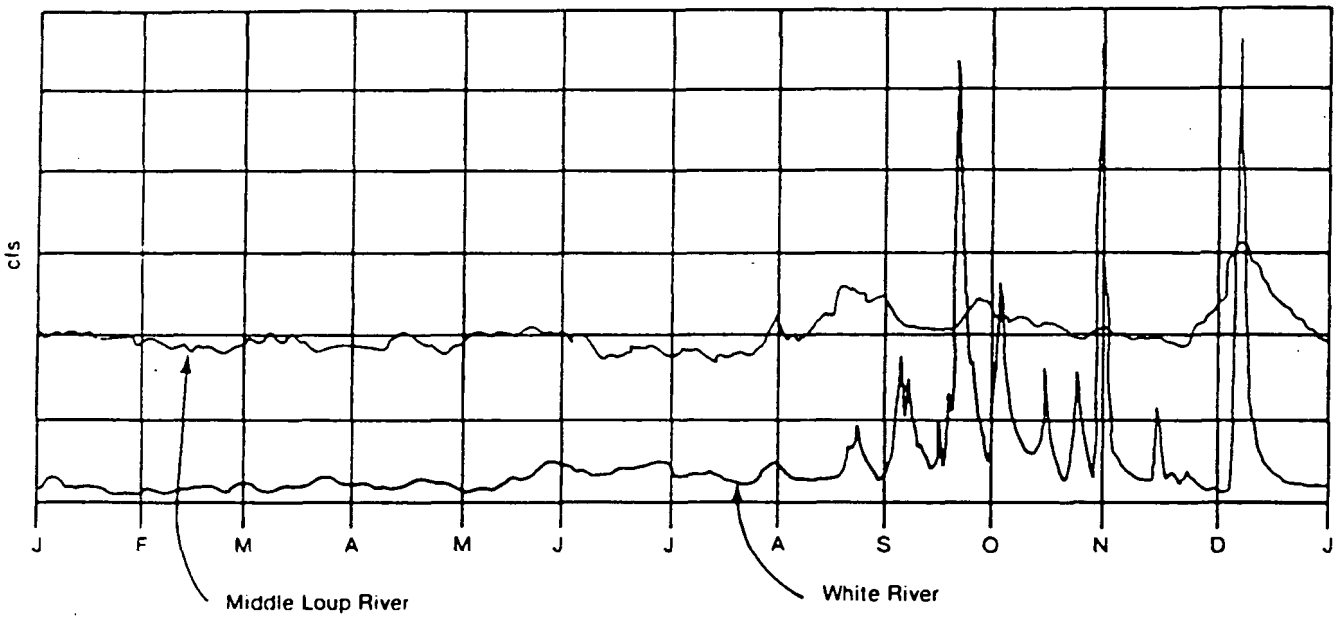


Figure 3-2. Hydrographs of Two Nearby Streams

serves as a major control on runoff. This further implies that in any regional hydrologic study, the investigation should begin with an examination of geologic maps.

Gaining and Losing Streams

Although the discharge of most streams increases downstream, the flow of some streams diminishes. These streams are referred to as gaining or losing, respectively. The hydrologic system, however, is even more complex, because a stream that may be gaining in one season, may be losing during another. Furthermore, various human activities may also affect a stream's discharge.

Under natural conditions a gaining stream is one where the water table is above the base of the stream channel. Of course the position of the water table fluctuates throughout the year in response to differences in ground-water recharge and discharge. Normally the water table is highest in the spring, which is the annual major period of ground-water recharge. From spring to fall, very little recharge occurs and the amount of ground water in storage is slowly depleted as it seeps into streams. Eventually, the water table may decline to the same elevation as a stream bottom, or even below it, at which time streamflow ceases except during periods of surface runoff. Following a period of recharge, caused either by infiltration of rainfall or seepage from a flood wave, the water table may again rise and temporarily contribute ground-water runoff.

Figure 3-3 shows a generalized diagram of the hydrology of a stream during two seasons of the year. During the spring, the water table is high and the gradient dips steeply towards the stream. If streamflow was measured at selected points, it would be found that the discharge increases downstream because of the addition of ground-water runoff. That is, it is a gaining stream. In the fall when the water table lies at or below the stream bottom, however, the same stream might become a losing stream. During a major runoff event the stage in the stream would be higher than the adjacent water table and water would migrate from the stream into the ground. The stream would continue to lose water until the water table and river stage were equal. When the stage declined, ground-water runoff would begin again.

In this case the stream changed from gaining to losing and back again to gaining. Similar situations may occur over longer intervals, such as during droughts. As a drought continues, the water table slowly declines as ground-water storage is depleted. A period of high flows, such as release from a dam, may cause tremendous amounts of water to flow from the stream channel into the ground, thus saturating the depleted streamside deposits. It may require weeks of high flow to replenish the ground-water reservoir, and until this is accomplished, the stream will be losing.

Some streams, particularly in arid and karst regions, are nearly always losing. Examples include those channels that cross coarse-grained alluvial fans. Even during

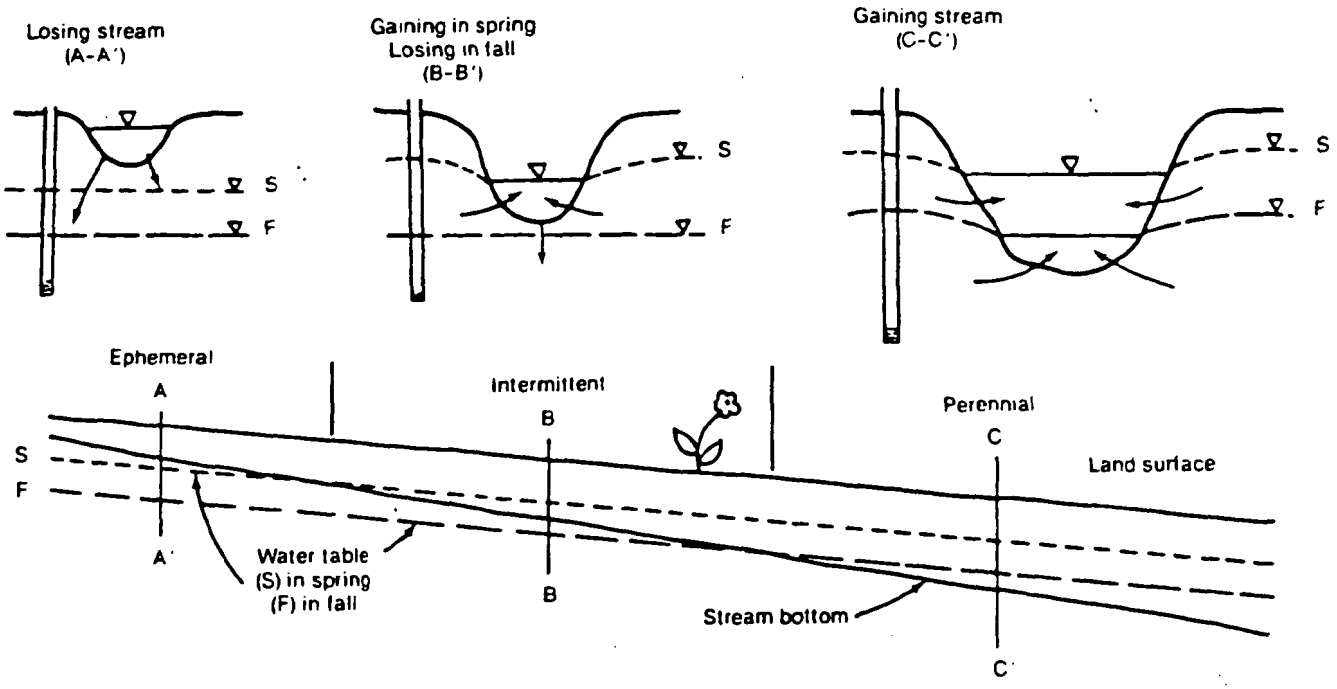


Figure 3-3. The Relation between the Water Table and Stream Types

flash floods, the great mass of flood water soon spreads out over the fan or adjacent desert to infiltrate or evaporate.

Because of the extensive network of solution openings in karst terrain, the water table may consistently lie below the bottom of all the streams. During a period of runoff, the water may rapidly flow into sink holes and solution openings or simply disappear into a swallow hole in a stream channel, only to appear again perhaps several miles downstream.

Gaining and losing streams also can be created artificially. Where well fields lie along stream channels and induce water to flow from the stream to the well, streaflow is diminished. In some cases stream depletion by pumping wells has proceeded to such an extent that the stream channels are dry throughout the year. Conversely, in some irrigated regions, so much infiltration occurs that the water table rises to near land surface. The underlying soil and ground water may become highly mineralized by the leaching of soluble salts. These highly mineralized waters may discharge into a stream, increasing its flow but deteriorating the chemical quality. In other places, municipal or industrial wastes may add considerably to a stream's flow, also deteriorating its quality. In fact, at certain times of the year, the entire flow may consist of waste water.

Bank Storage

Figure 3-4 shows that, as a flood wave passes a particular stream cross section, the water table may rise in the adjacent streamside deposits. The rise is caused by two phenomena. First, the stream stage, which is higher than the water table, will temporarily block ground-water runoff, thus increasing the amount of ground water in storage. Secondly, because of the increased head in the stream, water will flow from the stream channel into the ground, thus providing another component of water added to storage.

Once the flood wave begins to recede, which may occur quite rapidly, the newly added ground water will begin to flow back into the channel, rapidly at first and then more slowly as the hydraulic gradient decreases. This temporary storage of water in the near vicinity of the stream channel is called bank storage.

The rising and recession limbs of a hydrograph of a flood wave should provide clues concerning bank storage and streamside permeability. For example, where streamside deposits are of low permeability, such as clay or shale, the rising limb should be quite steep, but more gradual where the deposits are permeable. Since there would be little or no bank storage in the first case, recession curves also should

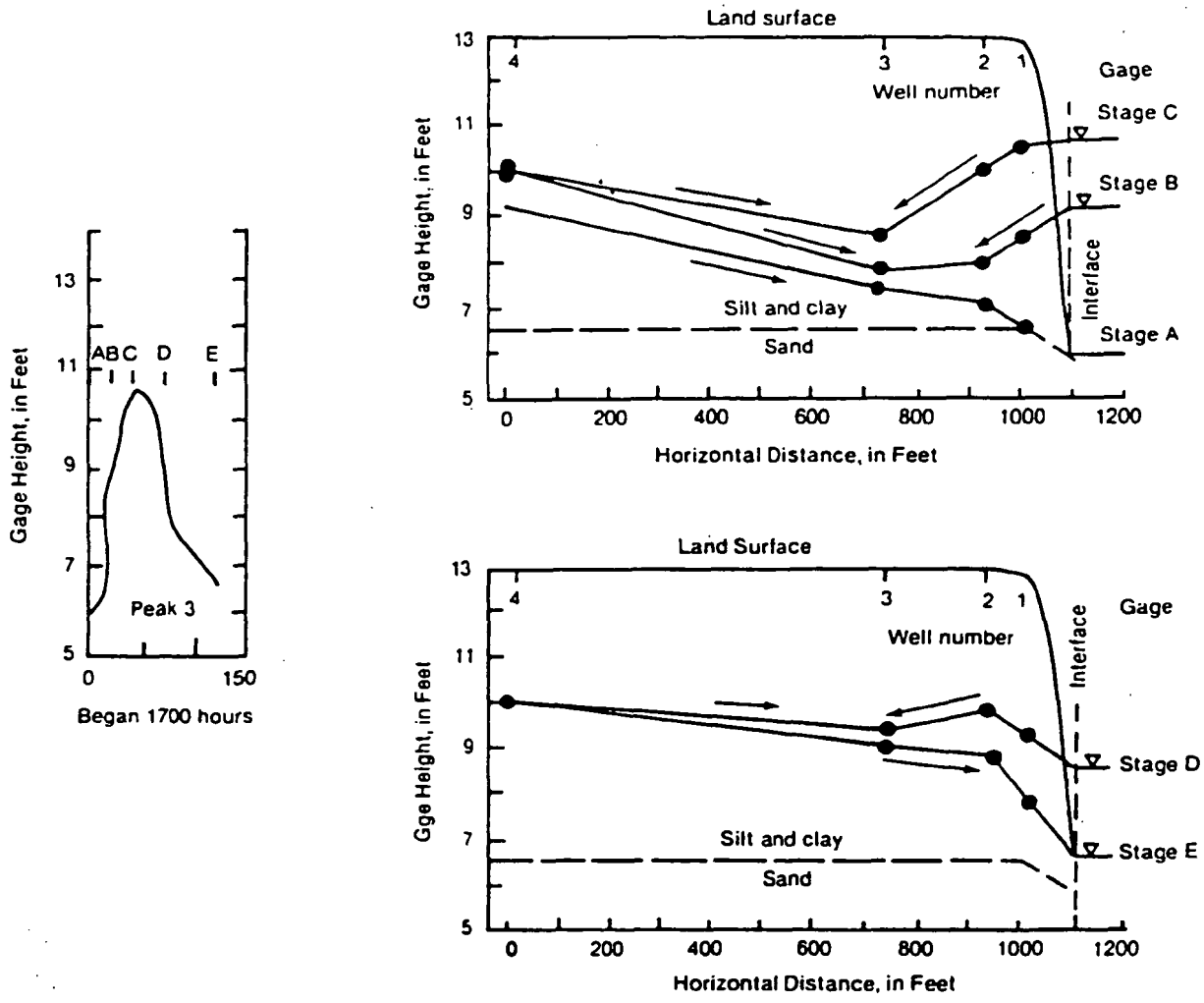


Figure 3-4. Movement of Water Into and Out of Bank Storage Along a Stream in Indiana

be steep, but the release from bank storage in a permeable basin should reduce the slope of the recession curve.

Effect of the Geologic Framework on Stream Hydrographs

Unfortunately, the discharge of ground water into a stream is not always as simple as has been implied from the above examples. As Figure 3-5 shows, an examination of the aquifer framework and its effect on a stream hydrograph is enlightening. Notice in Figure 3-5a that the stream channel is deeply cut into a shale that is overlain by sand. Ground water flows into the stream along a series of springs and seeps issuing at the sand-shale contact. During a runoff event the stream stage rises, but even at its peak, the stage remains below the top of the shale. In this case, the contribution of ground water remains constant despite the rise in stage. To separate the ground-water runoff component from the stream hydrograph, one merely needs to draw a straight line from the inflection points of the rising and falling limbs.

In Figure 3-5b the stream channel is cut into a deposit of sand that is underlain by shale. Ground water flows into the stream, but as the stage rises, ground-water runoff decreases and eventually stops. Surface water then begins to flow into the ground where it is retained as bank storage. As the stage declines, ground water again starts to discharge into the channel eventually providing the entire flow. This is the classic case of bank storage. Hydrograph separation is more difficult in this case.

Figure 3-5c is a combination of the previous two

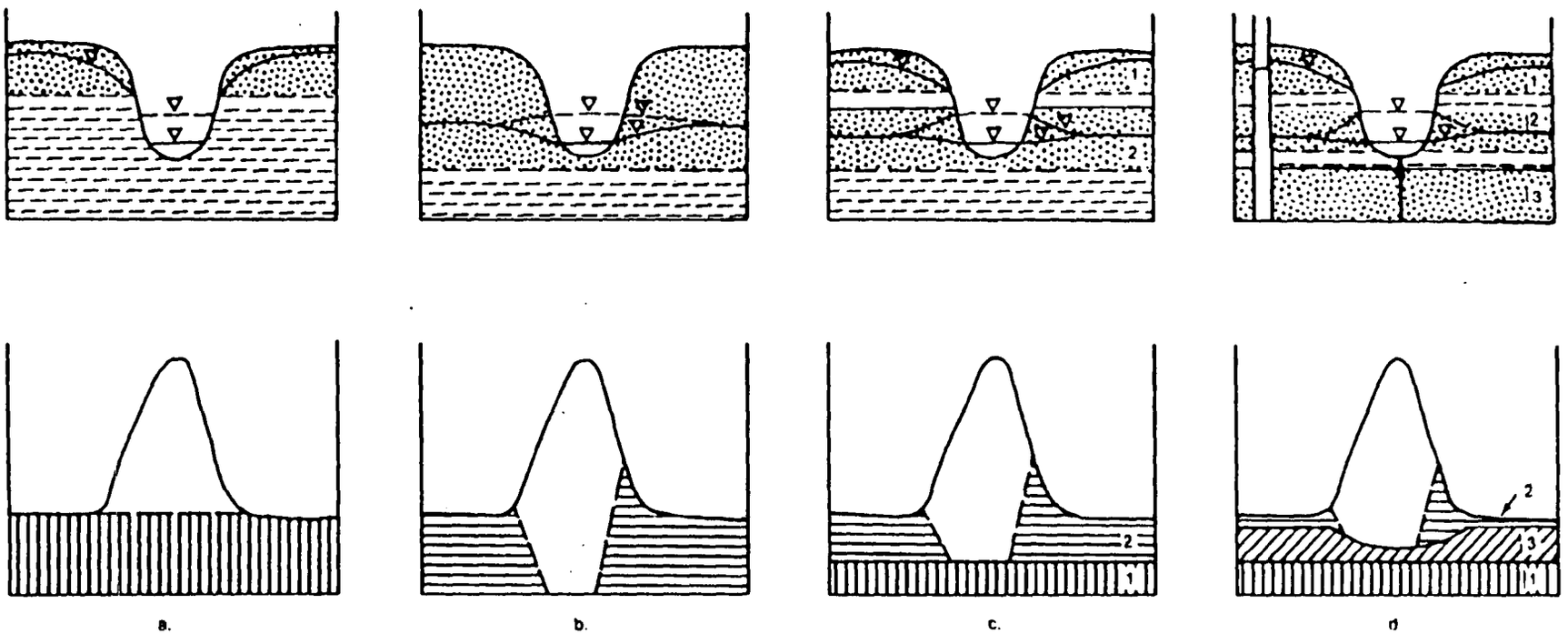


Figure 3-5. The Aquifer Framework in the Vicinity of a Stream Plays a Major Role in Ground-Water Runoff and Hydrograph Separation

examples. Ground water from a perched aquifer contributes a steady flow, while bank storage is gained and then released from the streamside aquifer. Hydrograph separation is even more difficult in this situation because of the contribution from both aquifers.

The final example, Figure 3-5d, consists of three aquifers—one perched, one in direct contact with the stream, and one deeper, confined aquifer. As the stream rises, there is a decrease in the head difference between the stream and the confined aquifer. The decrease in head difference will reduce upward leakage from the artesian aquifer, the amount depending on the thickness and vertical permeability of the confining bed and the head difference.

Single-Event Hydrograph Separation Techniques

Following a runoff event, the water held as bank storage begins to discharge into the channel. In the beginning the rate of bank storage discharge is high because of the steep water-level gradient, but as the gradient decreases so does ground-water runoff. The recession segment of the stream hydrograph gradually tapers off into what is called a depletion curve. To a large extent, the shape of the depletion curve is controlled by the permeability of the streamside deposits, although soil moisture and evapotranspiration may play important roles.

Depletion Curves

Intervals between surface runoff events are generally short and for this reason, depletion curves are plotted as a combination of several arcs of the hydrograph with the arcs overlapping in their lower parts, as shown in Figure 3-6. To plot a depletion curve, tracing paper is placed over a hydrograph of daily flows and, using the same horizontal and vertical scales, the lowest arcs of the hydrographs are traced, working backward in time from the lowest discharge to a period of surface runoff. The tracing paper is moved horizontally until the arc of another runoff event coincides in its lower part with the arc already traced; this arc is plotted on top of the first. The process is continued until all the available arcs are plotted on top of one another.

The upward curving parts of the individual arcs are disregarded because, presumably, they are affected by channel storage or surface runoff, or both. The resulting continuous arc is a mean or normal depletion curve that represents the trend that the hydrograph would have followed during a protracted dry period.

Even for the same stream, there may be appreciable differences in the shape of the depletion curve at different times of the year. This is largely due to evaporation, transpiration, and temperature effects. In cases such as these, a family of depletion curves may be constructed. One curve should represent winter when there is little or no evapotranspiration, another curve should represent the summer when evapotranspiration is at its maximum, and perhaps a third curve should be prepared to represent intermediate conditions.

Depletion curves are the basis for estimating ground-water runoff during periods of surface runoff. They also shed a great deal of light on the characteristics of a ground-water reservoir.

Hydrograph Separation

A flood hydrograph is a composite hydrograph consisting of surface runoff superimposed on ground-water runoff. When attempting to separate these two components of flow, however, some problems generally occur. Whatever method is employed, there is always some question as to the accuracy of the division. One can only say that, in any given case, ground-water runoff is probably not less than about "x" or more than about "y." Keeping in mind the complexities of a stream hydrograph brought about by variable parameters, and particularly the geology of the basin, an attempt will be made to develop some logical methods for hydrograph separation.

Using the flood hydrograph in Figure 3-7a, we can see that point A represents the start of surface runoff. Using a previously prepared depletion curve, the original recession can be extended to B. The area below AB represents the ground-water runoff that would have occurred had there been no surface runoff. Point D represents the end of surface runoff. A depletion curve can be matched with the recession limb, extending it from D to C. A partial envelope has now been formed that shows the upper and lower limits between which a line may reasonably be drawn to separate the two components of runoff. This assumption ignores possible effects brought about by difference in the geologic framework. This envelope forms a basis for the most commonly used separation methods which are described below.

Method 1. Using a depletion curve and starting at D in Figure 3-7b, extend the recession curve back to a line drawn vertically through the peak of the hydrograph (C). A second line is then extended from A, the start of surface runoff, to C. This method is more likely to be valid where ground-water runoff is relatively large and reaches the stream quickly.

Not uncommonly, the end of surface runoff is difficult to

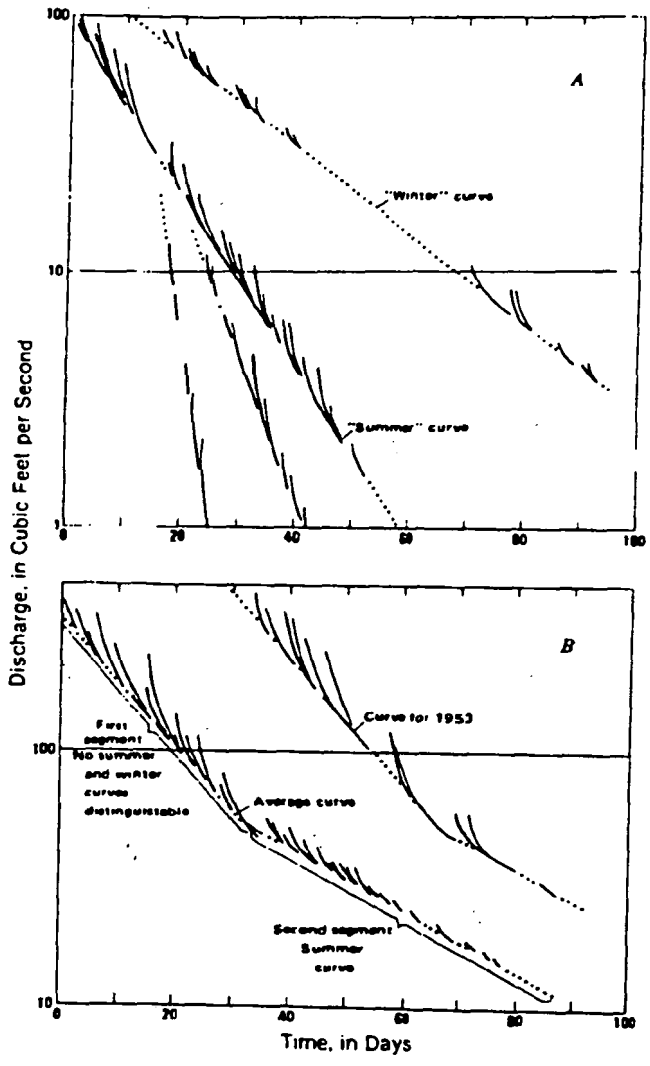


Figure 3-6 Ground-Water Depletion Curves Have Different Shapes that Reflect the Seasons

determine, but point D can be estimated by means of the equation

$$N=A^2 \quad (1)$$

where N = the number of days after a peak when surface runoff ceases and A = the basin area, in square miles. The distance N is measured directly on the hydrograph.

Method 2. In this example in Figure 3-7b, separation is accomplished merely by extending a straight line, originating at the start of surface runoff (A), to a point on the recession curve representing the end of surface runoff (D). This method of separation is certainly the simplest and is justifiable if little is known about the aquifer framework.

Method 3. In this example, also in Figure 3-7b, the prerunoff recession line is extended from A to a point directly under the hydrograph peak (B). From this point a second line is projected to D, the end of surface runoff.

The separation technique to be employed should be based on knowledge of the hydrogeology of the basin, keeping in mind the effect of the geologic framework on the hydrograph.

Separation of Complex Hydrographs

Commonly runoff events occur at closely spaced intervals and there is insufficient time for the recession curve to develop before runoff again increases. This complicates hydrograph separation.

Figure 3-7c shows two methods that can be used to determine ground-water runoff under a complex hydrograph, which represents two storms.

Method 1. The recession curve preceding the first runoff event is continued to its intersection with a line drawn through the first peak (A-B). The distance N is calculated and measured. The recession limb of the first event is continued to its intersection with the N-days line (C-D). Line B-D is then constructed. The first recession trend is continued to its intersection with a line drawn through the peak of the second runoff event (CD-E). From this point (E), the line is extended N days.

Method 2. As Figure 3-7c shows, the easiest method is to project a straight line from A to F. Although by far the simplest, this technique is not necessarily any less accurate than Method 1.

Hydrograph Separation by Chemical Techniques

Generally ground water is more highly mineralized than surface runoff. During baseflow the stream's natural quality is at or near its maximum concentration of dissolved solids, but as surface runoff reaches the channel and provides an increasing percentage of the flow, the mineral content is diluted. Following the discharge peak, surface runoff diminishes, ground-water runoff increases, and the mineral content again increases.

Several investigators have used the relation between runoff and water quality to calculate the ground-water contribution from one or more aquifers or to measure streamflow. This method of hydrograph separation, which requires the solution of a series of simultaneous equations, is based on the concentration of a selected chemical parameter that is characteristics of ground-water and surface runoff.

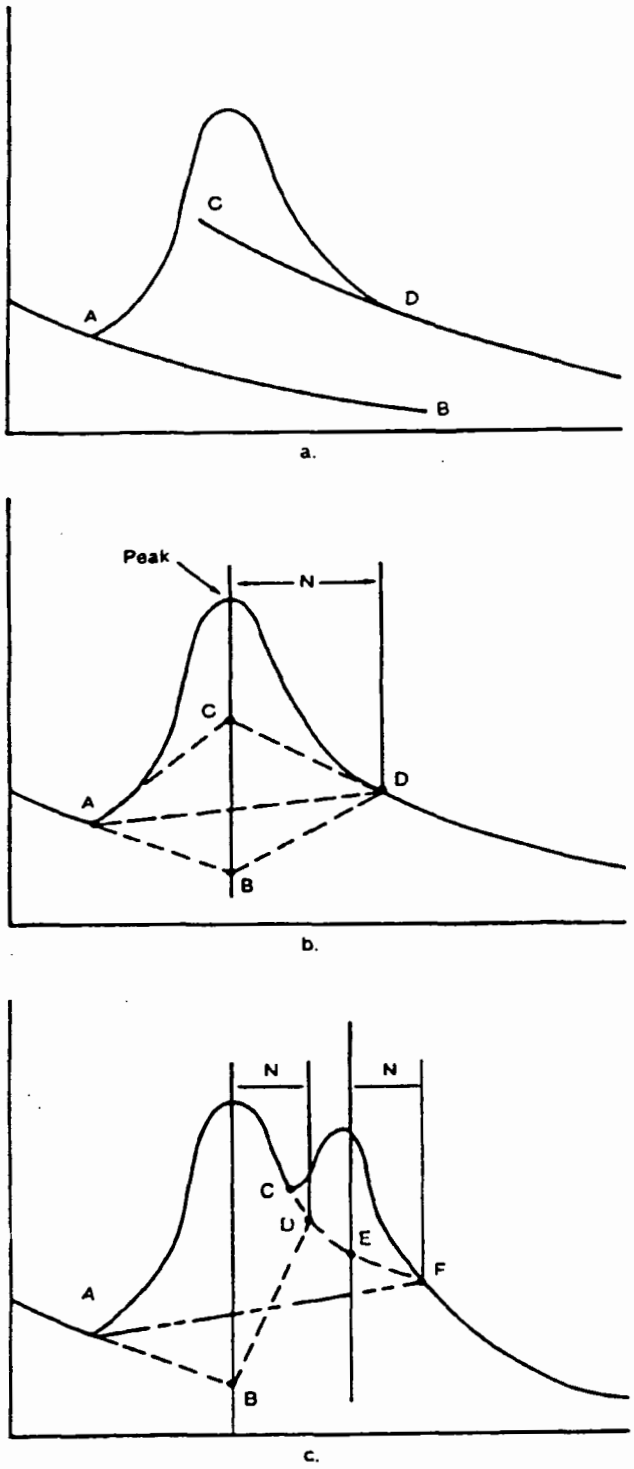


Figure 3-7. Separation of the Stream Hydrograph

The basic equations, which may take several forms, are as follows:

$$Qg + Qs = Q$$

$$CgQg + CsQs = CQ \quad (2)$$

where Qg, Qs, and Q are ground-water runoff, surface runoff, and total runoff, respectively; and Cg, Cs, and C represent the concentration of dissolved mineral species or specific conductance of ground water, surface runoff, and total runoff, respectively. Usually specific conductance is used as the C parameter because of the relative ease of obtaining it.

If Cg, Cs, C, and Q are known we can determine the quantity of ground-water runoff as follows:

$$Qg = Q\,(C—Cs)/(Cg—Cs) \quad (3)$$

C is determined by measuring the specific conductance in a well, in a series of wells, or during baseflow. The quality of surface runoff (Cs) is obtained from analysis of overland flow or, possibly in the case of small streams, at the period of peak discharge when the entire flow consists of surface runoff. It is assumed Cg and Cs are constant. C and Q are measured directly.

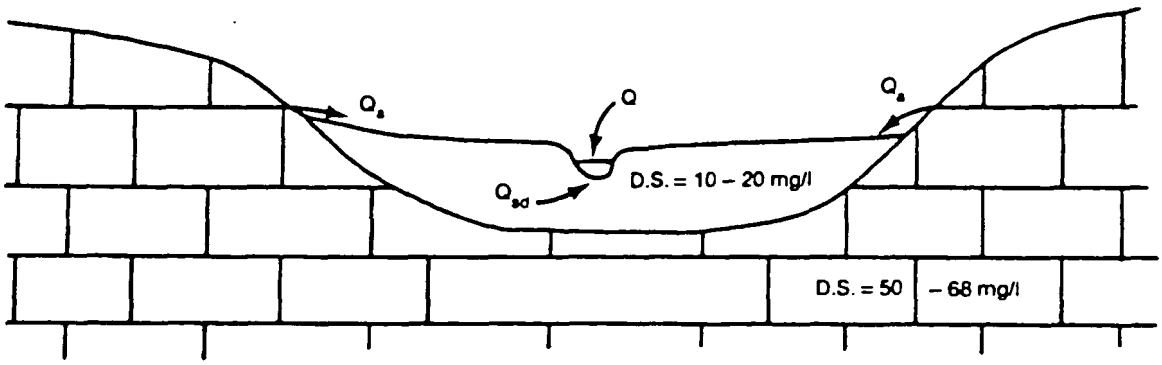
Visocky (1970) used continuous recording equipment to measure specific conductance and stage (water level) in the Panther Creek Basin in north-central Illinois. By using the equations given above, he calculated the ground-water runoff component of the stream on the basis of the relationship between discharge and specific conductance. He also calculated and compared ground-water runoff as determined from a ground-water rating curve and found that the chemical method provided a lower estimate under normal conditions than did the rating curve technique. On the other hand, the chemical method indicated more ground-water runoff following storms that were preceded by extended dry periods, which had caused considerable declines in water level in nearby observation wells.

During baseflow, the quantity of ground-water discharge from surficial sand and from limestone in the Floridan artesian aquifer into Econfina Creek in northwest Florida was distinguished by Toler (1965). In this case, as Figure 3-8 shows, the artesian water had a dissolved solids content of 50-68 mg/L, while that from the surficial sand was only 10-20 mg/L. The artesian water discharged through a series of springs along the central part of the basin and amounted to 70 to 75 percent of the stream's baseflow. The equation used for this analysis is as follows:

$$Qa = (C\text{-}Csd)/(Ca—Csd)\ Q \quad (4)$$

where Qa = artesian runoff, Q = runoff and Csd, Ca, and C represent the dissolved solids in water from the sand, the artesian aquifer, and during any instant in the stream, respectively. Of course,

$$Q\text{-}Qa = Qsd \quad (5)$$



$$Q_a + Q_{sd} = Q$$
$$CQ_a + CQ_{sd} = CQ$$

$Q_a = ?$ $C_a = 50$
$Q_{sd} = ?$ $C_{sd} = 10$
$Q = 18$ $C = 43$

$$Q_a + Q_{sd} = 18$$
$$50\,Q_a + 10\,Q_{sd} = 43 \times 18$$

$$-10Q_a - 10Q_{sd} = -180$$
$$50Q_a + 10Q_{sd} = 774$$
$$40Q_a = 594$$
$$Q_a = 14.85 \text{ fs}$$

$$Q_a = \frac{Q\,(C - C_{sd})}{C_a - C_{sd}} \quad \text{OR} \quad Q_a = \frac{18(43 - 10)}{50 - 10} = \frac{594}{40} = 14.85 \text{ cfs}$$

Figure 3-8. Contribution to Econfina Creek During a Period of Dry Weather Flow When the Stream Discharge Was 18 cfs and the Dissolved Solids Concentration Was 43 mg/L: From Sand Aquifer = 3.15 cfs, From Limestone Aquifer = 14.85 cfs

Continuous streamflow and conductivity measurements were collected at a gaging station on Four Mile Creek in east-central Iowa by Kunkle (1965). The basin above the gage, which contains 19.5 m^2, consists largely of till that is capped on the uplands by loess. As Figure 3-9 shows, the stream lies in a valley that contains as much as 30 feet of permeable alluvium. Ground water from the alluvium and loess, as well as the stream during low flow, has an average specific conductance of 520 micromhos (Cg) while surface runoff is about 160 micromhos (Cs).

Figure 3-10 shows continuous record of discharge and conductivity representing a storm in September 1963. Instantaneous ground-water runoff during this event was calculated for several points under the hydrograph by using the following formulas

$$Qg + Qs = Q$$
$$CgQg + CsQs = CQ \quad (6)$$

where Qg = ground-water runoff, Qs = surface runoff, Q = runoff, and Cg, Cs, and C = specific conductance of ground-water runoff, surface runoff, and runoff, respectively. As determined from the graphs in Figure 3-10, where Q = 2.3 cfs, C = 410; Cg = 520 and Cs = 160, then Qg is 1.6 cfs. Therefore, when the stream's discharge (Q) was 2.3 cfs, ground-water runoff was 1.6 cfs. This calculation provides one point under the hydrograph. Several other points need to be determined so that a separation line can be drawn.

Computer Separation Programs

Various methods of hydrograph separation have been described, all of which are laborious, time consuming, and, commonly, open to questions of accuracy and interpretation. In each case a mechanical technique is used to provide a number of points on a hydrograph through which a line can be drawn that separates ground-water runoff from surface runoff. Once this line is determined, one must measure, directly on the hydrograph, the daily components of streamflow and then sum the results.

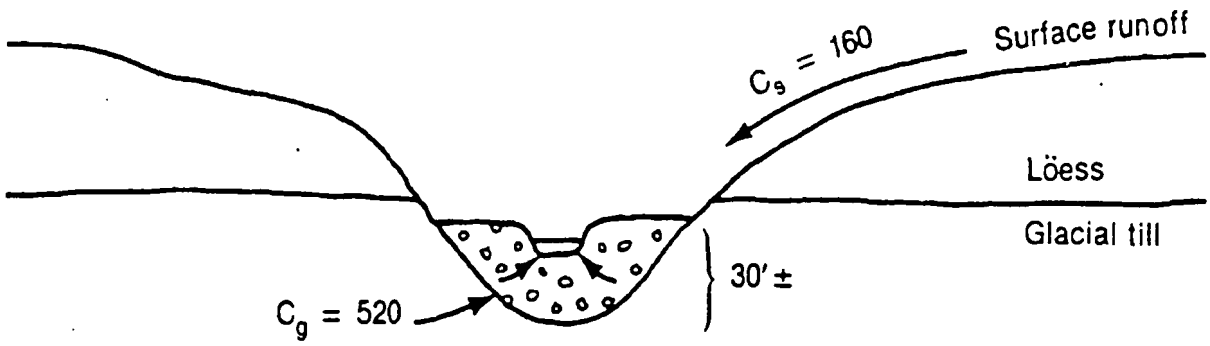


Figure 3-9. Four Mile Creek, Iowa

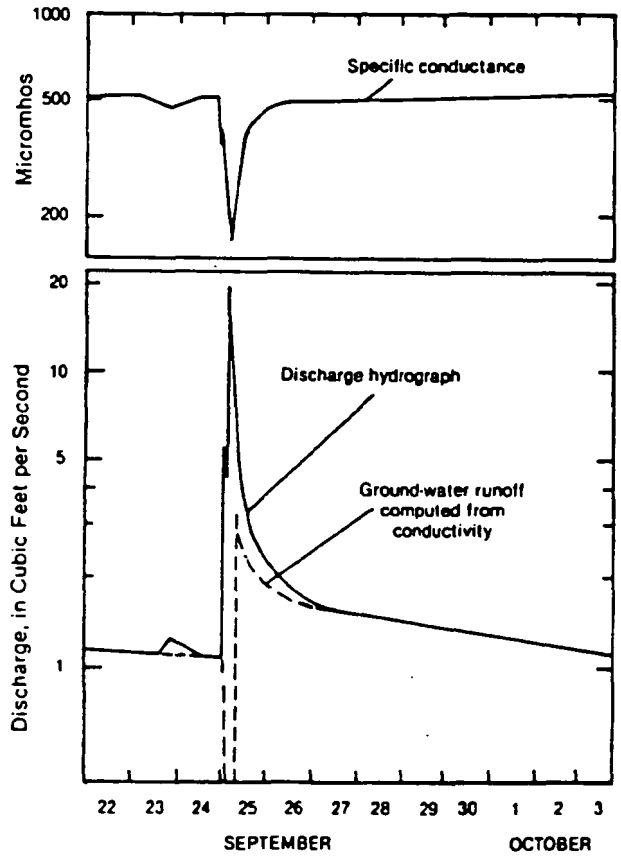


Figure 3-10. Hydrographs Showing Water Discharge, Specific Conductance, and Computed Ground-Water Runoff in Four Mile Creek near Traer, Iowa, September and October 1963

Annual ground-water runoff divided by total discharge provides the percentage of streamflow that consists of ground water. Effective ground-water recharge is that quantity of precipitation that infiltrates, is not removed by evapotranspiration, and eventually discharges into a stream.

Effective ground-water recharge rates can be easily estimated with a computer program described by Pettyjohn and Henning (1979). This program separates the hydrograph by three different methods, provides monthly recharge rates and an annual rate, produces a flow-duration table, and gives the operator the option of generating the separated hydrograph and a flow-duration curve with a line printer; as illustrated in Figure 3-11. The data base is obtained from annual streamflow records, which are published by the U.S. Geological Survey. The computer program will operate on a mainframe or microcomputer.

Ground-Water Rating Curve

A widely used technique to measure streamflow is the surface-water rating curve, which shows the relation between stage and discharge. Figure 3-12 shows a similar curve, called a ground-water rating curve, that illustrates the relation between the water table and streamflow. Prepared for those aquifer-stream systems that are hydrologically connected, the ground-water rating curve can be used to separate ground-water runoff from a stream hydrograph.

To prepare the curve, synchronous water table and stream discharge measurements are required. Ground-water levels are obtained either from: (1) a series of wells spread throughout the basin, (2) a series of wells, each of which represents an area of similar geology, or (3) a single near-stream well. Wells influenced by pumping should not be used. If more than one well is used, water levels, referred to some datum, such as sea level, must be averaged to form a composite curve. Furthermore, measurements of both ground water and stream stage should be made only during rainless

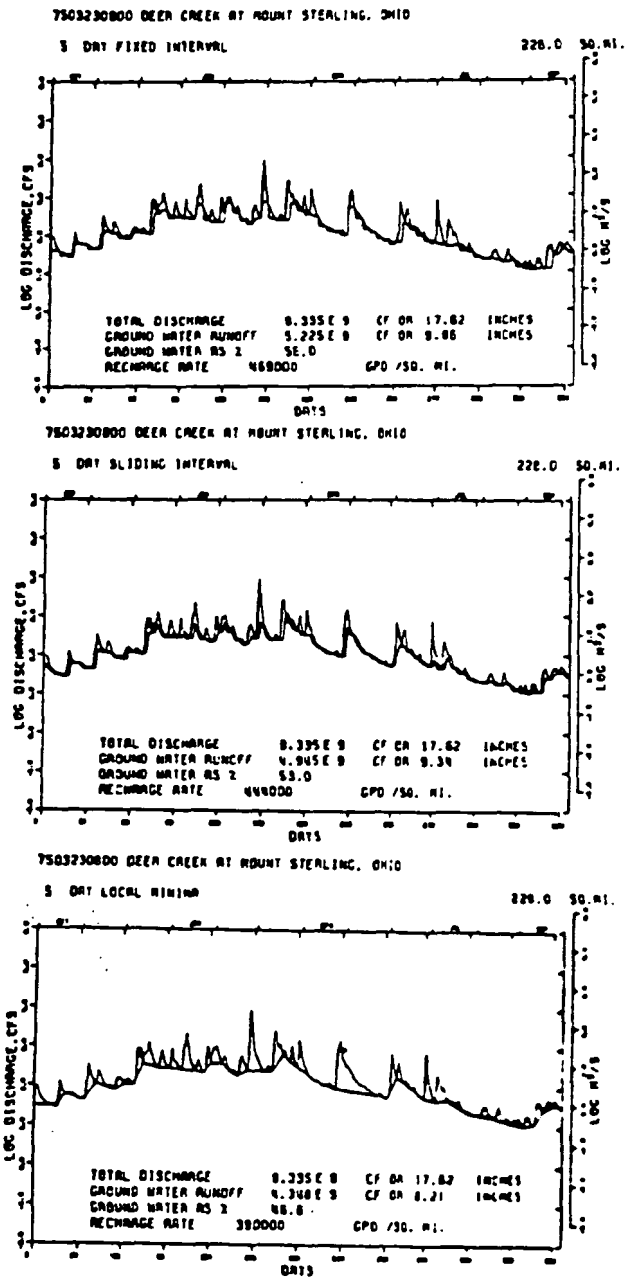


Figure 3-11. Deer Creek Hydrographs Separated by Three Methods and Statistical Data

intervals when streamflow consists entirely of ground-water runoff. Selected water-level measurements are plotted on a graph with the mean daily streamflow and a smooth curve is drawn through the points.

The graph is used by determining, either from individual measurements or water-level recorder data, the ground-water stage, reading across the graph to the curve, and then reading down to the stream discharge. For example, in Figure 3-12 when the mean ground-water stage is 44.5 feet, ground-water runoff is 10 cfs. Any flow in excess of this amount is surface runoff. Daily values of ground-water runoff are plotted on the stream hydrograph, eventually forming a continuous line throughout the period of record.

Although wells produce only limited yields from crystalline rocks in the Piedmont Upland part of the Delaware River Basin, streams have unusually high base flows. Olmsted and Hely (1962) used a ground-water rating curve to study this apparent inconsistency in a 287 m^2 part of the Brandywine Creek basin in southeastern Pennsylvania, as illustrated in Figure 3-13.

Bedrock units in the dissected upland basin, which consists largely of folded Precambrian and Paleozoic igneous and metamorphic rocks, have similar hydrologic characteristics. Weathered material of variable thickness mantles the area and the water table lies largely within this zone. Precipitation averages about 44 inches.

The 16 observation wells used in this study ranged from 12 to 40 feet in depth; all tapped a weathered schist aquifer. Six or seven wells were measured weekly or immediately after storms and wells De-3, Ch-13, and Ch-14 were selected as index wells. The average depth to water in all of the wells was 17.45 feet and the annual fluctuation was 5.75 feet.

Figure 3-14 shows a composite hydrograph of the three index wells and the discharge of Brandywine Creek. The curves have similar trends, differing only in amplitude following runoff events. This is to be expected because of the quick response time of a stream. Certainly the closer an observation well is to a stream, the more nearly the hydrographs will approach a similar shape.

The rating curve in Figure 3-15 shows the relation between ground-water runoff and ground-water stage in the Brandywine Creek basin. Notice the elliptical pattern of the data, which approach a straight line from October through March but then loop back during spring, summer, and early fall. Although confusing at first glance, the significance of the loop becomes readily apparent when the changes that occur in a ground-water reservoir throughout a 12-month period in a humid area are considered. From late fall to spring, the ground-water stage rises because there are little or no losses to evapotranspiration, soil moisture may be at or above field capacity, and ground-water recharge occurs. The water table reaches its peak during the spring runoff. From April to October, however, large quantities of ground water are removed by evapotranspiration, the soil moisture becomes so depleted there is little or no recharge, and the quantity of water in storage decreases because ground-water runoff exceeds recharge. Thus,

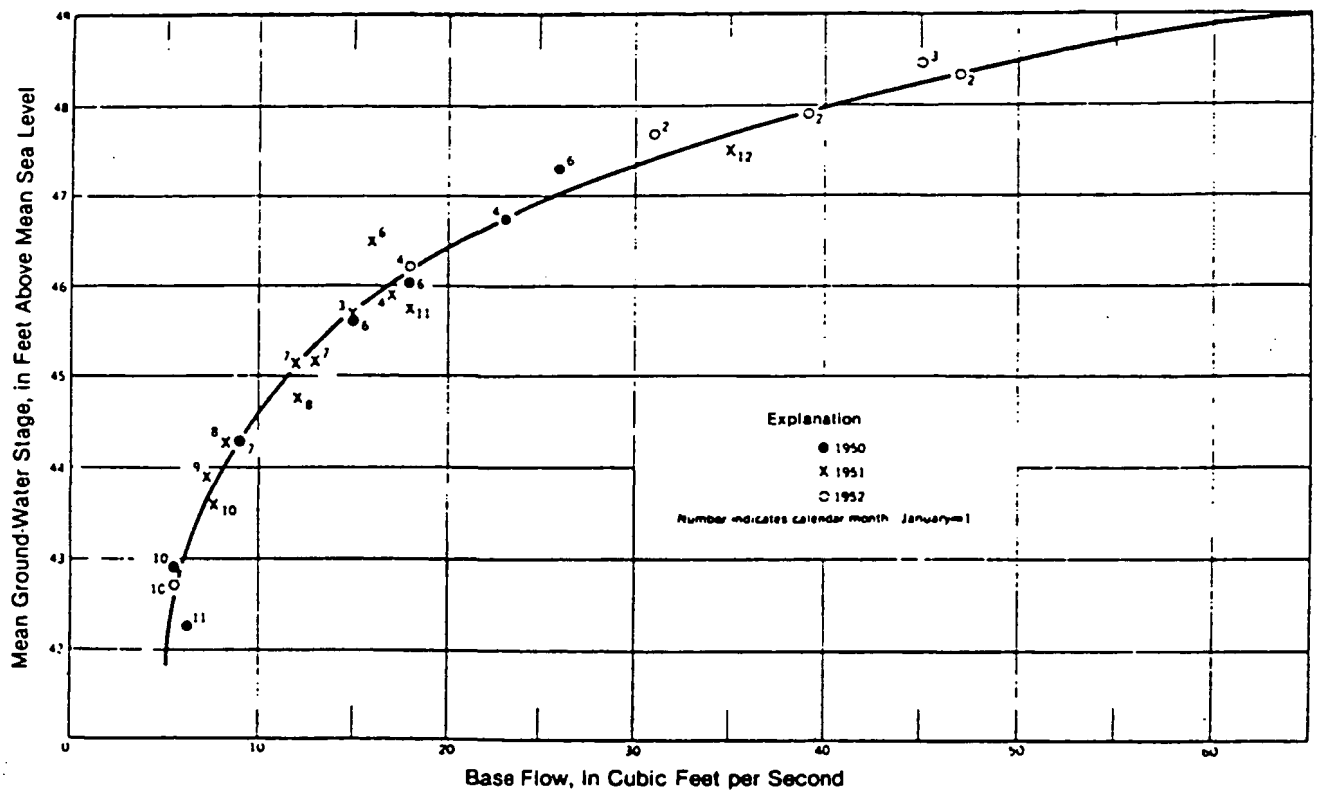


Figure 3-12. Rating Curve of Mean Ground-Water Stage Compared with Base Flow of Beaverdam Creek, Maryland

the elliptical shape of the data on the rating curve is controlled by evapotranspiration.

Using the rating curve, Olmsted and Hely (1962) separated the Brandywine Creek hydrograph, shown in Figure 3-16, and found that over a six year period, ground-water runoff accounted for 67 percent of the total flow. This compares favorably with the 64 percent determined for North Branch Rancocas Creek in the coastal plain of New Jersey; 74 percent for Beaverdam Creek in the coastal plain of Maryland (Rasmussen and Andreason, 1959); 42 percent for Perkiomen Creek, a flashy stream in the Triassic Lowland of Pennsylvania; and 44 percent for the Pomperaug River Basin, a small stream in Connecticut (Meinzer and Stearns, 1928).

During certain times of the year, when the water table lies at a shallow depth and large quantities of water are lost by evapotranspiration, a single rating curve cannot be used to separate a hydrograph with any degree of accuracy. As Figure 3-17 shows, Schicht and Walton (1961), in their study of Panther Creek basin in Illinois, developed two rating curves. One is used when evapotranspiration is very high and the other when evapotranspiration is small. Double rating curves also can be used to estimate evapotranspiration losses.

Evapotranspiration can also be calculated from the graph used by Olmsted and Hely (1962) in the case cited above. For example, when the ground-water stage was 80 inches, streamflow was expected to be about 550 cfs in February and March but only 400 cfs in June. In this case, the difference, about 150 cfs, is due to evapotranspiration.

Seepage or Dry Weather Measurements

Seepage or dry-weather measurements consist of flow determinations made at several locations along a stream during a short time interval. It is essential that there be no surface runoff during these measurements. Many investigators prefer to conduct seepage runs during the stream's 90 percent flow, that is, when the flow is so low that it is equaled or exceeded 90 percent of the time.

It is often implied that the 90 percent flow is the only time the flow consists entirely of ground-water runoff. This is not necessarily the case. The 90 percent flow-duration period, depending on geographic location and climate, commonly occurs during the late summer and fall when

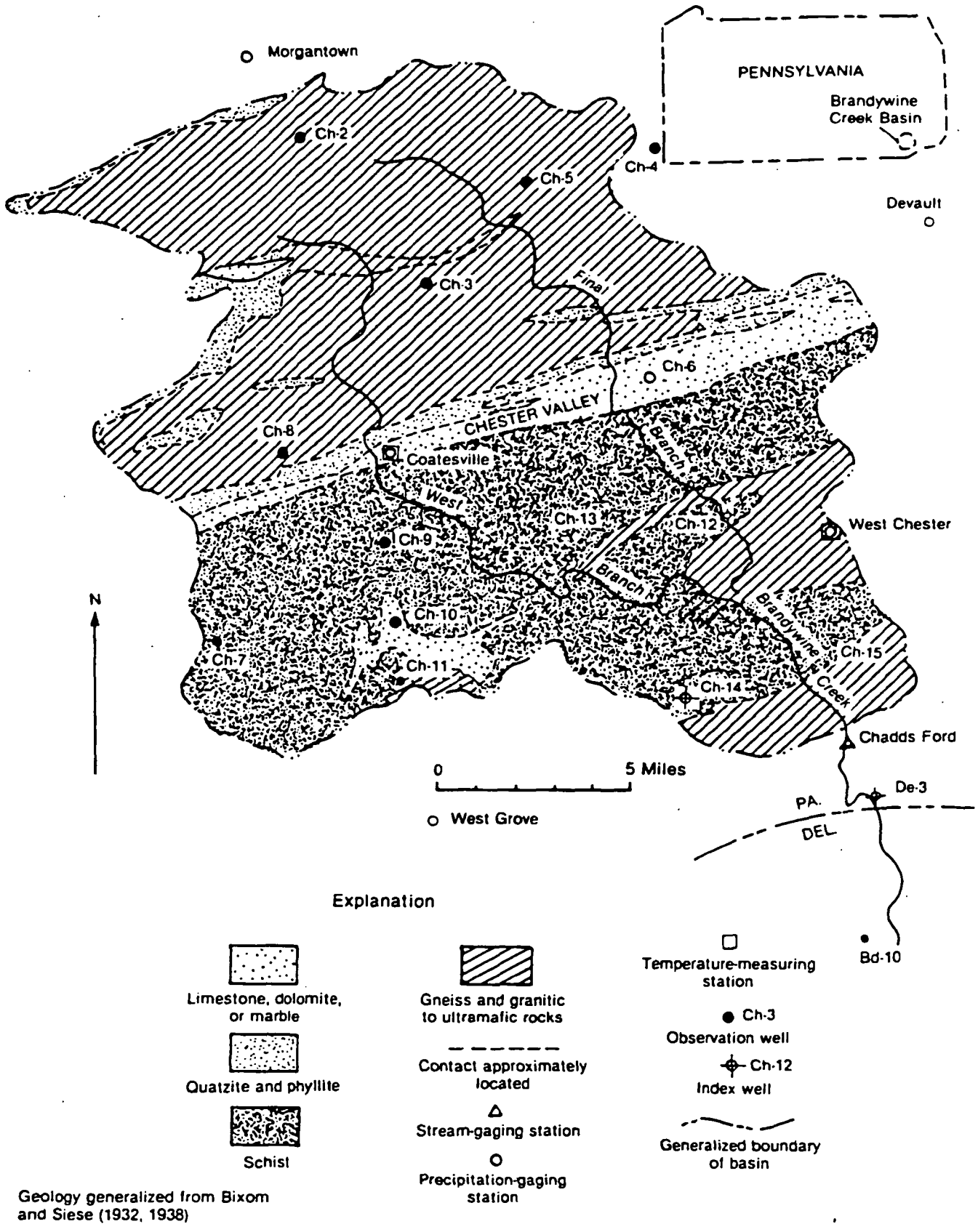


Figure 3-13. Sketch Map of Brandywine Creek Basin, Showing Generalized Geology and Location of Hydrologic and Meteorologic Stations Used in Report

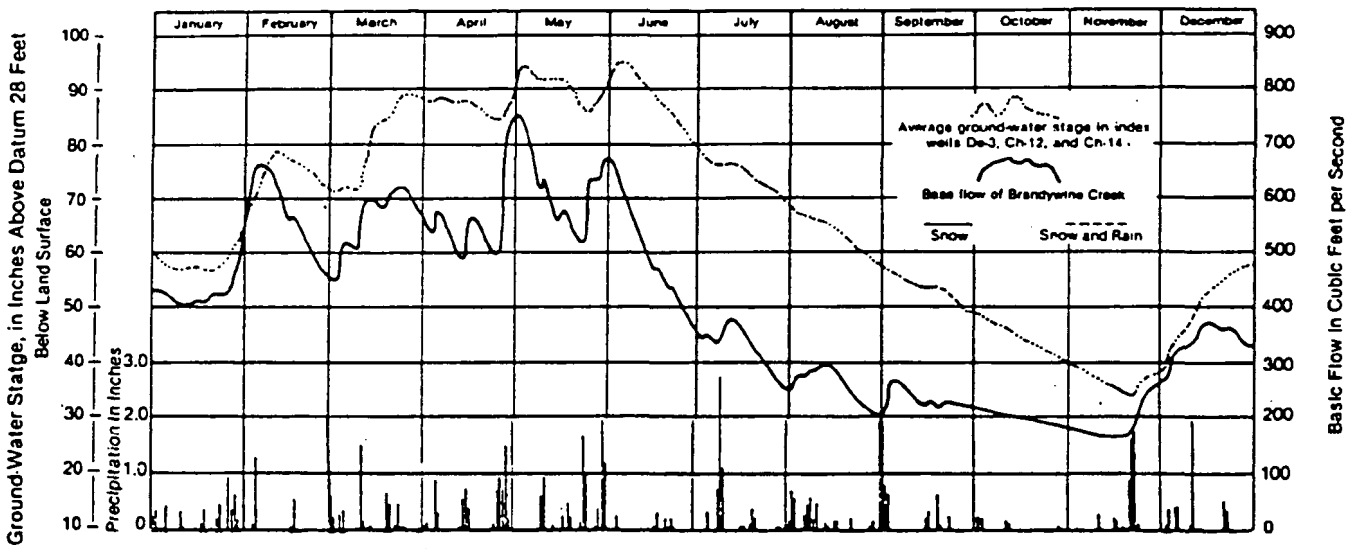


Figure 3-14. Composite Hydrograph of Three Index Wells and the Discharge of Brandywine Creek

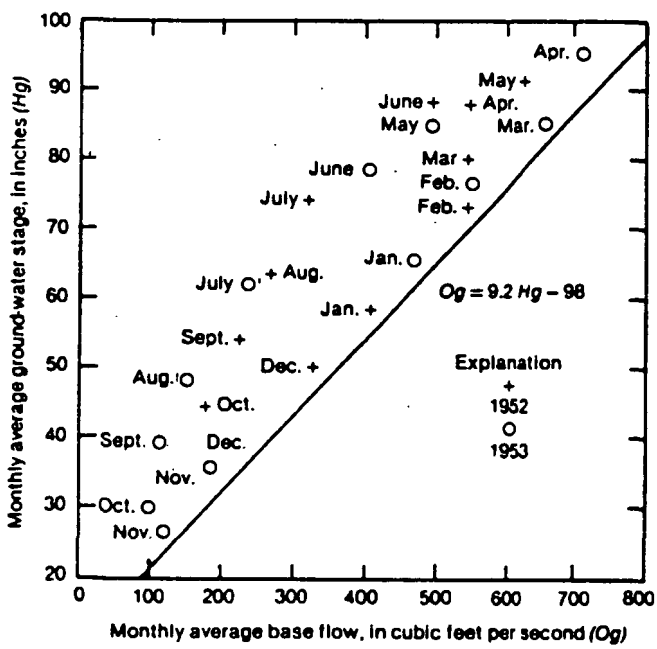


Figure 3-15. Relation of Monthly Average Base Flow to Ground-Water Stage in the Brandywine Creek Basin

soil moisture is depleted, there is little or no ground-water recharge, and the water table, having declined to its lowest level, has a low gradient. Under these conditions, ground-water runoff is minimal. However the physical aspect of the system may change following a recharge period and ground-water runoff may account for a substantial portion of the stream's flow. Hydrograph analyses, using techniques already described, may readily show that ground water provides 50 to 70 percent or more of the runoff. Therefore, the 90 percent flow may reflect only a small fraction of the total quantity of ground-water runoff.

Seepage measurements permit an evaluation of ground-water runoff (how much there is and where it originates) and provides clues to the geology of the basin as well. The flow of some streams increases substantially in a short distance. Under natural conditions this increase probably indicates deposits or zones of high permeability in or adjacent to the stream channel. These zones may consist of deposits of sand and gravel, fracture zones, solution openings in limestone or merely by local facies changes that increase permeability. In gaining stretches, ground water may discharge through a number of springs and seeps, along valley walls or the stream channel, or seep upward directly into the stream. Areas of significant ground-water discharge may cause the formation of quicksand.

In areas where the geology and ground-water systems are not well known, streamflow data can provide a means of testing estimates of the ground-water system. If the streamflow data do not conform to the estimates, then the geology must be more closely examined. For example, the northwest corner of Ohio is crossed by the Wabash and Fort Wayne moraines between which lies the St. Joseph River. As indicated by the Glacial Map of Ohio (Goldthwait and others, 1961), the St. Joseph basin consists mainly of till. However, low-flow measurements show that the discharge of the river increases more than 14 cfs along its reach in Ohio,

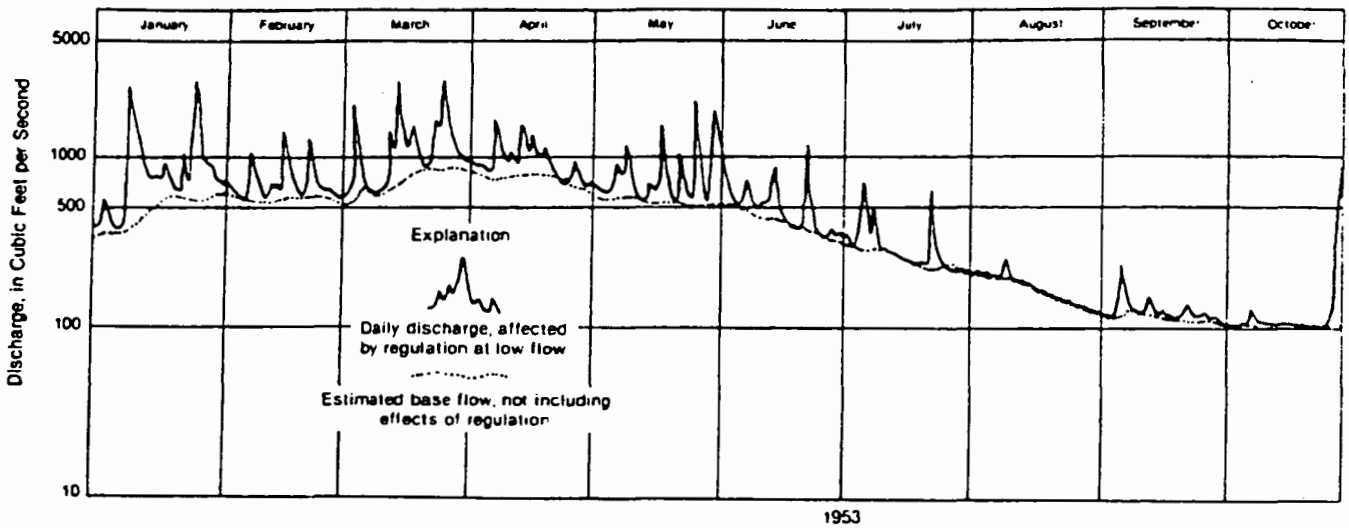


Figure 3-16. Hydrograph of Brandywine Creek at Chadds Ford, Pennsylvania, 1952-53

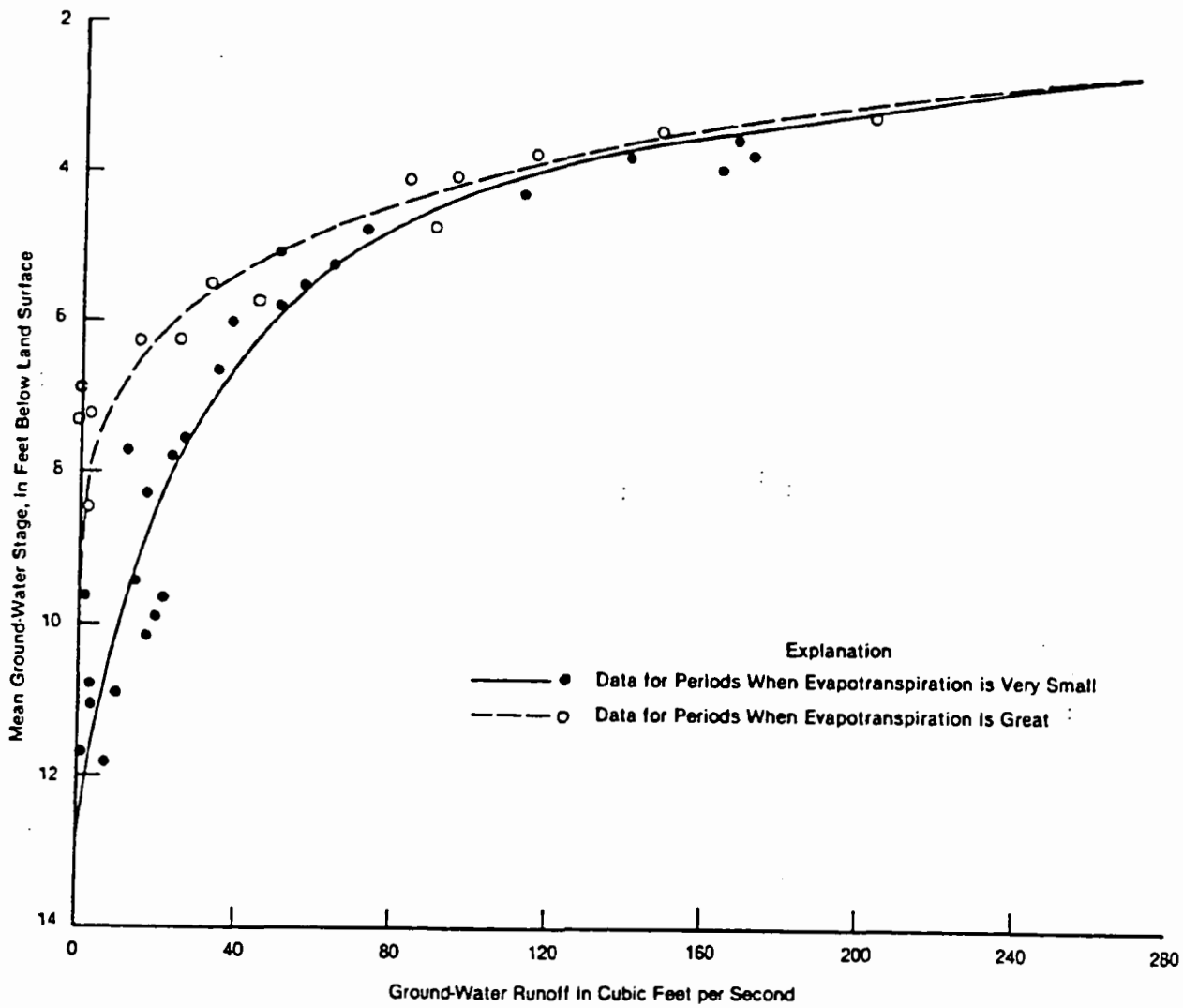


Figure 3-17. Rating Curves of Mean Ground-Water Stage vs. Ground-Water Runoff at Gaging Station 1, Panther Creek Basin, Illinois

indicating that the basin contains a considerable amount of outwash. Thus, hydrologic studies indicate the need for geologic map modification.

On the other hand, geologic maps may indicate reasonable locations for constructing stream gaging stations for hydrologic monitoring networks. The Auglaize River in northwestern Ohio rises from a mass of outwash that lies along the front of the Wabash moraine. The southwest-flowing river breaches the moraine near Wapakoneta and then flows generally north to its confluence with the Maumee River at Defiance. A gaging station is near Ft. Jennings in a till plain area and slightly above a reservoir on the Auglaize. In reality this gage measures, at a single point, the flow resulting as an end product of all causative hydrologic factors upbasin (ground-water runoff, surface runoff, slope, precipitation, use patterns, etc.)—it shows merely inflow into the reservoir. Low-flow measurements, however, indicate that nearly all of the baseflow is derived from outwash along the distal side of the Wabash moraine; there is no grain across the wide till plain downstream. It would seem that the most logical stream gage site for hydrologic evaluations would be at the breach in the Wabash moraine just downstream from the till-outwash contact.

Figure 3-18 shows a number of discharge measurements made in the Scioto River basin, which lies in a glaciated part of central Ohio. The flow measurements themselves are important in that they show the actual discharge, in this case at about 90 percent flow. In this case the discharge is reported as millions of gallons per day, instead of the usual cubic feet per second. The discharge at succeeding downstream sites on the Scioto River are greater than the flow immediately upstream. This shows that the river is gaining and that water is being added to it by ground-water runoff from the adjacent outwash deposits.

A particularly useful method for evaluating streamflow consists of relating the discharge to the size of the drainage basin (cfs/mi^2 or mgd/mi^2 of drainage basin). As Figure 3-18 shows, this technique can be used to relate the flow index (cfs/mi^2 or mgd/mi^2.) to the geology and hydrology of the area. A cursory examination of the data shows that the flow indices can be conveniently separated into three distinctive units. These units are arbitrarily called Unit I (0.01 to 0.020 mgd/mi^2), Unit 2 (0.021 to 0.035 mgd/mi^2) and Unit 3 (0.036 to 0.05 mgd/mi^2). The Olentangy River and Alum and Big Walnut Creeks fall into Unit I, Big Darby and Deer Creeks into Unit 2, and the Scioto River, Walnut Creek, and the lower part of Big Walnut Creek into Unit 3. Notice that even though the latter watercourses fall into Unit 3, the actual discharge ranges widely—from 3.07 to 181 mgd.

Logs of wells drilled along the streams of Unit I show a preponderance of fine-grained material that contains only a few layers of sand and gravel, and wells generally yield less than 3 gpm. Along Big Darby and Deer Creek, however, logs of wells and test holes indicate that several feet of sand and gravel underlie fine-grained alluvial material, the latter of which ranges in thickness from 5 to about 25 feet. Adequately designed and constructed wells that tap these outwash deposits can produce as much as 500 gpm. Glacial outwash, much of it coarse grained, forms an extensive deposit through which the streams and river of Unit 3 flow. The outwash extends from the surface to depths that exceed 200 feet. Industrial wells constructed in these deposits, most of which rely on induced infiltration, can produce more than 1,000 gpm. Formed by combining the seepage data and well yields with a map showing the areal extent of the deposits that are characteristic of each stream valley, the map in Figure 3-18 indicates potential well yields in the area. The potential ground-water yield map relies heavily on streamflow measurement, but nonetheless, provides, with some geologic data, a good first-cut approximation of ground-water availability.

Stream reaches characterized by significant increases in flow due to ground-water runoff, may also have unusual quality characteristics. In northern Ohio the discharge of a small stream, shown in Figure 3-19, that drains into Lake Erie increases over a 3-mile stretch from about 1 to more than 28 cfs and remains relatively constant thereafter. The increase begins at an area of springs where limestone, which has an abundance of solution openings, approaches land surface and actually crops out in the stream bottom. The till-limestone contact declines downstream eventually exceeding 90 feet in depth.

In the upper reaches of a stream, baseflow is provided by ground water that discharges from the adjacent till. Since this water has been in the ground but a short time, the mineral content is low. Electrical conductivity is probably in the range of 640 µmhos. Where streamflow begins to increase significantly, the limestone aquifer provides the largest increment. Furthermore, the bedrock water contains excessive concentrations of dissolved solids (electrical conductivity of about 2,400 µmhos), hardness, and sulfate, and in this stretch calcite precipitates on rocks in the stream channel. The fish population in the upper reaches is quite abundant until the stream reaches the limestone discharge zone. At this point, the population quickly diminishes and remains

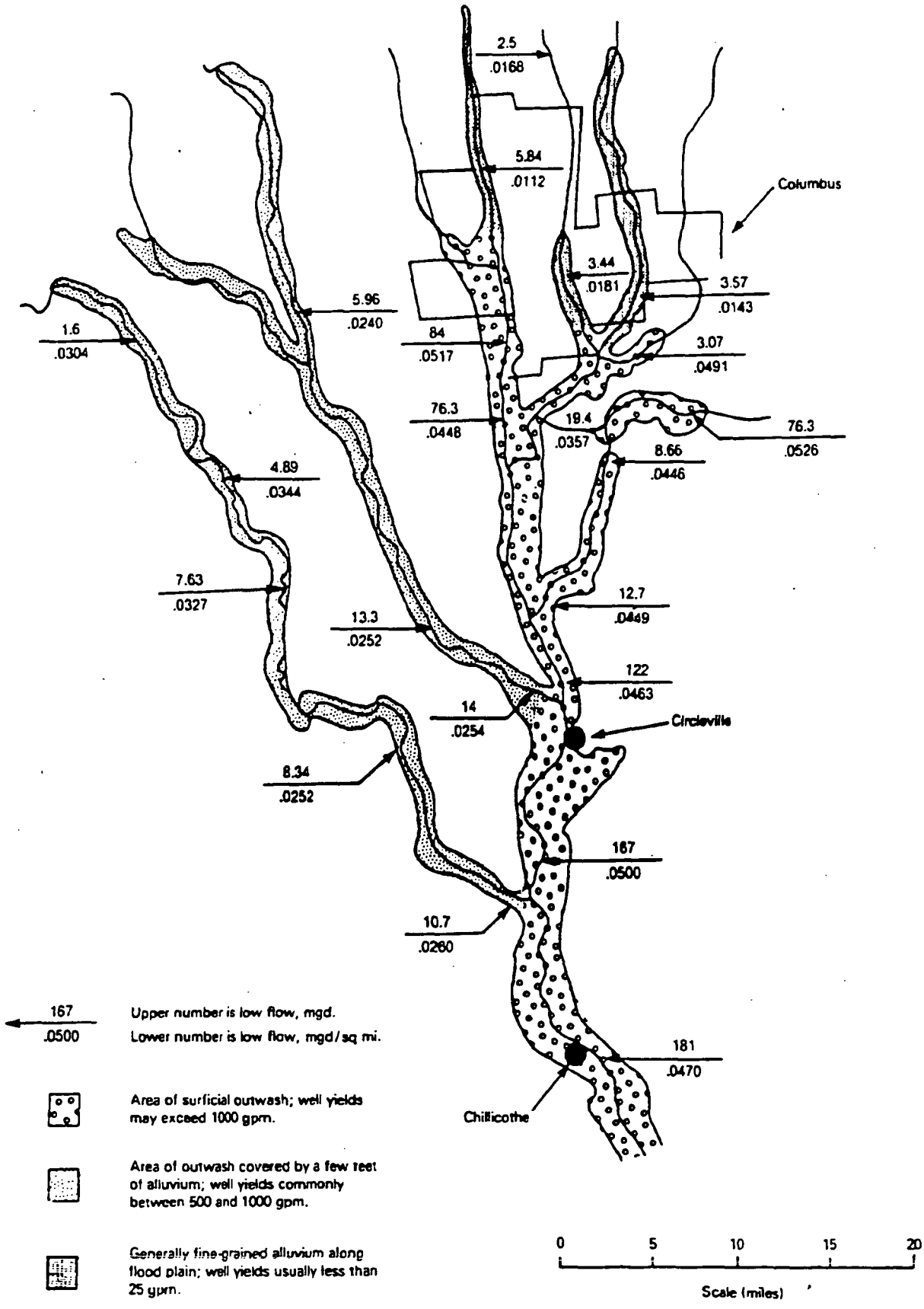


Figure 3-18. Discharge Measurements in the Scioto River Basin, Ohio

Station Number	1	2	3	4	5	6	7
No. species	10	13	10	14	13	13	2
No. Indiv.	980	1527	256	520	184	71	2
D.O.	10.8	9	9.5	11.3	9.6	9.0	8.7
Q	0	0	11.76	17.59	22.28	24.211	27.62
Temp C	21	20.5	13	16.5	16	17.0	17
pH	8.27	8.17	7.76	7.98	8.07	8.13	8.12
Alk	285	253	277	250	253	255	
CO_2	4	1.0	14	6	3	2	
Cond.	638	583	2410	2340	2370	2380	2340

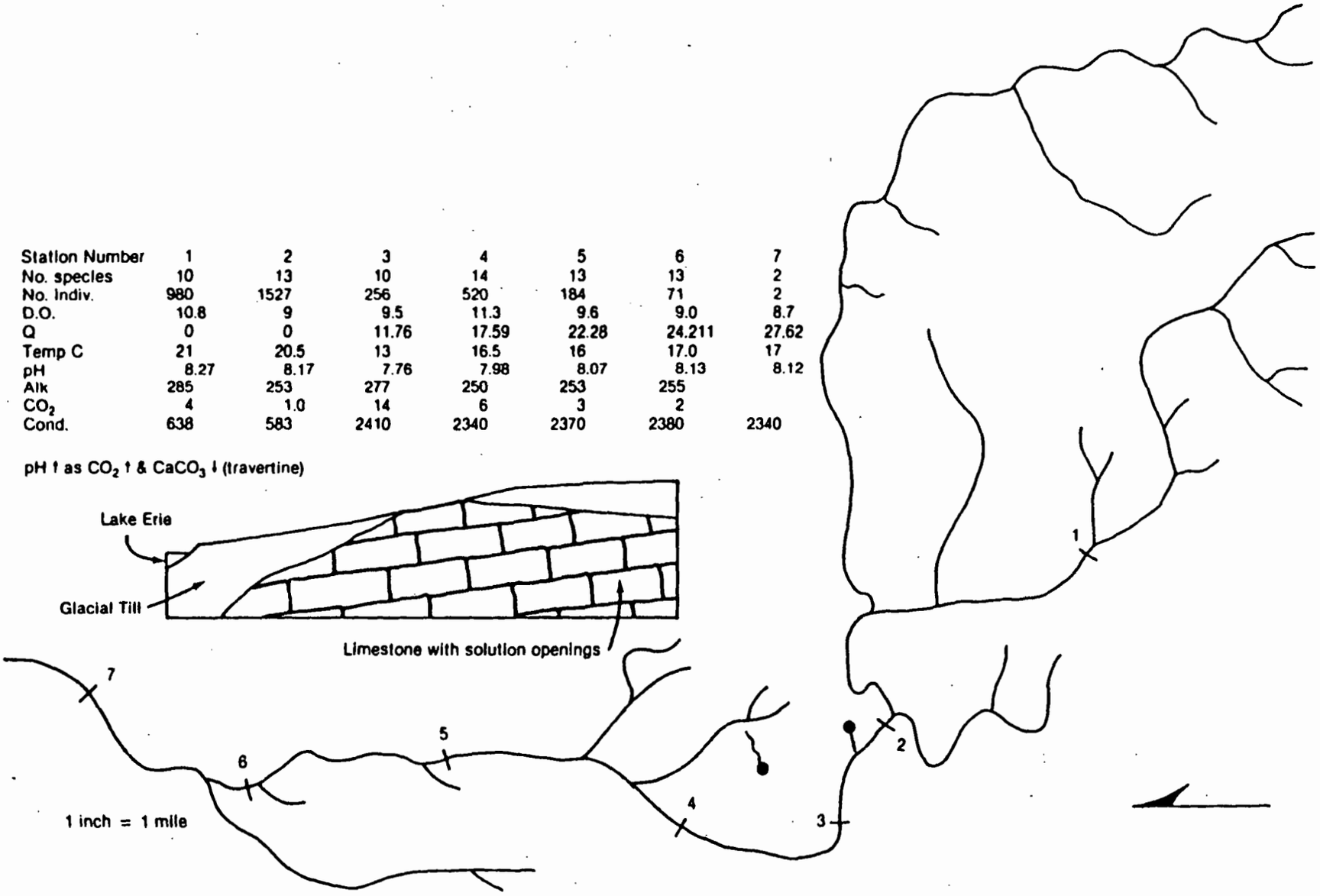


Figure 3-19. Green Creek Drainage Basin (Seneca and Sandusky Countys, Ohio)

in a reduced state throughout the remaining length of the stream. No doubt the reduction in fish population is related to the quality of the water.

In describing the hydrology of Wolf Creek in east-central Iowa, Kunkle (1965) used seepage measurements and water-quality data to determine the amount of ground-water runoff provided by alluvium and limestone. As Figure 3-20 shows, the 325 mi^2 basin is mantled by till and underlain by limestone and shale, but the valley itself contains about 40 feet of permeable alluvium. Well data show that the stream is hydraulically connected with the limestone aquifer along a 5-mile stretch and baseflow is provided by discharge from both the limestone and the alluvium. On either side of this reach the limestone potentiometric surface is below the stream bed.

Measurements were made at three stations during low-flow conditions. The discharge 8 miles upstream from the limestone discharge area was 16.4 cfs, midway along the reach was 29.8 cfs, and 7 miles downstream was 37.0 cfs. Water from the limestone has an average conductivity of 1,330 µmhos, while that from the alluvium and upstream-derived baseflow average 475 µmhos. After mixing, the surface water had a conductivity of 550 µmhos.

Using a slight modification of the equations given previously, it is possible to calculate the amount of ground-water runoff from the limestone in this reach under the given conditions.

$$Ci\,Qi + CaQa + CbQb = CQo \quad (7)$$

$$Qi + Qa + Qb = QO$$

where Qi, Qa, Qb, QO are the discharge from upstream (inflow), from the alluvium, from the limestone, and from the outflow respectively, and Ci, Ca, Cb and C represent the conductivity of the inflow from upstream, from the alluvium, from the limestone, and from the outflow water. Substituting:

$$475\,Qi + 475\,Qa + 1{,}330\,Qb = 20{,}350$$
$$-475Qi - 475\,Qa - 475\,Qb = -17{,}575$$

$$855\,Qb = 2{,}775 \quad \text{and} \quad Qb = 3\text{-}2 \text{ cfs} \quad (8)$$

Thus in this particular stretch, the limestone was providing about 3.2 cfs of the stream's total flow of 37 cfs.

Carrying the analyses a bit further, we could assume that since the limestone provides 3 to 4 cfs during baseflow, wells tapping the limestone in this stretch could provide a like amount without dewatering the system. Since 1 cfs = 450 gpm, wells could produce a total yield of 1,350 to 1,800 gpm. Using a similar approach we could predict the minimum yield of wells

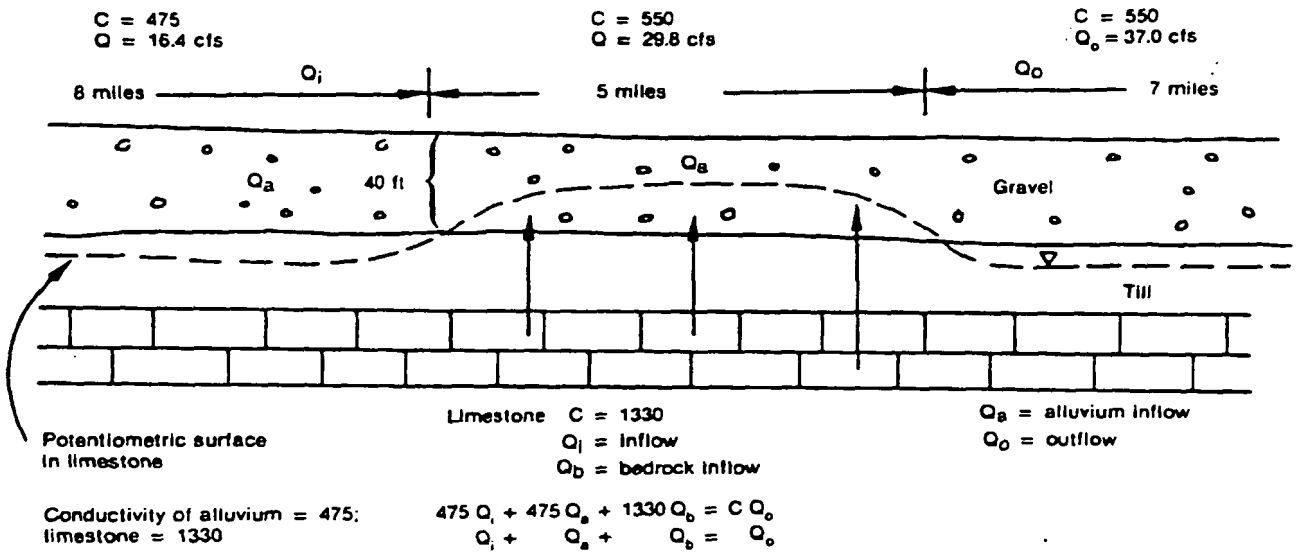


Figure 3-20. Generalized Hydrogeology of Wolf Creek, Iowa

tapping the alluvium, assuming that they would capture only the ground-water runoff.

Temperature Surveys

The temperature of shallow ground water is nearly uniform, reflecting the mean annual daily air temperature of the region. The temperature of shallow ground water ranges from a low of about 37 degrees in the north-central part of the U.S. to more than 77 degrees in southern Florida. Of course, at any particular site the temperature of ground water remains nearly constant. Surface-water temperatures, however, range within wide extremes—freezing in the winter in northern regions and exceeding 100 degrees during hot summer days in the south. Mean monthly stream temperatures during July and August range from a low of 55 in the northwest to more then 85 degrees in the southeast.

During the summer where ground water provides a significant increment of flow, the temperature of a stream in a gaining reach will decline. Conversely, during winter the ground water will be warmer than that on the surface and although ice will normally form, parts of a stream may remain open because of the inflow of the warmer ground water. In central Iowa, for example, winter temperatures commonly drop below zero and ice quickly forms on streams, ponds, and lakes. The ground-water temperature in this region, however, is about 52 degrees and, if a sufficient amount is discharging into a surface-water body, the temperature may remain above 32 degrees and the water will not freeze. In the summer, the relatively cold ground water (52 degrees) mixes with the warm surface water (more than 79 degrees) producing a mixture of water colder than that in non-gaining reaches.

Examination of winter aerial photography may show places where ice is either absent or thin. In the summer it is possible to float down a river, periodically measuring the temperature. Ground-water discharge areas are detected by temperature decrease. A third method of detection is by means of an aircraft-mounted thermal scanner. This sophisticated instrument is able to detect slight differences in temperature and would probably be more accurate than thermometry or low altitude aerial photography.

Flow-Duration Curves

A flow-duration curve shows the frequency of occurrence of various rates of flow. It is a cumulative frequency curve prepared by arranging all discharges of record in order of magnitude and subdividing them according to the percentages of time during which specific flows are equaled or exceeded; all chronologic order or sequence is lost (Cross and Hedges, 1959). Flow-duration curves may be plotted on either probability or semilog paper. In either case, the shape of the curve is an index of natural storage in a basin, including ground water. Since dry-weather flow consists entirely of ground-water runoff, the lower end of the curve indicates the general characteristics of shallow aquifers.

Figure 3-21 shows several flow-duration curves for Ohio streams. During low-flow conditions, the curves for several of the streams, such as the Mad, Hocking, and Scioto Rivers, and Little Beaver Creek, trend toward the horizontal, while Grand River, and Whiteoak and Home Creeks all remain very steep.

Mad River flows through a broad valley that is filled with very permeable sand and gravel. The basin has a large ground-water storage capacity and, consequently, the river maintains a high sustained flow. The Hocking Rver locally contains outwash in and along its floodplain, which provides a substantial amount of ground-water runoff. Above Columbus, the Scioto River crosses thin layers of limestone that crop out along the stream valley, and the adjacent uplands are covered with glacial till; ground-water runoff from this reach is relatively small. Immediately south of Columbus, however, the Scioto Valley widens and is filled with coarse outwash (see figure 3-18). The reason that Mad River has a higher low-flow index than the Scioto River at Chillicothe is because the Mad River receives ground-water runoff throughout its entire length, while the flow of the Scioto River increases significantly only in the southern part of the basin, that is, in the area of outwash south of Columbus.

Whiteoak and Home Creeks originate in bedrock areas where relatively thin alternating layers of sandstone, shale, and limestone crop out along the hill sides. The greater relief in these basins promotes surface runoff and the rocks are not very permeable. Obviously the ground-water storage characteristics and potential yield of these basins are far less than those filled or partly filled with outwash.

Figure 3-22 shows a geologic map of a part of southern Mississippi and northern Louisiana. Notice that gaging stations 1, 2, and 3 record the drainage from the Citronelle Formation, while stations 4, 5, and 6 represent the drainage from the older rocks. Respective flow-duration curves, illustrated in Figure 3-23, show that stations 1 and 2 have high low-flow indices, with station 3 a relatively close third. The high flow-duration indices indicate that the Citronelle Formation has a greater ground-water storage capacity, a higher rate of natural recharge, and presumably would provide larger yields to wells than the underlying strata. This formation

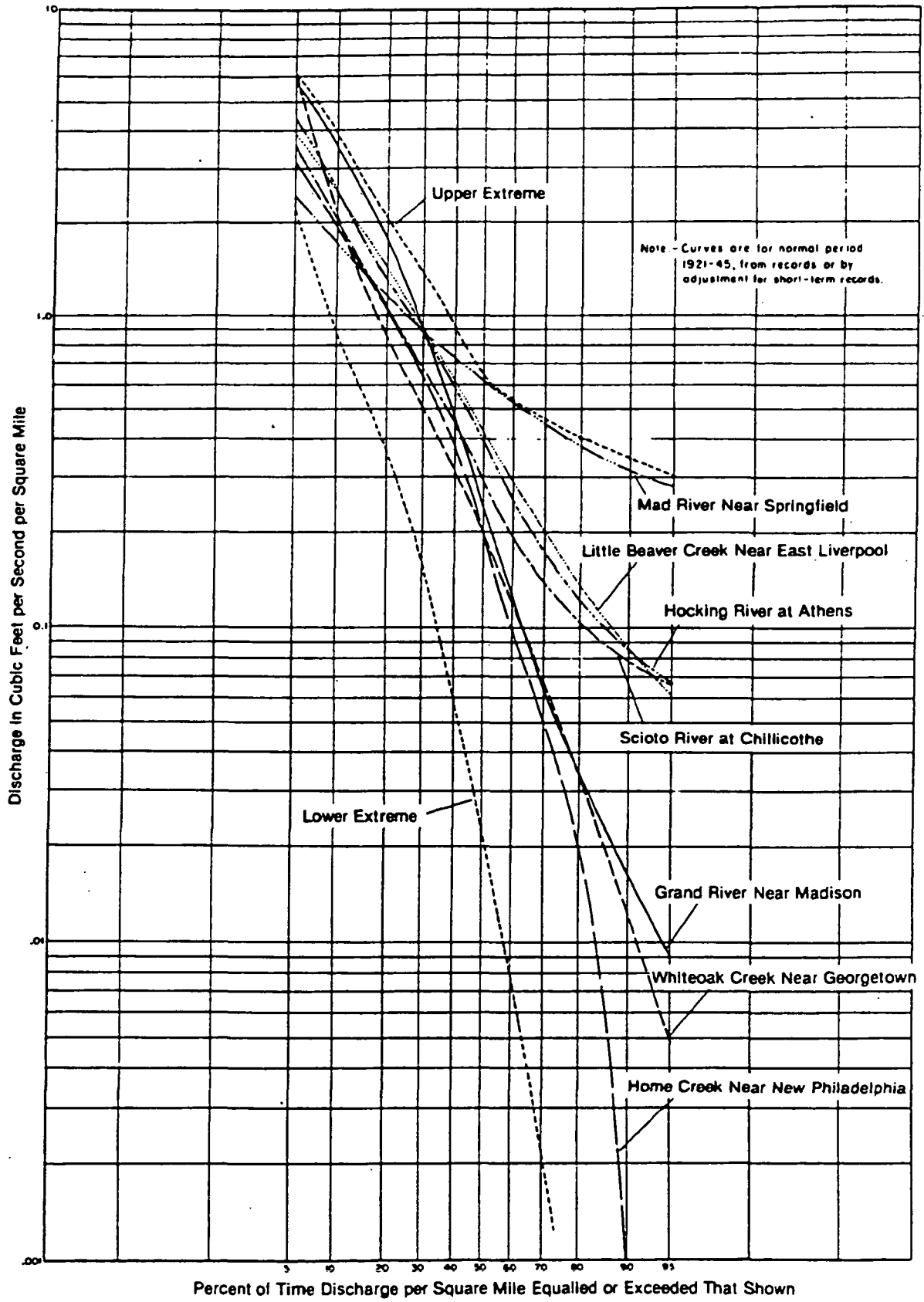


Figure 3-21. Flow-Duration Curves for Selected Ohio Streams

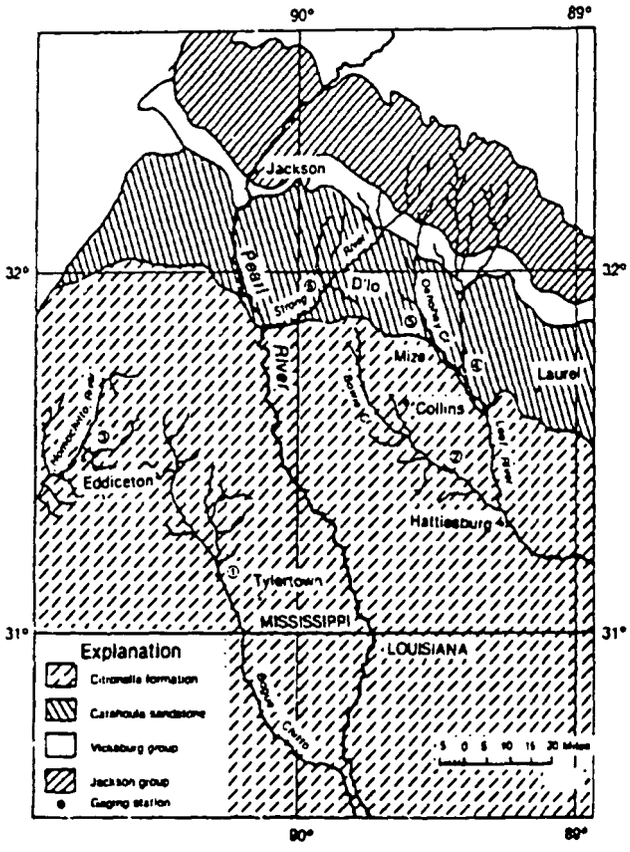


Figure 3-22. Geologic Map of Area in Southern Mississippi Having Approximately Uniform Climate and Altitude

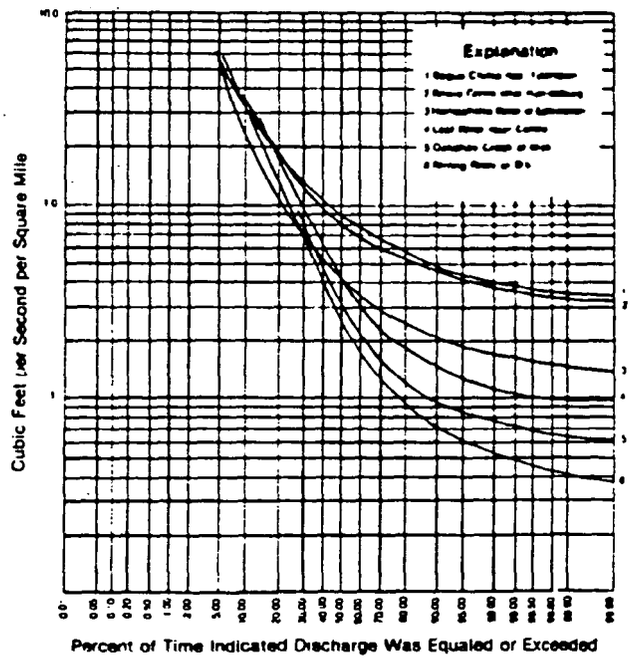


Figure 3-23. Flow-Duration Curves for Selected Mississippi Streams, 1939-48

consists of sand, gravel, and clay, while the other strata are generally composed of finer materials. Thus it would appear that streamflow data can be used as an aid to a better understanding of the permeability and infiltration capacity, as well as facies changes, of geologic units.

Flow Ratios

Walton (1970) reported that grain-size frequency distribution curves are somewhat analogous to flow-duration curves in that their shapes are indicative of water-yielding properties of deposits. He pointed out that a measure of the degree to which all of the grains approach one size, and therefore, the slope of the grain-size frequency distribution curve, is the sorting. One parameter of sorting is obtained by the ratio $(D25/D75)^{1/2}$. Walton modified this equation by replacing the 25 and 75 percent grain-size diameters with the 25 and 75 percent flow. In this case a low ratio is indicative of a permeable basin or one that has a large ground-water storage capacity.

The Q25 and Q75 data are easily obtainable from flow-duration curves. Using the data from Figure 113, Mad River has a flow ratio of 1.58 and the Scioto River's ratio is 2.58, while Home Creek, typifying a basin of low permeability, has the highest ratio which is 5.16.

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Chapter 4

BASIC HYDROGEOLOGY

Introduction

Hydrogeology is the study of ground water, its origin, occurrence, movement, and quality. Ground water is a part of the hydrologic cycle and it reacts in concert with all of the other parts. Therefore, it is essential to have some knowledge of the components, particularly precipitation, infiltration, and the relation between ground water and streams, as well as the impact of the geologic framework on water resources. This chapter provides a brief outline of these topics and interactions.

Precipitation

Much precipitation never reaches the ground; it evaporates in the air and from trees and buildings. That which reaches the land surface is variable in time, areal extent, and intensity. The variability has a direct impact on streamflow, evaporation, transpiration, soil moisture, ground-water recharge, ground water, and ground-water quality. Therefore, precipitation should be examined first in any hydrogeologic study in order to determine how much is available, its probable distribution, and when and under what conditions it is most likely to occur. In addition, a determination of the amount of precipitation is the first step in a water-balance calculation.

Seasonal Variations in Precipitation

Throughout much of the United States, the spring months are most likely to be the wettest owing to the general occurrence of rains of low intensity that often continue for several days at a time. The rain, in combination with springtime snowmelt, will saturate the soil, and streamflow is generally at its peak over a period of several weeks or months. Because the soil is saturated, this is the major period of ground-water recharge. In addition, since much of the total runoff consists of precipitation and snowmelt (surface runoff), streams most likely will contain less dissolved mineral matter than at any other time during the year.

Not uncommonly, the fall also is a wet period, although precipitation is not as great or prolonged as during the spring. Because ground-water recharge can occur over wide areas during spring and fall, one should expect some natural changes in the chemical quality of ground water in surficial or shallow aquifers.

During the winter in northern states, the ground is frozen, largely prohibiting infiltration and ground-water recharge. An early spring thaw coupled with widespread precipitation may lead to severe flooding over large areas.

Types of Precipitation

Precipitation is classified by the conditions that produce a rising column of unsaturated air, which is antecedent to precipitation. The major conditions are convective, orographic, and cyclonic.

Convectional precipitation is the result of uneven heating of the ground, which causes the air to rise, expand, the vapor to condense, and precipitation to occur. Much of the summer precipitation is convective, that is, high intensity, short duration storms that are usually of small areal extent. They often cause flash floods in small basins. Most of the rain does not infiltrate, usually there is a soil-moisture deficiency, and ground-water recharge is likely to be of a local nature. On the other hand, these typically small, local showers can have a significant impact on shallow ground-water quality because some of the water flows quickly through fractures or other macropores, carrying water-soluble compounds leached from the dry soil to the water table. In cases such as these, the quality of shallow ground water may be impacted as certain chemical constituents, and perhaps microbes as well, may increase dramatically within hours (see Chapter 5).

Orographic precipitation is caused by topographic barriers that force the moisture-laden air to rise and cool. This occurs, for example, in the Pacific Northwest,

where precipitation exceeds 100 inches per year, and in Bangladesh, which receives more than 425 inches per year, nearly all of which falls during the monsoon season. In this vast alluvial plain, rainfall commonly averages 106 inches during June for a daily average exceeding 3.5 inches.

Cyclonic precipitation is related to large low pressure systems that require 5 or 6 days to cross the United States from the northwest or Gulf of Mexico. These systems are the major source of winter precipitation. During the spring, summer, and fall, they lead to rainy periods that may last 2 or 3 days or more. They are characterized by low intensity and long duration, and cover a wide area. They probably have a major impact on natural recharge to shallow ground-water systems during the summer and fall, and influence ground-water quality as well.

Recording Precipitation

Precipitation is measured by recording and nonrecording rain gages. Many are located throughout the country but because of their inadequate density, estimates of annual, and particularly summer, precipitation probably are too low. Records can be obtained from Climatological Data, which are published by the National Oceanic and Atmospheric Administration (NOAA). Precipitation is highly variable, both in time and space. The areal extent is evaluated by means of contour or isohyet maps (fig. 4-1).

A rain gage should be installed in the vicinity of a site under investigation in order to know exactly when precipitation occurred, how much fell, and its intensity. Data such as these are essential to the interpretation of hydrographs of both wells and streams, and they provide considerable insight into the causes of fluctuations in shallow ground-water quality.

Infiltration

The variability of streamflow depends on the source of the supply. If the source of streamflow is from surface runoff, the stream will be characterized by short periods of high flow and long periods of low flow or no flow at all. Streams of this type are known as "flashy." If the basin is permeable, there will be little surface runoff and ground water will provide the stream with a high sustained, uniform flow. These streams are known as "steady." Whether a stream is steady or flashy depends on the infiltration of precipitation and snowmelt.

When it rains, some of the water is intercepted by trees or buildings, some is held in low places on the ground (depression storage), some flows over the ground to a stream (surface runoff), some is evaporated, and some infiltrates. Of the water that infiltrates, a part replenishes the soil-moisture deficiency, if any, while the remainder percolates deeper, perhaps becoming ground water. The depletion of soil moisture begins immediately after a rain due to evaporation and transpiration.

Infiltration capacity (f) is the maximum rate at which a soil is capable of absorbing water in a given condition. Several factors control infiltration capacity.

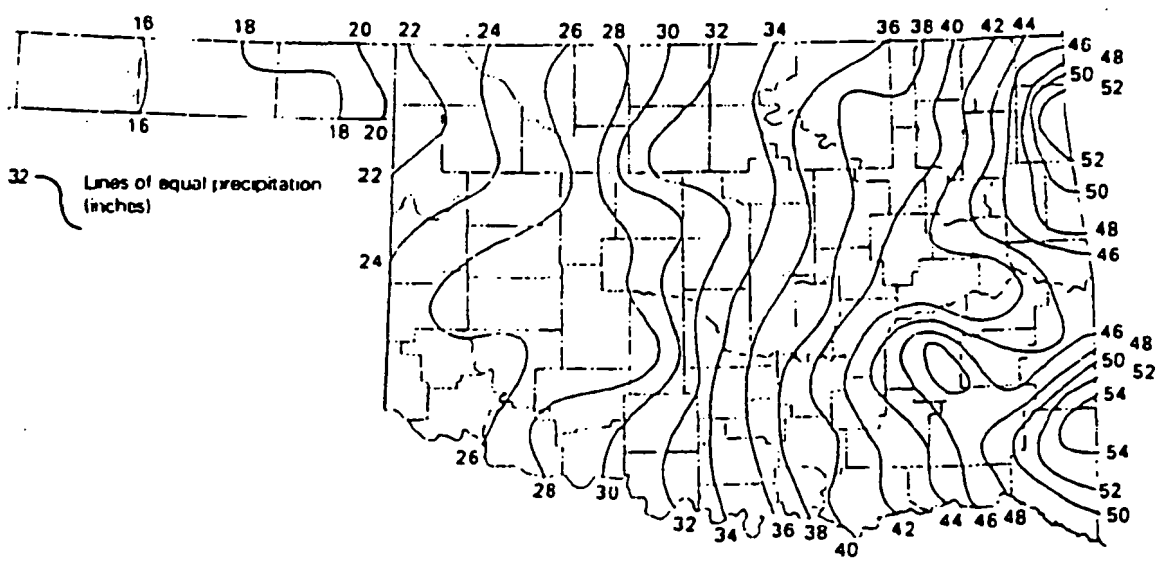

Figure 4-1. Distribution of Annual Average Precipitaiton in Oklahoma, 1970-79 (from Pettyjohn and others, 1983)

o Antecedent rainfall and soil-moisture conditions. Soil moisture fluctuates seasonally, usually being high during winter and spring and low during the summer and fall. If the soil is dry, wetting the top of it will create a strong capillary potential just under the surface, supplementing gravity. When wetted, the clays forming the soil swell, which reduces the infiltration capacity shortly after a rain starts.

o Compaction of the soil due to raindrop impact.

o Inwash of fine material into soil openings, which reduces infiltration capacity. This is especially important if the soil is dry.

o Compaction of the soil by animals, roads, trails, urban development, etc.

o Certain microstructures in the soil will promote infiltration, such as soil structure, openings caused by burrowing animals, insects, decaying rootlets and other vegetative matter, frost heaving, desiccation cracks, and other macropores.

o Vegetative cover, which tends to increase infiltration because it promotes populations of burrowing organisms and retards surface runoff, erosion, and compaction by raindrops.

o Decreasing temperature, which increases water viscosity, reducing infiltration.

o Entrapped air in the unsaturated zone, which tends to reduce infiltration.

o Surface gradient.

Infiltration capacity is usually greater at the start of a rain that follows a dry period, but it decreases rapidly (fig. 4- 2). After several hours it is nearly constant because the soil becomes clogged by particles and swelling clays. A sandy soil, as opposed to a clay-rich soil, may maintain a high infiltration capacity for a considerable time.

As the duration of rainfall increases, infiltration capacity continues to decrease. This is partly due to the increasing resistance to flow as the moisture front moves downward; that is, the resistance is a result of frictional increases due to the increasing length of flow channels and the general decrease in permeability owing to swelling clays. If precipitation is greater than infiltration capacity, surface runoff occurs. If precipitation is less than the infiltration capacity, all moisture is absorbed.

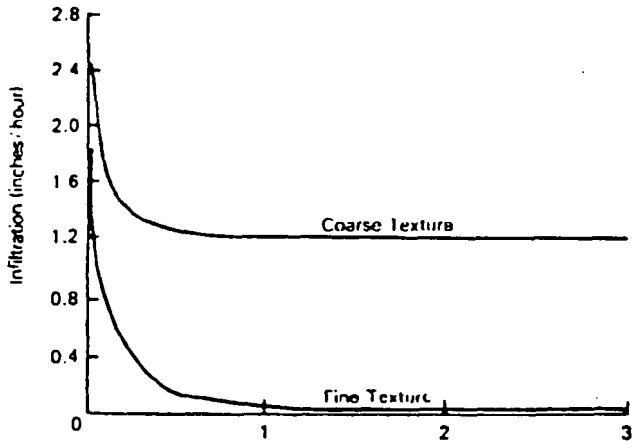

Figure 4-2. Infiltration Capacity Decreases with Time During a Rainfall Event

When a soil has been saturated by water then allowed to drain by gravity, the soil is said to be holding its field capacity of water. (Many investigators are opposed to the use and definition of the term field capacity because it does not account for the rapid flow of water through preferred paths, such as macropores.) Drainage generally requires no more than two or three days and most occurs within one day. A sandy soil has a low field capacity that is reached quickly; clay-rich soils are characterized by a high field capacity that is reached slowly (fig. 4-3).

The water that moves down becomes ground-water recharge. Since recharge occurs even when field

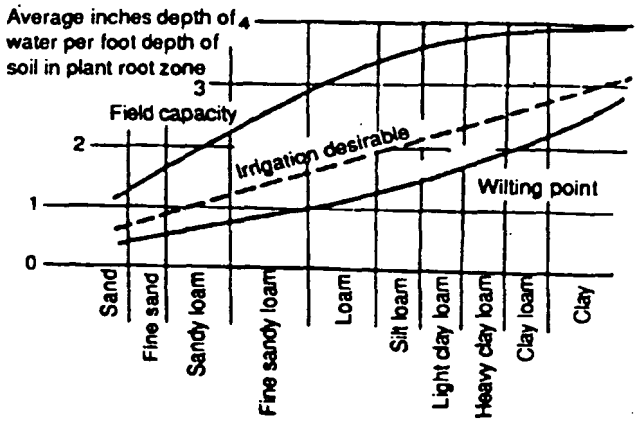

Figure 4-3. Relation Between Grain Size and Field Capacity and Wilting Point

capacity is not reached, there must be a rapid transfer of water through the unsaturated zone. This probably occurs through macropores (Pettyjohn, 1982). Figure 4-4 is a graph of the water table following a storm that provided slightly more than three inches of rain in about an hour in mid-July in north-central Oklahoma. At that time the water table in a very fine-grained aquifer was about 7.5 feet below land surface. Notice that the water table began to rise within a half hour of the start of the rain despite the very low soil-moisture content. The velocity of the infiltrating water through the unsaturated zone was about 15 feet per hour, and this only could have occurred by flow through fractures and other macropores. Clearly field capacity could not have been reached in this short period of time.

Surface Water

Streamflow, runoff, discharge, and yield of drainage basin are all nearly synonymous terms. Channel storage refers to all of the water contained at any instant within the permanent stream channel. Runoff includes all of the water in a stream channel flowing past a cross section; this water may consist of precipitation that falls directly into the channel, surface runoff, ground-water runoff, and effluent.

Although the total quantity of precipitation that falls directly into the channel may be large, it is quite small in comparison to the total flow. Surface runoff, including interflow or stormflow, is the only source of water in ephemeral streams and intermittent streams during part of the year. It is the major cause of flooding. During dry weather, ground-water runoff may account for the entire flow of a stream. It is the major source of water to streams from late summer to winter; at this time streams also are most highly mineralized under natural conditions. Ground water moves slowly to the stream, depending on the hydraulic gradient and permeability; the contribution is slow but the supply is steady. When ground-water runoff provides a stream's entire discharge, the flow is called dry-weather or base flow. Other sources of runoff include the discharge of industrial or municipal effluent or irrigation return flow.

Rates of Flow

Water courses are generally classified on the basis of their length, size of drainage basin, or discharge; the latter is probably the most significant index of a stream's utility in a productive society. Rates of flow generally are reported as cubic feet per second (cfs), millions of gallons per day (mgd), acre-feet per day, month, or year, cfs per square mile of drainage basin (cfs/mi^2), or inches depth on drainage basin per day, month, or year. In the United States, the most common unit of measurement is cfs. The discharge (Q) is determined by measuring the cross-sectional area of the channel (A), in square feet, and the average velocity of the water (v), in feet per second, so that:

$$Q=vA \quad (9)$$

Stream Discharge Measurements and Records

At a stream gaging site the discharge is measured periodically at different rates of flow, which are plotted against the elevation of the water level in the stream (stage or gage-height). This forms a rating curve (fig. 4-5). At a gaging station the stage is continuously

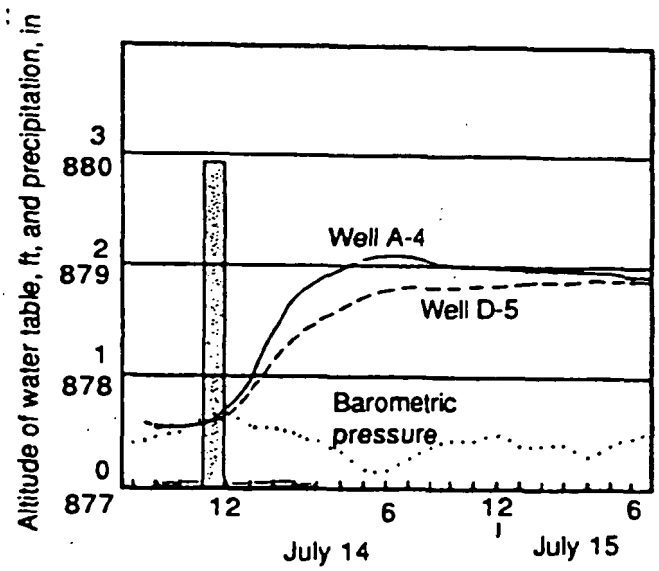

Figure 4-4. Response of the Water Table in a Fine-Grained, Unconfined Aquifer to a High Intensity Rain

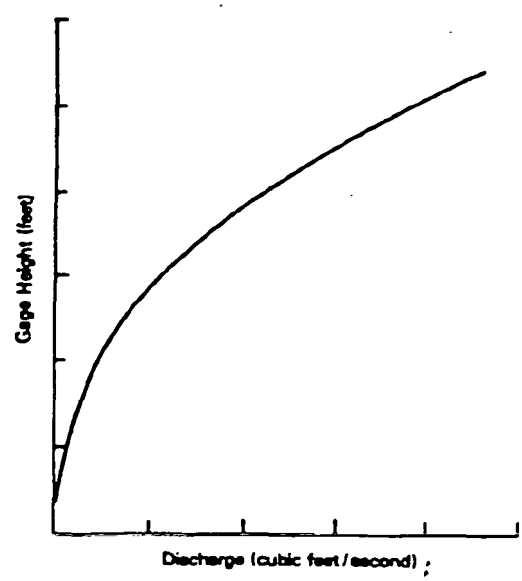

Figure 4-5. A Generalized Stream Stage vs. Discharge Rating Curve

measured and this record is converted, by means of the rating curve, into a discharge hydrograph. The terminology used to describe the various parts of a stream hydrograph are shown in Figure 4-6.

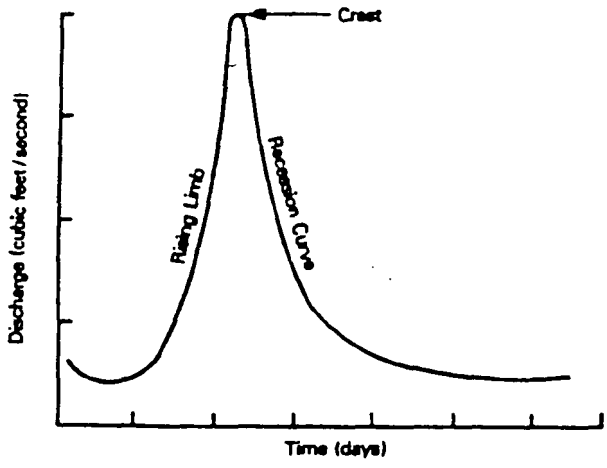

Figure 4-6. Stream Hydrograph Showing Definition of Terms

Discharge, water quality, and ground-water level records are published annually by the U. S. Geological Survey for each state. An example of the annual record of a stream is shown in Figure 4-7. Notice that these data are reported in "water years." The water year is designated by the calendar year in which it ends, which includes 9 of the 12 months. Thus, water year 1990 extends from October 1, 1989, to September 30, 1990.

The Relation between Ground Water and Surface Water

There are many tools for learning about ground water without basing estimates on the ground-water system itself, and one approach is the use of streamflow data (See Chapter 3). Analyses of streamflow data permit an evaluation of the basin geology, permeability, the amount of ground-water contribution, and the major areas of discharge. In addition, if chemical quality data are available or collected for a specific stream, they can be used to determine background concentrations of various parameters and locate areas of ground-water contamination as well.

Ground Water

The greatest difficulty in working with ground water is that it is hidden from view, cannot be adequately tested, and occurs in a complex environment. On the other hand, the general principles governing ground-water occurrence, movement, and quality are quite well known, which permits the investigator to develop a reasonable degree of confidence in his predictions. The experienced investigator is well aware, however, that these predictions are only estimates of the manner in which the system functions. Ground-water hydrology is not an exact science, but it is possible to develop a good understanding of a particular system if one pays attention to fundamental principles.

The Water Table

Water under the surface of the ground occurs in two zones, an upper unsaturated zone and the deeper saturated zone (fig. 4-8). The boundary between the two zones is the water table. In the unsaturated zone, most of the open spaces are filled with air, but water occurs as soil moisture and in a capillary fringe that extends upward from the water table. Water in the unsaturated zone is under a negative hydraulic pressure, that is, it is less than atmospheric. Ground water occurs below the water table and all of the pores and other openings are filled with fluid that is under pressure greater than atmospheric.

In a general way, the water table conforms to the surface topography, but it lies at a greater depth under hills than it does under valleys (fig. 4-8). In general, in humid and semiarid regions the water table lies at depths ranging from 0 to about 20 feet or so, but its depth exceeds hundreds of feet in some desert environments.

The elevation and configuration of the water table must be determined with care, and many such measurements have been incorrectly taken. The position of the water table can be determined from the water level in swamps, flooded excavations (abandoned gravel pits, highway borrow pits, etc.), sumps in basements, lakes, ponds, streams, and shallow wells. In some cases there may be no water table at all or it may be seasonal.

Measurement of the water level in drilled wells, particularly if they are of various depths, will more likely reflect the pressure head of one or more aquifers that are confined than the actual water table.

Figure 4-9 illustrates the difference in water levels in several wells, each of which is of a different depth. Purposely no scale has been applied to the sketch because the drawing is relative. That is, the same principle exists regardless of scale, and individual zones could be only a few inches or feet thick, or they might exceed several tens of feet. Notice that each well has a different water level but the water table can be determined only in Well 2. Wells 1, and 3-5, which tap confined aquifers, are deeper and each is screened in

07176500 BIRD CREEK NEAR AVANT, OK

LOCATION.--Lat 36°29'12", long 96°03'50", in SW 1/4 NW 1/4 sec.7 (revised), T.23 N., R.12 E., Osage County, Hydrologic Unit 11070107, 150 ft upstream from county road bridge at Avant, 1.5 mi upstream from Candy Creek, and at mile 54.2.

DRAINAGE AREA.--364 mi^2.

PERIOD OF RECORD.--August 1945 to current year.

GAGE.--Water-stage recorder. Datum of gage is 651.28 ft above National Geodetic Vertical Datum of 1929.

REMARKS.--Records fair. Several unpublished observations of water temperature, specific conductance, and pH were made during the year and are available at the District Office. Flow slightly regulated since 1958 by Bluestem Lake. Some regulation since March 1977 by Birch Lake (station 07176460), located on Birch Creek, 12.1 mi upstream. Small diversions upstream for municipal water supply for the cities of Pawhuska and Barnsdall.

AVERAGE DISCHARGE.--43 years, 221 ft^3/s, 160,100 acre-ft/yr.

EXTREMES FOR PERIOD OF RECORD.--Maximum discharge, 32,400 ft^3/s, Oct. 2, 1959, gage height, 31.40 ft; maximum gage height, 32.03 ft, Mar. 11, 1974; no flow at times.

EXTREMES FOR CURRENT YEAR.--Peak discharges greater than base discharge of 6,000 ft^3/s and maximum (*):

Date	Time	Discharge (ft^3/s)	Gage Height (ft)	Date	Time	Discharge (ft^3/s)	Gage Height (ft)
Nov. 24	1615	8,270	11.25	Mar. 3	1215	6,390	8.77
Dec. 19	2115	9,820	13.65	Apr. 1	2045	*16,200	*23.47

Minimum daily discharge, 3.1 ft^3/s, Oct. 17, 18.

DISCHARGE, CUBIC FEET PER SECOND, WATER YEAR OCTOBER 1987 TO SEPTEMBER 1988
MEAN VALUES

DAY	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP
1	416	14	157	457	101	69	13200	146	12	8.8	36	16
2	332	13	131	341	101	522	7040	144	142	9.6	30	17
3	303	12	115	297	120	4680	972	137	52	20	27	20
4	285	12	92	257	81	1460	633	133	55	22	21	17
5	277	12	45	163	118	1860	813	129	46	24	20	16
6	268	10	40	138	107	1890	711	125	26	24	16	16
7	234	10	36	132	105	1520	662	120	23	22	16	14
8	132	10	29	120	106	898	629	120	21	18	16	11
9	21	10	26	81	114	638	608	120	19	16	16	9.8
10	7.0	10	23	120	116	534	2180	108	17	16	16	9.6
11	6.3	10	20	120	111	478	1090	91	16	22	16	9.6
12	6.2	9.8	15	120	104	349	645	54	16	112	15	9.6
13	5.8	9.6	14	142	102	322	404	35	16	62	15	9.6
14	5.6	9.6	20	165	62	268	311	28	16	34	15	9.6
15	4.8	80	30	165	43	156	252	22	16	26	15	9.6
16	3.8	93	47	923	41	137	152	21	15	22	15	146
17	3.1	94	32	1340	41	170	145	23	12	34	15	208
18	3.1	51	141	519	42	555	2570	24	11	26	15	1410
19	3.3	36	5400	1020	195	405	831	24	11	234	15	1540
20	3.6	26	4770	785	212	333	484	24	11	158	15	450
21	4.1	16	943	397	147	272	383	24	11	50	15	219
22	4.7	10	778	344	117	232	317	24	11	41	15	160
23	8.5	8.2	595	253	96	209	186	29	11	31	15	327
24	9.6	3370	525	221	80	196	157	33	11	27	15	268
25	8.3	1450	876	191	73	188	200	59	11	26	16	168
26	7.6	336	1110	129	70	175	220	41	11	24	16	108
27	7.4	207	2160	91	69	170	200	29	10	22	16	80
28	8.4	227	1540	120	69	166	166	20	10	27	16	66
29	9.6	240	758	120	69	553	151	18	10	277	16	61
30	9.6	200	752	146	---	528	146	17	9.6	123	16	54
31	13	---	938	120	---	1170	---	15	---	56	16	---
TOTAL	2411.4	6596.2	22178	9737	2792	21103	36408	1937	658.6	1614.4	537	5459.4
MEAN	77.8	220	715	314	96.3	681	1214	62.5	22.0	52.1	17.3	182
MAX	416	3370	5400	1540	212	4680	13200	146	142	277	36	1540
MIN	3.1	8.2	14	81	41	69	145	15	9.6	8.8	15	9.6
AC-FT	4780	13080	43990	19310	5540	41860	72220	3840	1310	3200	1070	10830

CAL YR 1987 TOTAL 123598.2 MEAN 339 MAX 7340 MIN 3.1 AC-FT 245200
WTR YR 1988 TOTAL 111432.0 MEAN 304 MAX 13200 MIN 3.1 AC-FT 221000

Figure 4-7. Stream Discharge Record for Bird Creek near Avant, Oklahoma (From U.S. Geological Survey Water Resources Data for Oklahoma Water Year 1987, p. 90)

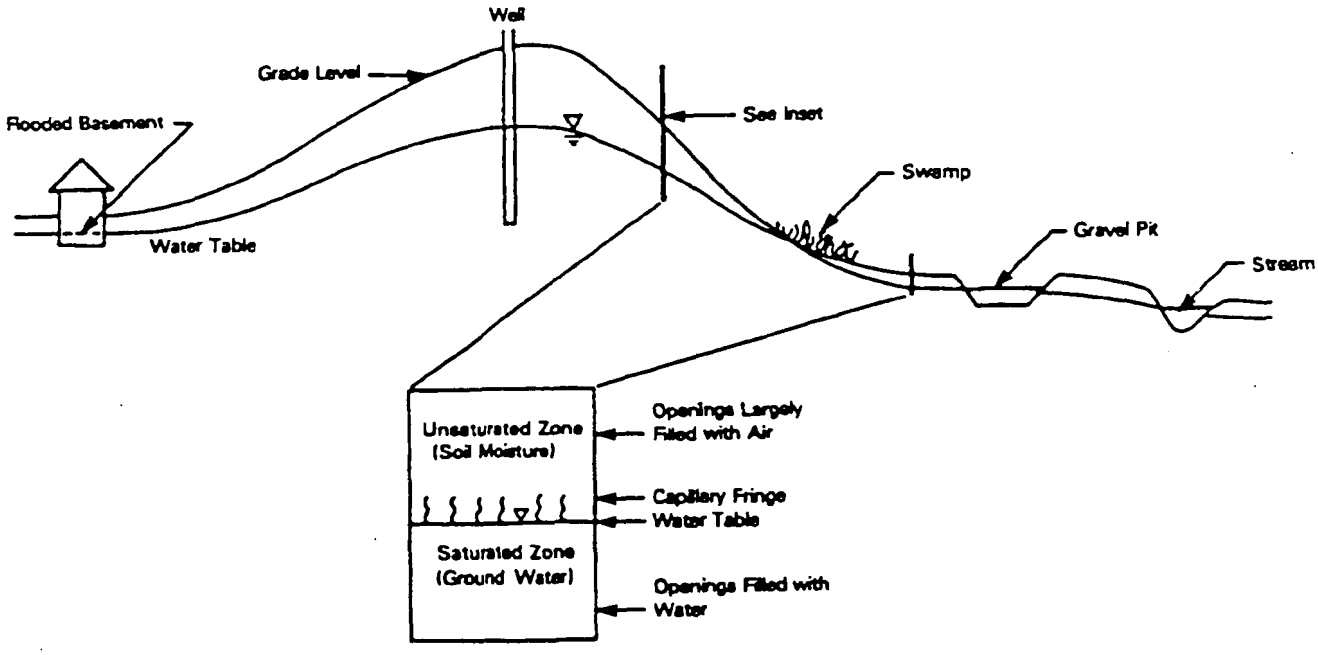

Figure 4-8. The Water Table Generally Conforms to the Surface Topography

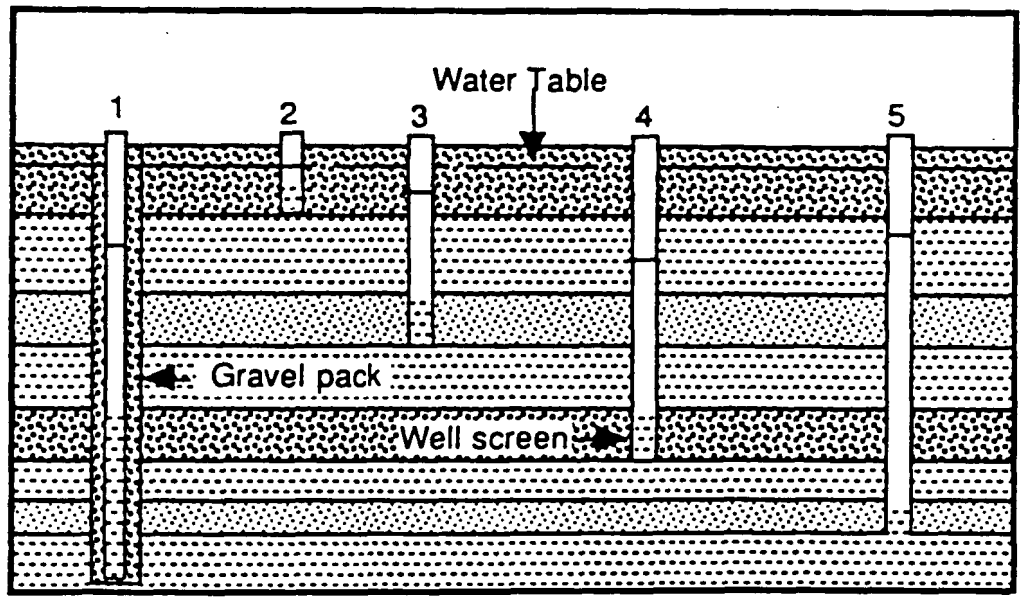

Figure 4-9. The Water Level in a Well Indicates the Pressure that Exists in the Aquifer that it Taps

a particular permeable zone that is bounded above and below by less permeable confining units. The water level in each well reflects the pressure that exists in the individual zone that is tapped by the well. A different situation occurs in Well 1, because the gravel pack surrounding the well casing and screen provides a high permeability conduit that connects all of the water-bearing zones. The water level in Well 1 is a composite of the pressure in all of the zones.

Because hydraulic head generally differs with depth, it is exceedingly important to pay attention to well depth and construction details when preparing water-level maps and determining hydraulic gradient and flow direction.

For example, water-level measurements in wells 1 and 2 or 5 and 4 would suggest a gradient to the left, but wells 2 and 3 or 3 and 4 would allude a gradient to the right of the drawing. In addition, the apparent slope of the gradient would depend on the wells being measured.

Accurately determining the position of the water table is important not only because of the need to determine the direction and magnitude of the hydraulic gradient, but, in addition, the thickness, permeability, and composition of the unsaturated zone exert a major control on ground-water recharge and the movement of contaminants from land surface to an underlying aquifer. Attempting to ascertain the position of the water table by measuring the water level in drilled wells nearly always will incorrectly suggest an unsaturated zone that is substantially thicker than actually is the case, and thus may provide a false sense of security.

Ground water has many origins, however, all fresh ground water originated from precipitation that infiltrated. Magmatic or juvenile water is "new" water that has been released from molten igneous rocks. The steam that is so commonly given off during volcanic eruptions is probably not magmatic, but rather shallow ground water heated by the molten magma. Connate water is defined as that entrapped within sediments when they were deposited. Ground water, however, is dynamic and probably in only rare circumstances does connate water meet this definition. Rather, the brines that underlie all or nearly all fresh ground water have changed substantially through time because of chemical reactions with the geologic framework.

Aquifers and Confining Units

In the subsurface, rocks serve either as confining units or aquifers. A confining unit or aquitard is characterized by low permeability that does not readily permit water to pass through it. Confining units do, however, store large quantities of water. Examples include shale, clay, and silt. An aquifer has sufficient permeability to permit water to flow through it with relative ease and, therefore, it will provide a usable quantity to a well or spring.

Water occurs in aquifers under two different conditions—unconfined and confined (fig. 4-10). An unconfined or water-table aquifer has a free water surface that rises and falls in response to differences between recharge and discharge. A confined or artesian aquifer is overlain and underlain by aquitards and the water is under sufficient pressure to rise above the base of the confining bed, if it is perforated. In some cases, the water is under sufficient pressure to rise above land surface. These are called flowing or artesian wells. The water level in an unconfined aquifer is referred to as the water table; in confined aquifers the water level is called the potentiometric surface.

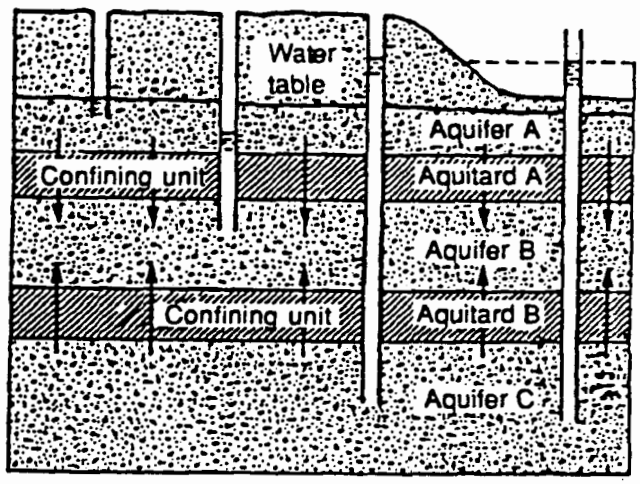

Figure 4-10. Aquifer A is Unconfined and Aquifers B and C are Confined, but Water May Leak Through Confining Units to Recharge Adjacent Water-Bearing Zones

Water will arrive at some point in an aquifer through one or several means. The major source is direct infiltration of precipitation, which occurs nearly everywhere. Where the water table lies below a stream or canal, the surface water will infiltrate. This source is important part of the year in some places (intermittent streams) and is a continuous source in others (ephemeral or losing streams). Interaquifer leakage, or flow from one aquifer to another, is probably the most significant source in deeper, confined aquifers. Likewise, leakage from aquitards is very important where pumping from adjacent aquifers has lowered the head or potentiometric surface sufficiently for leakage to occur. Underflow, which is the normal movement

of water through an aquifer, also will transmit ground water to a specific point. Additionally, water can reach an aquifer through artificial means, such as leakage from ponds, pits, and lagoons, from sewer lines, and from dry wells, among others.

An aquifer serves two functions, one as a conduit through which flows occurs, and the other as a storage reservoir. This is accomplished by means of openings in the rock. The openings include those between individual grains and those present in joints, fractures, tunnels, and solution openings. There are also artificial openings, such as engineering works, abandoned wells, and mines. The openings are primary if they were formed at the time the rock was deposited and secondary if they developed after lithification. Examples of the latter include fractures and solution openings.

Porosity and Hydraulic Conductivity

Porosity, expressed as a percentage or decimal fraction, is the ratio between the openings and the total rock volume. It defines the amount of water a saturated rock volume can store. If a unit volume of saturated rock is allowed to drain by gravity, not all of the water it contains will be released. The volume drained is the specific yield, a percentage, and the volume retained is the specific retention. Related to the attraction between water and earth materials, specific retention generally increases as sorting and grain size decrease. Porosity determines the total volume of water that a rock unit can store, while specific yield defines the amount that is available to wells. Porosity is equal to the sum of specific yield and specific retention. Typical values for various rock types are listed in Table 4-1.

Permeability (P) is used in a qualitative sense, while hydraulic conductivity (K) is a quantitative term They are expressed in a variety of units gpd/ft^2 (gallons per day per square foot) will be used in this section; see Table 4-2 for conversion factors) and both refer to the ease with which water can pass through a rock unit. It is the hydraulic conductivity that allows an aquifer to serve as a conduit. Hydraulic conductivity values range widely from one rock type to another and even within the same rock. It is related to grain size, sorting, cementation, and the amount of secondary openings, among others. Typical ranges in values of hydraulic conductivity for most common water-bearing rocks are shown in Table 4-3 and Figure 4-11.

Material	Porosity	Specific Yield (% by vol)	Specific Retention
Soil	55	40	15
Clay	50	2	48
Sand	25	22	3
Gravel	20	19	1
Limestone	20	18	2
Sandstone, semiconsolidated	11	6	5
Granite	0.1	0.09	0.01
Basalt, young	11	8	3

Table 4-1. Average Porosity, Specific Yield, and Specific Retention Values for Selected Earth Materials

Those rocks or aquifers in which the hydraulic conductivity is nearly uniform are called homogeneous and those in which it is variable are heterogeneous or nonhomogeneous. Hydraulic conductivity also can vary horizontally in which case the aquifer is anisotropic. If uniform in all directions, which is rare, it is isotropic. The fact that both unconsolidated and consolidated sedimentary strata are deposited in horizontal units is the reason that hydraulic conductivity is generally greater horizontally than vertically, commonly by several orders of magnitude.

Hydraulic Gradient

The hydraulic gradient (I) is the slope of the water table or potentiometric surface, that is, the change in water level per unit of distance along the direction of maximum head decrease. It is determined by measuring the water level in several wells. The water level in a well (fig. 4-12), usually expressed as feet above sea level, is the total head (ht), which consists of elevation head (z) and pressure head (hp).

$$ht=z+hp \quad (10)$$

The hydraulic gradient is the driving force that causes ground water to move in the direction of maximum decreasing total head. It is generally expressed in consistent units, such as feet per foot. For example, if the difference in water level in two wells 1000 feet apart is 2 feet, the gradient is 2/1,000 or 0.002 (fig. 4-13). Since the water table or potentiometric surface is a plane, the direction of ground-water movement and the hydraulic gradient must be determined by information from three wells (fig. 4-14). The wells must tap the same aquifer, and should be of similar depth and screened interval.

Using the three point method, water-level elevations are determined for each well, and their locations are plotted on a map. Lines are drawn to connect the wells in such a way that a triangle is formed. Using the elevations of the end points, each line is divided into a number of equal elevation segments. Selecting points of equal elevation on two of the lines, equipotential or potentiometric contours are drawn through the points. A flow line is then constructed so that it intersects the

Hydraulic Conductivity (K)

Meters per day ($m\ d^{-1}$)	Centimeters per second ($cm\ s^{-1}$)	Feet per day ($ft\ d^{-1}$)	Gallons per day per square foot ($gal\ d^{-1}\ ft^{-2}$)
1	1.16×10^{-3}	3.28	2.45×10^{1}
8.64×10^{2}	1	2.83×10^{3}	2.12×10^{4}
3.05×10^{-1}	3.53×10^{-4}	1	7.48
4.1×10^{-2}	4.73×10^{-5}	1.34×10^{-1}	1

Transmissivity (T)

Square meters per day ($m^2\ d^{-1}$)	Square feet per day ($ft^2\ d^{-1}$)	Gallons per day per foot ($gal\ d^{-1}\ ft^{-1}$)
1	10.76	80.5
.0929	1	7.48
.0124	.134	1

Recharge Rates

Unit depth per year	Volume		
	($m^3\ d^{-1}\ km^{-2}$)	($ft^3\ d^{-1}\ mi^{-2}$)	($gal\ d^{-1}\ mi^{-2}$)
(in millimeters)	2.7	251	1,874
(in inches)	70	6,365	47,748

Flow Rates

($m^3\ s^{-1}$)	($m^3\ min^{-1}$)	($ft^3\ s^{-1}$)	($ft^3\ min^{-1}$)	($gal\ min^{-1}$)
1	60.	35.3	2,120	15,800
.0167	1	.588	35.3	264
.0283	1.70	1	60	449
.000472	.0283	.0167	1	7.48
.000063	.00379	.0023	.134	1

Table 4-2. Conversion Factors for Hydraulic Conductivity, Recharge Rates, and Flow Rates

Material	Hydraulic conductivity (rounded values) (ft/day)	[(gal/day)/ft²]	(meters/day)
Coarse sand	200	1500	60
Medium sand	130	1000	40
Silt	1	5	0.2
Clay	0.001	0.01	0.0004
Limestone (Castle Hayne)	300	2000	80
Saprolite	5	50	2
Granite and gneiss	5	50	2
Slate	3	25	1

Table 4-3. Hydraulic Conductivity of Selected Rocks (Heath, 1980)

equipotential contours at a right angle. Ground water flows in the direction of decreasing head or water level.

Potentiometric Surface Maps and Flow Nets

Potentiometric surface or water-level maps are an essential part of any ground-water investigation because they indicate the direction in which ground water is moving and provide an estimate of the hydraulic gradient, which controls velocity. A potentiometric surface map is a graphical representation of the hydraulic gradient. They are prepared by plotting water-level measurements on a base map and then contouring them. The map should be drawn so that it actually reflects the hydrogeological conditions. Sample map is shown in Figure 4-15.

The water-level contours are called potentiometric lines, indicating that the water has the potential to rise to that

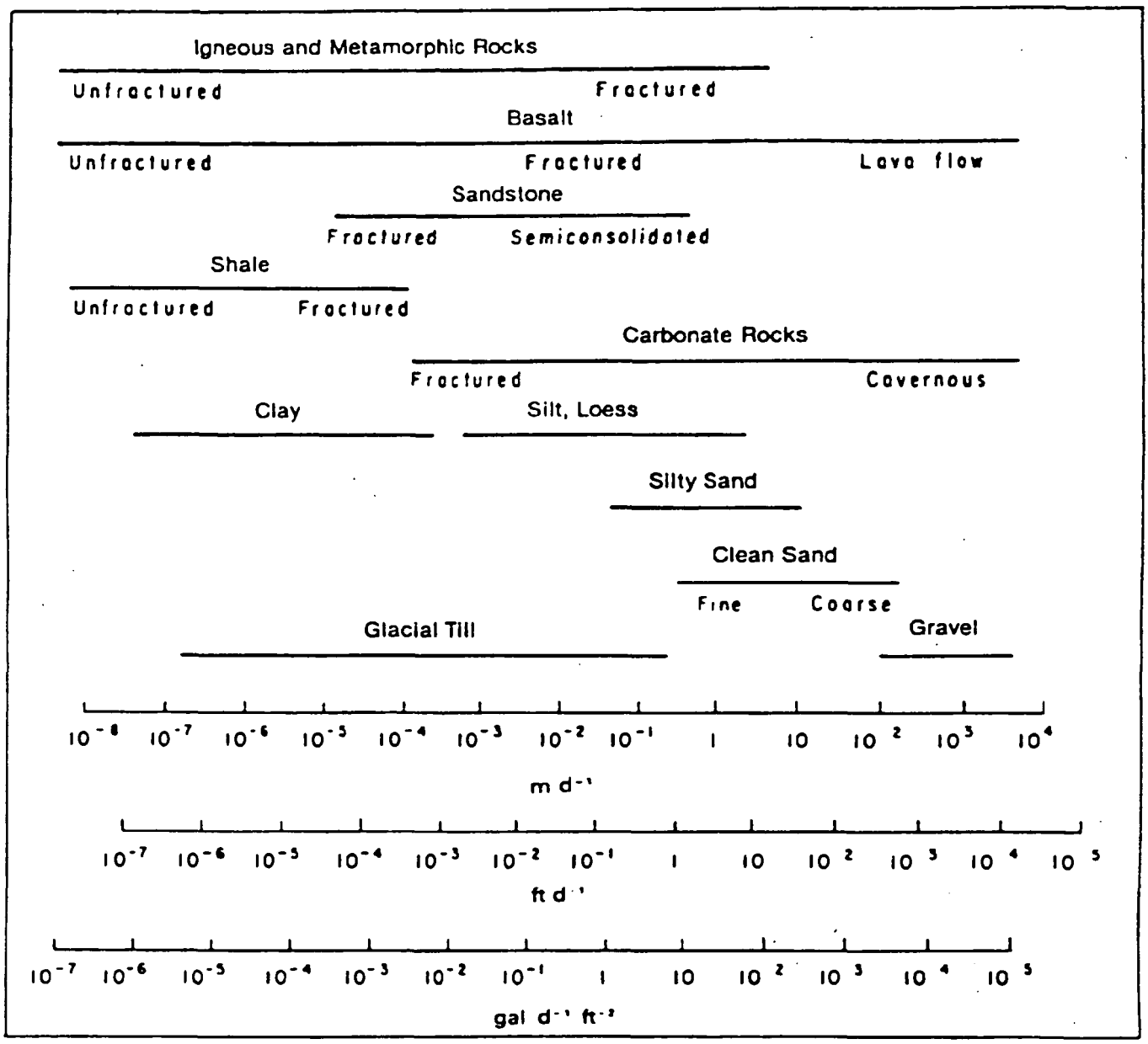

Figure 4-11. General Range in Hydraulic Conductivity for Various Rock Types

elevation. In the case of a confined aquifer, however, the water may have the potential to rise to a certain elevation, but it cannot actually do so until the confining unit is perforated by a well. Therefore, a potentiometric surface map of a confined aquifer represents an imaginary surface.

A potentiometric surface map can be developed into a flow net by constructing flow lines that intersect the equipotential lines at right angles. Flow lines are imaginary paths that would be followed by particles of water as they move through the aquifer. Although there is an infinite number of both equipotential and flow lines, the former are constructed with uniform differences in elevation between them and the latter so that they form, in combination with equipotential lines, a series of squares. A carefully prepared flow net in conjunction with Darcy's Law (discussed below) can be used to estimate the quantity of water flowing through an area.

A plan view flow net of an unconfined aquifer is shown in Figure 4-16. Notice that all of the water-table contours point upstream, and that the flow lines originate in the central part of the interstream divide (recharge area) and terminate at the streams (discharge line). A vertical

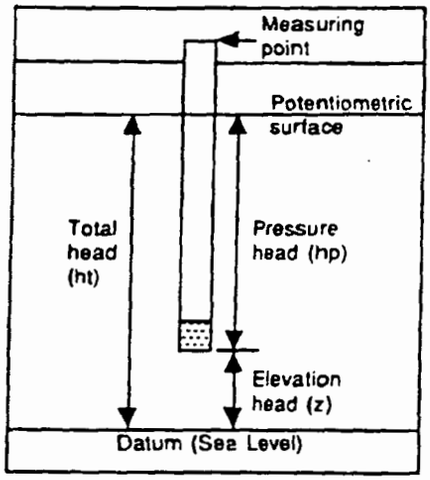

Figure 4-12. Relationship Between Total Head, Pressure Head, and Elevation Head

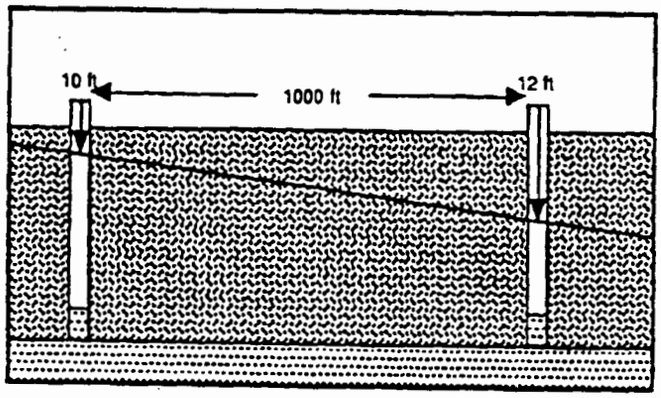

Figure 4-13. The Hydraulic Gradient Is Defined by the Decline in Water Level in Wells a Defined Distance Apart

flow net, representing the line A-A' in Figure 4-16, is shown in Figure 4-17. In this case, the curved flow lines illustrate that the ground water is moving in the same direction but not in the same manner as implied from the plan view.

A flow net that represents a different hydrologic situation is shown in Figure 4-18. In this case, the streams are gaining in the upper part of the map, while below their confluence the water-table contours begin to point downstream. This indicates that the water table is below the channel, the stream is losing water to the subsurface, and the flow lines are diverging from the line source of recharge. A vertical flow net is shown in Figure 4-19.

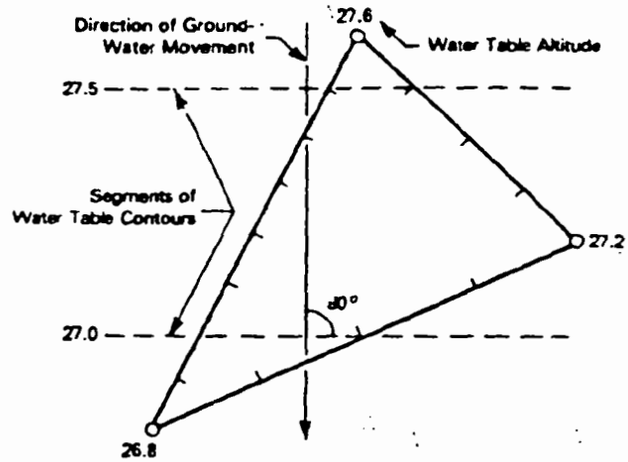

Figure 4-14. The Generalized Direction of Ground-Water Movement Can Be Determined by Means of the Water Level in Three Wells of Similar Depth (From Heath and Trainer, 1981)

Ground water flows not only through aquifers, but across confining units as well. Owing to the great differences in hydraulic conductivity between aquifers and confining units, most of the flow occurs through aquifers where the head loss per unit of distance is far less than in a confining unit. As a result, flow lines tend to parallel aquifer boundaries; they are less dense and trend nearly perpendicular through confining units (fig. 4-20). Consequently, lateral flow in units of low hydraulic conductivity is small compared to aquifers, but vertical leakage through them can be significant. Where an aquifer flow line intersects a confining unit the flow line is refracted to produce the shortest path. The degree of refraction is proportional to the differences in hydraulic conductivity.

Calculating Ground-Water Flow

Darcy's Law, expressed in many different forms, is used to calculate the quantity of underflow or vertical leakage. One means of expressing it is:

$$Q = KIA \quad (11)$$

where:

- Q = quantity of flow, in gpd
- A = cross-sectional area through which the flow occurs, in ft^2
- K = hydraulic conductivity, in gpd/ft^2
- I = hydraulic gradient, in ft/ft

The flow rate is directly proportional to the gradient and therefore the flow is laminar, which means the water will follow distinct flow lines rather than mix with other flow lines. Where laminar flow does not occur, as

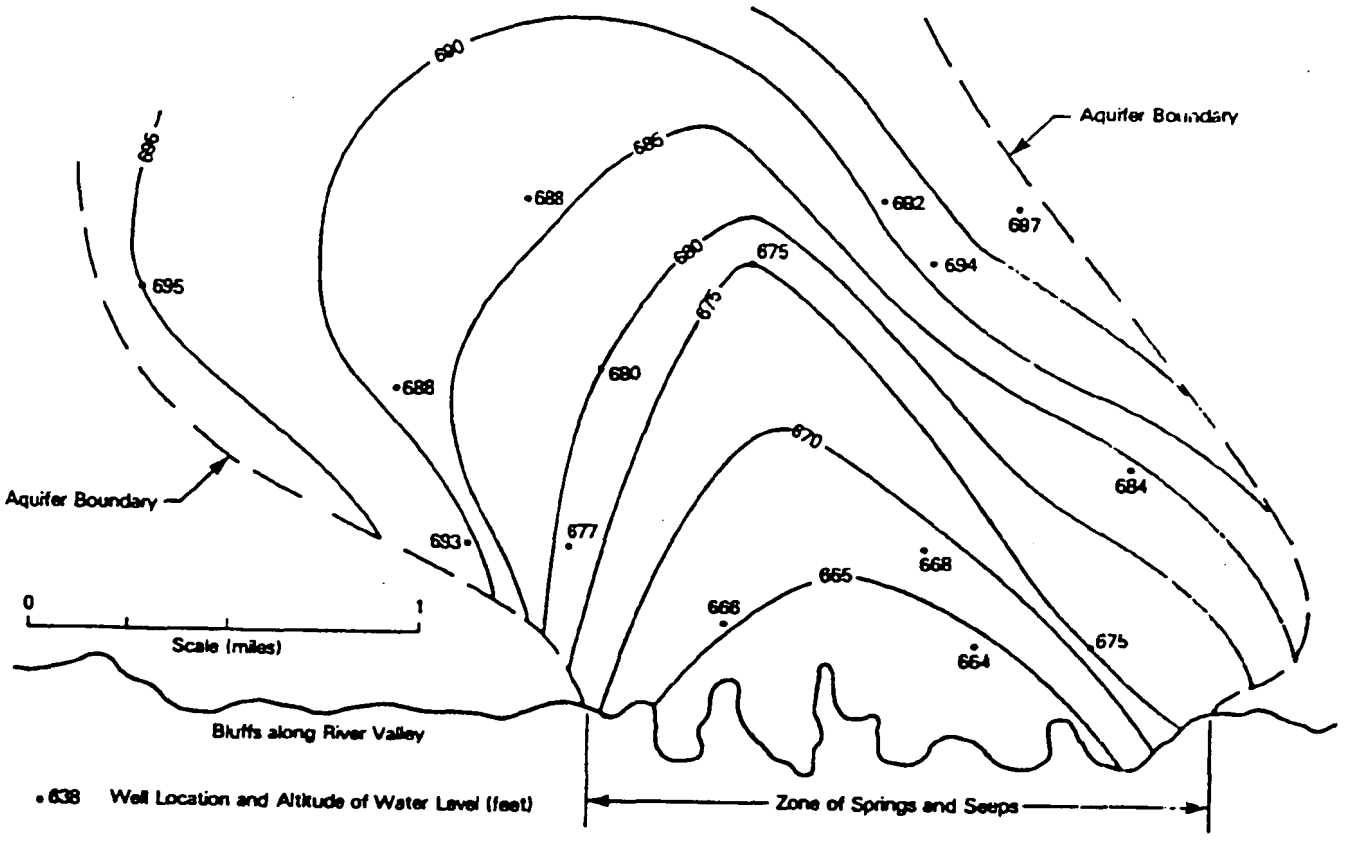

Figure 4-15. A Potentiometric Surface Map Representing the Hydraulic Gradient in an Aquifer that Crops Out Along the Bluffs of a River Valley

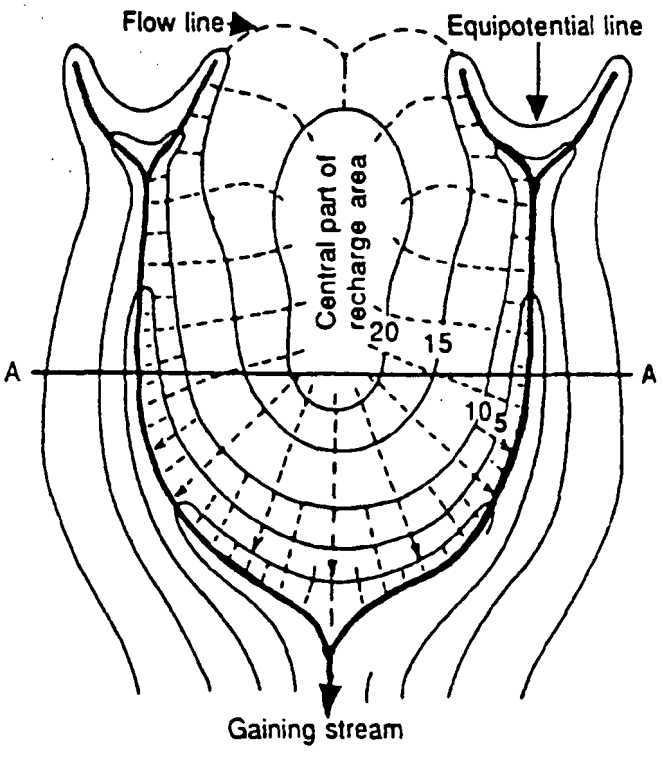

Figure 4-16. Plan View Flow Net of an Unconfined Aquifer (Modified from Heath, 1983)

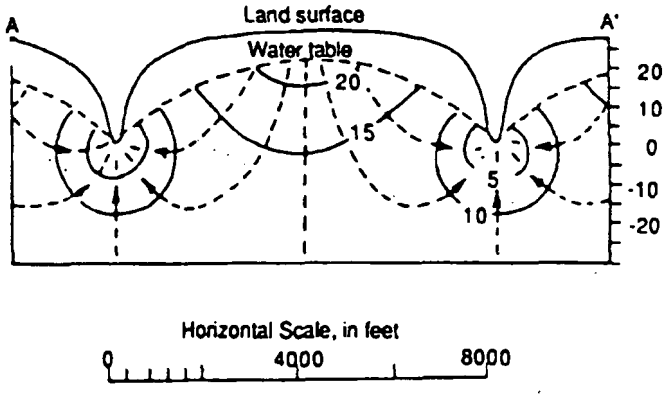

Figure 4-17. Vertical Flow Net of an Unconfined Aquifer (Modified from Heath, 1983)

in the case of unusually high velocity, which might be found in fractures, solution openings, or adjacent to some pumping wells, the flow is turbulent.

As an example of Darcy's Law, notice in Figure 4-21A that a certain quantity (Q) of fluid enters the sand-filled tube, with a cross section of A, and the same amount exits. The water level declines along the length of the flow path (L) and the head is higher in the manometer

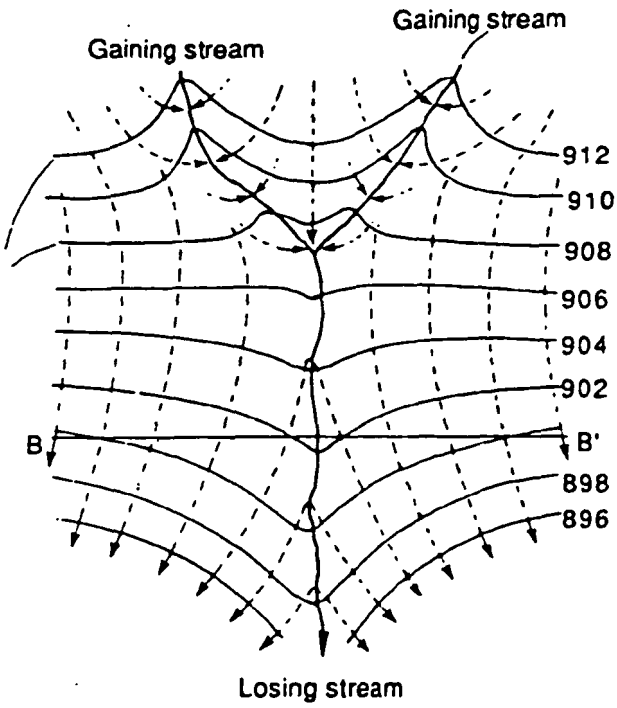

Figure 4-18. Plan View Flow Net of an Unconfined Aquifer Where Streams Change from Gaining to Losing (Modified from Heath, 1983)

at the beginning of the flow path than it is at the other end. The difference in head (H) along the flow path (L) is the hydraulic gradient (H/L or I). The head loss reflects the energy required to move the fluid this distance. If Q and A remain constant but K is increased, then the head loss decreases. It is important to keep in mind the fact that the head loss occurs in the direction of flow.

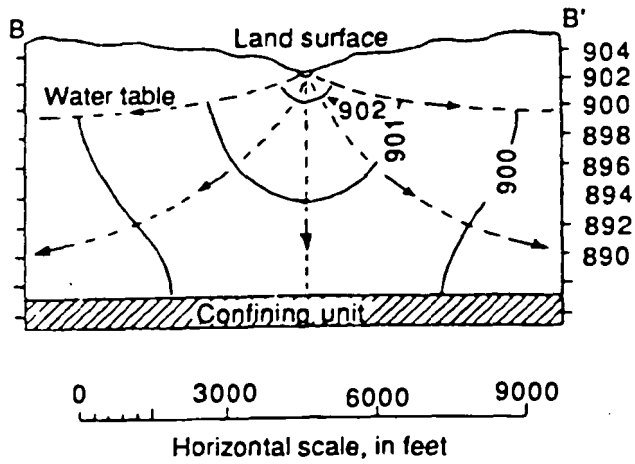

Figure 4-19. Vertical Flow Net of an Unconfined Aquifer with a Losing Stream (Modified from Heath, 1983)

In Figure 4-21B, the flow tube has been inverted and the water is flowing from bottom to top or top to bottom. Q, K, A, and I all remain the same. This illustrates an important concept when the manometers are considered as wells. Notice that the deeper well has a head that is higher than the shallow well when the water is moving upward, while the opposite is the case when the flow is downward.

Where nearby wells of different depths and water levels occur in the field, as shown in Figure 4-21C, it clearly indicates the existence of recharge and discharge areas. In recharge areas, shallow wells have a higher head than deeper wells; the difference indicates the energy required to vertically move the water the distance

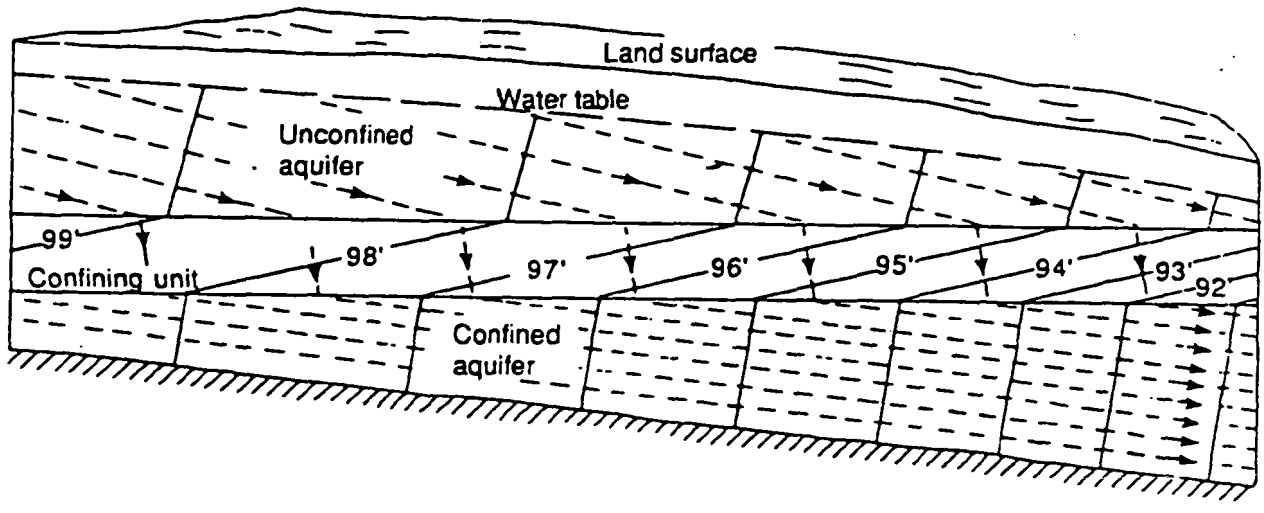

Figure 4-20. Flow Lines in Aquifers Tend to Parallel Boundaries but in Confining Units They are Nearly Perpendicular to Boundaries (Modified from Heath, 1983)

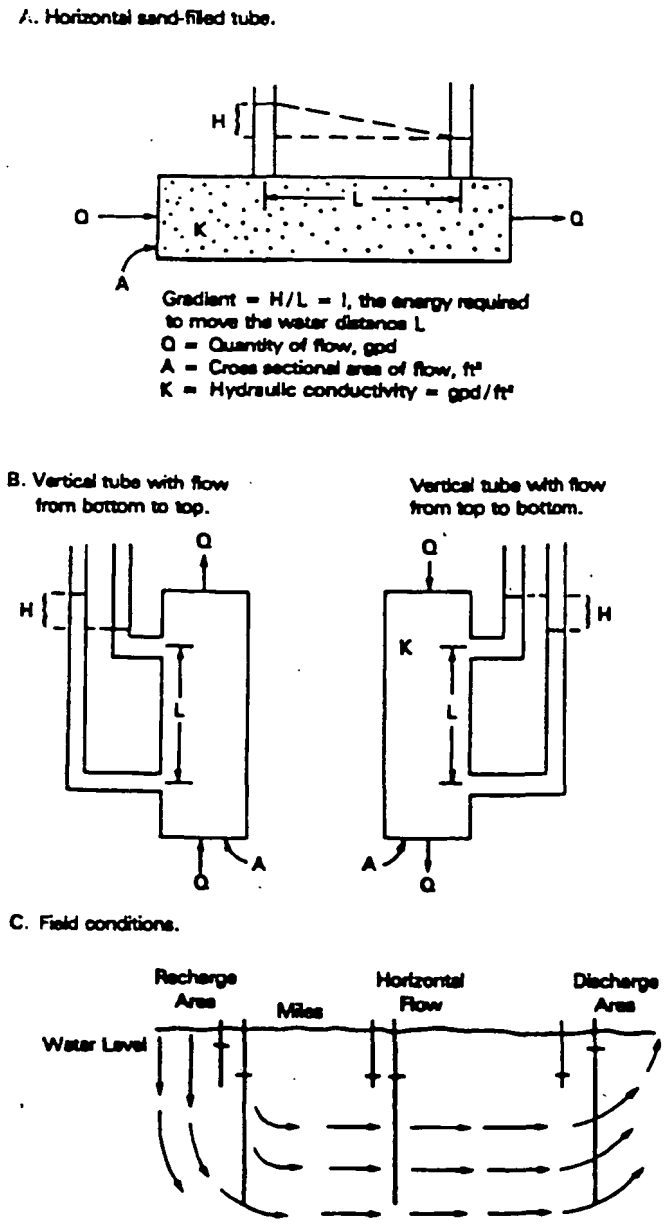

Figure 4-21. Graphical Explanation of Darcy's Law. Notice That the Flow in a Tube can be Horizontal or Vertical in the Direction of Decreasing Head

between the screens of the two wells. Where the flow is horizontal, there should be no difference in head. In discharge areas, the deeper well will have the higher head. Waste disposal in recharge areas might lead to the vertical migration of leachate to deeper aquifers and, from this perspective, disposal sites should be located in discharge areas.

An example of the use of Darcy's Law, consider a sand aquifer, about 30 feet thick, that lies within a mile wide flood plain of a river. The aquifer is covered by a confining unit of glacial till, the bottom of which is about 45 feet below land surface. The difference in water level in two wells a mile apart is 10 feet. The hydraulic conductivity of the sand is 500 gpd/ft^2. The quantity of underflow passing through a cross-section of the river valley is:

$$Q = KIA$$
$$= 500 \text{ gpd/ft}^2 \cdot (10 \text{ ft}/5280 \text{ ft}) \cdot (5280 \text{ ft} \cdot 30 \text{ ft})$$
$$= 150{,}000 \text{ gpd} \quad (12)$$

The quantity of flow from one aquifer to another through a confining unit can be calculated by a slightly modified form of Darcy's Law.

$$Q_L = (K'/m')AH \quad (13)$$

where

- Q_L = quantity of leakage, in gpd
- K' = vertical hydraulic conductivity of the confining unit, gpd/ft^2
- m' = thickness of the confining unit, ft
- A = cross-sectional area through which leakage is occurring, ft^2
- H = difference in head between the two wells tapping the upper and lower aquifers, ft

As illustrated in Figure 4-22, assume two aquifers are separated by a layer of silt. The silty confining unit is 10 feet thick and has a vertical hydraulic conductivity of 2 gpd/ft^2. The difference in water level in wells tapping the upper and lower aquifers is 2 feet. Let us also assume that these hydrogeologic conditions exist in an area that is a mile long and 2000 feet wide. The daily quantity of leakage that occurs within this area from the deep aquifer to the shallow aquifer is

$$Q = (2 \text{ gpd/ft}^2/10 \text{ ft}) \cdot (5{,}280\text{ft} \cdot 2{,}000 \text{ ft}) \cdot 2 \text{ ft} \quad (14)$$
$$= 4{,}224{,}000 \text{ gpd}$$

This calculation clearly shows that the quantity of leakage, either upward or downward, can be immense even if the hydraulic conductivity of the aquitard is small.

Interstitial Velocity

The interstitial velocity of ground water is of particular importance in contamination studies. It can be estimated by the following equation.

$$v = KI/7.48n \quad (15)$$

where:

- v = average velocity, in ft per day
- n = effective porosity

Other terms are as previously defined.

As an example, assume there is a spill that consists of a conservative substance, such as chloride. The

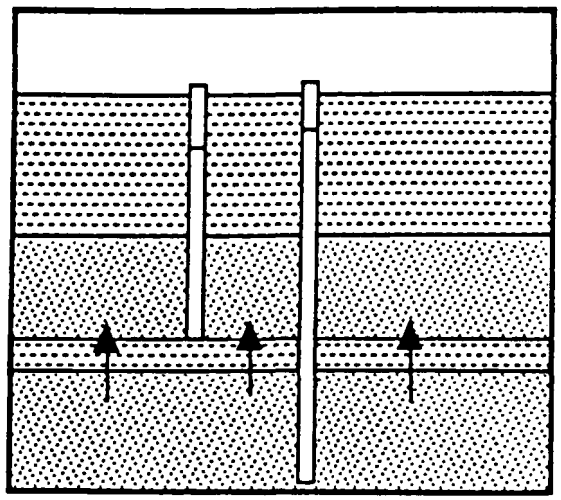

Figure 4-22. Example of Interaquifer Leakage

liquid waste infiltrates through the unsaturated zone and quickly reaches a water-table aquifer that consists of sand and gravel with a hydraulic conductivity of 2,000 gpd/ft^2 and an effective porosity of 0.20. The water level in a well at the spill lies at an altitude of 1,525 feet and at a well a mile directly downgradient it is at 1,515 feet (fig. 4-23). What is the velocity of the water and contaminant and how long will it be before the second well is contaminated by chloride?

$$v = (2{,}000 \text{ gpd/ft}^2 * (10 \text{ ft}/5{,}280 \text{ ft}))/7.48 * .20$$
$$= 2.5 \text{ ft/day} \quad (16)$$

$$\text{Time} = 5{,}280 \text{ ft}/2.5 \text{ ft/day}$$
$$= 2{,}112 \text{ days or } 5.8 \text{ years}$$

This velocity value is crude at best and can only be used as an estimate. Hydrodynamic dispersion, for

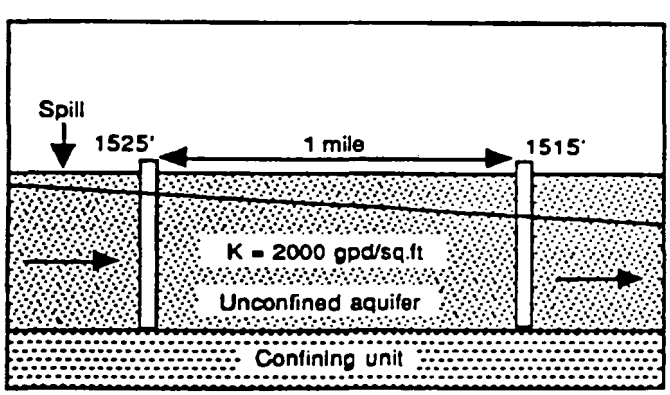

Figure 4-23. Using Ground-Water Velocity Calculations, It Would Require Nearly 6 Years for the Center of Mass of the Spill to Reach the Downgradient Well

example, is not considered in the equation. This phenomenon causes particles of water to spread transverse to the major direction of flow and move downgradient at a rate faster than expected. It is caused by an intermingling of streamlines due to differences in interstitial velocity brought about by the irregular pore space and interconnections.

Furthermore, most chemical species are retarded in their movement by reactions with the geologic framework, particularly with certain clays, soil-organic matter, and selected hydroxides. Only conservative substances, such as the chloride ion, will move unaffected by retardation (see Chapter 5).

In addition, it is not only the water below the water table that is moving, but also fluids within the capillary fringe. Here the velocity diminishes rapidly upward from the water table. Movement in the capillary fringe is important where the contaminant is gasoline or other substances less dense than water.

Transmissivity and Storativity

Hydrogeologists commonly use the term transmissivity (T) to describe an aquifer's capacity to transmit water. Transmissivity is equal to the product of the aquifer thickness (m) and hydraulic conductivity (K) and it is described in units of gpd/ft (gallons per day per foot of aquifer thickness).

$$T = Km \quad (17)$$

Another important term is storativity (S), which describes the quantity of water that an aquifer will release from or take into storage per unit surface area of the aquifer per unit change in head. In unconfined aquifers the storativity is, for all practical purposes, equal to the specific yield and, therefore, it should range between 0.1 and 0.3. The storativity of confined aquifers is substantially smaller because the water that is released from storage when the head declines comes from the expansion of water and compaction of the aquifer, both of which are exceedingly small. For confined aquifers the storativity generally ranges between 0.0001 and 0.00001, and for leaky confined aquifers it is in the range of 0.001. On method to estimate storativity for confined aquifers is to multiply the aquifer thickness by 0.000001 The small storativity for confined aquifers means that to obtain a sufficient supply from a well there must be a large pressure change throughout a wide area. This is not the case with unconfined aquifers because the water derived is not related to expansion and compression but comes instead from gravity drainage and dewatering of the aquifer.

Hydrogeologists have found it necessary to use transmissivity and storativity coefficients to calculate the response of an aquifer to stresses and to predict future water-level trends. These terms also are required as input for most flow and transport computer models.

Water-Level Fluctuations

Ground-water levels fluctuate throughout the year in response to natural changes in recharge and discharge, to changes in pressure, and to artificial stresses, such as pumping. Fluctuations brought about by changes in pressure are limited to confined aquifers. Most of these changes, which are short term, are caused by loading, such as a passing train compressing the aquifer or an increase in discharge from an overlying stream. Other water-level fluctuations are related to changes in barometric pressure, tides, earthtides, and earthquakes. None of these fluctuations reflect a change in the volume of water in storage.

An examination of the rise and fall of the water level in a well tapping flood plain deposits may lead to erroneous conclusions. If the aquifer is unconfined, a water-level rise implies ground-water recharge. On the other hand, a similar rise in a confined aquifer may be the result of loading brought about by the additional weight as the discharge of the stream increases. Generally ground-water recharge would lag behind an increase in stream discharge, while pressure loading would be concomitant (fig. 4-24).

Fluctuations that involve changes in storage are generally more long lived (fig. 4-25). Most ground-water recharge takes place during the spring and fall. Following these periods, which are a month or two long,

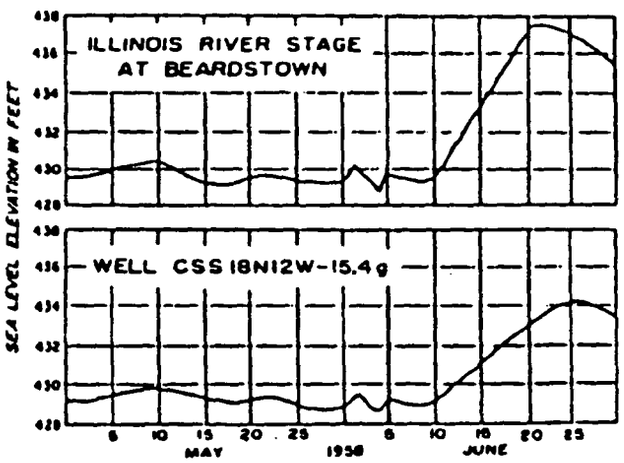

Figure 4-24. Effect of Increasing River Stage on the Water Level in a Well Tapping a Confined Aquifer (From Walton, 1970)

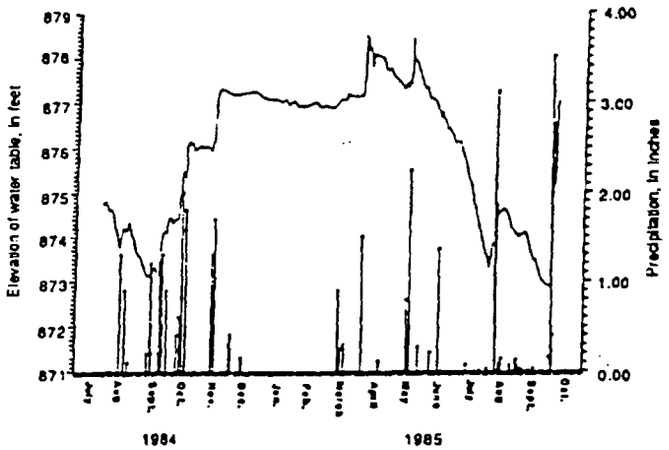

Figure 4-25. Relationship Between Precipitation and Water Level in a Well Tapping a Fine-Grained, Unconfined Aquifer

the water level declines in response to natural discharge, which is largely to streams. Although the major period of recharge occurs in the spring, minor events can happen any time there is a rain.

The volume of water added or removed from ground-water storage can be estimated by the following equation:

$$V_W = V_r S \quad (17)$$

where

V_W = the volume of water, in cubic feet
V_r = the volume of rock through which the water level has changed
S = storativity

For example, following a rain the water table rises a half a foot throughout an area of 10,000 square feet. If the aquifer has a storativity of 0 .2, then 1000 cubic feet or or nearly 7,500 gallons of water were added to storage. In this regard, Figure 4-25 shows an interesting relationship. Notice in April 1985 that the water table rose about 1.3 feet following 1.5 inches of rain, but in May the water table rose only about 1 foot after two storms provided more than twice the amount of rain (3.1 inches). This phenomenon suggests that the storativity changed, but actually the effect is related to soil moisture. When the unsaturated zone has a high soil-moisture content (April), the fillable porosity is less than it is when the moisture content is low (May); therefore, the greater the moisture content, the higher the water table rise.

Evapotranspiration effects on a surficial or shallow aquifer are both seasonal and daily. Trees, each serving as a minute pump, remove water from the

capillary fringe or even from beneath the water table during hours of daylight in the growing season (fig. 4-26). In turn, this results in a diurnal fluctuation in the water table and it might influence streamflow as well.

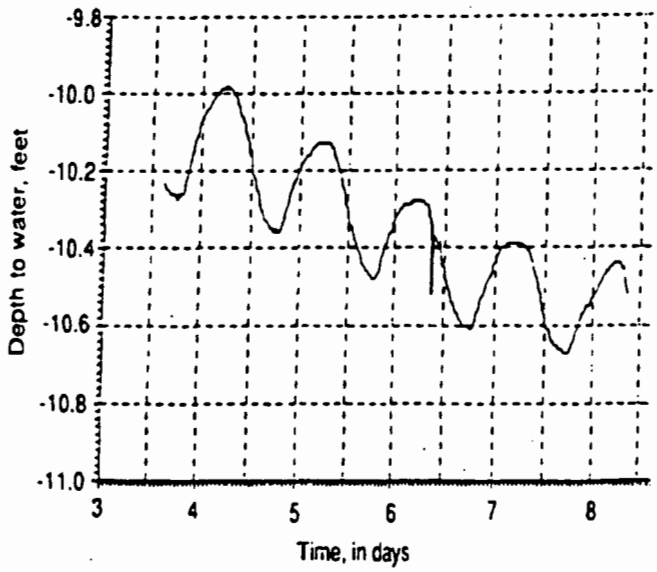

Figure 4-26. Hydrograph of a Well, 14 Feet Deep, that is Influenced by Transpiration

Cone of Depression

When a well is pumped, the water level in its vicinity declines to provide a gradient to drive water toward the discharge point. The gradient becomes steeper as the well is approached because the flow is converging from all direction, and the area through which the flow is occurring gets smaller. This results in a cone of depression around the well. Relatively speaking, the cone of depression around a well tapping an unconfined aquifer is small if compared to that around a well in a confined system. The former may be a few tens to a few hundred of feet in diameter, while the latter may extend outward for miles (fig. 4-27). By means of aquifer tests, which analyze the cone of depression, coefficients of transmissivity and storativity, as well as other hydraulic parameters can be determined.

Cones of depression from several pumping wells may overlap and, since their drawdown effects are additive, the water-level decline throughout the area of influence is greater than from a single cone (fig. 4-28). In ground-water studies, and particularly contamination problems, evaluation of the cone or cones of depression can be critical because they represent an increase in the hydraulic gradient, which in turn controls ground-water velocity and direction of flow. In fact, properly spaced and pumped wells provide a mechanism to control the migration of leachate plumes. Discharging and recharging well schemes are commonly used in attempts to restore contaminated aquifers (see Chapter 7).

Specific Capacity

The decline of the water level in a pumping well, or any well for that matter, is called the drawdown and the pre-

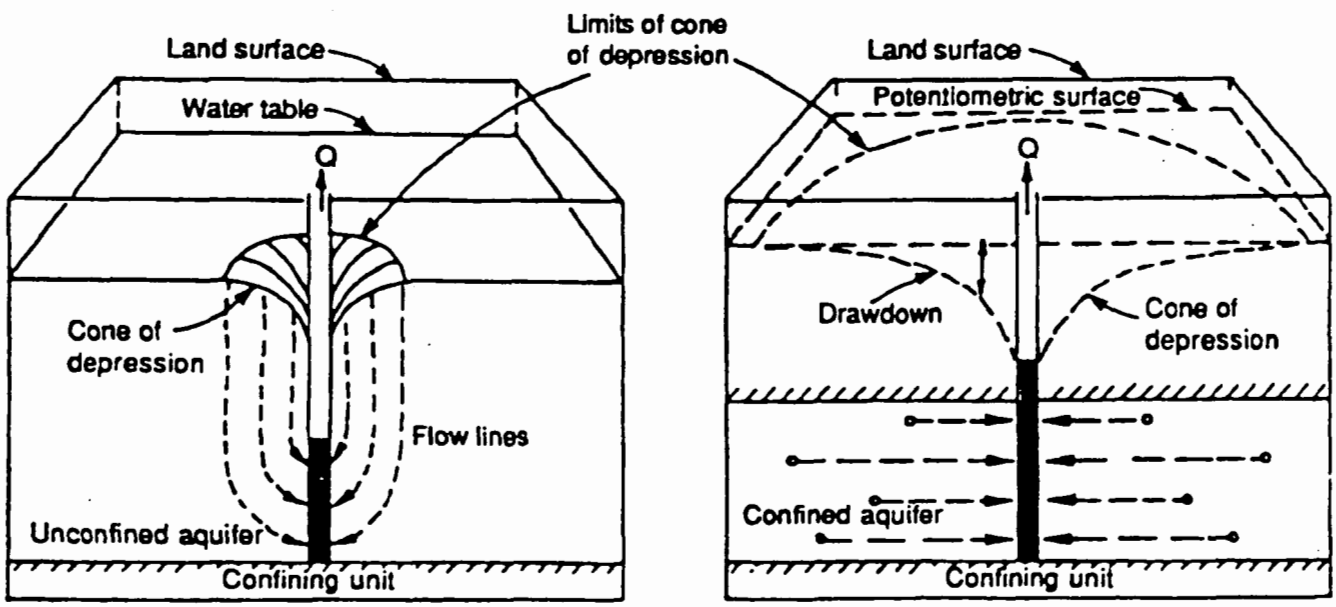

Figure 4-27. Cones of Depression in Unconfined and Confined Aquifers (From Heath, 1983)

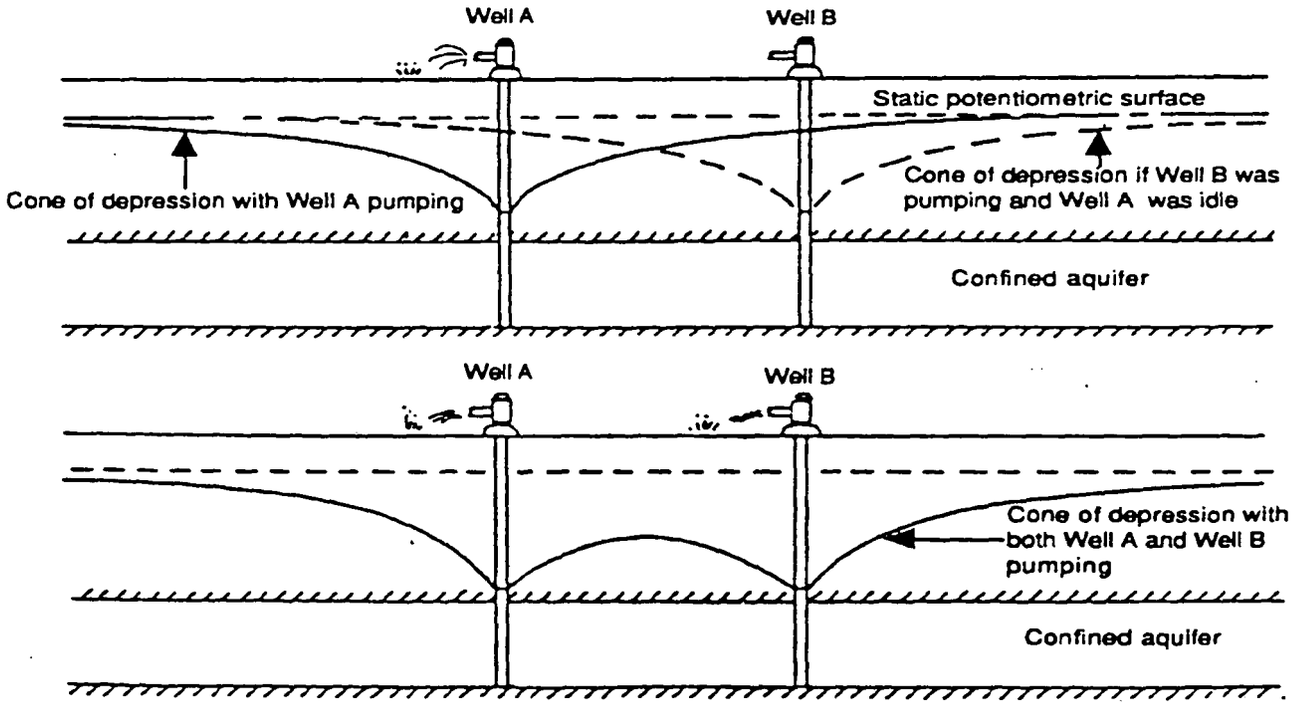

Figure 4-28. Overlapping Cones of Depression Result in More Drawdown Than Would Be the Case for a Single Well (Modified from Heath, 1983)

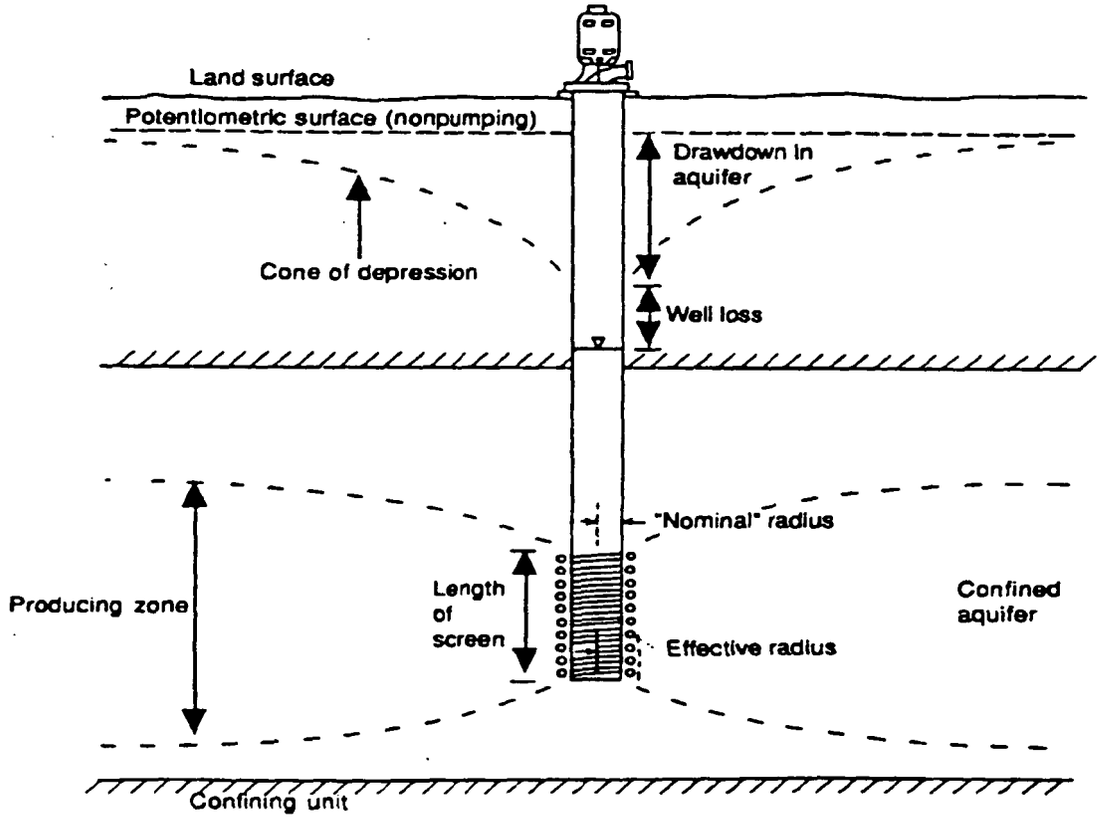

Figure 4-29. Values of Transmissivity Based on Specific Capacity Commonly are Too Low Because of Well Construction Details that Increase Well Loss (Modified from Heath, 1983)

pumping level is the static water level (fig. 4-29). The discharge rate of the well divided by the difference between the static and the pumping level is the specific capacity. The specific capacity indicates how much water the well will produce per foot of drawdown.

$$\text{Specific capacity} = Q/s \quad (18)$$

where
Q = the discharge rate, in gpm
s = the drawdown, in ft

If a well produces 100 gpm and the drawdown is 8 feet, the well will produce 12.5 gallons per minute for each foot of available drawdown. One can rather crudely estimate transmissivity of confined aquifers by multiplying specific capacity by 2,000 and by 1,550 in the case of unconfined systems.

The material presented in this chapter is both brief and generalized, but it should provide sufficient information and general principles to allow one to develop some understanding of hydrogeology. Greater detail can be obtained from the literature mentioned in the references.

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Chapter 5

GROUND-WATER CONTAMINATION

Introduction

For millennia, hu mans have disposed of waste products in a variety of ways. The method might reflect convenience, expedience, expense, or best available technology, but in many instances, leachates from these wastes have come back to haunt later generations. Ground-water contamination may lead to problems of inconvenience, such as taste, odor, color, hardness, or foaming, but the problems are far more serious when pathogenic organisms, flammable or explosive substances, or toxic chemicals and their by-products are present.

Presently, most regulatory agencies are concerned with ground-water contamination cases that involve organic compounds, and this is the result of the rapid growth of the synthetic organic chemical industry in the United States during the last 50 years. At least 63,000 synthetic organic chemicals are in common industrial and commercial use in the United States, and the number increases by 500 to 1,000 each year. Furthermore, health effects brought about by long term, low level exposures are not well known.

More than 200 chemical constituents in ground water have been documented, including approximately 175 organic compounds and more than 50 inorganic chemicals and radionuclides (OTA, 1984). The sources of these chemicals are both natural and human-induced. In a survey conducted by the U.S. EPA, volatile organic compounds (VOCs) were detected in 466 randomly selected public ground-water supply systems. One or more VOCs were detected in 16.8 percent of small systems and 28.0 percent of the larger systems sampled. Those occurring most often were trichloroethylene (TCE) and tetrachloroethylene (PCE).

In the lesser developed countries, contamination of water supplies by organic compounds is of minor concern, or of no concern at all. In such places the major health problems are the result of poor sanitary conditions and illness brought about by pathogenic organisms. In Mexico, for example, 10 percent of the the individuals who perish each year die from diarrhea, which is caused by the ingestion of contaminated food, water, and air. The primary health-related goal of water treatment is disinfection, and the emphasis over the past several years on synthetic organic compounds in drinking water in the United States has overshadowed this goal.

Individual contaminated sites generally are not large, but once degraded, ground water may remain in an unusable or even hazardous condition for decades or even centuries (Pettyjohn, 1979). The typically low velocity of ground water prevents a great deal of mixing and dilution; consequently, a contaminant plume may maintain a high concentration as it slowly moves from points of recharge to zones of discharge.

Sources of Ground-Water Contamination

As water moves through the hydrologic cycle, its quality changes in response to differences in the environments through which it passes. The changes may be either natural or human-influenced; in some cases they can be controlled, in other cases they cannot, but in most instances they can be managed in order to limit adverse water-quality changes.

The physical, chemical, and biological quality of water may range within wide limits. In fact, it is often impossible or at least difficult to distinguish the origin (human-made or natural) of many water-quality problems. Natural quality reflects the types and amounts of soluble and insoluble substances with which the water has come in contact. Surface water generally contains less dissolved solids than ground water, although at certain times where ground-water runoff is the major source of streamflow, the quality of both surface water and ground water is similar. During periods of surface runoff, streams may contain large quantities of suspended materials and, under some circumstances, a large amount of

dissolved solids. Most commonly, however, during high rates of flow streams have a low dissolved-mineral concentration.

Although the chemical quality of water in surficial or shallow aquifers may range within fairly broad limits from one time to the next, deeper ground water is characterized by nearly constant chemical and physical properties, at least on a local scale where the aquifer is unstressed by pumping. As a general rule, dissolved solids increase with depth and with the time and distance the water has traveled in the ground. A few uncommon water-quality situations exist throughout the country, reflecting peculiar geologic and hydrologic conditions. These include, among others, thermal areas and regions characterized by high concentrations of certain elements, some of which may be health hazards.

For centuries humans have been disposing of waste products by burning, placing them in streams, storing them on the ground, or putting them in the ground. Human-induced influences on surface-water quality reflect not only waste discharge directly into a stream, but also include contaminated surface runoff. Another major influence on surface-water quality is related to the discharge of ground water into a stream. If the adjacent ground water is contaminated, stream quality tends to deteriorate. Fortunately in the latter case because of dilution, the effect in the stream generally will not be as severe as it is in the ground.

The quality of ground water most commonly is affected by waste disposal and land use. One major source of contamination is the storage of waste materials in excavations, such as pits or mines. Water-soluble substances that are dumped, spilled, spread, or stored on the land surface eventually may infiltrate. Ground water also can become contaminated by the disposal of fluids through wells and, in limestone terrains, through sinkholes directly into aquifers. Likewise, infiltration of contaminated surface water has caused ground-water contamination in several places. Irrigation tends to increase the mineral content of both surface and ground water. The degree of severity in cases such as these is related to the hydrologic properties of the aquifers, the type and amount of waste, disposal techniques, and climate.

Another cause of ground-water quality deterioration is pumping, which may precipitate the migration of more mineralized water from surrounding strata to the well. In coastal areas pumping has caused seawater to invade fresh-water aquifers. In parts of coastal west Florida, wild-flowing, abandoned artesian wells have salted, and consequently ruined, large areas of formerly fresh or slightly brackish aquifers.

Ground-Water Quality Problems that Originate on the Land Surface

1. Infiltration of contaminated surface water
2. Land disposal of solid and liquid waste materials
3. Stockpiles, tailings, and spoil
4. Dumps
5. Disposal of sewage and water-treatment plant sludge
6. Salt spreading on roads
7. Animal feedlots
8. Fertilizers and pesticides
9. Accidental spills
10. Particulate matter from airborne sources

Ground-Water Quality Problems that Originate Above the Water Table

1. Septic tanks, cesspools, and privies
2. Surface impoundments
3. Landfills
4. Waste disposal in excavations
5. Leakage from underground storage tanks
6. Leakage from underground pipelines
7. Artificial recharge
8. Sumps and dry wells
9. Graveyards

Ground-Water Quality Problems that Originate Below the Water Table

1. Waste disposal in wet excavations
2. Agricultural drainage wells and canals
3. Well disposal of wastes
4. Underground storage
5. Secondary recovery
6. Mines
7. Exploratory wells and test holes
8. Abandoned wells
9. Water supply wells
10. Ground-water development

Table 5-1. Sources of Ground-Water Quality Deterioration

Table 5-1 shows that ground-water quality problems are most commonly related to: (1) water-soluble products that are stored or spread on the land surface, (2) substances that are deposited or stored in the ground above the water table, and (3) material that is stored, disposed of, or extracted from below the water table. Many of the contamination problems related to these activities are highly complex, and some are not well understood.

Ground-Water Quality Problems that Originate on the Land Surface

Infiltration of Contaminated Surface Water. The yield of many wells tapping streamside aquifers is sustained by

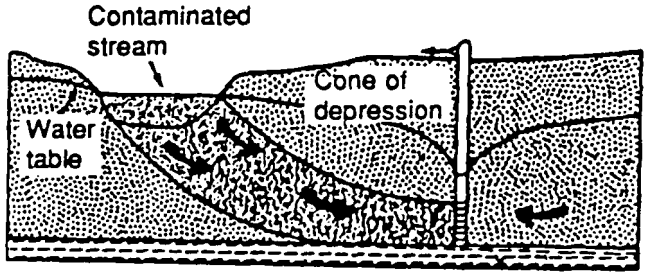

Figure 5-1. Induced Infiltration from a Contaminated Stream Will Degrade Ground-Water Quality

infiltration of surface water (fig. 5-1). In fact, more than half of the well yield may be derived directly by induced recharge from an adjacent surface-water source, which may be contaminated. As the induced water migrates through the subsurface, a few substances are diluted or removed, particularly where the water flows through filtering materials, such as sand and gravel, or organic matter. Filtration is not likely to occur if the water flows through large openings, such as those in some carbonate aquifers. Chloride, nitrate, and several organic compounds, are highly mobile, move freely with the water, and are not removed by filtration.

Examples of the degradation of ground-water supplies by induced infiltration of contaminated surface water are both numerous and widespread. In the greatest number of cases, the contamination originated from the disposal of municipal or industrial waste directly into a stream, which was then induced by pumping into adjacent aquifers. In hydrologic situations such as these, months or even years may be required for the contaminant to reach a well, but once there, all of the intervening area may be completely degraded.

Land Disposal of Solid and Liquid Waste Materials. One cause of ground-water contamination is the disposal of waste materials directly onto the land surface. Examples include manure, sludges, garbage, and industrial wastes. The waste may occur as individual mounds or it may be spread over the land. If the waste material contains soluble substances, they may infiltrate. Similar problems occur in the vicinity of various types of stockpiles.

Stockpiles, Tailings, and Spoil. Perhaps the prime example of ground-water contamination caused by stockpiles is unprotected storage of de-icing salt (sodium and calcium chloride), commonly mixed with sand, at highway maintenance lots. The salt readily dissolves to either infiltrate or run off. An average sized stockpile may contain 150 to 250 tons of salt, with anticaking additives, such as ferric ferrocyanide and sodium ferrocyanide, and perhaps phosphate and chromate to reduce corrosivity (Williams, 1984).

Other stockpiles include coal, metallic ores, phosphates, and gypsum. Both coal and metal sulfide ores, when weathered, may cause acid drainage, and the resulting low pH water may dissolve additional constituents from the ore or from other earth materials that it contacts.

Tailings, which consist of ore of a grade too low for further treatment, also may generate acid waters. They are commonly associated with ponds used for the disposal of mining wastes from cleaning and ore concentration. As a general rule, tailings ponds are unlined and, when eventually filled with slurry, are abandoned; they may serve as sources of acid, metals, dissolved solids, and radioactivity.

The debris or waste material produced during mining is called spoil. For over a century, iron-sulfide-rich spoil has served as a major source of acid-mine drainage in the eastern coal fields and at metal sulfide mines throughout the country.

Dumps. During the past two decades, investigators have taken a serious look at the environmental effects of dumps. As rainwater infiltrates through trash in a dump, it accumulates an ample assortment of chemical and biological substances. The resulting fluid, or leachate, may be highly mineralized, and as it infiltrates, some of the substances it contains may not be removed or degraded.

Disposal of Sewage and Water Treatment Plant Sludge. Sludge is the residue of chemical, biological, and physical treatment of municipal and industrial wastes. They include lime-rich material from water treatment plants, as well as sewage sludge from wastewater treatment plants. Sludges typically contain partly decomposed organic matter, inorganic salts, heavy metals, bacteria, and perhaps viruses. Nitrogen in municipal sludge may vary from 1 to 7 percent. Land application of wastewater and sewage sludge is an alternative to conventional treatment and disposal, and is in common usage by the canning and vegetable industry, petroleum refining, pulp and paper, and the power industry. Contamination results from the infiltration of partly treated wastewaters that have not undergone sufficient attenuation.

Infiltration from wastewater stabilization ponds also can cause ground-water contamination. Ponds of this type primarily are used for settlement of suspended solids and biological treatment of primary and secondary effluent.

Salt Spreading on Roads. Especially since the construction of the interstate highway system, water contamination due to wintertime road salting has become

an increasing problem. From a quality viewpoint, the salting may bring about deterioration of streams due to surface runoff, and infiltration causes ground-water contamination. Numerous instances of contamination have been reported in the New England states and Michigan.

On the outskirts of Muskegon, MI, which lies on a sandy plain adjacent to Lake Michigan, a class action suit was filed against a county wastewater treatment operation that uses large upground lagoons, alleging contamination of several domestic wells. Evidence presented at a pretrial hearing clearly showed, however, that a few of the domestic wells had been contaminated from time to time, but the source was de-icing salt spread on high crowned roads that were bordered by wide, deep ditches cut into the sand of an unconfined aquifer. All of the domestic wells were adjacent to the road ditches and, when pumping, induced salty water to the well.

Accidental Spills. A large volume of toxic materials are transported throughout the country by truck, rail, and aircraft, transferred at handling facilities, and stored in tanks; accidental spills of these materials are commonplace. It has been estimated that about 16,000 spills, ranging from a few to several million gallons, occur each year, and these include hydrocarbons, paint products, flammable materials, acids and anhydrous ammonia, among many others (National Academy of Sciences, 1983). Virtually no methods are available to quickly and adequately clean up an accidental spill or those caused by explosions or fires. Furthermore, immediately following an accident, the usual procedure is to spray the area with water. The resulting fluid may either flow into a stream or infiltrate. In a few cases, the fluids have been impounded by dikes, causing even more infiltration.

Fertilizers and Pesticides. Increasing amounts of both fertilizers and pesticides are being used in the United States each year. Reportedly, there are more than 32,000 different compounds consisting of an excess of 1,800 active ingredients used in agricultural applications (Houzim and others, 1986). Many are highly toxic and, in countless cases, quite mobile in the subsurface. Numerous compounds, however, become quickly attached to fine-grained sediment, such as organic matter and clay and silt particles. A part of this attached material is removed by erosion and surface runoff. In many heavily fertilized areas, the infiltration of nitrate, a decomposition product of ammonia fertilizer, has adversely affected ground water. The consumption of nitrate-rich water leads to a disease in infants known as "blue babies" (methemoglobinemia).

In some irrigated regions, automatic fertilizer feeders are attached to irrigation sprinkler systems. When the pump is shut off, water flows back through the pipe into the well bore, creating a partial vacuum that may cause fertilizer to flow from the feeder into the well. It is possible that some individuals even dump fertilizers (and perhaps pesticides) directly into the well to be picked up by the pump and distributed to the sprinkler system.

Aurelius (1989) described an investigation in Texas where 188 wells were sampled for nitrate and pesticides in 10 counties where aquifer vulnerability studies and field characteristics indicated the potential for ground-water contamination from the normal use of agricultural chemicals. Nine pesticides (2,4,5-T, 2,4-DB, metolachlor, dicamba, atrazine, prometon, bromacil, picloram, and triclopyr) were found present in 10 wells, nine of which were used for domestic supply. Also, 182 wells were tested for nitrate and of these, 101 contained more than the recommended limit. Of the high nitrate wells, 87 percent were used for household purposes. In addition, 28 wells contained arsenic at or above the limit of 0.05 mg/L, and 23 of these were domestic wells.

Animal Feedlots. Feedlots, used for cattle, hogs, sheep, and poultry, cover relatively small areas but provide a huge volume of wastes. These wastes and seepage from lagoons have contaminated both surface and ground water with large concentrations of nitrate, phosphate, chloride, and bacteria.

Particulate Matter from Airborne Sources. A relatively minor source of ground-water contamination is caused by acid rain and the fallout of particulate matter originating from smoke, flue dust, or aerosols, and from automobile emissions. Some of the particulate matter is water-soluble and toxic. Deutsch (1963) described an example of ground-water contamination by chromium-rich dust discharged through roof ventilators at a factory in Michigan. Accumulating on the downwind side of the plant, the highly soluble hexavalent chromium infiltrated, contaminating a local municipal water supply. Along the Ohio River in the vicinity of Ormet, Ohio, the airborne discharge of fluoride from an aluminum processing plant seriously affected dairy operations, and fluoride concentrations in ground water at the plant exceeded 1,000 mg/L in the mid 1970s.

Ground-Water Quality Problems that Originate Above the Water Table

Many different types of materials are stored, extracted, or disposed of in the ground above the water table. Table 5-1 shows that contamination can originate from many of these operations.

Septic Tanks, Cesspools, and Privies. Probably the major cause of ground-water contamination in the United States is effluent from septic tanks, cesspools, and privies. Individually of little significance, these devices are important in the aggregate because they are so abundant and occur in every area not served by municipal or privately owned sewage treatment systems. Onsite sewage disposal systems number approximately 22 million and discharge an estimated one trillion gallons of effluent. Conventional septic tanks and their associated leach fields account for 85 percent of the systems in use.

The area that each point source affects is generally small, since the quantity of effluent is small, but in some limestone areas effluent may travel long distances in subterranean cavern systems. Biological contamination of ground water is widely recognized. In areas where the density of septic tanks is unusually high and the soils are permeable, this form of waste disposal has caused regional ground-water contamination (Nassau and Suffolk counties, NY, and Dade County, FL).

Surface Impoundments. Surface impoundments, including ponds and lagoons, generally consist of relatively shallow excavations that range in area from a few square feet to many acres. They are used in agricultural, municipal, and industrial operations for the treatment, retention, and disposal of both hazardous and nonhazardous wastes. During the Surface Impoundment Assessment (EPA, 1983), more than 180,000 impoundments were located at approximately 80,000 sites. Nearly half of the sites were located over zones that are either very thin or very permeable, and more than half of these contained industrial waste. In addition, 98 percent of the sites on thick, permeable aquifers were located within a mile of potential drinking water supplies.

Special problems develop with surface impoundments in limestone terrain with extensive near-surface solution openings. In Florida, Alabama, Missouri, and elsewhere, municipal sewage lagoons have collapsed into sinkholes draining raw effluent into widespread underground openings. In some cases the sewage has reappeared in springs and streams several miles away. Wells producing from the caverns could easily become contaminated and cause epidemics of waterborne diseases.

Oil-field brines, which are highly mineralized salt solutions, are particularly noxious and without doubt they have contaminated both surface and ground water in every state that produces oil. The brine, an unwanted by-product, is produced with the oil, as well as during drilling. In the latter case, drilling fluids and brines are stored in reserve pits, which are filled some time after completion or abandonment of the well. Customarily, produced oil-field brines are temporarily stored in holding tanks or placed in an injection well. Owing to the corrosive nature of the brine, leaky tanks and pipelines are not uncommon.

Landfills. Lehman (1986) reported that there are approximately 18,500 municipal and 75,700 industrial landfills that are subject to RCRA Subtitle D regulations. Of the 94,000 known landfills recorded during a 1979 inventory, only about 5,600 facilities were licensed, and the remainder were open dumps (Peterson, 1983).

Sanitary landfills generally are constructed by placing wastes in excavations and covering the material daily with soil—thus the term "sanitary" to indicate that garbage and other materials are not left exposed to produce odors or smoke or attract vermin and insects. Even though a landfill is covered, leachate may be generated by the infiltration of precipitation and surface runoff. Fortunately many substances are removed from the leachate as it filters through the unsaturated zone, but leachate may contaminate ground water and even streams if it discharges at the surface as springs and seeps.

Waste Disposal in Excavations. Following the removal of clay, limestone, sand, and gravel, or other material, the remaining excavations are traditionally left unattended and often used as unregulated dumps. The quantity and variety of materials placed in excavations are almost limitless. They have been used for the disposal of liquid wastes, such as oil-field brines and spent acids from steel mill operations, and for snow removed from surrounding streets and roads—snow that commonly contains a large amount of salt.

Leakage from Underground Storage Tanks. A growing problem of substantial potential consequence is leakage from underground storage tanks and from pipelines leading to them. These facilities store billions of gallons of liquids that are used for municipal, industrial, and agricultural purposes. Corrosion is the most frequent cause for leakage. It has been estimated that at least 35 percent of all underground storage tanks are now leaking (EPA, 1986). Gasoline leakage has caused severe hazardous difficulties throughout the nation. Since gasoline will float on the water table, it tends to leak into basements, sewers, wells, and springs, causing noxious odors, explosions, and fires.

Leakage from Underground Pipelines. Literally thousands of miles of buried pipelines cross the U.S. Leaks, of course, do occur, but they may be exceedingly difficult to detect. Leaks are most likely to develop in

lines carrying corrosive fluids. An example occurred in central Ohio where a buried pipeline carried oil-field brine from a producing well to a disposal well. The corrosive brine soon weakened the metal pipe, which then began to leak over a length of several tens of yards. The brine infiltrated, contaminating the adjacent ground water, then flowed down the hydraulic gradient to a stream. During the ensuring months, nearly all of the vegetation between the leaking pipeline and the stream was killed. The leaking area of the pipe was detected only because of the dead vegetation and salty springs. A pipeline that cut through a municipal unconfined well field in south-central Kansas ruptured, spilling a substantial amount of hydrocarbons. Restoration has been both expensive and time consuming.

Sewers are used to transport wastes to a treatment plant. Rarely watertight, fluids leak out of sewers if they are above the water table, and into them if they are in the saturated zone. In many places the water table fluctuates to such a degree that the sewer is gaining in discharge part of the time and losing at other times. Figure 5-2 shows the chloride content in wells 10.5 (D-3) and 14 feet (D-4) deep that are a few feet from a sewer and, upgradient, a 14 feet deep control well (A-4). While the shallower well reached a peak of nearly 175 mg/L, the concentration is much reduced in the deeper well. Even the lowest concentrations near the sewer are 50 percent or more higher than the average background concentration, which is less than 25 mg/L.

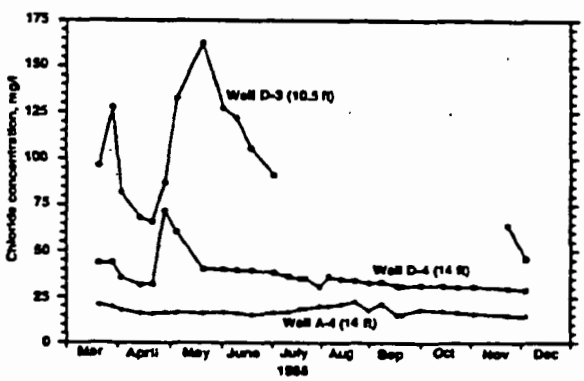

Figure 5-2. Leakage from a Sewer Increases the Chloride Concentration of the Ground Water

Artificial Recharge. Artificial recharge includes an assortment of techniques used to increase the amount of water infiltrating an aquifer. Methods consists of spreading the water over the land or placing it in pits or ponds, or injecting water through wells directly into the aquifer. Waters used for artificial recharge consist of storm runoff, excess irrigation water, streamflow, cooling water, and treated sewage effluent, among others. The quality of water artificially recharged can effect the quality of that in the ground. In several places this has led to increased concentrations of nitrates, metals, detergents, synthetic organic compounds, bacteria, and viruses.

Sumps and Dry Wells. Sumps and dry wells are used for drainage, to control storm runoff, for the collection of spilled liquids, and disposal. They are usually of small diameter and may be filled with pea gravel, coarse sand, or large rocks.

Orr (1990) described several storm water drainage wells in Ohio that receive a variety of contaminants through intentional dumping, illegal disposal, and inadvertent collection of leaks and spills. At Fairfield and Fairborn, dry wells serve as runoff collection wells (an estimated 2,900 in Fairfield) and discharge into very permeable deposits that serve as the major source of domestic, municipal, and industrial water supply. In addition to typical storm water, other contaminants have included used oil and filters, antifreeze, and, in one well, a considerable number of dead catfish. At Fairfield an accidental release of 21,000 gallons of fuel oil from a surface tank flowed into two storm drainage wells in March 1989. Although approximately 16,000 gallons were recovered, by September 1989 ,product thickness in monitoring wells was as much as eight feet.

Graveyards. Leachate from graveyards may cause ground-water contamination, although cases are not well documented. In some of the lightly populated glaciated regions in the north-central part of the U.S., graveyards are commonly found on deposits of sand and gravel, because these materials are easier to excavate than the adjacent glacial till and are better drained so that burials are not below the water table. Unfortunately, these same sand and gravel deposits also may serve as a source of water supply. Graveyards also are possible sources of contamination in many hard rock terrains where there are sinkholes or a thin soil cover.

Ground-Water Quality Problems that Originate Below the Water Table

Table 5-1 lists a number of causes of ground-water contamination produced by the use and misuse of space in the ground below the water table.

Waste Disposal In Wet Excavations. Following the cessation of various mining activities, the excavations usually are abandoned; eventually they may fill with water. These wet excavations have been used as dumps for both solid and liquid wastes. The wastes, being directly connected to an aquifer, may cause extensive contamination. Furthermore, highly

concentrated leachates may be generated from the wastes due to seasonal fluctuations of the water table. In the late 1960s at a lead-zinc mine in northwestern Illinois, processing wastes were discharged into an abandoned mine working. The wastes, moving slowly in the ground water, contaminated several farm wells. Analyses of water from several of the wells showed high concentrations of dissolved solids, iron, sulfate, and, more importantly, heavy metals and cyanide.

Agricultural Drainage Wells and Canals. Where surficial materials consist of heavy clay, flat-lying land may be poorly drained and contain an abundance of marshes and ponds. Drainage of this type of land generally is accomplished with field tiles and drainage wells. A drainage well is merely a vertical, cased hole in the ground or in the bottom of a pond that allows the water to drain into deeper, more permeable materials. The drainage water may contain agricultural chemicals and bacteria.

Deepening of stream channels may lower the water table. Where the fresh-saltwater interface lies at shallow depths, lowering of the water table (whether by channelization, pumping, or other causes) may induce upward migration of the saline water; it may even flow into the deepened channel. Under these circumstances, reduction of the depth to fresh water can result in a rise in the level of saline water several times greater than the distance the freshwater level is lowered.

In some coastal areas, the construction of extensive channel networks has permitted tidal waters to flow considerable distances inland. The salty tidal waters infiltrate, increasing the salt content of the ground water in the vicinity of the canal. Some canals are used for the disposal of urban runoff and sewage effluent

Well Disposal of Wastes. For decades, humans have disposed of liquid wastes by pumping them into wells. Since World War II, a considerable number of deep well-injection projects (Class I wells) have come into existence, usually at industrial sites. Industrial disposal wells range in depth from a few tens of feet to several thousand feet. The injection of highly toxic wastes into some of these wells has led to ground-water contamination. The problems are caused by direct injection into an aquifer, by leakage of contaminants from the well head, through the casing, or via fractures in confining beds.

Exclusive of oil-field brine, most deep well-injection operations are tied to the chemical industry. Well depths range from 1,000 to 9,000 feet and average 4,000 feet. The deepest wells are found in Texas and Mississippi. As of October 1983, EPA reported the existence of at least 188 active hazardous waste injection wells in the United States. There were an additional 24,000 wells used to inject oil-field brine (Class II wells).

Properly managed and designed underground injection systems can be effectively used for storage of wastes deep underground and may permit recovery of the wastes in the future. Before deep well disposal of wastes is permitted by state regulatory agencies and the EPA, however, there must be an extensive evaluation of the well system design and installation, the waste fluids, and the rocks in the vicinity of the disposal well.

Underground Storage. The storage of material underground is attractive from both economic and technical viewpoints. Natural gas is one of the most common substances stored in underground reservoirs. However, the hydrology and geology of underground storage areas must be well understood in order to insure that the materials do not leak from the reservoir and degrade adjacent water supplies.

Secondary Recovery. With increased demands for energy resources, secondary recovery, particularly of petroleum products, is becoming even more important. Methods of secondary recovery of petroleum products commonly consist of injection of steam or water into the producing zone, which either lowers the viscosity of the hydrocarbon or flushes it from the rocks, enabling increased production. Unless the injection well is carefully monitored and constructed, fluids can migrate from a leaky casing or through fractures in confining units.

Mines. Mining has instigated a variety of water contamination problems. These have been caused by pumping of mine waters to the surface, by leaching of the spoil material, by waters naturally discharging through the mine, and by milling wastes, among others. Literally thousands of miles of stream and hundreds of acres of aquifers have been contaminated by highly corrosive mineralized waters originating in coal mines and dumps in Appalachia. In many western states, mill wastes and leachates from metal sulfide operations have seriously affected both surface water and ground water.

Many mines are deeper than the water table, and in order to keep them dry, large quantities of water are pumped to waste. If salty or mineralized water lies at relatively shallow depths, the pumping of freshwater for dewatering purposes may cause an upward migration, which may be intercepted by the well. The mineralized water most commonly is discharged into a surface stream.

Many abandoned underground mine workings serve as

a source of water supply for homes, cities, and industry. They also are used as waste receptacles. Orr (1990) described an Ohio situation where combined sanitary sewers and storm water are discharged through wells tapping an abandoned mine. An additional 200 drainage wells were drilled through septic tank leach fields to the underground workings. In the same town, a nationally recognized food processing plant uses the mine for water supply and the city installed their standby well field in it. Water samples from borings into the mine were opaque with sewage, and strong raw sewage and diesel fuel odors were present, along with a strong flow of methane.

Exploratory Wells and Test Holes. Literally hundreds of thousands of abandoned exploratory wells dot the countryside. Many of these holes were drilled to determine the presence of underground mineral resources (seismic shot holes, coal, salt, oil, gas, etc.). The open holes permit water to migrate freely from one aquifer to another. A freshwater aquifer could thus be joined with a contaminated aquifer or a deeper saline aquifer, or contaminated surface water could drain into freshwater zones.

Abandoned Wells. Another cause of ground-water contamination is the migration of mineralized fluids through abandoned wells, and dumping wastes directly into them. In many cases when a well is abandoned the casing is pulled (if there is one) or the casing may become so corroded that holes develop. This permits ready access for fluids under higher pressure to migrate either upward or downward through the abandoned well and contaminate adjacent aquifers. In other cases, improperly cased wells allow high-pressure artesian saline water to spread from an uncased or partly cased hole into shallower, lower-pressure aquifers or aquifer zones.

Although confined aquifers, to some extent, are protected by overlying confining units, abandoned wells make the seal ineffective. In addition, some individuals, probably through a lack of awareness, use abandoned wells to dispose of used motor oil and other liquid wastes, permitting direct access to a drinking water supply.

Water Supply Wells. Improperly constructed water-supply wells may either contaminate an aquifer or produce contaminated water. Dug wells, generally of large diameter, shallow depth, and poorly protected, commonly are contaminated by surface runoff flowing into the well. Other contamination has been caused by infiltration of water through contaminated fill around a well or through the gravel pack. Still other contamination has been caused by barnyard, feedlot, septic tank, or cesspool effluent draining directly into the well. Many contamination and health problems can arise because of poor well construction.

Although well construction standards institute rigid guidelines, they may not be strictly adhered to during the installation of domestic and livestock wells. Furthermore, a great number of water supply wells were constructed long before well standards were established.

Ground Water Development. In certain situations pumping of ground water can induce significant water-quality problems. The principal causes include interaquifer leakage, induced infiltration, and landward migration of sea water in coastal areas. In these situations, the lowering of the hydrostatic head in a freshwater zone leads to migration of more highly mineralized water toward the well site. Undeveloped coastal aquifers are commonly full, the hydraulic gradient slopes towards the sea, and freshwater discharges from them through springs and seeps into the ocean. Extensive pumping lowers the freshwater potentiometric surface permitting sea water to migrate toward the pumping center. A similar predicament which occurs in inland areas where saline water is induced to flow upward, downward, or laterally into a fresh water aquifer due to the decreased head in the vicinity of a pumping well. Wells drilled adjacent to streams induce water to flow from the streams to the wells. If the stream is contaminated, induced infiltration will lead to deterioration of the water quality in the aquifer.

Natural Controls on Ground-Water Contamination

As Deutsch (1965) clearly pointed out, there are four major natural controls involved in shallow ground-water contamination. The first includes the physical and chemical characteristics of the earth materials through which the liquid wastes flow. A major attenuating effect for many compounds is the unsaturated zone. Many chemical and biological reactions in the unsaturated zone lead to contaminant degradation, precipitation, sorption, and oxidation. The greater the thickness of the unsaturated zone, the more attenuation there is likely to take place. Below the water table, the mineral content of the medium probably becomes more important because assorted clays, hydroxides, and organic matter take up some of the contaminants by exchange or sorption. Many of the other minerals have no effect on the contaminants with which they come into contact.

The second major control includes the natural processes that tend to remove or degrade a contaminant as it flows through the subsurface from areas or points of recharge to zones or points of discharge. These processes include filtration, sorption, ion-exchange, dispersion,

oxidation, and microbial degradation, as well as dilution. The third control relates to the hydraulics of the flow system through which the waste migrates, beginning with infiltration and ending with discharge. The contaminant may enter an aquifer directly, by flowing through the unsaturated zone, by interaquifer leakage, by migration in the zone of saturation, or by flow through open holes.

The final control is the nature of the contaminant. This includes its physical, chemical, and biological characteristics and, particularly, its stability under varying conditions. The stability of the more common constituents and the heavy metals are fairly well known. On the other hand, the stability of organic compounds, particularly synthetic organic compounds, has only recently come under close inspection and actually little is known of their degradation and mobility in the subsurface. This fact has been brought clearly to the attention of the general public by the abundance of reported incidences of contamination by EDB, TCE, and DBCP.

To a large extent, it is the aquifer framework that controls the movement of ground water and contaminants. Of prime importance, of course, is the hydraulic conductivity, both primary and secondary. In the case of consolidated sedimentary rocks, primary permeability, in many respects, is more predictable than secondary permeability. In sedimentary rocks, similar units of permeability tend to follow bedding planes or formational boundaries, even if the strata are inclined. Permeable zones most often are separated by layers of fine-grained material, such as clay, shale, or silt, which serve as confining beds. Although leakage through confining beds is the rule rather than the exception, both water and contaminants are more likely to remain in a permeable zone than to migrate through units of low permeability. The movement of ground water and contaminants through larger openings, such as fractures, complicates the assumed picture. Not only can the velocity change dramatically, but in fracture flow, much of the attenuation capacity is lost, and it is difficult to predict local directions of flow.

The geologic framework, in conjunction with surface topography, also exerts a major control on the configuration of the water table and the thickness of the unsaturated zone. Generally speaking, the water table would be relatively flat in a deposit of permeable surficial sand and gravel. In contrast, the water table in glacial till, which is typically fine-gained, would more closely conform to the surface topography. The position of the water table is important not only because it is the boundary between the saturated and unsaturated zones, but also because it marks the bottom and, therefore, the thickness of the unsaturated material.

In many, if not most, contaminated areas, the water table has been or is intermittently affected by pumping. The resulting cone of depression on the water table changes both the hydraulic gradient and ground-water velocity. A change in gradient and velocity also occurs in the vicinity of recharge basins (lagoons, pits, shafts, etc.), because the infiltrating water forms a mound in the water table. As Figure 5-3 shows, the mound causes radial flow and, therefore, contaminants can move in directions that are different than the regional hydraulic gradient, at least until the mounding effects are overcome by the regional flow.

Ground-water or interstitial velocity is controled by the hydraulic conductivity, gradient, and effective porosity. Water movement through a permeable gravel with a gradient of 10 feet per mile averages about 60 feet per day, but in a clay with the same gradient and no secondary permeability the water movement would be only about 1 foot in 30,000 years. In most aquifers, ground-water velocity ranges from a few feet per day to a few feet per year.

Carlston (1964) determined that the mean residence time of ground water in a basin in Wisconsin was about 45 days and in New Jersey about 30 days. This study shows that ground water may discharge into closely spaced streams in humid areas within a few days to a few months. On the other hand, in less permeable terrains ground water and contaminants may remain in the subsurface for years or even decades.

Leachate

The causes of ground-water contamination are many, but it is the source that needs special consideration. For example, an accidental spill from a ruptured tank may provide a considerable volume of liquid with an extremely high concentration that is present only during a short time, but leachate continuously generated from a landfill may consist of a large volume of low concentration that spans many years. Once it reaches the water table, a spill might move largely as a conservative contaminant because of its high concentration, despite the fact that it might be degradable in smaller concentrations. Leachate is more likely to be attenuated by microbial degradation, sorption, dilution, and dispersion.

In the case of landfills and similar sources, leachate is a liquid that has formed as infiltrating water migrates through the waste material extracting water-soluble compounds and particulate matter. The mass of leachate generated is directly related to precipitation, assuming the waste lies above the water table. Much of the annual precipitation, including snowmelt, is removed by surface runoff and evapotranspiration; it is only the remainder

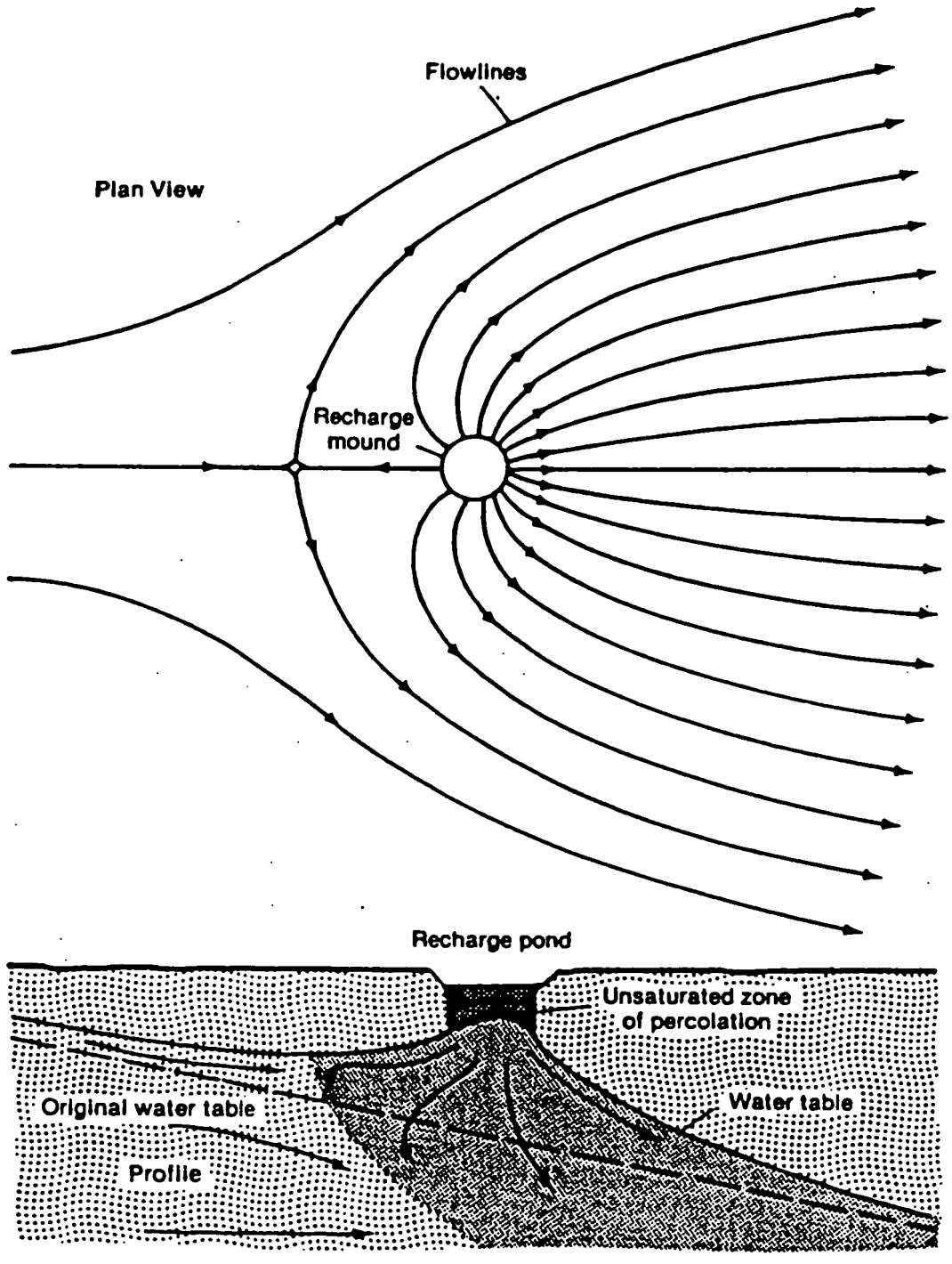

Figure 5-3. Infiltration from a Surface Impoundment Will Create a Mound on the Water Table

that is available to form leachate. Since the landfill cover, to a large extent, controls leachate generation, it is exceedingly important that a cover be properly designed, maintained, and monitored.

The physical, chemical, and biological characteristics of leachate are influenced by: (1) the composition of the waste, (2) the stage of decomposition, (3) microbial activity, (4) the chemical and physical characteristics of the soil cover and of the landfill, and (5) the time rate of release (recharge). Since all of the above can range within remarkably wide limits, it is possible to provide only a general range in concentration of leachate constituents, as Table 5-2 shows.

Constituents	Operating Landfill	Abandoned Landfill
BOD_5, mg/L	1,800	18
COD, mg/L	3,850	246
Ammonia-N, mg/L	160	100
Hardness, mg/L as $CaCO_3$	900	290
Total iron, mg/L	40.4	2.2
Sulfate, mg/L	225	100
Specific Conductance, µmhos	3,000	2,500

Table 5-2. Comparison of Chemical Characteristics of Leachate from an Operating Landfill and a 20-Year-Old Abandoned Landfill in Southeastern Pennsylvania (From Wu and Ahlert, 1976)

It also is important to account for the fact that materials placed in landfills may vary seasonally. For example, many municipal landfills are used to dispose of snow and ice, which may contain calcium, sodium, and chloride from de-icing salts. This could lead to the generation of leachate that changes throughout the year, particularly in regard to the chloride concentration. In addition, leachate collected from a seep at the base of a landfill should be more highly mineralized than that present in the underlying ground water, which is diluted.

Changes in Ground-Water Quality

It is often assumed that natural ground-water quality is nearly constant at any particular site. Field data substantiate this assumption, and logic leads to the same conclusion, if the aquifer is confined and not subjected to a stress. Multiple samples from a single well, however, are likely to show slight changes in concentrations of specific constituents owing to differences in sample collection, storage, and analytical technique.

Deeper or confined aquifers in which ground-water flow is lethargic, generally have a nearly constant chemical quality that, at any particular place, reflects the geochemical reactions that occurred as the water migrated through confining layers and aquifers from recharge area to points of collection or discharge.

The quality of deeper water can change, but generally not abruptly, in response to stresses on the aquifer system. Changes in hydrostatic head brought about by pumping, for example, may cause migration of other types of waters from adjacent units into the producing zone. As shown in Figure 5-4, the sulfate content of a municipal well in north-central North Dakota increased fivefold, from 200 to 1,000 mg/L, over a period of a few years.

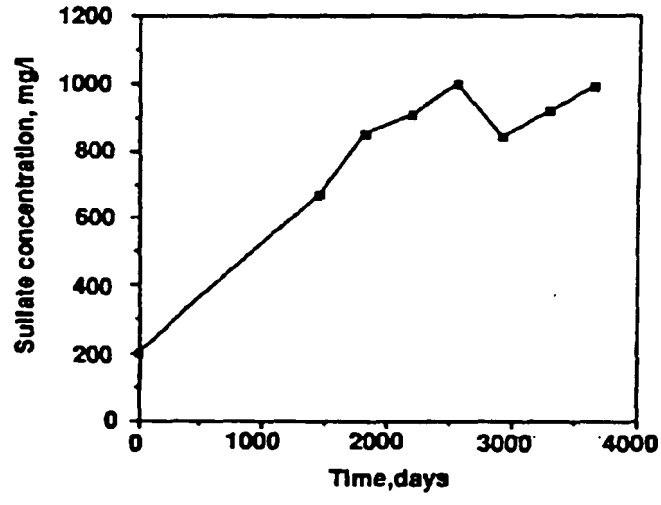

Figure 5-4. The Increase in Sulfate Concentration Was Related to Natural Causes Brought About by Pumping

In this instance, the first sample was collected in 1974 when the well was first pumped for an acceptance test. The well, one of six in a new field, tapped a previously unused and confined ground-water system. By 1978 the entire well field was in operation, overlapping cones of depression had spread out several miles along the trend of the buried glacial valley, and the sulfate concentration had increased to nearly 700 mg/L. From 1982 through 1985 sulfate fluctuated between about 850 and 1,000 mg/L, and the slow change, either an increase or a decrease, was in response to the pumping durations and rates of all of the wells in the field. The source of the naturally occurring sulfate was several hundred feet from the nearest production well and, fortunately, only one other well was affected, and then to a far smaller degree. Consequently, it was possible to blend the water from all of the production wells, and

the concentration of sulfate in the mixed water consistently was less than 250 mg/L.

Changes in water quality in confined aquifers also may be due to fluid migration along the well casing or gravel pack, or by leakage through confining beds, abandoned wells, or exploration holes, and by well injection of waste fluids.

In contrast to confined aquifers, ground-water quality, in shallow and surficial aquifers, can change considerably within a few hours or days. These aquifers are not well protected from changes brought about by natural events occurring at the land surface or from human-induced contamination. Surficial aquifers, in fact, are highly susceptible to rapid and sometimes dramatic changes in quality.

In the majority of cases, neither water levels nor water samples are measured or collected at regular intervals. Annual or quarterly measurements or samples may be satisfactory for most purposes, but they are likely to be far too infrequent in ground-water studies if an investigator is attempting to develop an understanding of the manner in which a system functions. Figure 5-5 shows the March through December 1988 fluctuation of the water table in a well 14 feet deep. Quarterly measurements provide a good indication of the annual

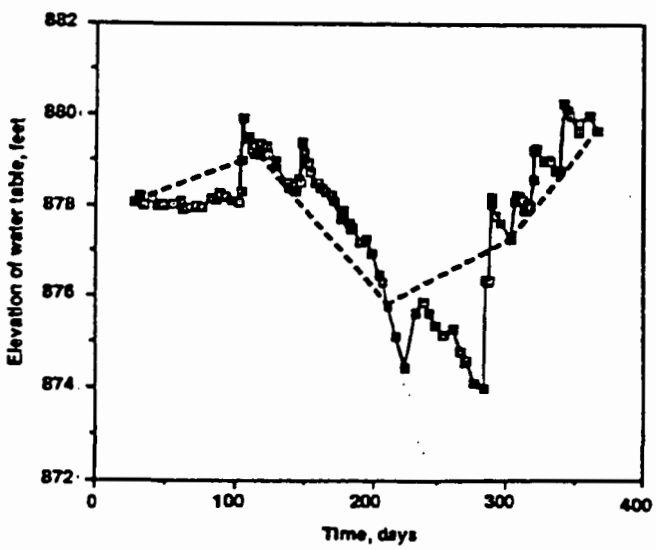

Figure 5-5. Weekly Water-Table Measurements More Accurately Show the Aquifer Response than do Quarterly Measurements

change, but they do not display the complexity of the hydrograph, as shown by weekly measurements. It is the short-term rise in water level that is most likely to indicate changes in ground-water quality.

Figure 5-6 shows the annual range in electrical conductivity in the same well described above. Again, quarterly measurements provide a general impression of the change throughout the year, which in this example is from about 925 to 1,070 µmhos. On the other hand, an average of five measurements per month reveal that the electrical conductivity changed considerably from one time to the next, and that the annual range is from 800 to 1,175 µmhos.

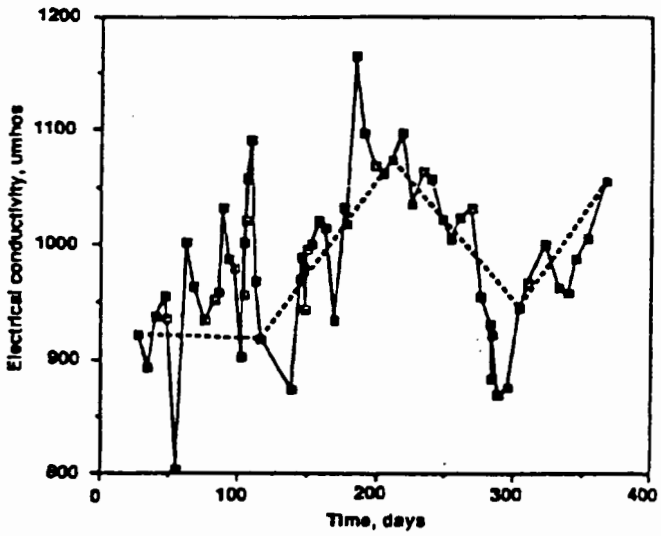

Figure 5-6. Weekly Measurements of Electrical Conductivity are more useful for Determining Changes in Chemical Quality than are Quarterly Determinations

The Concept of Cyclic Fluctuations

Several years ago, Pettyjohn (1971,1976, 1982) described cyclic fluctuations of ground-water quality. The mechanisms that lead to cyclic fluctuations will be discussed in greater detail here because both the cause and effect can have a significant impact on: (1) ground-water quality monitoring and determination of background quality; (2) transport and fate of organic and inorganic compounds, as well as bacteria and viruses; and (3) monitoring well design and installation.

The contaminated site that Pettyjohn used to develop the concept of cyclic fluctuation lies on the flood plain of the Olentangy River in central Ohio where precipitation averages about 38 inches per year (fig. 5-7). Underlain by shale, the alluvial deposits consist of 15 to 35 feet of sand, gravel, silt, and clay. The water table, 1.5 to 5 feet below land surface, oscillates a foot or so annually.

Oil production began at this site in mid-1964, but by July 1965, all wells had been plugged. Ground-water contamination occurred because of leakage of oil-field brine, containing about 35,000 mg/L of chloride, from

three holding ponds. When samples were first collected from 23 monitoring wells in July 1965, the aquifer locally contained more than 35,000 mg/L of chloride.

Of particular importance in the monitoring of this site is a cluster of three wells, one screened at a depth of 7 to 9 feet and another from 21 to 23 feet, while a third, gravel-packed through much of its length (23 feet), receives water from the entire aquifer (fig. 5-8). It is assumed that the third well provides a composite sample of the reservoir and that when it had a higher concentration than both the deep and shallow wells, the most highly mineralized water was between 9 and 23 feet, and vice versa.

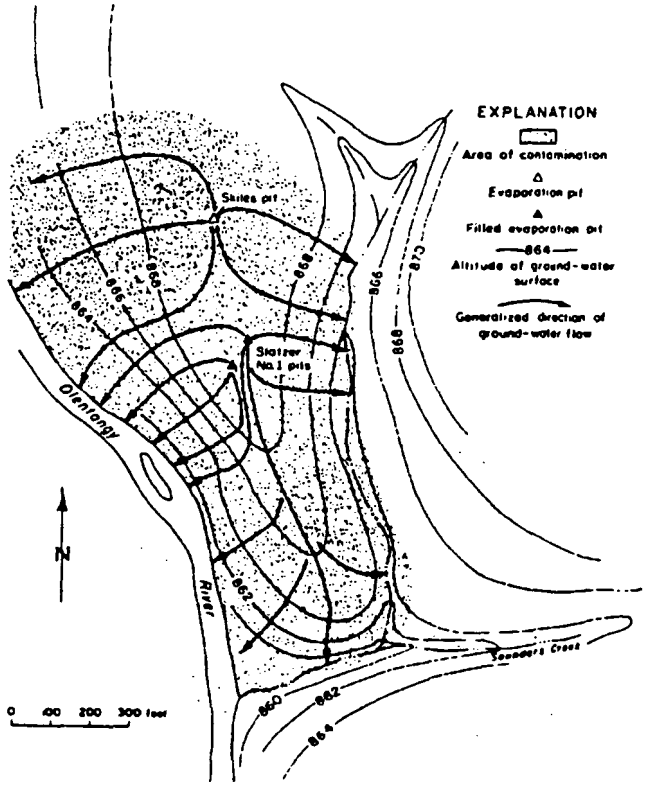

Figure 5-7. Water-Table Map of the Contaiminated Olentangy River Site

Figure 5-9 shows the chloride fluctuations in the three ns occurred at the shallowest depths, at other times at the greatest depth, and at still other times the greatest concentration was somewhere in the middle of the aquifer. Figures 5-10 and 5-11 show the vertical distribution of chloride in the aquifer. The only means for accounting for the variable distribution, both in space and time, is intermittent recontamination, which is puzzling in view of the fact that oil-field activities ceased in June 1965 before any of the samples were collected.

The chloride fluctuations that occurred during 1965 to

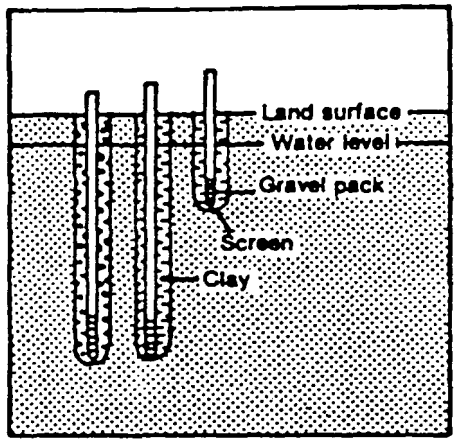

Figure 5-8. Construction Details of Three Wells in a Cluster

1966 and 1969 are shown schematically in Figure 5-12. The October 1965 samples apparently were collected shortly after a recharge event, which leached salt from the unsaturated zone. This slowly sinking mass (1) was subsequently replaced with less mineralized water. A month later, the first mass had reached and was migrating along the bottom of the aquifer when another recharge event occurred (2). By December, the second mass had reached the bottom of the aquifer and was moving toward the river. Recharge events also occurred in January 1966 (3), and in February 1966 (4). Figure 5-12 shows that the aquifer was recontaminated several times during 1969, particularly during January, February, and March. On the average, it appears that the chloride concentration in the ground water at the Olentangy River site was reduced by half every 250 or so days. This was not a linear decline, but rather intermittent flushing of the source.

Findings similar to those in the Olentangy River study have been reported by Hagen (1986), Hoyle (1987), Ross (1988), Pettyjohn (1987a, 1987b, 1988), Pettyjohn and others (1986), Nelson (1989), and Froneberger

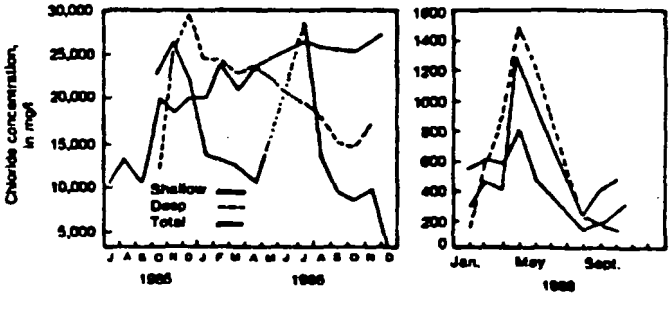

Figure 5-9. Variations in Chloride Concentration in Cluster Wells at the Olentangy River Site

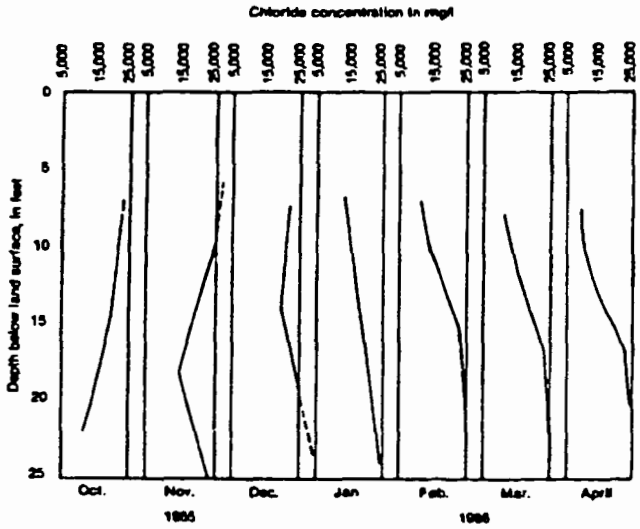

Figure 5-10. Vertical Distribution of Chloride During 1965 and 1966

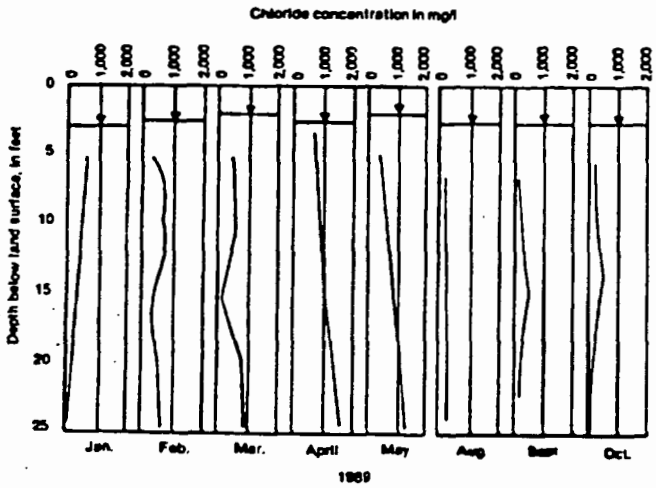

Figure 5-11. Vertical Distribution of Chloride During 1969

(1989). The investigations were conducted in an urban area in north-central Oklahoma at a small but intensely monitored field site. The site lies on the flood plain of a small stream, and the alluvium consists of a fine-grained silt loam that contains soil structures throughout the entire thickness of 43 feet.

At the Oklahoma site, fertilizer application, followed by rain, has a short lived but significant effect on the concentration of nitrogen in ground water, regardless of the soil-moisture content. As Figure 5-13 shows, nitrate concentrations increased in one well (14 feet deep) from about 4 to to 16 mg/L within a two-day period following 1.3 inches of rain. The concentration then decreased to about 2 mg/L during the next three days. At this time (September 1985) the soil-moisture content was very low. The change in nitrate concentration over

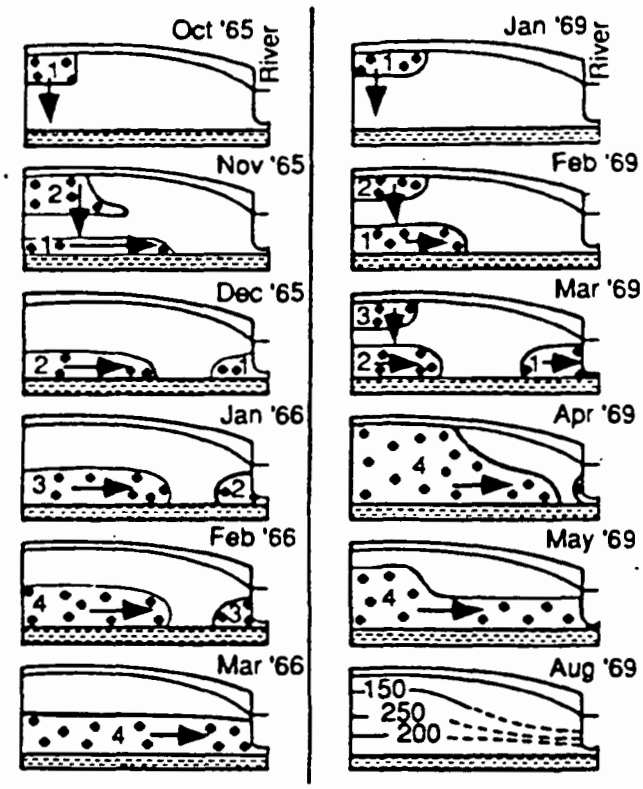

Figure 5-12. Conceptual Model Showing Leaching and Recontamination of Ground Water at the Olentangy River Site

the five-day interval appears to suggest the infiltration of a relatively small volume of highly concentrated water that is followed about two days later with a large volume of water with a very low nitrate content. Flow through the unsaturated zone was greater than 5 feet per day, suggesting early flow through macropores that is followed by piston-type flow.

A similar phenomenon occurred in April 1986 when the soil-moisture content was twice as great as it had been in September, 1985. Shown in Figure 5-14 is the nitrate concentration in three wells at a cluster; the wells are 8.5 (A-1), 9.5 (A-2), and 14 (A-4) feet deep. The change in concentration in all of the wells follows the same pattern, but the concentration decrease with depth. Following a rain, the concentration at a depth of 8.5 feet, for example, increased only about 3 mg/L and this was followed during the next two days by a decrease of 15 mg/L and then nitrate again slowly increased.

During the fall and spring events, nitrate accounted for only a small percentage of the dissolved solids content, and the concentration of the other major constituents in the water followed a different pattern. As Figure 5-15 shows; there was a small decrease in electrical

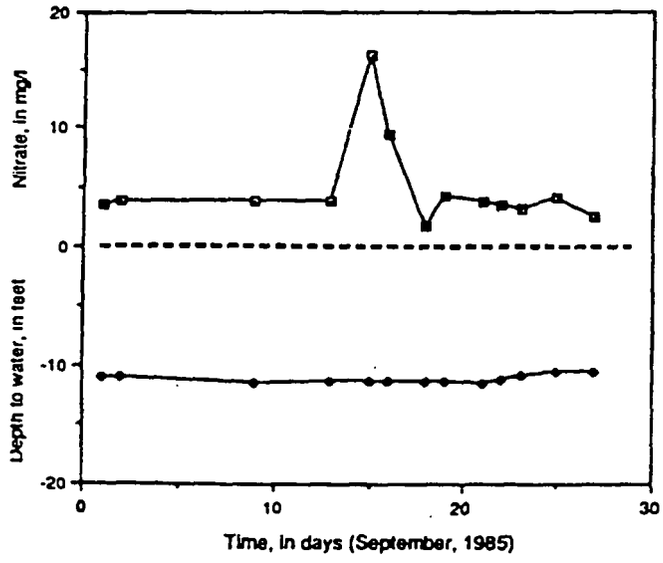

Figure 5-13. During Dry Weather the Water-Table Aquifer Responded Quickly to Rain and Flushed Nitrate into the Ground

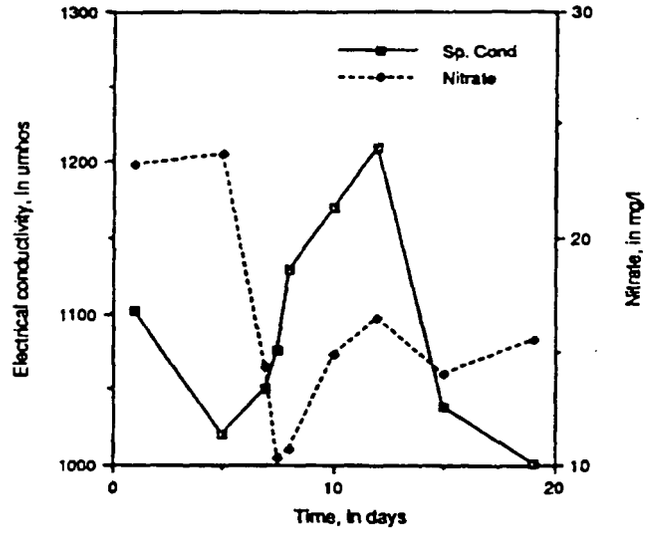

Figure 5-15. The Increase in Nitrate Was Caused by a Small Volume of Rapidly Infiltrating Water, While the Later Increase in Electrical Conductivity Was Caused by a Large Mass of Slowing Moving Water

conductivity at the peak nitrate concentration, and as the nitrate decreased, electrical conductivity began to increase, reaching a maximum about 11 days later.

At the same site, another well cluster, adjacent to a building, receives runoff from the roof that infiltrates in the vicinity of the wells. The runoff has a low dissolved mineral content and, when it infiltrates during a prolonged wet period, the electrical conductivity of the ground water decreases from around 1,000 to about 400 µmhos (fig. 5-16).

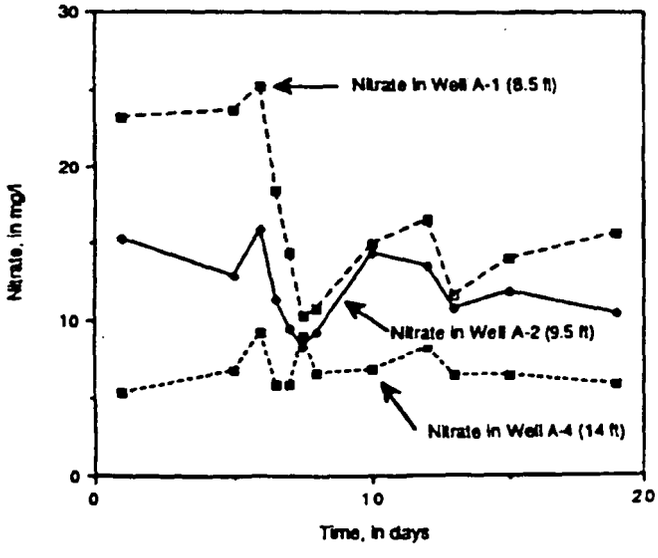

Figure 5-14. Although Decreasing with Depth, Nitrate Concentrations in all of the Wells Followed a Similar Pattern after a Rain

The Ohio and Oklahoma studies indicate that water soluble substances on the land surface or in the unsaturated zone may be intermittently introduced into a shallow aquifer, changing its quality, for many years. The rate of introduction or leaching is dependent upon the chemical and physical properties of the waste and the soil, and the frequency of the recharge events.

Throughout most of the year in humid and semiarid regions, the quantity of water that infiltrates and the amount of contaminants that are flushed into an aquifer are relatively small. During summer months, ground-water quality changes would be expected to occur more rapidly, perhaps in a matter of hours, because of the large size and abundance of the macropores and fractures. These changes, however, may occur only over a relatively small area because of the local nature of convective storms.

On the other hand, during the spring recharge period and, in many places, during the fall as well, noteworthy quantities of contaminants may infiltrate over wide areas. Although the quantity of leached substances is larger than at any other time during the year, the change may occur more slowly and the resulting concentration in ground water may not be at a maximum because of the diluting effect brought about by the major influx of water. Therefore, the major infusion of contaminants, which is strongly influenced by climate, occurs twice a year, although minor recharge events may occur at any time.

This phenomenon has important implications in

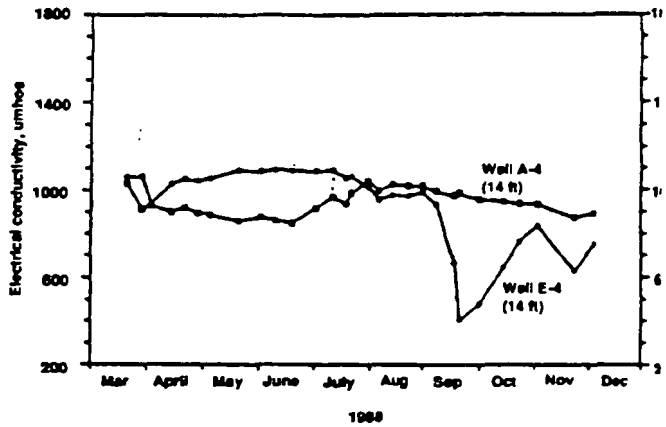

Figure 5-16. Runoff from a Roof Tends to Reduce the Electrical Conductivity of the Underlying Ground Water

monitoring and sampling. Since the natural quality of shallow ground water ranges fairly widely, background concentration is not a finite number but, rather, a range that may encompass an order of magnitude for major constituents, such as dissolved solids, and two or three orders of magnitude for minor forms, such as nitrate. In addition, the concentration might increase several fold a day or two after a rain, or decrease even more three to five or so days later. The question then arises as to the most appropriate time to sample. Available data suggest that the least biased sample could be obtained at least two weeks after a recharge event, but the interval is strongly influenced by the physical and chemical characteristics of the unsaturated zone and the depth to the water table.

In order to account for cyclic fluctuations in ground-water quality it is assumed that: (1) the unsaturated zone may store a considerable volume of water-soluble substances for long periods of time, and (2) the main paths along which contaminants rapidly move through the unsaturated zone to the water table consist largely of fractures and macropores.

Most macropores may be barely detectable without a close examination. Ritchie and others (1972) suggested that the interfaces between adjacent soil peds also serve as macropores. Moreover, these openings need not extend to the land surface in order for flow to occur in them (Quisenberry and Phillips, 1976). Nonetheless, water can flow below the root zone in a matter of minutes. Thomas and Phillips (1973) suggested that this type of flow does not appear to last more than a few minutes or perhaps, in unusual cases, more than a few hours after "cessation of irrigation or rain additions."

Even though there may be a considerable influx of contaminants through macropores and fractures to the water table following a rain, the concentration of solutes in the main soil matrix may change little, if at all. This is clearly indicated in studies by Shuford and others (1977) and again shows the major role of large openings. On the other hand, in the spring, when the soil-moisture content is high, some of the relatively immobile or stagnant soil water may percolate to the water table transporting salts with it. A similar widespread event may occur during the fall as a result of decreasing temperature and evapotranspiration, and of wet periods that might raise the soil-moisture content.

Ecologic conditions in fractures and macropores should be quite different from those in the main soil matrix, largely because of the greater abundance of oxygen and smaller moisture content. As a result, one might expect different microbial populations and densities, as well as chemical conditions in macropores and fractures than in the bulk soil matrix. Coupled with their far greater fracture permeability, this may help to explain why some biodegradable organic compounds or those that should be strongly sorbed actually reach the water table and move with the ground water.

Prediction of Contaminant Migration

In any ground-water contamination investigation it is essential to obtain the background concentration of the chemical constituents of concern, particularly those that might be common both to the local ground water and a contaminant. As mentioned previously, the water in shallow or surficial aquifers can undergo substantial fluctuations in chemical quality. Therefore, it is not always a simple task to determine background concentrations, particularly of the more conservative constituents, such as chloride or nitrate.

The severity of ground-water contamination is partly dependent on the characteristics of the waste or leachate, that is, its volume, composition, concentration of the various constituents, time rate of release of the contaminant, the size of the area from which the contaminants are derived, and the density of the leachate, among others. Data describing these parameters are difficult to obtain and commonly are lumped together into the term "mass flow rate," which is the product of the contaminant concentration and its volume and recharge rate, or leakage rate.

Once a leachate is formed it begins to migrate downward through the unsaturated zone where several physical, chemical, and biological forces act upon it. Eventually, however, the leachate may reach saturated strata where it will then flow primarily in a horizontal direction as defined by the hydraulic gradient. From this point on, the

leachate will become diluted due to a number of phenomena, including filtration, sorption, chemical processes, microbial degradation, dispersion, time, and distance of travel.

Filtration removes suspended particles from the water mass, including particles of iron and manganese or other precipitates that may have been formed by chemical reaction. Dilution by sorption of chemical compounds is caused largely by clays, metal oxides and hydroxides, and organic matter, all of which function as sorptive material. The amount of sorption depends on the type of contaminant and the physical and chemical properties of the solution and the subsurface material.

Chemical processes are important when precipitation occurs as a result of excess quantities of ions in solution. Chemical processes also include volatization as well as radioactive decay. In many situations, particularly in the case of organic compounds, microbiological degradation effects are not well known, but it does appear, however, that a great deal of degradation can occur if the system is not overloaded and appropriate nutrients are available (see Chapter 7).

Dispersion of a leachate in an aquifer causes the concentration of the contaminants to decrease with increasing length of flow. It is caused by a combination of molecular diffusion, which is important only at very low velocities, and dispersion or hydrodynamic mixing, which occurs at higher velocities in laminar flow through porous media. In porous media, different macroscopic velocities and flow paths that have various lengths are to be expected. Leachate moving along a shorter flow path or at a higher velocity would arrive at an end point sooner than that part following a longer path or a lower velocity; this results in hydrodynamic dispersion.

Dispersion can be both longitudinal and transverse and the net result is a conic form downstream from a continuous contamination source. As Figure 5-17 shows, the concentration of the leachate is less at the margins of the cone and increases toward the source. Because dispersion is directly related to ground-water velocity, the size of a plume of contamination tends to increase with more rapid flow.

Since dispersion is affected by velocity and the configuration of the aquifer's pore spaces, coefficients must be determined experimentally or empirically for a given aquifer. There is considerable confusion regarding the quantification of the dispersion coefficient. Selection of dispersion coefficients that adequately reflect conditions that exist in an aquifer is a problem that can

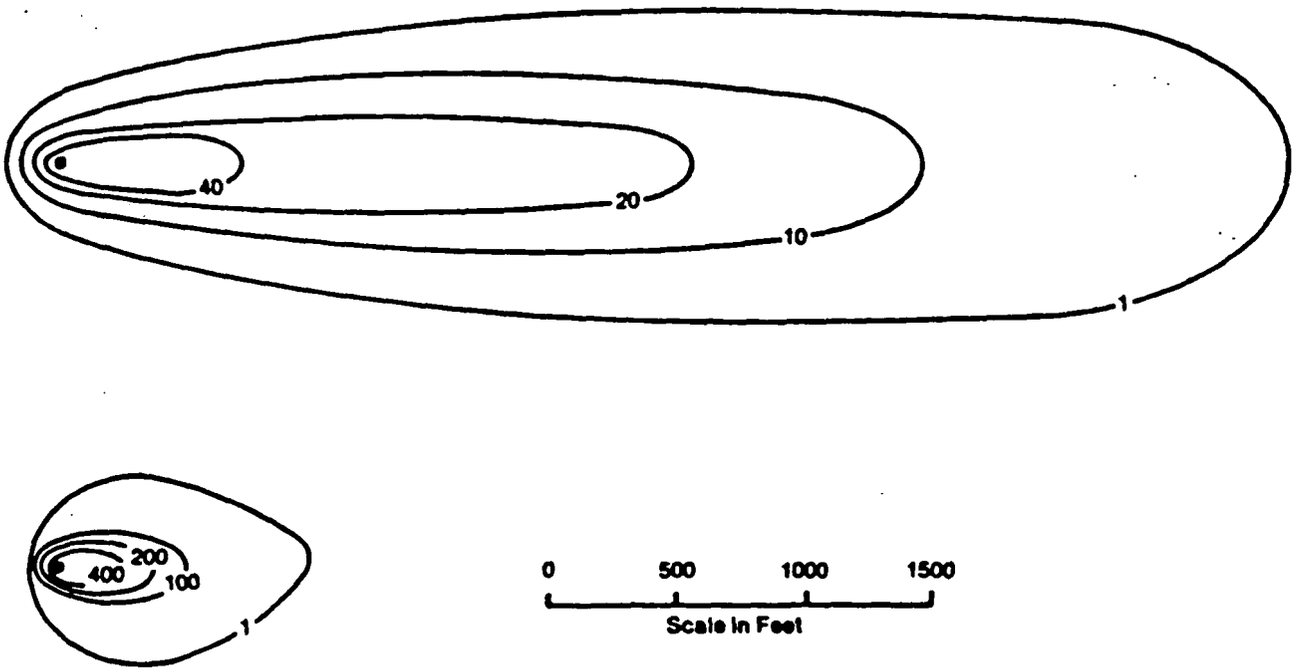

Figure 5-17. The Size and Concentration Distribution in a Contaminant Plume is Related to Ground-Water Velocity. Upper Plume Velocity is 1.5 feet/day; in lower Plume Velocity is 0.5 feet/day

not be readily solved and herein lies one of the major stumbling blocks of chemical transport models.

Often confused with the term dispersion (D_x = longitudinal dispersion and D_y = transverse dispersion) is dispersivity. Dispersion includes velocity: to transform from one to another requires either division or multiplication by velocity.

The rate of advance of a contaminant plume can be retarded if there is a reaction between its components and ground-water constituents or if sorption occurs. This is called retardation (R_d). The plume in which sorption and chemical reactions occur generally will expand more slowly and the concentration will be lower than the plume of an equivalent nonreactive leachate.

Hydrodynamic dispersion affects all solutes equally while sorption, chemical reactions, and microbial degradation affects specific constituents at different rates. As Figure 5-18 shows, a leachate source that contains a number of different solutes can have several solutes moving at different rates due to the attenuation processes.

The areal extent of plumes may range within rather wide extremes depending on the local geologic conditions, influences on the hydraulic gradient, such as pumping, ground-water velocity, and changes in the time rate of release of contaminants.

The many complex factors that control the movement of leachate and the overall behavior of contaminant plumes are difficult to assess because the final effect represents several factors integrated collectively. Likewise, concentrations for each constituent in a complex waste are difficult to obtain. Therefore, predictions of concentration and plume geometry, at best, can only be used as estimates, principally to identify whether or not a plume might develop at a site and, if so, to what extent. Models can be used to study plume migration, and as an aid in determining potential locations for monitoring wells, and to test various renovation or restoration schemes.

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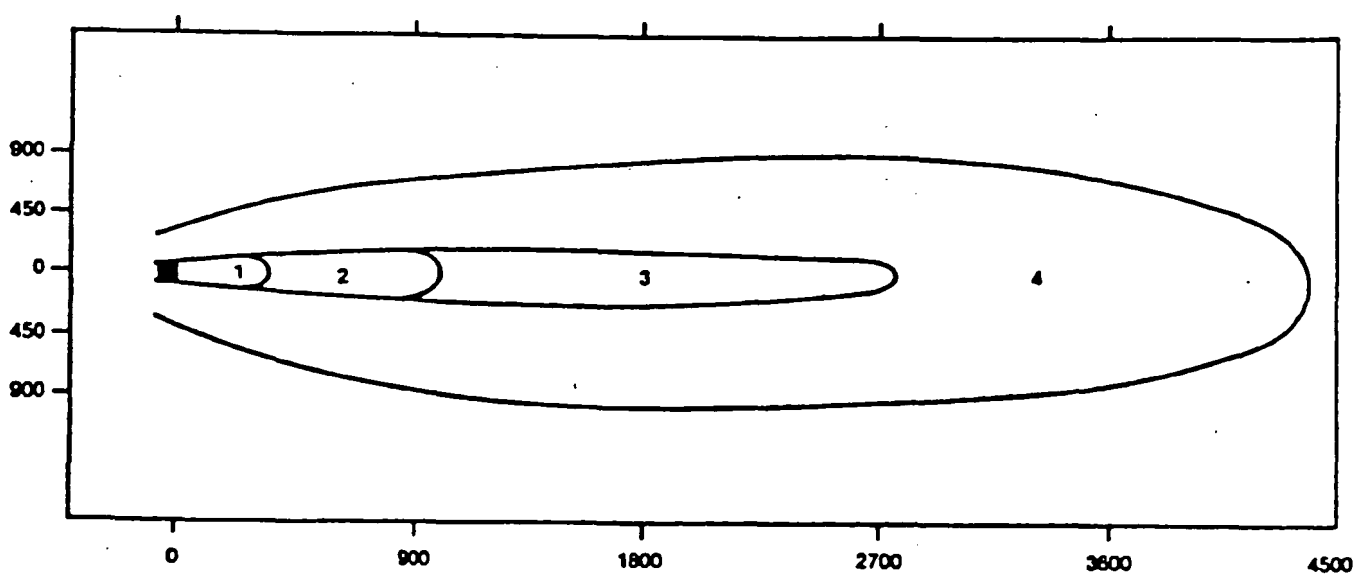

1. Chlorobenzene R_d = 35
2. Unknown = 15
3. Chloroform = 3
4. Chloride = 1

Plumes after 2800 days

Figure 5-18. Constituents Move at Different Rates Because of Retardation

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Chapter 6

GROUND-WATER INVESTIGATIONS

Introduction

Within the last decade, a substantial number of ground-water investigations have been conducted. Many of these have been centered at specific contaminated sites in response to federal legislation concerned with sources of drinking water, threats to human health and the environment posed by toxic and hazardous waste, and the restoration of contaminated aquifers. In general, most of the sites consist of only several acres or a few square miles, but a number of reconnaissance studies have focused on thousands of square miles.

In most cases the cost of these investigations has been excessively high, due in large measure to the expense of analytical services. The most disconcerting feature of many of these investigations is that their results were found to be inadequate, and additional work and expense were required. It must be understood that a data base will almost always be inadequate to some and its resolution will eventually be dictated by time, common sense, and budgetary constraints. Although these constraints will always be present to one degree or another, it is imperative that the most reliable and applicable information be collected commensurate with the available resources.

The reason many field investigations are both inadequate and costly is that a comprehensive work plan was not developed before the project was initiated, or that it was not followed. Any type of an investigation must be carefully planned, keeping in mind the overall purpose, time limitations, and available resources. Moreover, the plan must use a practical approach based on sound, fundamental principles. As far as ground-water quality investigations are concerned, the basic questions are (1) is there a problem, (2) where is it, and (3) how severe is it? A subsequent question may relate to what can be done to reduce the severity, that is, aquifer restoration.

Ground-water quality investigations can be divided into three general types: regional, local, and site evaluation. The first, which may encompass several hundred or even thousands of square miles, is reconnaissance in nature, and is used to obtain an overall evaluation of the ground-water situation. A local investigation is conducted in the vicinity of a contaminated site, may cover a few tens or hundreds of square miles, and is used to determine local ground-water conditions. The purpose of the site evaluation is to ascertain, with a considerable degree of certainty, the extent of contamination, its source or sources, hydraulic properties, and velocity, as well as all of the other related controls on contaminant migration.

Ground-water investigations can be quite varied in terms of purpose as well as scale and duration. Although a few of these variations will be discussed briefly, the main topic of this chapter will be site specific ground-water investigations involving contamination with toxic and hazardous wastes.

Purposes of Ground-Water Investigations

Ground-water investigations are conducted for a variety of purposes. One is for reconnaissance or the establishment of background quality, such as those done by the U. S. Geological Survey for many years, which resulted in a historical documentation of the quality and quantity of both surface and subsurface waters. Usually these investigations are made using existing private, municipal, industrial, and irrigation wells. The data are useful for determining fluctuations, trends, and cycles in water levels and chemical quality.

Another purpose may be to monitor a variety of ground-water parameters in order to establish cause and effect relationships, as for example, an assessment of the design, construction and operation of a hazardous waste disposal facility on areal ground-water quality. Monitoring may be done to assure the integrity of lagoon liners or, in general, prove compliance with any of the regulatory standards dealing with waste disposal,

storage, or treatment facilities. Ground-water quality monitoring also is increasing with respect to possible contaminant sources, such as underground storage tanks, the application of agricultural chemicals, and mining, just to mention a few.

Ground-water investigations traditionally have played an important role in litigation. Under the civil laws of trespass or negligence, the information obtained during a study may be used to determine the source of ground-water contamination in order to establish liability, or be used in response to federal legislation, such as RCRA and Superfund. In these cases particular attention must be given to the handling of samples as well as the documentation of field and laboratory procedures. During the beginning of the study, legal counsel should be obtained to assure that proper procedures are built into the work plan.

Perhaps the most specialized types of ground-water investigations are those driven by research objectives. The goals of these studies are as varied as the nature of research itself, and may range from model validation to determining the rates and daughter products of contaminant degradation. Specialized field equipment and technologies often are required to obtain representative samples of subsurface materials for use in column and microcosm studies. Usually more observation points are required for research studies than for other types of ground-water investigations, as are the demands for more stringent quality control.

Types of Ground-Water Investigations

Regional Investigations

Ground-water investigations can be carried out on a regional, local, or site-specific scale. The first, which may encompass hundreds or even thousands of square miles, is reconnaissance in nature, and is used to obtain an overall evaluation of a ground-water situation.

This broad-brush reconnaissance study can be the starting point for two general types of investigations. First, it can be carried out with the purpose of locating potential sources of contamination, or it may provide an understanding of the occurrence and availability of ground water on a regional scale. The underlying objectives are first, to determine if a problem exists, and second, if necessary, to ascertain prevalent hydrologic properties of earth materials, generalized flow directions of both major and minor aquifers, the primary sources and rates of recharge and discharge, the chemical quality of the aquifers and surface water, and the locations and yields of wells. These data can be useful in more detailed studies because they provide information on the geology and flow direction, both of which affect studies of smaller scale.

Local Investigations

A local investigation, which is conducted in the vicinity of a contaminated site, may cover a few tens or hundreds of square miles, and is used to determine local ground-water conditions. The purpose is to define, in greater detail, the geology, hydrology, and water quality in the area surrounding a specific site or sites of concern. This information is important in designing and carrying out more detailed site investigations.

Site Investigations

The goals of an investigation at a contaminated site are to ascertain, with considerable certainty, the nature and extent of contamination, its source or sources, and the relative movement of different contaminants and their degradation products. The end result is to provide information leading to an effective and cost-efficient remediation plan.

The site investigation is usually the most detailed, complex, costly, and, from the legal and restoration viewpoint, the most critical of the three types of ground-water studies. A site investigation must address a myriad of pertinent parameters affecting contaminant transport and transformation, including geology and hydrogeology, geochemical interactions, biotic and abiotic degradation processes, and the rate of movement of contaminants through the unsaturated and saturated zones. It also is important, when appropriate, to locate and determine the effect of phenomena influencing the movement of contaminant plumes such as nearby pumping wells, multiaquifer interactions, and local streams.

At the same time ground-water studies are being carried out there are usually auxiliary investigations. These may include tank inventories, toxicological evaluations, air pollution monitoring, manifest scrutiny, and manufacturing procedures, as well as other information gathering, all of which eventually combine in the development of a comprehensive report.

Organization and Development of the Investigation

Regardless of the complexity or detail of the investigation, a logical series of steps should be followed. Although each investigation is unique, these general rules are:

1. Establish objectives.
2. Prepare work plan
3. Data collection.

4. Data interpretation.
5. Develop conclusions.
6. Present results.

Establish Objectives

Establishing the major goal or goals of an investigation is paramount to a successful and cost-effective project. The exact goals should be clearly defined and agreed upon by all interested parties. They should be clearly expressed in writing and referred to often during the life of the study. Otherwise as the work progresses, there may be a tendency for the study to drift from the stated objectives, resulting in the collection of costly superfluous information, perhaps at the expense of required information.

The approach, time requirements, and funding can be vastly different between a regional reconnaissance evaluation and a site-investigation. The former, which deals with gross features, may only require a few days, while the latter, which necessitates minute detail, may demand years. In either case the time and resource requirements are dictated by the goals, and the success of the work is measured by how directly the investigation pursues those goals.

In one case the objective statement may be to "measure the water levels in a given township using existing wells." Another might be "evaluate the degradation rate of tetrachloroethylene at a specific spill site, define the plumes of the parent and degradation contaminants, and predict the location and concentrations of these contaminants after 10 years." In both of these examples the objective is clearly stated and the complexity is evident. In the first case, the caveat "using existing wells" states that the study, for whatever purpose, is very limited. Clearly the second set of goals is vastly more complicated and will undoubtedly require many observation points; a detailed knowledge of the site's soils, geochemistry, geology, and hydrogeology, sophisticated analytical capabilities; predictive models and the information necessary to drive them.

Once the general objective of the ground-water study is established, a number of secondary purposes must be considered. These involve the physical system and the chemical aspects. Secondary objectives include the following:

1. Determination of the thickness, soil characteristics, infiltration rate, and water-bearing properties of the unsaturated zone.
2. Determination of the geologic and hydrologic properties and dimensions of each geologic unit that potentially could be affected by ground-water contamination. This includes rock type, thickness of aquifers and confining units, their areal distribution, structural configuration, transmissivity, hydraulic conductivity, storativity, water levels, infiltration or leakage rate, and rate of evapotranspiration, if appropriate.
3. Determination of recharge and discharge areas, if appropriate.
4. Determination of the direction and rate of ground-water movement in potentially affected units.
5. Determination of the ground water and surface water relationships.
6. Determination of the background water-quality characteristics of potentially affected units.
7. Determination of potential sources of contamination and types of contaminants.

Prepare Work Plan

The preparation of the work plan or method of approach should be made in direct response to the stated goals, using existing data and information to the fullest extent possible. The investigative plan needs to be flexible in a practical way. For example, the position of all test wells, borings, and monitoring points cannot be determined in the office at the start of an investigation. Rather, these locations should be adjusted on the basis of information obtained as each hole is completed. In this way, one can maximize the data acquired from each drill site and more appropriately locate futures holes in order to develop a better understanding of the ground water and contamination situation at the site under study.

Similarly, the exact contaminants of target, appropriate analytical methods, detailed sampling techniques, and the required number of samples cannot be accurately estimated at the beginning of a project. These must be refined as data are collected and the statistics of those data interpreted.

The early development of a flexible plan of investigation occasionally may be required to include, at least in part, guidelines established by the Environmental Protection Agency, such as the Ground Water Technical Enforcement Guidance Document. State regulatory agencies may have even more stringent requirements. Also, in the case of Superfund and RCRA sites, the investigator probably will be required to work with or at least use data collected by consultants for the defendant.

In almost all cases, as the work progresses, it is necessary to adjust the work plan to one degree or another. In the event changes must be made, it is important that they do not cause the work to drift from the original objectives.

Even fairly simple ground-water investigations can result in large amounts of data, adjustment of the project

approach, statistical evaluations, interpretations and conclusions, the preparation of graphics for presentations, and the final report. The work plan should contain provisions for dealing with data either by developing an automatic data processing program or selecting one from the software market. Also, if the project requires the use of mathematical models, data storage and retrieval systems should be developed in concert with these needs.

One section of the work plan should be dedicated to the health and safety of those actively participating in the investigation, as well as the general public. Health monitoring tests, performed before, during, and after the field work is completed, are necessarily predicated on estimates of the toxicity and concentration of contaminants at the site. Protective clothing and other safety considerations also must be based on these estimates until collected information becomes available. Access to the site should be limited to project personnel, particularly when drilling or other heavy equipment is in use.

Another section of the work plan should deal with chain-of-custody requirements when working at Superfund, RCRA, or other sites where litigation is involved. As discussed in EPA's Technical Enforcement Guidance Document, this section should include instructions concerning:

1. Sample labels to prevent misidentification of samples.
2. Sample seals to preserve the integrity of samples from the time they are collected until opened in the laboratory.
3. Field logbook to record information about each sample collected during the ground-water monitoring program.
4. Chain-of-custody record to document sample possession from time of collection to analysis.
5. Sample analysis request sheets, which serve as official communications to the laboratory of the particular analyses required for each sample and provide further evidence that the chain of custody is complete.
6. Laboratory logbook and analysis notebooks, which are maintained at the laboratory and record all pertinent information about the sample.

Data Collection

Existing Information. Data collection forms the basis for the entire investigation, consequently, time must be allocated and care exercised in addressing this part of the project. As mentioned above, all existing information should be collected, analyzed, and used to prepare a work plan before field activities are begun. The amount and types of data to be collected are dictated by the objectives of the study. Materials that should be collected, when available, include soil, geologic, topographic, county and state maps, geologic cross-sections, aerial photographs, satellite imagery, the location of all types of wells with discharge rates, well logs, climatological and stream discharge records, chemical data, and the location of potential sources of ground-water contamination.

Many of these data are readily available in the files and reports of local, state, and federal agencies. Personnel with these agencies also can be of great help because of their knowledge with the area and available literature. Examples include the U.S. Geological Survey, which has at least one office in each state, the state geological survey, and several state agencies that deal with water, such as the state water survey, water resources board, or a water commission. Other sources of information include the state or federal departments of agriculture, soil conservation, and the weather service, among others.

It often is useful to talk with long-term local residents, realizing that their information may be biased because of prejudices involving the cause of the investigation. Their historical knowledge often can assist in defining possible sources of contamination. For example, "there used to be a service station on that corner about 30 years ago," or "that company buried trash out in that field until after World War II." Often their memory is of events that are not available in the literature.

Climatological data are important because they indicate precipitation events and patterns, which influence surface runoff and ground-water recharge. Additionally, these data include temperature measurements that can be used for an evaluation of evapotranspiration, which, for shallow ground water can produce a significant effect on the water-table gradient, causing it to change in slope and direction, both seasonally and diurnally.

Soil types are related to the original rock from which they were derived. Consequently, soils maps can be used as an aid in geologic mapping, and they are valuable for estimating infiltration. Soil information also is necessary to evaluate the potential for movement of organic and inorganic compounds through the unsaturated zone.

Exceedingly useful tools, both in office and field study, are aerial photographs and satellite imagery. The latter should be examined first in an attempt to detect trends of lineaments, which may indicate the presence of faults and major joints or joint systems. These may reflect zones of high permeability that exert a strong influence

on fluid movement from the land surface or through the subsurface. Satellite imagery also can be used to detect the presence of shallow ground water owing to the subtle tonal changes and differences in vegetation brought about by a higher moisture content. Rock types also may be evident on imagery.

Aerial photographs, particularly stereoscopic pairs, should be an essential ingredient of any hydrologic investigation. They are necessary to further refine the trends of lineaments, map rock units, determine the location of cultural features and land use, locate springs and seeps, as well as potential drilling sites, and detect possible sources of contamination. Topographic and state and county road maps also are useful for many of these purposes.

Geologic reports, maps, and cross sections provide details of the surface and subsurface, including the areal extent, thickness, composition, and structure of rock units. These sources of information should be supplemented, if possible, by an examination of the logs of wells and test holes. Depending on the detail of the logs, they may provide a clear insight into the complexities of the subsurface.

Logs of wells and test holes are essential in ground-water investigations. They provide first-hand information on subsurface strata, their thickness, and areal extent. They also may allow inferences as to relative permeability, well-construction details, and water-level depths.

Inorganic chemical data may be available from reports, but the most recent information is probably stored in local, state, and federal files. Concentrations of selected constituents, such as dissolved solids, specific conductance, chloride, and sulfate, may be plotted on base maps and used to estimate background quality and, perhaps, indicate areas of contamination.

Sources of information that report concentrations of organic compounds usually are scarce and should be questioned, particularly if they are old. It only has been within the last decade or so that organic compounds have become of concern in ground water. The cost of analysis is high, and much remains to be learned about appropriate sampling methods, storage, and interpretation. Consequently, when using existing data, investigators normally will need to rely on inorganic substances to detect contaminated ground-water sites. In some cases both organic and inorganic substances are present in a leachate. On the other hand, reliance on concentrations of inorganic constituents to evaluate contamination by organic compounds may not be appropriate, possible, or desirable.

Field Investigations. Several generalized methods have been available for a number of years to evaluate a possible or existing site relative to the potential for ground-water contamination. These rating techniques are valuable, in a qualitative sense, for the formulation of a detailed investigation. One of the most noted is the LeGrand (1983) system, which takes into account the hydraulic conductivity, sorption, thickness of the water-table aquifer, position and gradient of the water table, topography, and distance between a source of contamination and a well or receiving stream. The LeGrand system was modified by the U.S. Environmental Agency (1983) for the Surface Impoundment Assessment study.

Fenn and others (1975) formulated a water balance method to predict leachate generation at solid waste disposal sites. Gibb and others (1983) devised a technique to set priorities for existing sites relative to their threat to health. An environmental contamination ranking system was developed by the Michigan Department of Natural Resources (1983). On a larger scale DRASTIC, prepared by the National Water Well Association for EPA (1987), is a method to evaluate the potential for ground-water contamination based on the hydrogeologic setting. A methodology for the development of a ground-water management and aquifer protection plan was described by Pettyjohn (1989).

The field phase of a ground-water investigation is the most intensive and important part of the project. The data collected during this phase will determine its success. Some of the main factors affecting the quality of the field data include an understanding of the hydrogeology at the site, a knowledge of the types of contaminants involved and their behavior in the subsurface, the location and construction of monitoring wells, and how they are sampled and analyzed.

In order to detect and outline areas of contamination in the subsurface, an understanding of the movement of ground water is necessary. In soils the important parameters to quantify by field investigations include soil-moisture characteristic curves, soil texture, unsaturated hydraulic conductivity curves and preferential flow paths, such as fractures and macropores, and the spatial and temporal variability of these factors.

In the saturated zone it is important to determine the hydraulic properties of the aquifer including gradient, direction of flow and velocity, storativity, transmissivity, and hydraulic conductivity.

With respect to water levels and flow patterns, the factors that affect seasonal and temporal variations should be identified. Such factors include onsite and offsite pumping and recharge, tidal and stream stage fluctuations, construction, changes in land use, and waste disposal practices.

In addition to determining the gross hydraulic conductivity, its distribution also should be determined. Variations in hydraulic conductivity, both within and between strata, affect ground-water flow paths, including magnitude and direction, and must be identified in order to isolate major zones of contaminant migration.

If the aquifer is composed of a fractured media, the nature of the fractures is required for use in flow models. Although this information can be very difficult to obtain, it can be helpful to collect information describing fracture density, location, orientation, roughness, and the degree to which they are connected.

A number of techniques are available to measure the hydraulic properties of aquifers. A few of the techniques include the following:

1. Aquifer tests are performed by pumping from one well and observing the resulting drawdown in nearby wells. These analyses can be used to determine an aquifers coefficients of storativity and transmissivity.
2. Slug tests are conducted by suddenly removing or adding a known volume of water from a well and observing the return of the water level to its original location. Slug tests are used to determine hydraulic conductivity.
3. Flow-net analyses, both horizontal and vertical, also permit an evaluation of hydraulic parameters and flow directions, and aid in the understanding of the role played by strata of different permeability.
4. Tracer tests can be used to determine if two locations are hydraulically connected, measure flow velocities, and determine the variability of hydraulic conductivity within an aquifer system.
5. Borehole dilution tests can be used to determine the hydraulic conductivity in a single well by introducing a tracer and measuring the dilution with time caused by the inflow of water into the well that is brought about by the natural hydraulic gradient in the vicinity of the well.
6. Rock cores taken during the drilling of wells and test holes can be analyzed in the laboratory to determine a number of physical and chemical properties, including porosity, hydraulic conductivity, and mineralogy. Care must be used in an evaluation of hydraulic parameters determined by laboratory analyses of unconsolidated materials.
7. Surface and borehole geophysics, aerial photography, and imagery are particularly helpful in working with fractured media.

After the site has been described in terms of ground-water movement, the work plan can be adjusted to sample for contaminants in the unsaturated and saturated zones. Nested lysimeters can be used to detect contaminants in the unsaturated zone; however, great care must be taken to assure that the collected samples are representative and not affected by sorption and volatilization. The placement of nested piezometers in closely spaced, separate boreholes of different depths generally is preferred to determine vertical head differences and the vertical movement of contaminants, while monitoring wells with appropriately located screens are used to determine the lateral movement of contaminants in the saturated zone.

As discussed above, an understanding of the variability or distribution of hydraulic conductivity, in both the vertical and horizontal dimension, allows one to isolate the major zones of water transmission and, therefore, to select the proper lengths and depths for well screens. This follows for offsite, upgradient, and downgradient observation points.

The length and position of well screens also must be predicated on the nature of the contaminant. For example, if the contaminants are miscible with the liquid phase, it may be possible to use only one well per sampling point. It also may be possible to use only one well if the transmissive zone is very thin. If the contaminants are immiscible with the liquid phase (sinkers or floaters), the well screens must be located appropriately.

In carrying out a ground-water investigation it is not uncommon for at least part of the chemical species of concern to be dictated by state or federal regulations, such as the RCRA list of priority pollutants. Beyond this one must be aware of contaminant transformation phenomena in the design and implementation of a ground-water sampling program. For example, when selecting proper contaminant targets it is imperative to realize that the original species may have been reduced in concentration, altered, or eliminated by chemical, physical, or biological processes taking place in the subsurface environment. Aerobic and anaerobic biological degradation, and hydrolysis and redox reactions are among these processes. The sampling protocol also should be influenced by alterations in the transport of contaminants caused by immiscible compounds, sorption-desorption phenomena, and the facilitated transport of hydrophobic compounds.

Data Interpretation

Data interpretation should begin with the development of the work plan. Today's widespread availability and use of computers allow the application of data processing to the results of almost all investigations and software exists for a wide variety of data handling requirements. To the extent possible, the amounts and types of data should be anticipated early in the project and provisions made for the continuous input of collected information as the work progresses. Also, the quality assurance and quality control program should be built into the data handling system so that the quality of the data can be continuously monitored.

If predictive models are required at some point in the investigation, or later in the development of an aquifer remediation project, they should be formulated or selected from existing models as early as possible so that requirements for acquisition of the appropriate data can be built into the work plan. Steps also should be taken for model calibration and validation as the investigation proceeds.

Even at a moderately sized site, a ground-water investigation of limited scope can result in the collection of a great deal of information. The amount of time saved and the amount of frustration avoided during data interpretation is directly proportional to the skill with which one anticipates (1) the types and amounts of data collected; (2) the calculations required to determine contaminant transformation process rates, support conclusions, and make projections; (3) the correlations required to prove cause and effect, define relationships, and determine reaction coefficients; and (4) prepare the graphic displays needed for reports and presentations.

Develop Conclusions

In a very real sense the development of conclusions, like preparing for the interpretation of data, should be done in the early stages of the project by establishing hypotheses. These hypotheses must be proposed in direct response to the objectives of the investigation, then, as in hypothesis testing in statistics, the project designed around their acceptance or rejection. If done correctly, this approach can play a significant role in assuring that the project design is an efficient response to the project goals, and that the collection of extraneous information is kept to a minimum.

To carry this point further, assume that gasoline fumes are detected in the basement of a small house. A service station is located immediately to the east at a slightly higher elevation. There is another service station about 200 feet south of the first, across a street. The goal of an investigation would be to determine the source of gasoline so that negligence could be proven. If one assumed that the shallow water table followed the surface topography, the first hypothesis would be that the gasoline originated from the closest, upgradient buried tanks. After drilling only three shallow wells, water-level data might prove that the ground water was moving due west from the closest station and the first hypothesis could be accepted.

On the other hand, the water-level data could show that the gradient did not follow the lay of the land but was about 30 degrees west of north. In this case the first hypothesis must be rejected with the conclusion now being that the second service station is at fault. At this point the investigation might be ended or additional proof provided by drilling wells to delineate the plume and show that no other sources existed.

A more complicated example might involve the need to define the plume of contamination at a Superfund site so that a remediation plan could be developed. The goal would be to locate the plume horizontally, as well as vertically, and provide concentration isograms. If the parent contaminant were trichloroethylene, the hypothesis must be made that biodegradation is taking place and that the well placements, sampling, and analytical procedures must be designed to also locate dichloroethylene and vinyl chloride.

Many of today's ground-water contamination problems are extremely complex, particularly those associated with hazardous waste sites. It is very important, therefore, that conclusions be based on the collective wisdom and experience of interdisciplinary teams to the fullest extent possible.

Present Results

All investigations usually result in a report and commonly other types of presentations as well. Their style and content are determined by the type of study and can vary in as many ways as the investigations themselves. However, some general traits can be suggested.

Those studies designed to "Establish Background" and those for "Monitoring" cause-and-effect relationships should consist predominantly of field data appropriately grouped and tabulated for easy access. Reports prepared for use in litigation are usually brief with only the essentials of the study highlighted along with the essence of the findings—most often in proof or disproof of a legal argument. "Site Characterization" reports generally are more complete and detailed than other reports because they generally serve as the basis of

other activities, such as the design and implementation of a remedial action plan or a complex and costly compliance monitoring system.

Examples of Ground-Water Investigations

Regional Examples

Regional investigations are conducted for many different purposes. One type is to detect potential sources and locations of ground-water contamination. Another type of exceedingly broad scope includes library searches. Examples entail an early EPA effort to evaluate ground-water contamination throughout the United States (van der Leeden and others, 1975, Miller and Hackenberry, 1977, Scalf and others, 1973, Miller and others, 1974, and Fuhriman and Barton, 1971). The reports are useful for obtaining a general appreciation of the major sources and magnitude of contamination over a regionally extensive area.

In 1980, individuals in EPA Region VII became aware of what appeared to be a large number of wells that contained excessive concentrations of nitrate. Suspecting a widespread problem, a regional reconnaissance investigation was initiated. The general approach consisted of a literature search, a meeting in each state with regulatory and health personnel, an evaluation of existing data, and an interpretation of all of the input values.

The fundamental principle guiding this study was the fact that abnormal concentrations of nitrate can arise in a variety of ways, both from natural and human-made sources or activities. The degradation may encompass a large area if it results from the over-application of fertilizer and irrigation water on a coarse textured soil, from land treatment of waste waters, or from a change in land use, such as converting grasslands to irrigated plots. On the other hand, it may be a local problem affecting only a single well if the contamination is the result of animal feedlots, municipal and industrial waste treatment facilities, or improper well construction or maintenance.

Most of the data base for this study was obtained from STORET. First, nitrate concentrations in well waters were placed in a separate computer file. Two maps were generated from the file, the first showing the density of wells that had been sampled for nitrate, and the second showing the density of wells that exceeded 10 mg/L of nitrate (fig. 6-1). These maps indicated the areas of the most significant nitrate problems. In turn, the nitrate distribution maps were compared to geologic maps, which allowed some general identification of the physical system that was or appeared to be impacted (fig. 6-2).

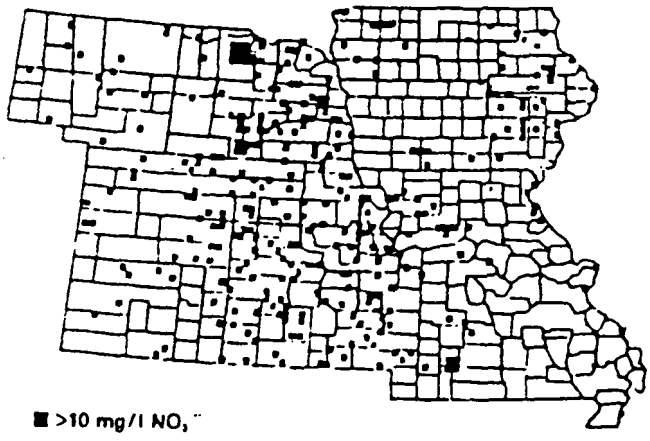

Figure 6-1. Location of Wells with Nitrate Exceeding 10mg/L in Region VII

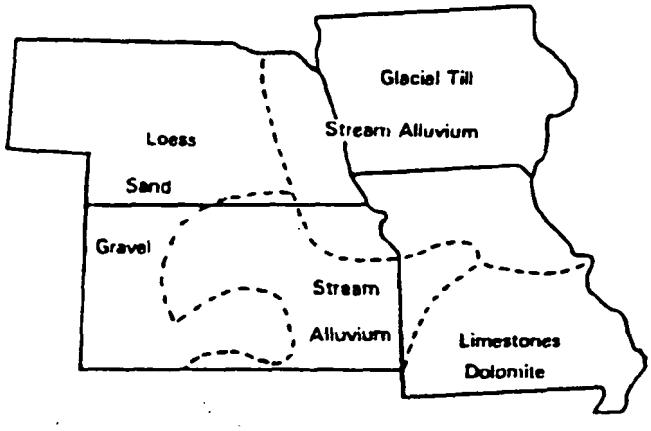

Figure 6-2. Generalized Rock Types with High Nitrate Concentrations in Region VII

Iowa, eastern Nebraska, northeastern Kansas, and the northern third of Missouri are characterized by glacial till interbedded with local deposits of outwash. Throughout the area are extensive deposits of alluvium. Many of the aquifers are shallow and wells are commonly dug, bored, or jetted. This area contained the greatest number of domestic wells with high nitrate concentrations. It also contained the greatest number of municipal wells that exceeded the nitrate Maximum Contaminant Level (MCL). The cause of contamination in the shallow domestic wells was suspected to be poor well construction and maintenance, but this was possibly not the case for many of the generally deeper municipal wells, where the origin appeared to be from naturally occurring sources in the glacial till.

Most of Nebraska and western Kansas are mantled by sand, gravel, and silt, which allow rapid infiltration. The water table is relatively shallow. The irrigated part of this region, particularly adjacent to the Platte River and in areas of Holt County, NB, contained the greatest regional nitrate concentrations in the four state area. This was brought about by the excessive application of fertilizers and irrigation waters in this very permeable area.

The remaining area in Kansas and an adjacent part of Missouri is underlain by sedimentary rocks across which flow many streams and rivers with extensive flood plains. Most of the contaminated wells tapped alluvial deposits. The primary cause of high nitrate in domestic wells was suspected to be poor well construction and maintenance, or poor siting with respect to feedlots, barnyards, and septic tanks.

The southern part of Missouri is represented by carbonate rocks containing solution openings. Aquifers in these rocks are especially susceptible to contamination and the contaminants can be transmitted great distances with practically no change in chemistry other than dilution. The carbonate terrain is not easily managable nor is monitoring a simple technique because of the vast number of possible entry sites whereby contaminants can enter the subsurface.

The STORET file also was used to generate a number of graphs of nitrate concentration versus time for all of the wells that were represented by multiple samples. The graphs clearly showed that the nitrate concentration in the majority of wells ranged within wide limits from one sampling period to the next, suggesting leaching of nitrate during rainy periods from the unsaturated zone.

The state seminars were exceedingly useful because the personnel representing a number of both state and federal agencies had a good working knowledge of the geology, water quality, and land-use activities of their respective states.

Although the study extended over several months, the actual time expended amounted to only a few days. The conclusions, for the most part, were straightforward and, in some cases, pointed out avenues for improvement in sample collection and data storage/ access. The major conclusions are as follows:

1. High levels of nitrate in ground water appeared to be randomly distributed throughout the region.

2. The most common cause of high nitrate concentration in wells was the result of inadequate well construction, maintenance, and siting. Adequate well construction codes could solve this problem. Dug wells, those improperly sealed, and wells that lie within an obvious source of contamination, such as a pig lot, should probably be abandoned and plugged.

3. In areas of extensive irrigation where excess water was applied to coarse textured soils, the nitrate concentration in ground water appeared to be increasing.

4. In the western part of the region, changes in land use, particularly the cultivation or irrigation of grasslands, had resulted in leaching of substantial amounts of naturally occurring nitrate from the unsaturated zone.

5. The population that was consuming high-nitrate water supplies was small, accounting for less than 2 per cent of the population.

6. There had been no more than two reported cases of methemoglobinemia in the entire Region within the preceeding 15 years despite the apparent increase in nitrate concentration in ground-water supplies. This implied a limited health hazard.

7. State agency personnel were convinced that they did not have significant nitrate-related health problems

8. Many of the wells used in state and federal monitoring networks are of questionable value because little or nothing is known about their construction.

9. The volume of chemical data presently in the files of most of the state agencies within the region is not adequately represented in the STORET data system.

This cursory examination provided only a general impression of the occurrence, source, and cause of abnormal nitrate concentrations in ground water in the Region. Nonetheless, it furnished a base for planning local or site investigations, was prepared quickly, and did not require field work or extensive data collection.

As mentioned previously, the source of excessive nitrate in many municipal wells could not be readily explained. There could be multiple sources related to naturally occurring high nitrate concentrations in the unsaturated zone or the glacial till, to contamination, or to poor well constructioin. Definitive answers would require more detailed local or site studies.

The overall effect of changing from grazing land to irrigated agriculture, in view of the great mass of nitrate in the unsaturated zone, warrants additional local investigation. Although the concentration of nitrate in underlying ground water would increase following irrigation, it is likely that some control on the rate of leaching could be implemented by limiting the amount of water applied to the fields.

The obvious relationship between the application of excessive amounts of fertilizer and water on a coarse

textured, as was the case in Nebraska, shows the need for experimental work on irrigation techniques in order to reduce the loading. Also implied is the necessity for the development of educational materials and seminars to offer means whereby irrigators can reduce water, pesticide, and fertilizer applications, and yet maintain a high yield.

Local Example

Local investigations can be as varied in scope and areal extent as regional evaluations and the difference between the two is relative. For example, one might desire to obtain some knowledge on the hydrogeology of an area encompassing a few tens or several hundred square miles in order to evaluate the effect of oil-field brine production and disposal. Examples of this scope include Kaufmann (1978) and Oklahoma Water Resources Board (1975). The other extreme may center around a single contaminated well. In this case the local investigation would most likely focus on the area influenced by the cone of depression, the size of which depends of the geology, hydraulic properties, and well discharge.

Consider an area in the Great Plains where a number of small municipalities have reported that some of their wells tend to increase in chloride content over a period of months to years. The increase in a few wells has been sufficient to cause abandonment of one or more wells in the field. Additionally, a number of wells when drilled yielded brackish or salty water necessitating additional drilling elsewhere. This is an expensive process that strains the operating budget of a small community.

In this case, a local investigation covered an area of about 576 mi^2. A review of files and reports and discussions with municipal officials and state and federal regulatory agencies indicated that the entire area had produced oil and gas for more than 30 years. Inadequate brine disposal appeared to be the most likely cause of the chloride problem.

During the initial stage of the investigation, all files dealing with the quality of municipal well water were examined. This task was followed by a review of the geology, which included a assessment of all existing maps, cross sections, and well logs, both lithologic and geophysical.

The chemical data clearly showed that the chloride content in some wells increased with time, although not linearly. The geologic phase of the study showed that the rocks consist largely of interbedded layers of shale and sandstone and that the sandstone deposits, which serve as the major aquifers, are lenticular and range from 12 to about 100 feet in thickness. The sandstones are fine-grained and cemented to some degree and, as a result, each unit will not yield a large supply. Resultingly, all sandstone strata are screened.

Trending north-south through the east-central part of the area is an anticline (fig. 6-3) that causes the rocks to dip about 50 feet per miles either to the east or west of the strike of the structure (fig. 6-4). This means that a particular sandstone will lie at greater depths with increasing distances from the axis of the anticline.

In this example, the subsurface geology was examined by an evaluation of geophysical and geologists logs of wells and test holes, including oil and gas wells and tests. As shown in Figure 6-4, interpretation of the logs, in the form of a geologic cross section, brings to light an abundance of interesting facts. The municipal wells range in depth from 400 to 900 feet, but greater depth does not necessarily indicate a larger yield nor does depth imply a particular chemical quality. The difference in well depth and yield is related to the thickness and permeability of the sandstone units encountered within the well bore. Secondly, the volume of the sandstone components ranges widely, but the thinnest and most discontinuous units increase in abundance westward. More importantly, the mineral content of the ground

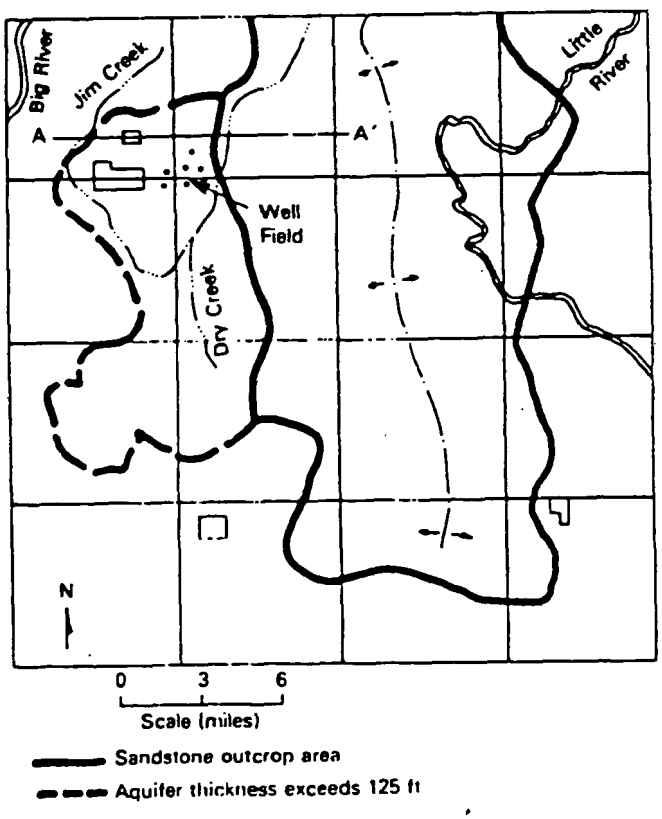

Figure 6-3. Generalized Geologic Map of a Local Investigation

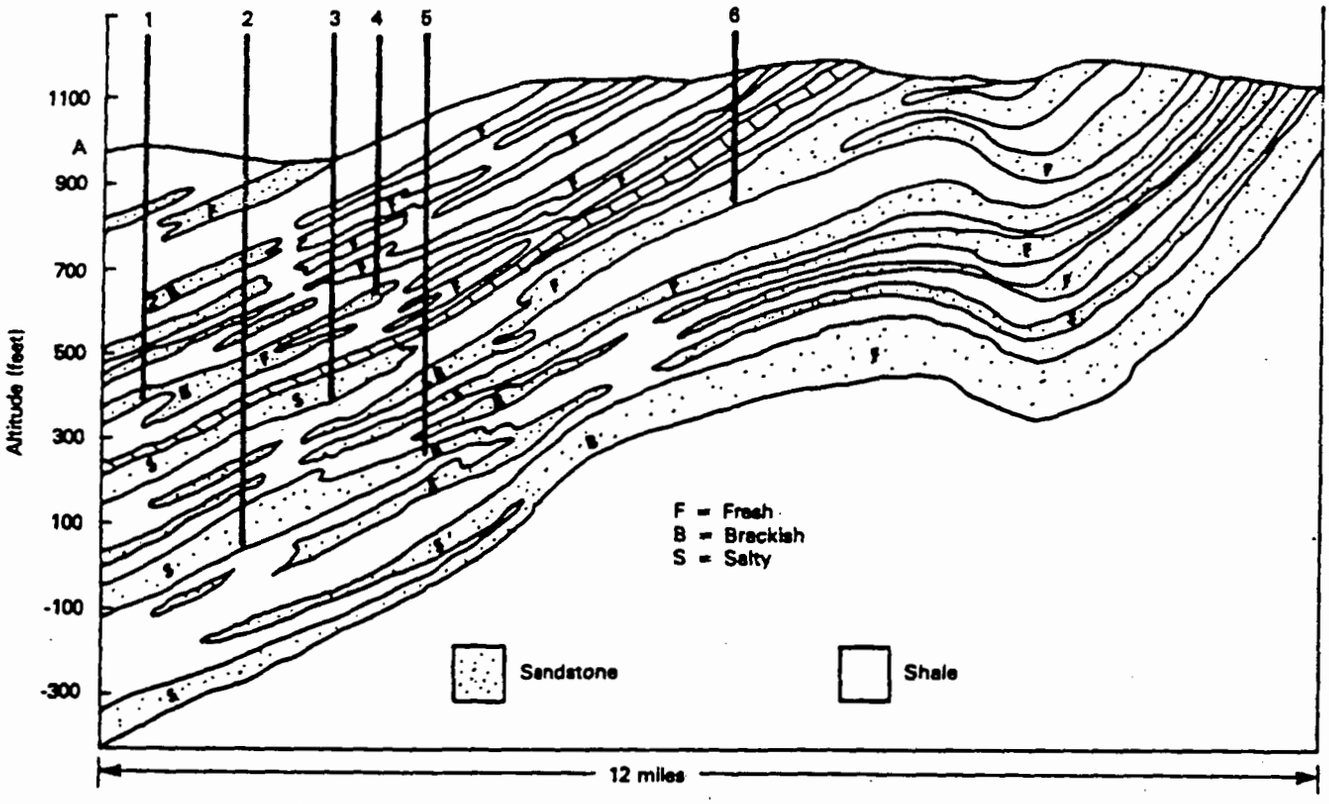

Figure 6-4. Geologic Cross-Section Showing Downdip Change in Water Quality

water, which can be determined from geophysical logs, increases down the dip of the sandstone, from fresh in the outcrop area, to brackish, and finally to salt water (fig. 6-4). Notice also that brackish and saline water lie at increasingly shallower depths to the west of the outcrop area.

The position and depth of a few municipal wells and test holes are also shown on the cross section. Well 1 would be expected to have a small yield of brackish water. Well 2 is an abandoned test hole that penetrated a thick saline zone and a thick brackish water zone. In the case of Well 3, the freshwater derived from the thin, shallower sandstones is sufficient to dilute water derived from the more mineralized zones. On the other hand, as the artesian pressure in the shallow sandstones decreases with pumping and time, an increasing amount of the well yield might be derived from the deeper brackish layer, causing the quality to deteriorate.

The major conclusion derived from this study is that the most readily apparent source of high chloride content in municipal wells, that is, inadequate oil-field brine disposal, is not the culprit. Rather all of the problems are related to natural conditions in the subsurface, brought about by the downdip increase in the dissolved solids content as freshwater grades into brackish and eventually into saline water. Deterioration of municipal well water quality is related to the different zones penetrated by the well and to a decrease in artesian pressure in freshwater zones brought about by pumping. The latter allows updip migration of brackish or saline water. Furthermore, the migration of mineralized water could occur through the well bore or by lateral or vertical leakage from one aquifer to another, which again is the result of a pressure decline in the freshwater zones. The problem could be diminished by constructing future wells eastward toward the axis of the anticline, limiting them to those areas either within the outcrop or where the thickness of the freshwater aquifers comprise a total thickness that exceeds 125 feet (fig. 6-3).

Site Example

Site investigations are ordinarily complex, detailed, and expensive. Furthermore, the results and interpretations are likely to be thoroughly questioned in meetings, interrogatories, and in court, because the expenditure of large sums of money may be at stake. The investigator must exercise extreme care in data collection and interpretation. The early development of a flexible plan of investigation is essential and it must be based, at least in part, on guidelines established by the EPA, such as the Ground-Water Monitoring Technical Enforcement Guidance Document. State regulatory agencies may have even more stringent requirements. In the case of Superfund and RCRA sites, the regulatory

investigator probably will be required to work with or at least use data collected by consultants for the defendant. In some cases, the defendant conducts and pays for the entire investigation; regulatory personnel only modify the work plan so that it meets established guidelines. There are two points to consider in these situations. First, the consultant is hired by the defendant and should act in his best interest. This means that his interpretations may be biased toward his client and concepts detrimental to the client are not likely to be freely given. Second, even though the regulatory investigator and the consultant, to some degree, are adversaries, this does not mean that the consultant is dishonest, ignorant, or that his ideas are incorrect. It must always be remembered that the entire purpose of the investigation is to determine, insofar as possible, what has or is occurring so that effective and efficient corrective action can be undertaken. In the long run cooperation leads to success.

As an example of a ground-water quality site investigation, consider a rather small refinery that has been in existence for several decades. For some regulatory reason an examination of the site is required. The facility, which has not been in operation for several years, includes an area of about 245 acres. The geology consists of alternating layers of sandstone and shale that dip slightly to the west; the upper 20 to 30 feet of the rocks are weathered.

Potential sources of ground-water contamination include wastewater treatment ponds, a land treatment unit, a surface runoff collection pond, and a considerable number of crude and product storage tanks. Line sources of potential contaminants include unimproved roads, railroad lines, and a small ephemeral stream that carries surface runoff from the plant property to a holding pond.

After considering the topography and potential sources of contamination, the location of 11 test borings was established. The purpose of the holes was to determine the subsurface geologic conditions underlying the site. Following completion, the holes were geophysically logged and then plugged to the surface with a bentonite and cement slurry. The borehole data were used to determine drilling sites for 20 observation wells, in order to ascertain the quality of the ground water, to establish the depth to water, and to determine the hydraulic gradient. Eight of the observation wells were constructed so that they could be used later as a part of the monitoring system. Two of the wells tapped the weathered shale, their purpose being to monitor the water table, evaluate the relation between precipitation and recharge, and ascertain the potential fluctuation of water quality in the weathered material in order to determine if it might serve as a pathway for contaminant migration from the surface to the shallowest aquifer. (From a technical perspective, the weathered shale and sandstone is not an aquifer, but from a regulatory point of view it could be considered a medium into which a release could occur and, therefore, would fall under RCRA guidelines.)

Regulations required that the uppermost aquifer be monitored, which in this case was a relatively thin, saturated sandstone. After the initial investigative information was available, all of the findings were used to design a ground-water monitoring system. This plan called for an additional 12 monitoring wells.

Graphics based on all of the drilling information (geologic and geophysical logs) included several geologic cross sections (fig. 6-5) and maps showing the thickness of shale overlying the aquifer (fig. 6-6), thickness of the aquifer, and the hydraulic gradient (fig. 6-7). The major purpose of the first map was to show the degree of natural protection that the shale provided to the aquifer relative to infiltration from the surface. The aquifer thickness map was needed for the design of monitoring wells. The water-level gradient map was necessary to estimate ground-water velocity and flow direction. During the drilling phases, cores of the aquifer and the overlying shale were obtained for laboratory analyses of hydraulic conductivity, porosity, specific yield, grain size, mineralogy, and general description. Aquifer tests were conducted on two of the wells.

The cross sections and maps indicate that the sandstone dips gently eastward and nearly crops out in a narrow band along the western margin of the facility. Elsewhere, owing to the change in topography and the dip of the aquifer, the sandstone is overlain by 25 feet or more of shale; throughout nearly all of the site the shale exceeds 50 feet in thickness. Consequently, only one small part of the aquifer, its outcrop and recharge area, is readily subject to contamination.

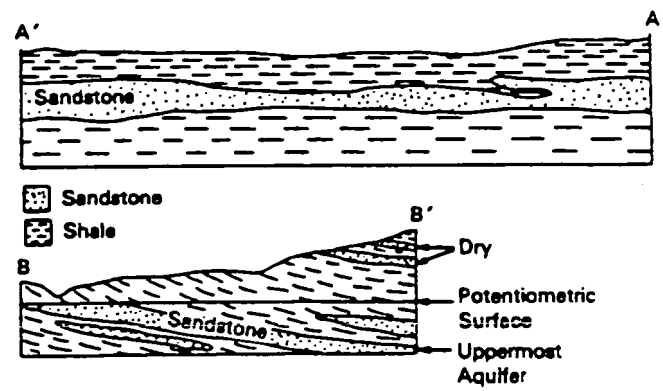

Figure 6-5. Geologic Cross-Section for the Site Investigation

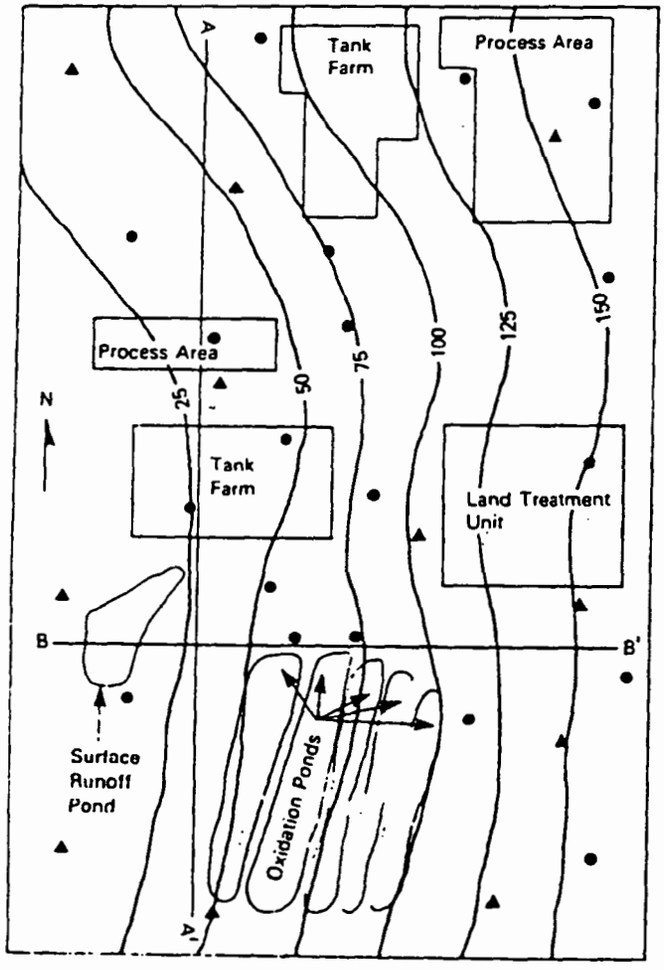

Figure 6-6. Map Showing Thickness of Shale Overlying the Uppermost Aquifer

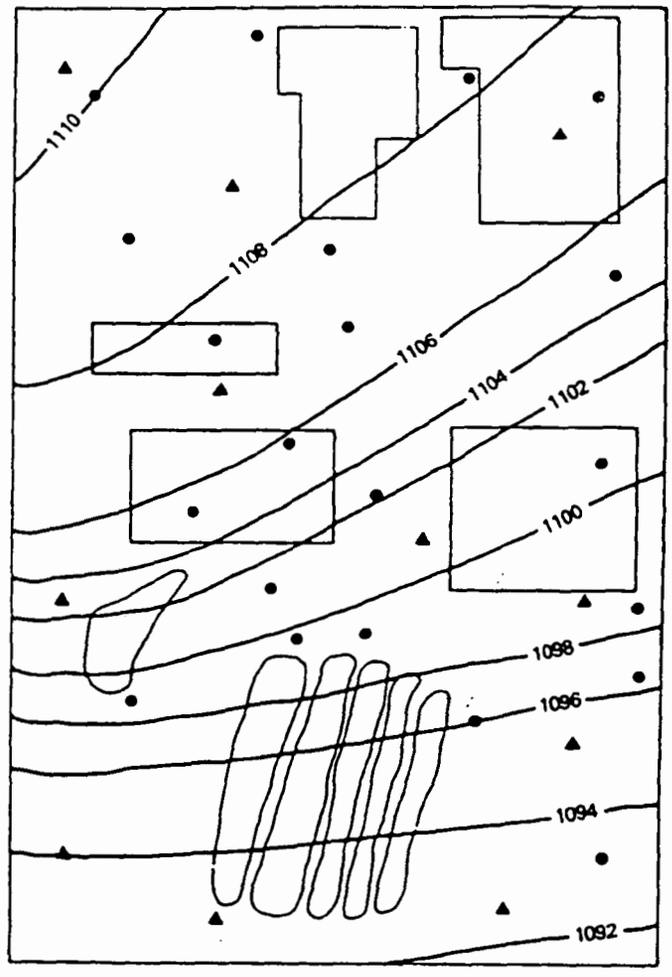

Figure 6-7. Potentiometric Surface of the Uppermost Aquifer

The water-level map indicates that the hydraulic gradient is not downdip but rather about 55 degrees from it. It is controlled by the topography off site. The average gradient is about 0.004 ft/ft, but from one place to another it differs to some extent, reflecting changes in aquifer thickness and hydraulic conductivity.

The topographic map indicates that surface runoff from the entire facility is funneled down to a detention pond. The pond and the lower part of the drainage way lie in the vicinity of the aquifer's recharge or outcrop area.

Logs of the drill holes list specific depths in six of the holes in which highly viscous hydrocarbons were present. All were reported in the unsaturated zone at depths of 2 to 9 feet with thicknesses ranging from a half inch to nearly a foot. At these locations the shale overlying the aquifer exceeded 55 feet in thickness.

Chemical analyses of water from the observation wells indicated, with one exception, that the quality was within background concentrations and no organic compounds were present. The exception was an observation well near the surface runoff retention pond.

Evaluation of all of the data indicated two potential problems—hydrocarbons in the unsaturated zone and ground-water contamination in the vicinity of the surface runoff detention pond. Since the plant had been in operation more than 50 years, the hydrocarbons had migrated from the surface into the weathered shale no more than 9 feet, and there was a minimum of at least 45 feet of tight, unfractured shale between the hydrocarbons and the shallowest aquifer, it did not appear that the soil contamination would present a hazard to ground water.

The existence of contaminated ground water, however, was a problem that needed to be addressed even though the sandstone aquifer is untapped and is never likely to serve as a source of supply. Four additional monitoring wells were installed downgradient in order to

determine the size of the plume and its concentration. Corrective action called for removal of sediment and sludge from the pond, backfilling with clean material, a cap, and pumping to capture the plume. The contaminated water was treated on site with existing facilities.

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Chapter 7

GROUND-WATER RESTORATION

Introduction

Prevention of ground-water contamination is far more logical, simple, and cost-effective than attempting to correct a problem—a problem that may have been in existence for years. A great deal of time, effort, and money are presently being expended to develop remedial measures to counteract the effects of contaminated aquifers and public water supplies. These include traditional as well as innovative construction techniques, water management, and research initiatives.

Several options or combinations of options are available to restore a contaminated aquifer: (1) provide inground treatment/containment, (2) provide aboveground treatment, (3) remove or isolate the source of contamination, (4) abandon the source of supply, or (5) ignore the problem. Generally, several techniques are coupled in order to achieve the desired results.

Restoration of contaminated aquifers to former background or near background conditions or to contain contaminated ground water in certain locations is generally accomplished through one of two overall approaches. One approach involves natural or induced in situ treatment, while the other approach uses engineered systems to contain the contaminated ground water. In the latter case pumping wells or engineered structures are installed in order to develop hydraulic gradients that cause the contaminated water to remain in a specified, general location from which it may be removed for later treatment.

Regardless of the restoration approach, any source or sources that continue to contaminate the ground water should be removed, isolated, or treated. Treatment or removal of an existing contamination source eventually may result in restoration of ground-water quality through natural processes. In other situations, contaminated ground water is removed from the aquifer by pumping or is allowed to discharge to a stream in which the flow is sufficient to dilute the contaminant to nondetectable concentrations. Natural replacement of the ground water is relied upon to eventually restore the quality of the water in the aquifer. Typically the natural restoration processes require many years or perhaps even decades for completion. As a result, ground-water restoration commonly requires a combination of approaches that involve ground-water removal and treatment or, if necessary, induced in situ treatment coupled with source control (removal, isolation, treatment). Site-specific conditions, properly defined and understood, provide the ground-water investigator with the basic information needed for the determination of a viable approach and for selecting and designing a cost-effective restoration scheme.

This chapter provides an overview of aquifer restoration technologies utilizing techniques derived from interrelated disciplines of geology, hydrology, geochemistry, engineering, construction, biology, and agronomy. The major emphasis of the chapter is on ground-water pumping systems and in situ biological treatment for organic contaminants, which are found at almost all hazardous waste sites. Many of the technologies have been developed by demonstration and research in conjunction with remedial activities in the Superfund program. Detailed information on selected techniques can be obtained from the references.

Contaminant Mobility

The design of a ground-water restoration program is complicated by the fact that all contaminants do not behave in the same manner. Although discussed previously, it is important to briefly redescribe the significance of contaminant mobility in developing and designing a ground-water restoration program.

The movement of most ground-water contaminants is controlled by gravity, the permeability and wetness of the geological materials, and the miscible character of the contaminants in ground water. When a material, particularly a hydrocarbon, is released to the soil, capillary

attraction and gravity actively draw it into the soil. As the main body of material moves downward into the more moist regions of the soil, capillary forces become less important as the contaminants move through the more favorable channels by displacing air.

When the contaminants reach the water table, those less dense than water tend to spread laterally along the air-water interface or capillary fringe, while the heavier ones continue to move downward in the saturated zone. In both cases, the contaminants tend to migrate in the direction of ground-water flow. In unusual circumstances very dense contaminants may be more affected by gravity than by advective flow and move in directions other than that of the ground water.

The amount of a contaminant that reaches the water table depends on the quantity involved, the characteristics of the contaminant, the chemical and biological properties of the unsaturated zone, precipitation (ground-water recharge), and the physical and chemical characteristics of the earth materials. In general, the more permeable the earth material, the greater the quantity of contaminant that is likely to reach the ground water. The entire amount of a contaminant may be temporarily immobilized in the unsaturated zone so that it only migrates downward after rainfall events, becoming a continual or long-term contamination source. Material so immobilized in the unsaturated zone may remain there unless physically, chemically, or biologically removed.

A hydrocarbon liquid phase, for example, generally is considered to be immiscible with both water and air. Residual hydrocarbons can occupy from 15 to 40 percent of the available pore space. However, it is important to realize that various components of the hydrocarbon may slowly volatilize into the vapor phase and then dissolve into the liquid phase. A halo of dissolved components of the hydrocarbon precedes the immiscible phase, some of which becomes trapped in the pore spaces and is left behind as isolated masses. Even when the so-called residual phase is entirely immobile, ground water coming into contact with the trapped material leaches soluble components and continues to contaminate ground water.

Interaction of the contaminant and the aquifer materials is another consideration in the evaluation of contaminant mobility. Some contaminants tend to partition between the liquid, solid, and vapor phases in amounts dictated by the characteristics of each contaminant, the nature of the aquifer material, particularly the amount of organic carbon, and other geochemical parameters. For many contaminants, these associations are not fixed but can be completely reversible. In addition, these compounds may move freely from one phase to another, depending upon their concentration in each phase. The processes of ion-exchange and sorption, chemical precipitation, and biotransformation all result in retardation or transformation of the contaminants. Ground water can become contaminated as freshwater moves through or past the aquifer material where contaminants are attached, or as infiltrating water moves through the unsaturated zone, which contains contaminants in the vapor phase. The subsurface transport of hydrophobic compounds is an active field of research.

Highly soluble contaminants, such as salts, some metal species, and nitrates, have little affinity for sorption to the solid phase. For aquifer restoration purposes, these contaminants can be considered to move essentially in the same direction and velocity as the ground water and are ideal candidates for pump-and-treat technology.

Site Characterization

In most restoration schemes, all too often the physical features of the subsurface are largely ignored and little understood, and most of the effort is involved with the design and construction of engineering structures. The important point to consider, however, is that the physical features of the subsurface, that is, the distribution of permeability and porosity, and the resulting hydrogeologic characteristics control the movement and storage of fluids in the subsurface.

Ground-water restoration activities require dedication of sufficient resources to collect and understand site conditions. An adequate amount of field data must be collected to provide a detailed understanding of the geology, hydrology, and geochemistry of the site, as well as the types of contaminants to be removed, their concentrations, and distribution. The literature should be reviewed to determine, to the fullest extent possible, the contaminants characteristics of sorption, volatilization, partitioning, and ability to be degraded. Finally, laboratory investigations, including treatability studies, development of sorption isotherms, and column and microcosm examinations to determine contaminant transport and transformation parameters, assist in developing a full understanding of the site conditions, and potential alternatives for ground-water remediation.

Many ground-water texts and reports, particularly the older ones, show ground-water flow nets to be homogeneous in both the horizontal and vertical dimensions—at least on a regional scale. In reality such depictions are rare and the actual water movement is much more complicated. Flow lines drawn on a water-table map, for example, imply that the fluids are moving directly downgradient when, in fact, the flow actually

follows curvilinear paths (see Chapter 4). All too often significant amounts of the flow may be through limited parts of the aquifer, both horizontally and vertically. This could result from the spatial variability of permeability for water, or it could result from density or other considerations for contaminants. In other words, neither the bulk of the water flow nor the distribution of the contaminants can be assumed as homogeneous.

Figure 7-1 is the map of a contaminated waste disposal site that shows the location of a number of monitoring wells and the altitude of the water surface in them. Notice that there is as much as 100 feet of difference in head in wells that are relatively close. The reason for this difference is well depth, with the deeper wells having the greatest depth to water. Figure 7-2 is a water-level map of the same area; contours were based on shallow wells of nearly the same depth and screen length. Flow lines depict the general direction of ground-water movement. Figure 7-3 is a hydrologic cross section, that is, a vertical flow net, constructed along the line A-A'. Notice in this example that in the upper 50 feet or so the ground water is flowing across different geologic units with little loss in head. This indicates that secondary permeability (fractures), rather than the primary permeability of the various geologic units, is the major control on ground-water flow. In the lower part of the cross section the water-level contours or equipotential lines are closely spaced and roughly parallel land surface. This reflects the depth at which the fractures tend to disappear. The hydrologic cross section shows that fluid movement, both contaminants and ground water, is largely limited to the upper 50 feet of the strata.

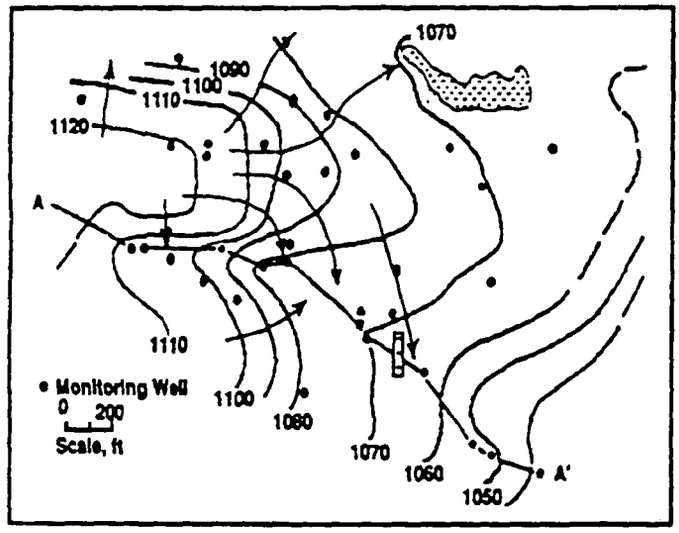

Figure 7-2. Map Showing Configuration of the Water Table and Flow Lines

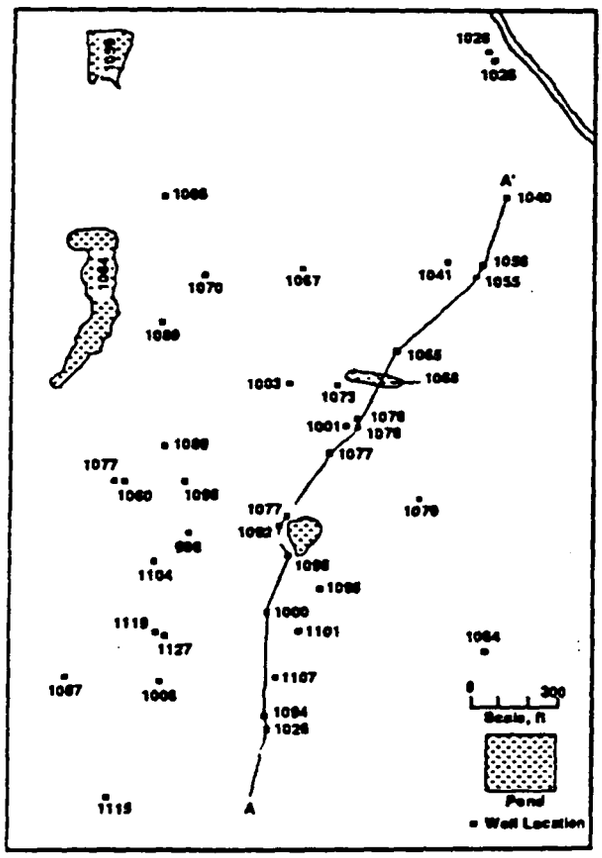

Figure 7-1. Map of a Contaminated Area Showing Location of Monitoring Wells and Elevation of Water Levels

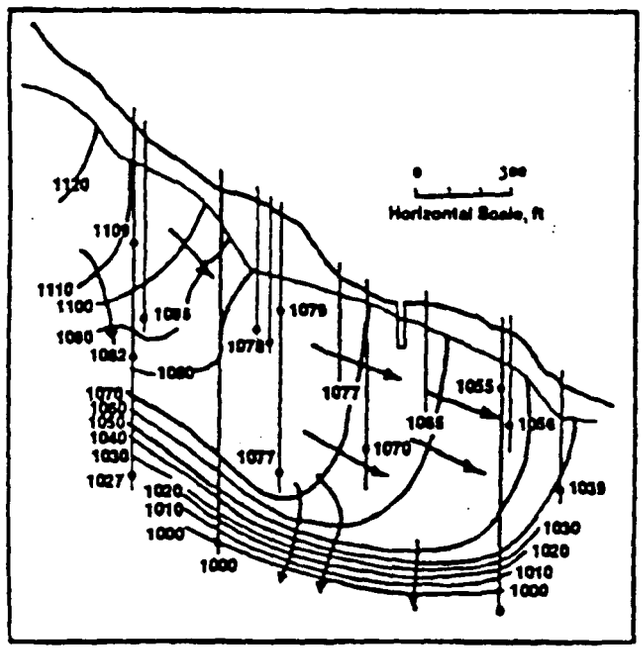

Figure 7-3. Hydrologic Cross Section Showing Equipotential and Flow Lines. Numbers Represent Total Head

Obviously, in either pump-and-treat or in situ restoration systems, or in ground-water monitoring, the location, depth, and length of the screens of monitoring or extraction wells are of paramount importance. If the wells are improperly located, monitoring results would not adequately represent the aquifer being studied, and its restoration would be more costly and less effective than necessary. Therefore, in planning and carrying out ground-water restoration activities, it is essential to dedicate adequate resources to the collection of background information. In designing remediation activities, it is more important to describe the most permeable zones so that it can be determined where the water can go, under a remediation system, rather than its natural state.

Source Control

The objective of source- control strategies is to reduce or eliminate the volume of waste, thereby removing or minimizing ongoing contamination of the ground-water environment. Source-control techniques include removal of the source(s), surface-water controls, ground-water barriers, interceptors, and hydrodynamic controls.

Source Removal

Soil and water at a hazardous waste site may be removed for treatment or relocation to a site that is more acceptable from an engineering or environmental viewpoint. While the removal and treatment or reburial of contaminated materials at a more controlled site may appear to solve a contamination problem, various factors need to be evaluated before excavation commences. These factors include:

(1). Problems associated with the excavation of bulky, partially decomposed or hazardous waste.
(2). Distance to an acceptable treatment/reburial site.
(3). Road conditions between sites.
(4). Accessibility of both sites.
(5). Political, social, and economic factors associated with locating a new site.
(6). Disposition of contaminated ground water.
(7). Control of nuisances and vectors during excavation.
(8). Reclamation of excavated site.
(9). Costs.

These considerations suggest that excavation and relocation may be a viable alternative only where costs are not significant compared to the importance of the resource being protected. In some cases, removal and reburial in an approved facility transfers a problem from one location to another, and possibly creates additional problems.

Surface Runoff Controls

Surface runoff control measures are used to minimize the infiltration and percolation of overland flow or precipitation at a waste site. It is the infiltration of these waters that serve as the moving or driving force that leaches contaminants from the surface or unsaturated zone to the water table. According to an EPA estimate (Schuller and others, 1983), a disposal site consisting of 17 acres with 10 inches per year of infiltration could produce 4.6 million gallons of leachate each year for 50 to 100 years. This estimate, of course, is site-specific. Reduction of infiltration through a contaminated site can be accomplished by contouring the site, providing a cap or barrier to infiltration, and revegetating the site.

Several standard engineering techniques can be used to change the topographic configuration of the land surface in order to control the movement of overland flow. Some of the more common techniques are dikes and berms, ditches, diversion waterways, terraces, benches, chutes, downpipes, levees, sedimentation basins, and surface grading.

A mounded and maintained cover or cap of low permeability material greatly reduces or even prevents water from entering the source, thus reducing leachate generation. Covers also can control vapors or gases produced in a landfill. They may be constructed of native soils, clays, synthetic membranes, soil cement, bituminous concrete, or asphalt, a combination of these materials.

Revegetation can be a cost-effective method of stabilizing the surface of a waste site, especially when preceded by capping and contouring. Vegetation reduces raindrop impact and the velocity of overland flow, and strengthens the soil mass, thereby reducing erosion by wind and water. It also improves the site aesthetically.

Schuller and others (1983) described the effect of regrading, installation of a PVC topseal, and revegetation of a landfill in Windham, Connecticut. As Figures 7-4 and 7-5 illustrate, field data clearly indicate that the cover reduced infiltration and leachate generation, which caused a reduction in the size and concentration of the leachate plume.

Ground-Water Barriers

Subsurface barriers are designed to prevent or control ground-water flow into, through, or from a certain location. Barriers keep fresh ground water from coming into contact with a contaminated aquifer zone or ground

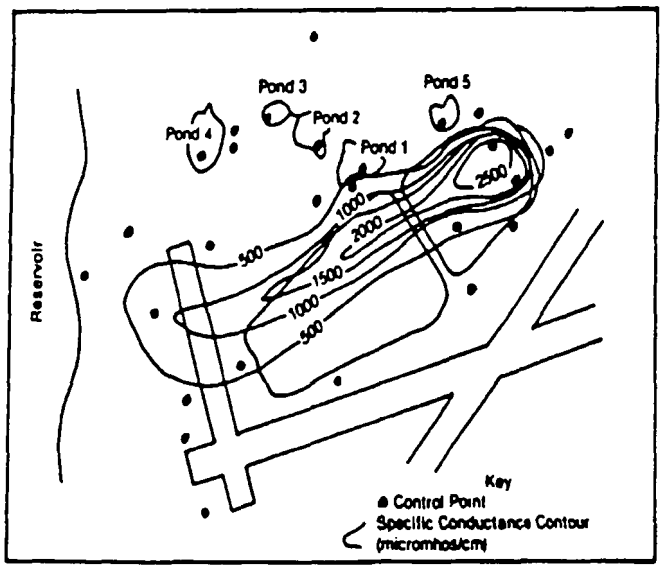

Figure 7-4. Distribution of Specific Conductance, May 19, 1981

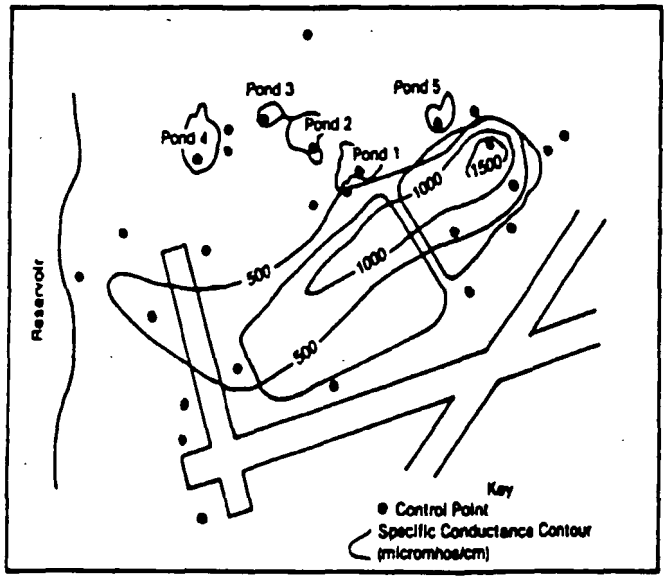

Figure 7-5. Distribution of Specific Conductance, November 12, 1981

water from existing areas of contamination from moving into areas of clean ground water. Usually it is necessary to incorporate other technologies, such as pump-and-treat systems, with ground-water barriers.

The types of barriers commonly used include:

1. Slurry trench walls
2. Grout curtains
3. Vibrating beam walls
4. Bottom sealing
5. Block displacement

Slurry trench walls are placed either upgradient from a waste site to prevent flow of ground water into the site, downgradient to prevent offsite flow of contaminated water, or around a source to contain the contaminated ground water. A slurry wall may extend through the water-bearing zone of concern, or it may extend only several feet below the water table to act as a barrier to floating contaminants. In the former case, the foundation should lie on, or preferably in, an underlying unit of low permeability so that contaminants do not flow under the wall. A slurry wall is constructed by excavating a trench at the proper location and to the desired depth, while keeping the trench filled with a clay slurry composed of a 5 to 7 percent by weight suspension of bentonite in water. The slurry maintains the vertical stability of the trench walls and forms a low permeability filter cake on the walls of the trench. As the slurry trench is excavated, it is simultaneously backfilled with a material that forms the final wall. The three major types of slurry backfill mixtures are soil bentonite, cement bentonite, and concrete. Slurry walls, under proper conditions, can be constructed to depths of 100 feet or so.

Slurry trench walls are reported to have a long service life and short construction time, cause minimal environmental impact during construction, and be a cost-effective method for enclosing large areas under certain conditions (Nielsen, 1983). A concern regarding the use of a slurry wall where contaminated materials are in direct contact with the wall is the long-term integrity of the wall (Wagner and others, 1986). In such cases, the condition of the wall needs to be verified over time by ground-water monitoring.

Two separate slurry walls were constructed along parts of the margin of the Rocky Mountain Arsenal near Denver in order to contain plumes that originate on the plant property (Shukle, 1982, Pendrell and Zeltinger, 1983, and Hager and others, 1983). Along the north boundary, where surficial, unconsolidated sand and gravel occur with a thickness that averages about 30 feet, the slurry wall, about 2 feet thick, is 6,800 feet long. On the upgradient side are a series of 35 12-inch-diameter discharging wells on 200 foot centers that pump contaminated ground water into a treatment facility. After flowing through a carbon filtration system the water is reinjected into 50 6-inch diameter recharge wells on 100 foot centers on the opposite side of the barrier.

Along the northwest boundary of the Arsenal is another bentonite slurry barrier, 1,425 feet long, that extends southwestward from a bedrock high. The wall, excavated into the sand and gravel with the bentonite slurry trench method, is 30 inches wide and extends 3 feet into the underlying bedrock. The barrier contains about 7,000 cubic yards of backfill that were obtained from a borrow

pit and blended with the bentonite prior to emplacement. The barrier was constructed where the saturated thickness of the permeable material is less than 10 feet. Paralleling the downgradient side of the barrier is a series of 21 recharge wells, stretching nearly 2,100 feet along the Arsenal boundary. Directly behind (upgradient) the barrier and extending into the thicker part of the surficial aquifer are 15 discharge wells. The contaminated ground water is pumped to a treatment plant and then reinjected into the recharge wells, thus forming a hydraulic barrier. Farther southeast along the boundary is another hydraulic barrier system, about 1,500 feet long, that consists of two parallel rows of discharge wells with 15 wells per row and, downgradient, a row of 14 recharge wells. The contaminated water, originating from a spill, is pumped, treated, and then reinjected. This system and the one along the north boundary was put into operation in late 1981 and the system along the northwest boundary began operation in 1984.

Grouting is the process of pressure -injecting stabilizing materials into the subsurface to fill and, thereby, seal voids, cracks, fissures, or other openings. Grout curtains are underground physical barriers formed by injecting grout through tubes. The amount of grout needed is a function of the available void space, the density of the grout, and the pressures used in setting the grout. Two or more rows of grout are normally required to provide a good seal. The grout used may be either particulate (i.e., Portland cement) or chemical (i.e., sodium silicate) depending on the soil type and the contaminant present. Grouting creates a fairly effective barrier to ground-water movement, although the degree of completeness of the grout curtain is difficult to ascertain (Nielsen, 1983). Incomplete penetration of the grout into the voids of the earth material permits leakage through the curtain.

A variation of the grout curtain is the vibrating beam technique for placing thin (approximately 4 inches) curtains or walls. Although this type of barrier is sometimes called a slurry wall, it is more closely related to a grout curtain since the slurry is injected through a pipe in a manner similar to grouting. A suspended I-beam connected to a vibrating driver-extractor is vibrated through the ground to the desired depth. As the beam is raised at a controlled rate, slurry is injected through a set of nozzles at the base of the beam, filling the void left by the beam's withdrawal. The vibrating beam technique is most efficient in loose, unconsolidated deposits, such as sand and gravel.

Another method that uses grouting is bottom sealing, where grout is injected through drill holes to form a horizontal or curved barrier below the site to prevent downward migration of contaminants.

Block displacement is a relatively new plume management method, in which a slurry is injected so that it forms a subsurface barrier around and below a specific mass or "block" of material. Continued pressure injection of the slurry produces an uplift force on the bottom of the block, resulting in a vertical displacement proportional to the slurry volume pumped. Brunsing and Cleary (1983) described an example of slurry-induced block displacement. Demonstrated in Whitehouse, Florida, a slurry wall was constructed around a small area, 60 feet in diameter, to a depth of 23 feet in unconsolidated material. Injection wells were then used to force a soil bentonite slurry outward along the bottom of the cell. Subsequent test holes indicated that the new floor of the cell contained 5 to 12 inches of slurry.

Sheet pile cutoff walls have been used for many years for excavation bracing and dewatering. Where conditions are favorable, depths of 100 feet or more can be achieved. Sheet piling cutoff walls can be made of wood, reinforced concrete, or steel, with steel being the most effective material for constructing a ground-water barrier. The construction of a sheet pile cutoff wall involves driving interlocking sheet piles down through unconsolidated materials to a unit of low permeability. Individual sheet piles are connected along the edges with various types of interlocking joints. Unfortunately, sheet piling is seldom water-tight and individual plates can move laterally several to several tens of feet while being driven. Acidic or alkaline solutions, as well as some organic compounds, can reduce the expected life of the system.

Membrane and synthetic sheet curtains can be used in applications similar to grout curtains and sheet piling. With this method, the membrane is placed in a trench surrounding or upgradient of the plume, thereby enclosing the contaminated source or diverting ground-water flow around it. Placing a membrane liner in a slurry trench application also has been tried on a limited basis. Attaching the membrane to an underlying confining layer and forming perfect seals between the sheets is difficult but necessary in order for membranes and other synthetic sheet curtains to be effective. Arlotta and others (1983) described a system that consists of a trench lined with 100 mil high density polyethylene and backfilled with sand. It was installed by the slurry trench construction method in New Brunswick, New Jersey, in the fall of 1982.

Hydrodynamic Controls

Hydrodynamic controls are used to isolate a plume of contamination from the normal ground-water flow regime to prevent the plume from moving into a well field, another aquifer, or to surface water. Controlling the

movement of ground water by means of recharge and discharge wells has been practiced for several years. The major disadvantages include the commonly long pumping periods, well construction and maintenance costs, and the fact that the subsurface geology dictates system design.

The extent of the cone of depression around a pumping well can be controlled by the discharge rate and thus the cone, which is a change in the hydraulic gradient, can be used to control ground-water flow directions and velocity. Management of the cone or cones permits the operator to capture contaminants, which can then be diverted to a treatment plant. Well placement is particularly important since proper spacing and pumping rates are required to capture the contaminants. Moreover, well placement should be optimized so that as little uncontaminated water as possible is produced in order to reduce treatment costs.

Recharge wells are used to develop a hydraulic barrier (an inverted cone of depression) or pressure ridge. In this way, recharge wells can be used to force the contaminant plume to move in preferred directions, such as toward a drain or discharging well.

The design of well systems is, in large part, based on trial and error methods coupled with experience. Herein also lies one of the more useful exercises of computer simulations, because with this approach one can quickly and easily evaluate different well location and pumping schedules, and estimate costs.

Gradient-control techniques are used at a great number of sites undergoing restoration and nearly always play some role in containment methods, as is the case at the Rocky Mountain Arsenal.

A well point system, which is a common technique used for dewatering at construction sites, consists of several closely spaced shallow wells connected to a main header pipe. The header pipe is connected to a suction lift pump. Well point systems are used only for shallow aquifers and are designed so that the drawdown produced by the system completely intercepts the plume of contamination.

Deep wells are similar to well point systems except they are generally deeper and normally are pumped individually. This system commonly is used in places where the ground-water surface is too deep for the use of a suction lift system.

A thorough knowledge of the hydrogeological conditions of a site is required for the development of a hydrodynamic control system. The effect of the injection wells on the drawdown and the radius of influence of the pumping wells must be analyzed. Of particular importance are the potential well yield or injection rate, and the effect of hydrologic flow boundaries. Monitoring of the system is essential.

Ground-Water Collection and Treatment

The cleanup of a contaminated ground-water site involves the collection and treatment of the contaminated water. Some of the techniques used for source control often are used as part of a ground-water cleanup program, including pumping well systems, interceptor systems, and some of the techniques used for source control. In addition, in situ treatment, enhanced desorption, encapsulation, and biodegradation may be part of a cleanup plan.

Pumping Systems

A ground-water pumping scheme combined with a treatment procedure, also called a pump-and-treat system, is usually designed for a specific ground-water contamination problem. The use of pump-and-treat systems is probably more widespread and successful than all other restoration techniques combined. Large expenditures are made each year to prepare for and operate pump-and-treat remediation of ground-water contamination (Keely, 1989). The hydrogeology of the site, the source of the contaminant, and the characteristics of the contaminant must be understood if an efficient and cost-effective program is to be conducted.

The operation of a well field to remove ground water causes the formation of stagnation zones downgradient from the extraction wells, which must be considered in the system design. For example, if remedial action wells are located within the bounds of a contaminant plume, the portion of the plume lying within the stagnation zones will not be effectively remediated because the contaminants are removed only from the zone of advective ground-water flow. In this case, the only remediation in the stagnation zone will result from the process of chemical diffusion and degradation, which may be very slow. Proper location of wells based on pumping rates and drawdown tends to mitigate this effect.

The tailing effect also can affect the removal and renovation of ground water containing a low solubility contaminant. Tailing is the slow, nearly asymptotic decrease in contaminant concentration in ground water moving through contaminated geologic material. The

contaminants migrate into the finer pore structures of the earth materials and are slowly exchanged with the bulk water present in larger pores and this results in "tailing."

Many human-made and natural organic compounds found in ground water tend to adsorb to the organic and mineral components of the aquifer material. When water is removed by pumping, contaminants can remain on the aquifer material, the amount depending on the geologic materials and characteristics of the contaminants. Once sorbed to the geologic material, contaminants may desorb slowly into the ground water, thus requiring extended periods of pumping and treating to attain desired levels of restoration.

The removal of a water-insoluble liquid, such as gasoline, can be difficult since the product may become trapped in the pores of earth materials and is not easily removed by pumping. Pumping ground water to remove the components of a residual phase initially may reduce the concentration, but this reduction may only be the result of dilution or lowering of the water table below the level of contamination. A contaminant will not be removed faster than it is released into the ground water, so if the pumping stops for a period of time, water-soluble residual phase components again will dissolve into the ground water bringing the concentrations back to the previous level.

An innovation in pump-and-treat technology is pulsed pumping. This technique involves alternating the periods of pumping, allowing contaminants time to come to equilibrium with the ground water in each cycle. Equilibrium is achieved by diffusion from stagnant zones or zones of lower permeability, and by partitioning of sorbed contaminants or those associated with residual contaminant phases. Alternating pumping among wells also can establish active flow paths in the stagnant zones.

Another innovation is the use of pump-and-treat systems in conjunction with other remediation technologies. Examples are the use of extraction wells with barrier walls to limit plume expansion while reducing the amount of clean water pumped, and the use of surface ponds or flooding to flush contaminants from the unsaturated zone prior to collection by a pumping system.

Interceptor Systems

Interceptor systems may be an alternate to pumping systems. The subsurface drains used in interceptor systems essentially function as an infinite line of extraction wells, and can perform many of the same functions. Subsurface drains create a continuous zone of influence in which ground water flows towards the drain. Subsurface drains are installed perpendicular to the direction of ground-water flow and collect ground water from an upgradient source for treatment. Interceptor systems prevent leachate or contaminated ground water from moving downgradient toward wells or surface water.

Two types of interceptor systems used for source control are the passive system, which relies on gravity flow, and the active system, which uses pumps. An interceptor system consists of a trench excavated to a specified depth below the water table in which a perforated collection pipe is installed in the bottom. Active interceptor systems have vertical removal wells spaced along the interceptor trench or a horizontal removal pipe in the bottom of the trench. Active systems are usually backfilled with a coarse sand or gravel to maintain the stability of the wall. These interceptor systems can be used as preventive measures, such as leachate collection systems, as abatement measures, such as interceptor drains, or in product recovery from ground water, such as the removal of gasoline or oil. Interceptor drains generally are used to either lower the water table beneath a contamination source or to collect contaminated ground water from an upgradient source. Interceptor systems are relatively inexpensive to install and operate, but they are not well suited for soils with a low permeability.

In stratified soils with variable hydraulic conductivities, the drain is normally installed on a layer with a low hydraulic conductivity to minimize leachate leakage under the drain. An impermeable liner placed in the bottom of a trench also can be used to control underflow. The design, spacing, and location of drains for various soil and ground water conditions are described further in Wagner and others (1986).

A combined interceptor and ground-water dam installation was described by Giddings (1982). In this case, a landfill that began as a burning dump, was found to be discharging leachate both to the surface and to the ground water, much of which eventually flowed into an adjacent river. A leachate interceptor trench was constructed on the downgradient side of the disposal area, as shown in Figure 7-6. In the trench on the upgradient side was placed a perforated pipe in a gravel envelope that was covered with permeable material. The remainder of the trench on the downgradient side was then backfilled with fine-grained materials as shown in Figure 7-7. Leachate from the landfill flows into the filled trench, seeps into the perforated pipe, and then is collected for treatment. In this case, the main

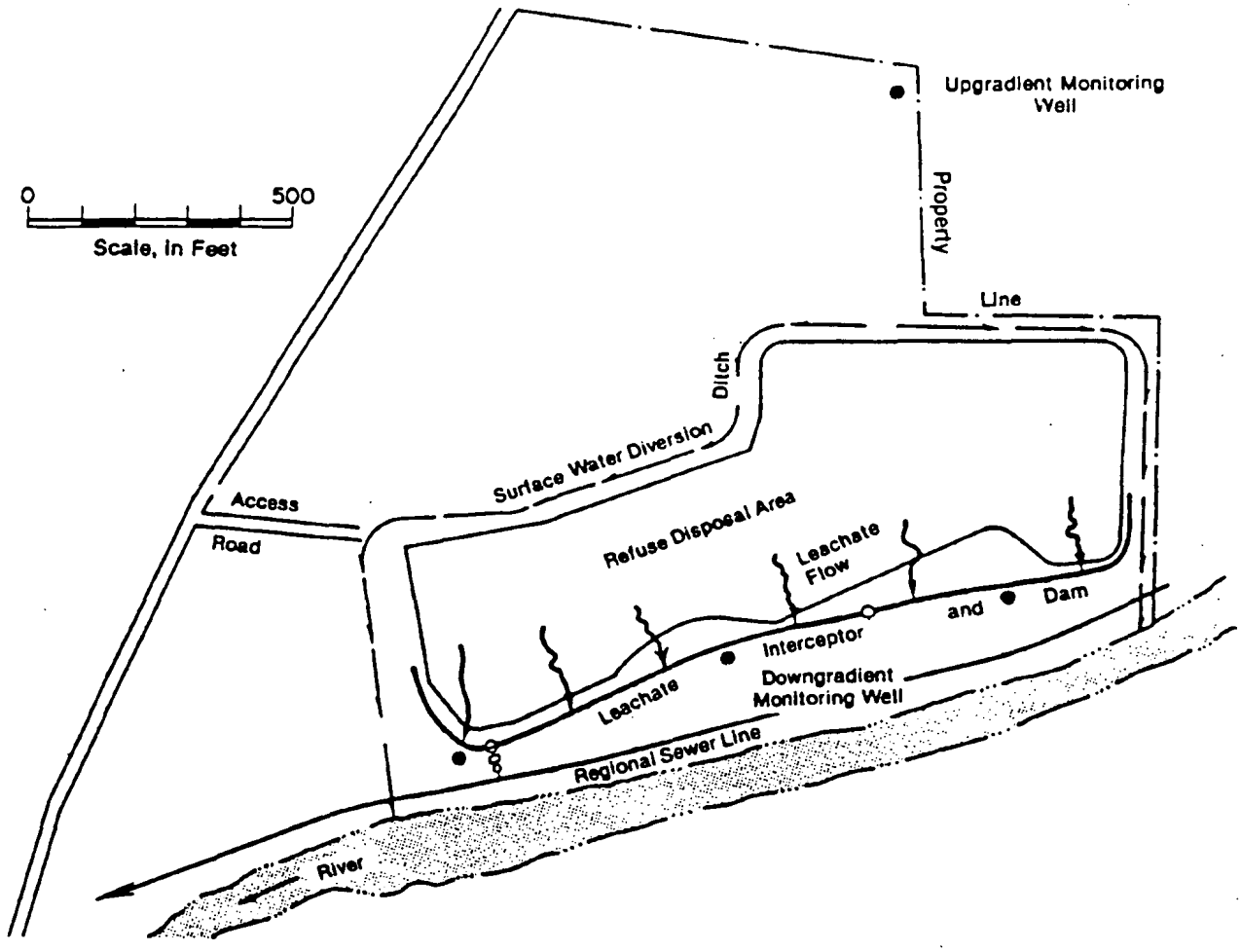

Figure 7-6. Site Layout

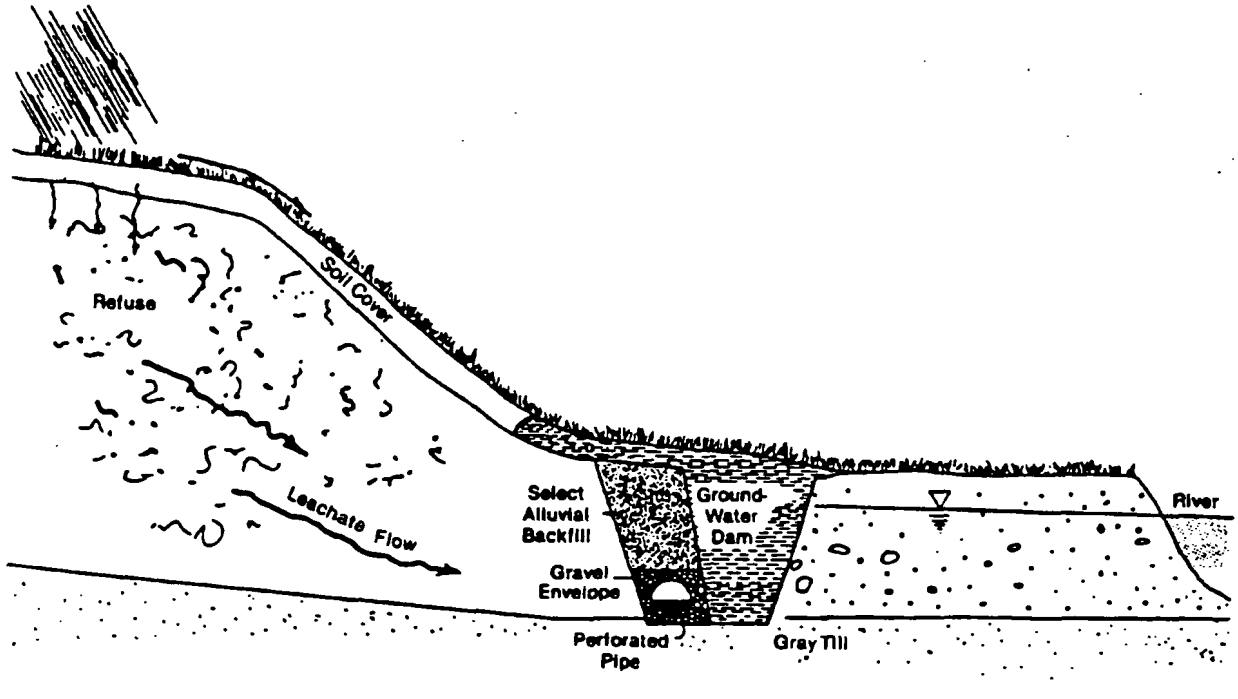

Figure 7-7. Site Cross Section

purpose of the ground-water dam was to prohibit water originating in the adjacent river from flowing into the trench, which would have substantially increased the volume of wastewater.

Ground-Water Treatment after Removal

Of course the technology of pumping and treating of ground water implies that a cadre of engineering processes are available for treating the extracted water at the surface. A detailed discussion of these is beyond the scope of this document. They will only be mentioned to give the reader a familiarity with the processes so that detailed searches can be made elsewhere.

Treatment technologies for pumped or intercepted ground water can be grouped into three broad areas: physical, chemical, and biological. Physical treatment methods include adsorption, density separation, filtration, reverse osmosis, air and steam stripping, and incineration. Precipitation, oxidation/reduction, ion exchange, and neutralization are commonly used chemical treatment methods. Biological treatment methods include activated sludge, aerated surface impoundments, anaerobic digestion, trickling filters, and rotating biological discs.

In Situ Treatment

In situ treatment is an alternative to the removal and subsequent treatment of contaminated ground water. This method requires minimal surface facilities and reduces exposure to the contaminant. The success of various treatment methods is highly dependent on physical factors including aquifer permeability, the characteristics of the contaminants involved, and the geochemistry of the aquifer material.

In situ treatment technology has not yet been developed to the extent of other currently available technologies for restoring contaminated aquifers. However, some in situ treatment technologies have demonstrated success in actual site remediations (Wagner and others, 1986). Laboratory and pilot-scale testing generally must be performed to evaluate the applicability of a particular technology to a specific site.

In situ treatment may be grouped into two broad categories: physical/chemical and biological. Brief descriptions follow of the available technologies that have potential for success at hazardous waste sites.

In Situ Physical/Chemical Treatment

Organic and inorganic contaminants may be treated chemically to cause immobilization, mobilization for extraction, or detoxification. The application of oxidation and reduction reactions to in situ treatment is largely conceptual, but potentially may be used to accomplish immobilization by precipitation, mobilization by solubilizing metals or organics, or detoxification of metals and organics (Wagner and others, 1986). The chemicals used in these processes, however, have the potential to degrade compounds other than those targeted and to form degradation products that may be more toxic than the original ones.

Precipitation, chelation, and polymerization are three methods used to immobilize a contaminant. Precipitation using caustic solutions is effective in immobilizing dissolved metals in ground water. Chelation also may be effective in immobilizing metals, although considerable research is needed (Wagner and others, 1986). Polymerization is effective in immobilizing organic monomers. However, the chemicals added to the contaminants in the ground water may react to form toxic by-products. Solidification methods used for treatment of soils also can immobilize contaminants. Mobilization of contaminants is accomplished by soil flushing or vacuum extraction. Neutralization, hydrolysis, and permeable treatment bed technologies may be used for detoxification. Precipitation and polymerization will lower the hydraulic conductivities near the injection wells making closely spaced wells necessary for effective treatment.

One interesting example of polymerization, reported by Williams (1982), involved a 4,200 gallon leak of acrylate monomer from a corroded pipeline at a small plant in Ohio. The contaminant migrated through a layer of fill, consisting largely of cinders, and then downward through a storm sewer trench into a thin sand and gravel aquifer. A test boring and soil sampling program delineated the plume and indicated that the contaminant was slowly beginning to undergo polymerization and, therefore, immobilization. To increase the rate of reaction, 2-inch-diameter perforated PVC pipe was buried, about 2 feet below land surface, in four narrow trenches that trended across the plume. A riser and manifold header connected each pipe to solution tanks containing a catalyst in one and an activator in the other. Both solutions contained a wetting agent. A total of 8,000 gallons of solution were injected during the two treatment operations and 1,000 gallons had been injected previously during the investigative phase. On the basis of pre- and post-treatment soil borings, it was estimated that 85 to 90 percent of the liquid monomer contaminant was solidified, and in some places it exceeded 99 percent polymerization. It was assumed that the remaining material would polymerize naturally.

In situ physical/chemical treatment processes generally entail the installation of a series of injection wells at the head of or within the plume of contaminated ground

water. An alternative technique that has been used in shallow aquifers, is the installation of in situ permeable treatment beds. Trenches are filled with a reactive permeable medium and contaminated ground water entering the trench reacts with the medium to produce a nonhazardous soluble product or a solid precipitate. Among the materials commonly used in permeable bed trenches are limestone to neutralize acidic ground water and remove heavy metals, activated carbon to remove nonpolar contaminants, such as carbon tetrachloride, polychlorinated biphenyls, and benzene, and zeolites and other ion exchange resins for removing solubilized heavy metals.

Permeable treatment beds are applicable only in relatively shallow aquifers because the trench must be constructed down to a layer of low permeability. They also are often effective for only a short time because they lose their reactive capacity or become plugged with solids. An overdesign of the system or replacement of the reactive medium can lengthen the time during which permeable treatment is effective.

Mobilization for Extraction

Pump-and-treat remediation techniques often are inefficient when a preponderance of the contaminants are sorbed to the solid phase of the aquifer. The same can be said for in situ treatment if the reactive chemicals are unable to come into contact with the contaminants. In these cases, the enhanced desorption or mobilization of contaminants would be of considerable interest in aquifer restoration activities.

Soil flushing is the process of flooding a contaminated area with water or a solvent to mobilize the contaminant, followed by the collection of the elutriate The process is based on the solvent solubilizing or chemically reacting with the contaminants and mobilizing them into the solvent phase. Water is used if the contaminant is readily soluble. Acid solutions tend to flush metals and basic organics.

The mobilization of contaminants by injecting surfactants into the aquifer matrix is possible. Techniques used for the secondary recovery of oil are being used experimentally, with moderate success. Both surfactant and alkaline floods have been attempted. Most oil-field surfactants are expensive, while alkaline floods produce lye; therefore, this approach promises little benefit to aquifer restoration.

In the recovery of hydrocarbons, there are three possible physical-chemical methods. At shallow depths, thermal or steam flooding may be helpful while on a larger scale, alcohol flooding may at some future date prove to be helpful. Alcohol is easily produced and dissolves the hydrocarbon, but tentative research results indicate that the required alcohol-water ratio must be so high as to make the technique questionable.

Another emerging technology, which is increasingly being used, is alternately called in situ vacuum extraction or in situ volatilization. It is used to extract volatile organic contaminants from the unsaturated zone where contaminants exist as a result of underlying contaminated ground water, or free product riding on top of the ground water, or from leaks or spills. The technology has enjoyed considerable success in this and other industrialized countries.

The plumbing associated with this type of remediation is obviously dictated by site conditions, including the thickness of the unsaturated zone, the volatility of the contaminants involved as well as their source and extent, and the porosity and permeability of the unsaturated zone (Pacific Environmental Services, 1989).

Generally these vapor extraction projects consists of a series of slotted PVC wells configured to span the area of contamination. Air inlet wells located both inside and outside of the plume increase the introduction of air from the atmosphere (fig. 7-8).

Like pump-and-treat remediation techniques, vacuum extraction projects usually require some type of surface treatment facility to deal with the collected vapors. When surface treatment is required, activated carbon columns are widely in operation, however the use of biologically active columns is being studied, which will allow the introduction of oxygen or other gases needed for biodegradation.

Vacuum extraction is best suited for areas of high, relatively homogeneous, permeability. There should be no underground structures, and great care must be given to the explosive nature of the extracted vapors. The unit cost, which appears to be very promising, varies widely according to the size of the area under remediation and the specific site characteristics.

Radio frequency heating has been under development since the mid-1970s and the concept is being applied to in situ decontamination of uncontrolled hazardous waste landfills and sites (Rich and Cherry, 1987). In this process, the ground is heated with radio frequency waves that vaporize the hazardous contaminants. The vapors emanating from the soil are then treated.

Detoxification

Neutralization of ground water may be accomplished by injecting dilute acids or bases into the aquifer through

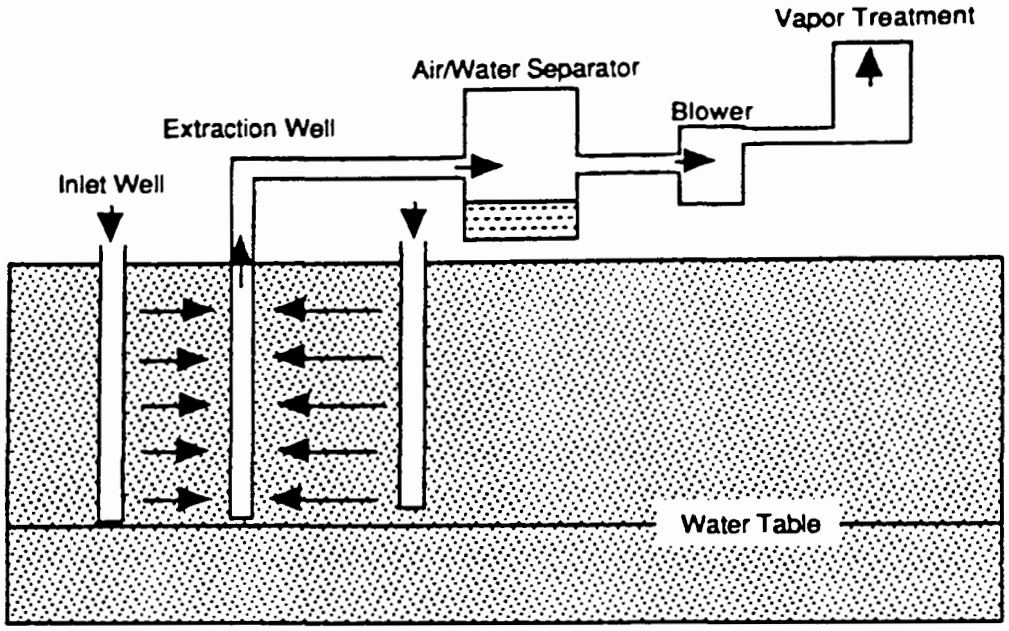

Figure 7-8. Schematic of a Vaccum Extraction System

injection wells to adjust the pH to the desired level. Tolman and others (1978) recommended that neutralization only be applied to ground water at industrial waste disposal sites since municipal landfills, which constantly generate anaerobic decomposition products, would require neutralization over a long period of time.

Hydrolysis may be used for detoxification, however, the intermediate products formed during hydrolysis of a particular compound must be known since they may be more toxic than the targeted compound. Esters, amides, carbamates, phosphoric and phosphonic acid esters, and pesticides are potentially degradable by hydrolysis (Wagner and others, 1986).

Biodegradation

There are two basic approaches to in situ biodegradation. The first relies on the natural biological activity in the subsurface. The second approach, called enhanced biorestoration, involves the stimulation of the existing microorganisms by adding nutrients.

Natural Subsurface Biological Activity

Biological treatment in the subsurface involves the use of microorganisms to break down hazardous organic compounds into nonhazardous materials. The site hydrology, environmental conditions, and the biodegradability of the contaminants are factors that determine the potential effectiveness of in situ biological treatment. Most compounds are more rapidly degraded aerobically, however some compounds will only degrade under anaerobic conditions. Biodegradation in ground water and solids can be a slow process and may take several years for completion depending on the compounds present. In situ biodegradation, however, is a desirable method of treatment because the contaminants are destroyed, thus, removal of ground water for external treatment and residual handling possibly can be avoided.

In situ biorestoration of the subsurface is a relatively new technology that has recently gained considerable attention. Scarcely more than a decade ago, conventional wisdom assumed that the subsurface below the root zone of plants was, for all practical purposes, sterile. Research during the last decade has indicated that the deeper subsurface is not sterile, but in fact, harbors significant populations of microorganisms. Bacterial densities of around a million organisms per gram of dry soil have been found in several uncontaminated aquifers. Water-table aquifers examined so far exhibit considerable variation in the rate of biodegradation of specific contaminants and rates can vary two or three orders of magnitude from one aquifer to another or over a vertical separation of only a few feet in the same aquifer. Although extremely variable, the rates of biodegradation are fast enough to protect ground-water quality in many aquifers.

Although not clearly defined, several environmental factors are known to influence the capacity of indigenous microbial populations to degrade contaminants. These

factors include dissolved oxygen, pH, temperature, oxidation-reduction potential, availability of mineral nutrients, salinity, soil moisture, the concentration of specific contaminants, and the nutritional quality of dissolved organic carbon in ground water.

Natural biorestoration does occur in the subsurface environment. Contaminants in solution in ground water, as well as vapors in the unsaturated zone, can be completely degraded or transformed to new compounds. Undoubtedly, thousands of contamination events are remediated naturally before the contamination reaches a point of detection. On the other hand, methods are needed to determine when natural biorestoration is occurring, the stage the restoration is in, whether enhancement of the process is possible or desirable, and what will happen if natural processes are allowed to run their course.

For information on in situ biorestoration of specific compounds and conditions see Bower and McCarty (1983), Jhaveri and Mazzacca (1983), Lee and Ward (1984), Parsons and others (1985), Parsons and others (1984), Sulflita and Gibson (1985), Sulflita and Miller (1985), Wilson (1985), Wilson and Rees (1985), Wood and others (1985), and Young (1984).

Enhanced Biorestoration

In the subsurface environment, populations of organisms capable of degrading contaminants increase until limited by metabolic requirements, such as mineral nutrients or oxygen. Once this point is reached, the rate of biodegradation or transformation of organic compounds is controlled by the transport mechanisms that supply the limiting nutrients.

The majority of microbes in the subsurface are firmly attached to soil particles. As a result, nutrients must be brought to the active sites by advection and diffusion of water in the saturated zone, or by soil gas, in the unsaturated zone. In the simplest and perhaps most common case, the compounds to be degraded for microbial energy and cell synthesis are transported in the aqueous phase by infiltrating water or by advective flow through the ground water. In the unsaturated zone, volatile organic compounds can move readily as vapors in the soil gas where oxygen is present. Below the water table, aerobic metabolism is limited by the low solubility of oxygen in water. Factors that control the rate of biological activity are the stoichiometry of the metabolic process, the concentration of the required nutrients in the mobile phases, the flow of the mobile phases, the opportunity for colonization in the subsurface by metabolically capable organisms, and the toxicity of the waste.

Much of the development work in the area of ground-water and soil remediation by biodegradation has been performed using petroleum products. The number of gasoline stations, underground tanks, and gasoline pipelines throughout the country and the potential for ground-water contamination have prompted considerable laboratory and field studies on in situ biodegradation of hydrocarbons.

Many of the enhanced biorestoration techniques now in use are variations on those developed by Raymond and his coworkers (Raymond, 1974; Raymond and others, 1986). This process reduces hydrocarbon contaminants in aquifers by enhancing the indigenous hydrocarbon-utilizing microflora. Nutrients and oxygen are introduced through injection wells and circulated through the contaminated zone by pumping one or more producing wells. The increased supply of nutrients and oxygen stimulates biodegradation of the hydrocarbons.

Raymond's process has been used with reasonable success to restore aquifers contaminated with gasoline. The overall removal of total hydrocarbons using this technology usually ranges from 70 to 80 percent. Some of the sites treated by this technique have been restored to the point where no dissolved gasoline was present in the ground water, and state regulatory standards were satisfied. State agencies charged with restoring other sites, however, have required that the operation continue until no trace of liquid gasoline could be detected. Most of the sites restored in this manner have had appropriate monitoring programs installed following remediation.

Usually the first step in the process is to use physical methods to recover as much of the gasoline as possible and then a detailed investigation of the hydrogeology is undertaken to determine the extent of the contamination. Laboratory studies are conducted to determine if the native microbes can degrade the contaminants and to determine the combination of minerals required to promote maximum cell growth at the ambient ground-water temperature and under aerobic conditions.

Considerable variations in nutrient requirements among aquifers have been noted. One aquifer required only the addition of nitrogen and phosphorus, while another was best stimulated by the addition of ammonium sulfate, mono- and disodium phosphate, magnesium sulfate, sodium carbonate, calcium chloride, and manganese and ferrous sulfate. It was found that a chemical analysis of the ground water was not helpful in estimating the nutrient requirements of the system.

Field investigations and laboratory studies guide the

design and installation of a system of wells for injecting the nutrients and oxygen, and for the control of ground-water flow. Controlling the ground-water flow is critical to moving oxygen and nutrients to the contaminated zone and optimizing the degradation process.

The technique developed by Raymond does not provide for treatment above the water table. Soils contaminated by leaking underground storage tanks may be physically removed during the process of removing the tank, however, this may not be practical with deep water tables or large areas of contamination. An alternative to soil removal is the construction of one or more infiltration galleries, which are used to recirculate the treated water back through the contaminated unsaturated zone. Oxygen may be added to the infiltrated water during an in-line stripping process for volatile organic contaminants or through aeration devices placed in the infiltration galleries.

The rate of biorestoration of hydrocarbons, either above or below the water table, is effectively the rate of supply of oxygen. Table 7-1 compares the number of times the water in the aquifer, or the air above it, must be replaced to restore subsurface materials of various textures. The calculations assume typical values for the volume occupied by air, water and hydrocarbons (De Pastrovich and others, 1979, Clapp and Horberger, 1978). The calculations further assume that the oxygen content of the water is 10 mg/L, that of the air is 200 mg/L and that the hydrocarbons are completely metabolized to carbon dioxide. These values are provided only to exemplify the processes involved and would differ at an actual site. The oxygen concentration in the water can be increased by using oxygen rather than air, which also would reduce the volumes of recirculated water required.

Hydrogen peroxide is an alternative source of oxygen in biorestoration and Raymond and others (1986) have patented a process of treatment with hydrogen peroxide. Iron or an organic catalyst may be used to decompose the hydrogen peroxide to oxygen. The rate at which hydrogen peroxide decomposes to oxygen must be controlled to limit the formation of bubbles that could lead to gas blockage and the loss of permeability. Hydrogen peroxide may mobilize metals, such as lead and antimony, and, if the water is hard, magnesium and calcium phosphates can precipitate and plug the injection well or infiltration gallery. To determine the microorganism's hydrogen peroxide tolerance level laboratory studies are performed.

Treatment Trains

In most contaminated hydrogeologic systems, the remediation process may be so complex, in terms of contaminant behavior and site characteristics, that no single system or unit is capable of meeting all requirements. Consequently, several unit operations may be combined in series or in parallel to effectively restore ground-water quality to the required level. Barriers and hydrodynamic controls may serve as temporary plume control measures, however, hydrodynamic processes are integral parts of any withdrawal and treatment or in situ treatment process.

Most remediation projects typically are started by removing the source. The next step may be the installation of pumping systems to remove free product floating on the water surface or the removal of soluble contaminants for treatment at the surface. Barriers also might be constructed to slow an advancing plume or to reduce the amount of water requiring treatment.

	Proportion of Total Subsurface Volume Occupied by:			Volumes Required to meet Hydrocarbons Oxygen Demand	
Texture	Hydrocarbons (when drained)	Air (when drained)	Water (when flooded)	Air	Water
Stone to Coarse Gravel	0.005	0.4	0.4	250	5,000
Gravel to Coarse Sand	0.008	0.3	0.4	530	8,000
Coarse to Medium Sand	0.015	0.2	0.4	1,500	15,000
Medium to Fine Sand	0.025	0.2	0.4	2,500	25,000
Fine Sand to Silt	0.040	0.2	0.5	4,000	32,000

Table 7-1. Estimated Volumes of Water or Air Required to Completely Renovate Subsurface Material that Contained Hydrocarbons at Residual Saturation

Enhanced biorestoration techniques may be feasible in some of the more diluted areas of the plume. In some circumstances, a site may reach final restoration goals using natural chemical and biological processes. An adequate monitoring program would be required to establish data on the progress of the restoration program.

Steps in treatment of contaminated ground water include the removal, collection, and delivery of the contaminated water to the treatment units, and in the case of in situ processes, delivery of the treatment materials to the contaminated areas in the aquifer. A thorough knowledge and understanding of the hydrogeologic and geochemical characteristics of the site are required to design a system that will optimize the remediation techniques selected, maximize the predictability of restoration effectiveness, and allow for the development of a cost-effective and lasting remediation program.

Institutional Limitations on Controling Ground-Water Contamination

The principal criteria for selecting remediation procedures are the water-quality level to which to restore an aquifer, and the most economical technology available to reach that level. Institutional limitations, however, sometimes override these criteria in determining if, when, and how remediation will be selected and carried out.

Response to a ground-water contamination problem is likely to require compliance with several local, state, and federal pollution control laws and regulations. If the response involves handling hazardous wastes, discharging substances into the air or surface waters, or injecting wastes underground, federal and state pollution control laws will apply. These laws do not exempt the activities of federal, state, or local officials or other parties attempting to remediate contamination problems. They apply to both generators and responding parties, and it is not unusual for these pollution control laws to conflict. A hazardous waste remediation project must meet RCRA permit requirements governing the transport and disposal of hazardous wastes, which can influence the selection of the remediation plan and the scheduling of cleanup activities.

In situ remediation procedures may be subject to permitting or other requirements under federal or state underground injection control programs. Withdrawal and treatment approaches may be subject to regulation under federal or state air pollution control programs or to pretreatment requirements if contaminated ground water is to be discharged to a surface water or to a municipal wastewater treatment system. A remediation plan involving pumping from an aquifer may be subject to state ground-water regulations on well construction and well spacing, and may need to consider various competing legal rights to extract ground water.

Other factors influencing selection and design of a ground-water remediation program include the availability of alternative sources of water supply, political and judicial constraints, and the availability of funds. Where alternate water supplies are plentiful and economical, there may not be a demand for total remediation; adequate remediation to protect human health and the environment may be sufficient. In the final analysis, responsible agencies can pursue remediation measures to the extent that resources are made available.

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*U.S. GOVERNMENT PRINTING OFFICE: 1994-550-001/80346

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EPA/625/6-90/016a

VOL. II

United States
Environmental Protection
Agency

Office of
Research and Development
Washington, DC 20460

EPA/625/6-90/016b
July 1991

EPA

Handbook

Ground Water

Volume II: Methodology

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Handbook

Ground Water
Volume II: Methodology

U.S. Environmental Protection Agency
Office of Research and Development

Center for Environmental Research Information
Cincinnati, OH 45268

Printed on Recycled Paper

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This document has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

This document is not intended to be a guidance or support document for a specific regulatory program. Guidance documents are available from EPA and must be consulted to address specific regulatory issues.

Contents

		Page
Chapter 1.	Monitoring Well Design and Construction	1
Chapter 2.	Ground-Water Sampling	22
Chapter 3.	Transport and Fate of Contaminants in the Subsurface	41
Chapter 4.	Ground-Water Tracers	67
Chapter 5.	Introduction to Aquifer Test Analysis	96
Chapter 6.	Models and Computers in Ground-Water Investigations	117

Acknowledgment

Many individuals contributed to the preparation and review of this handbook. The document was prepared by Eastern Research Group, Inc., for EPA's Center for Environmental Research Information, Cincinnati, OH. Contract administration was provided by the Center for Environmental Research Information.

Volume I, Ground Water and Contamination, was published in September 1990 (EPA/625/6-90/016a). Volume II, Methodology describes various investigative approaches and techniques. Although extensively revised, part of Volume II was obtained from previous publications, "Handbook: Ground Water" (EPA/625/6-87/016) and "Protection of Public Water Supplies from Ground-Water Contamination" (EPA/625/4-85/016).

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Preface

The subsurface environment of ground water is characterized by a complex interplay of physical, geochemical and biological forces that govern the release, transport and fate of a variety of chemical substances. There are literally as many varied hydrogeologic settings as there are types and numbers of contaminant sources. In situations where ground-water investigations are most necessary, there are frequently many variables of land and ground-water use and contaminant source characteristics which cannot be fully characterized.

The impact of natural ground-water recharge and discharge processes on distributions of chemical constituents is understood for only a few types of chemical species. Also, these processes may be modified by both natural phenomena and man's activities so as to further complicate apparent spatial or temporal trends in water quality. Since so many climatic, demographic and hydrogeologic factors may vary from place to place, or even small areas within specific sites, there can be no single "standard" approach for assessing and protecting the quality of ground water that will be applicable in all cases.

Despite these uncertainties, investigations are underway and they are used as a basis for making decisions about the need for, and usefulness of, alternative corrective and preventive actions. Decision makers, therefore, need some assurance that elements of uncertainty are minimized and that hydrogeologic investigations provide reliable results.

A purpose of this document is to discuss measures that can be taken to ensure that uncertainties do not undermine our ability to make reliable predictions about the response of contamination to various corrective or preventive measures.

EPA conducts considerable research in ground water to support its regulatory needs. In recent years, scientific knowledge about ground-water systems has been increasing rapidly. Researchers in the Office of Research and Development have made improvements in technology for assessing the subsurface, in adapting techniques from other disciplines to successfully identify specific contaminants in ground water, in assessing the behavior of certain chemicals in some geologic materials and in advancing the state-of-the-art of remedial technologies.

An important part of EPA's ground-water research program is to transmit research information to decision makers, field managers and the scientific community. This publication has been developed to assist that effort and, additionally, to help satisfy an immediate Agency need to promote the transfer of technology that is applicable to ground-water contamination control and prevention.

The need exists for a resource document that brings together available technical information in a form convenient for ground-water personnel within EPA and state and local governments on whom EPA ultimately depends for proper ground-water management. The information contained in this handbook is intended to meet that need. It is applicable to many programs that deal with the ground-water resource. However, it is not intended as a guidance or support document for a specific regulatory program.

GUIDANCE DOCUMENTS ARE AVAILABLE FROM EPA AND MUST BE CONSULTED TO ADDRESS SPECIFIC REGULATORY ISSUES.

Chapter 1

MONITORING WELL DESIGN AND CONSTRUCTION

The principal objective of constructing monitoring wells is to provide access to an otherwise inaccessible environment. Monitoring wells are used to evaluate topics within various disciplines, including geology, hydrology, chemistry, and biology. In ground-water quality monitoring, wells are used for collecting ground-water samples, which upon analysis may allow description of a contaminant plume, or the movement of a particular chemical (or biological) constituent, or ensure that potential contaminants are not moving past a particular point.

Ground-Water Monitoring Program Goals

Each purpose for ground-water monitoring-ambient monitoring, source monitoring, case preparation monitoring, and research monitoring-must satisfy somewhat different requirements, and may necessitate different strategies for well design and construction (Barcelona and others, 1984). At the outset,the goals of the intended monitoring program must be clearly understood and thought should be given to the potential future use of the wells in other, possibly different, monitoring programs.

Regional investigations of ground-water quality involve ambient monitoring. Such investigations seek to establish an overall picture of the quality of water within all or parts of an aquifer. Generally, sample collection is conducted routinely over a period of many years to determine changes in quality over time. Often, changes in quality are related to long-term changes in land use (e.g., the effects of urbanization). Monitoring conducted for Safe Drinking Water Act compliance generally falls in this category.

Samples commonly are collected from a variety of public and private water supply wells for ambient quality investigations. Because of the diversity of sources, the data obtained through some ambient monitoring programs may not meet the strict well design and construction requirements imposed by the three other types of monitoring. However, such programs are important for detecting significant changes in aquifer water quality over time and space and protecting public health.

Regulatory monitoring at potential contaminant sources is considered source monitoring. Under this type of program, monitoring wells are located and designed to detect the movement of specific pollutants outside of the boundaries of a particular facility (e.g., treatment, storage, or disposal). Ground-water sampling to define contaminant plume extent and geometry would fall into this monitoring classification. Monitoring well design and construction are tailored to the site geology and contaminant chemistry. With source monitoring, quantitative aspects of analytical results become most important because the level of contaminant concentration may require specific regulatory action.

Monitoring for case preparation, such as legal proceedings in environmental enforcement, requires a level of detail similar to source monitoring. Source monitoring, in fact, often becomes a part of legal proceedings to establish whether or not environmental damage has occurred and to identify the responsible party. This is a prime example of one type of monitoring program evolving into another. The appropriateness and integrity of monitoring well design and construction methods will come under close scrutiny in legal proceedings. In such cases, the course of action taken during the monitoring investigation, the decisions made concerning well design and construction, and the reasons for those decisions must be clearly established and documented.

Monitoring for research generally requires a level of sophistication beyond that required of any other type of monitoring (this, of course, depends upon the types and concentrations of constituents being sought and the overall objectives of the research). Detailed information is often needed to support the basic concepts and expand understanding of the complex mechanisms of

ground-water movement and solute/contaminant transport.

The goals of any proposed ground-water monitoring program should be clearly stated and understood before making any decisions on the types and numbers of wells needed, their locations, depth, constituents of interest, and methods of collection, storage, transportation, and analysis.

As each of these decisions is made, consideration must be given to the costs involved in each step of the monitoring program and how compromises in one step may affect the integrity and outcome of the other steps. For example, cost savings in well construction materials may so severely limit the usefulness of a well that another well may need to be constructed at the same location for the reliable addition of a single chemical parameter.

Monitoring Well Design Components

Monitoring well design and construction methods follow production well design and construction techniques; a monitoring well ,however, is built specifically to give access to the ground water so that a "representative" sample of water can be withdrawn and analyzed. While well efficiency and yield is important, the ability to produce large amounts of water for supply purposes is not the primary objective.

Emphasis is placed instead on constructing a well that will provide easily obtainable ground-water samples that will give reliable, meaningful information. Therefore, materials and techniques used for constructing a monitoring well must not materially alter the quality of the water being sampled. An understanding of the chemistry of suspected pollutants and the geologic setting in which the monitoring well is to be constructed play a major role in the drilling technique and well construction materials used.

Several components need to be considered in monitoring well design: location and number of wells, diameter, casing and screen material, screen length and depth of placement, sealing material, well development, and well security. Often, discussion of one component will impinge upon other components.

Location and Number

Locating monitoring wells spatially and vertically to ensure that the ground-water flow regime of concern is being monitored is obviously one of the most important components in ground-water quality monitoring design. Monitoring well locations (sites) and the number of wells in the monitoring program are closely linked. The number of wells and their location are principally determined by the purpose of the monitoring program. In most monitoring situations, the goal is to determine the effect that some surface or near-surface activity has had on nearby ground-water quality. Most dissolved constituents will descend vertically through the unsaturated zone beneath the area of activity and then, upon reaching the saturated zone, move horizontally in the direction of ground-water flow. Therefore, monitoring wells are normally completed downgradient in the first permeable water-bearing unit encountered. Consideration should be given to natural (seasonal) fluctuations, which can amount to several feet throughout the year and from one year to the next, and artificial fluctuations brought about largely by pumping, which can amount to several tens of feet in only a few hours. Artificial fluctuations also are caused by lagoon operation, which can cause a rise or "mound" in the water table.

Preliminary boreholes and monitoring wells can be constructed to collect and analyze geologic material samples, to measure ground-water levels, and to collect water-quality samples, all of which provide a guide to the future placement of additional wells. Accurate water-level information must be obtained to determine if local ground-water flow paths and gradients differ significantly from the regional appraisal.

The analysis of water-quality samples from the preliminary wells can direct the placement of additional wells. Such data are helpful in the vertical arrangement of sampling points, especially for a contaminant that is denser than water. Without some preliminary chemical data, it is usually very difficult to determine the location of the most contaminated zone.

Site geology, site hydrology, source characteristics, contaminant characteristics, and the size of the area under investigation all help determine where and how many wells should be constructed. Certainly, the more complicated the geology and hydrology, the more complex the contaminant and source, and the larger the area being investigated, the greater the number of monitoring wells that will be required.

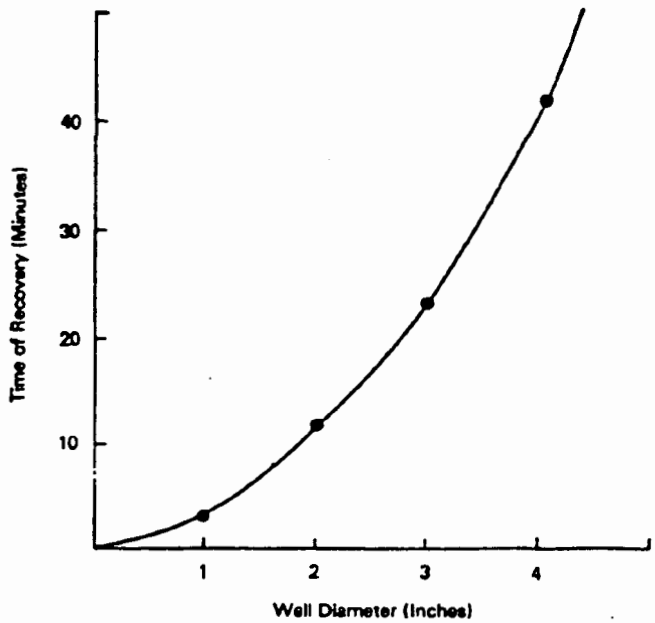
Diameter

In the past, the diameter of a monitoring well was based primarily on the size of the device (bailer, pump, etc.) used to withdraw the water samples. This practice was similar to that followed for water supply well design. For example, a domestic water well is commonly 4 to 6 inches in diameter, which is of sufficient size to accommodate a submersible pump capable of delivering from 5 to 20 gallons per minute. Municipal, industrial,

and irrigation wells have greater diameters to handle larger pumps, and to increase the available screen open area so the well can produce water efficiently.

This practice worked well in very permeable formations, where an aquifer capable of furnishing large volumes of water was present. However, unlike most water-supply wells, monitoring wells are quite often completed in very marginal water-producing zones. Pumping one or more well volumes of water (the amount of water stored in the well casing under nonpumping conditions) from a well built in low-yielding materials (Gibb and others, 1981) may present a serious problem if the well has a large diameter.

Figure 1-1 illustrates the amount of water in storage per foot of casing for different well casing diameters. Well casings with diameters of 2 and 6 inches will contain 0.16 and 1.47 gallons of water per foot of casing, respectively. Purging four well volumes from a well containing 10 feet of water would require removal of 6.4 gallons of water from a 2-inch well and 58.8 gallons of water from a 6-inch well. Under low-yielding conditions, it can take considerable time to recover enough water from the well to collect a sample (see Figure 1-2).



Assumptions: $K = 1 \times 10^{-5}$ cm/sec, well screen = 10′, 10′ of water above screen, 6′ of water instantaneously removed

Figure 1-2. Time Required for Recovery After Slug of Water Removed (from Rinaldo-Lee, 1983)

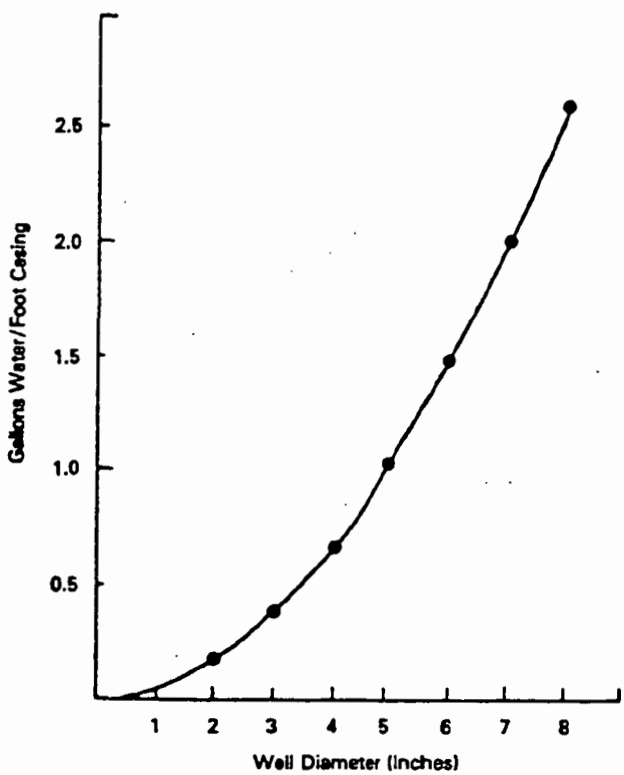


Figure 1-1. Volume of Water Stored Per Foot of Well Casing for Different Diameter Casings (from Rinaldo-Lee, 1983)

In addition, when hazardous constituents are present in the ground water, the purged water must be properly disposed. Therefore, the quantity of water pumped from the well should be minimized for reasons of safety, as well as disposal cost. Cost of well construction also is a consideration. Wells less than 4 inches in diameter are much less expensive than large diameter wells in terms of both cost of materials and cost of drilling.

For these reasons and with the advent of a variety of commercially available small-diameter pumps (less than 2 inches OD) capable of lifting water over 100 feet, 2-inch ID wells have become the standard in monitoring well technology.

Large diameter wells can be useful in situations where monitoring may be followed by remedial actions involving reclamation and treatment of the contaminated ground water. In some instances, the "monitoring" well may become a "supply" well to remove contaminated water for treatment. Larger diameter wells also merit consideration when monitoring is required at depths of hundreds of feet and in situations where the additional strength of large diameter casing is needed. For sampling at several depths beneath one location, several monitoring wells have been nested in a single borehole (Johnson, 1983). This type of technique will require drilling a larger diameter hole to accommodate the multiple well casings. Again, the use of smaller diameter

casing provides advantages by allowing more wells to be nested in the borehole, thus easing construction and reducing drilling expenses.

Casing and Screen Material

The type of material used for a monitoring well can have a distinct effect on the quality of the water sample to be collected (Barcelona and others, 1985; Gillham and others, 1983; and Miller, 1982). The materials of choice should retain their structural integrity for the duration of the monitoring program under actual subsurface conditions. They should neither adsorb nor leach chemical constituents that would bias the representativeness of the samples collected.

Galvanized steel casing can impart iron, manganese, zinc, and cadmium to many waters, and steel casing may contribute iron and manganese to a sample. PVC pipe has been shown to release and adsorb trace amounts of various organic constituents to water after prolonged exposure (Miller, 1982). PVC solvent cements used to attach sections of PVC pipe also have been shown to release significant quantities of organic compounds.

Teflon^R and glass are among the most inert materials considered for monitoring well construction. Glass, however, is difficult and expensive to use under most field conditions. Teflon^R also is very expensive; with technological advances, Teflon^R-coated casings and screens may become available. Stainless steel also offers desirable properties for monitoring, but it too is expensive.

A reasoned strategy for ground-water monitoring must consider the effects of contaminated water on well construction materials as well. Unfortunately, there is limited published information on the performance of specific materials in varied hydrogeologic settings (Pettyjohn and others, 1981). The following is a preliminary ranking of commonly used materials exposed to different solutions representing the principal soluble species present in hazardous waste site investigations (Barcelona and others, 1984). They are listed in order of best to worst in terms of chemical resistance:

Teflon^R
Stainless Steel 316
Stainless Steel 304
PVC Type 1
Lo-Carbon Steel
Galvanized Steel
Carbon Steel

Polyvinyl chloride (PVC Type I) is very chemically resistant except to low molecular weight ketones, aldehydes, and chlorinated solvents. Generally, as the organic content of a solution increases, direct attack on the polymer matrix or solvent absorption, adsorption, or leaching may occur. This reaction, however, has not been observed with Teflon^R. Provided that sound construction practices are followed, Teflon^R can be expected to outperform all other casing and sampling materials (Barcelona and others, 1984).

Stainless steels are the most chemically resistant of the ferrous materials. Stainless steel, however, may be sensitive to the chloride ion, which can cause pitting corrosion, especially over long-term exposures under acidic conditions. Given the similarity in price, workability, and performance, the remaining ferrous materials (lo-carbon, galvanized steel, and carbon) provide little advantage over one another for casing/screen construction.

Significant levels of organic components found in PVC primers and adhesives (such as tetrahydrofuran, methylethylketone, cyclohexanone, and methylisobutylketone) were detected in well water several months after well installation (Sosebee and others, 1982). The presence of compounds such as these can mask the presence of other similar volatile compounds (Miller, 1982). Therefore, when using PVC and other similar materials, such as ABS, polypropylene, or polyethylene, for well construction, threaded joints are the preferred means for connecting sections together.

In many situations, it may be possible to compromise accuracy or precision for initial cost, depending on the objectives of the monitoring program. For example, if the contaminants of interest are already defined and they do not include substances that might bleed or sorb, it may be reasonable to use wells cased with a less expensive material.

Wells constructed of less than optimum materials might be used for sampling if identically fabricated wells are constructed in uncontaminated parts of the monitored aquifer to provide ground-water samples for use as "blanks" (Pettyjohn and others, 1981). Such blanks, however, may not adequately address problems of adsorption on, or leaching from, the casing material induced by contaminants in the ground water. It may be feasible to use two or more kinds of casing materials in the saturated zone and above the seasonal high water table, such as Teflon^R or stainless steel, and use a more appropriate material, such as PVC or galvanized steel casing, above static water level.

Trying to save money by compromising on material quality or suitability, however, may eventually increase program cost by creating the need for reanalysis, or

worse, monitoring well reconstruction. Each case requires careful consideration and the analytical laboratory should be fully aware of the construction materials used.

Care also must be taken in preparing the casing and well screen materials prior to installation. At a minimum, materials should be washed with detergent and rinsed thoroughly with clean water. Steam-cleaning and high pressure, hot water cleaners provide excellent cleaning of cutting oils and lubricants left on casings and screens after their manufacture (particularly for metal casing and screen materials). To ensure that these and other sampling materials are protected from contamination prior to placement down-hole, materials should be covered (with plastic sheeting or other material), and kept off the ground.

All wells should allow free entry of water. The water produced should be as clear and silt-free as possible. For drinking water supplies, sediment in the raw water can create additional pumping and treatment costs and lead to the general unpalatability of the water. With monitoring wells, sediment-laden water can greatly lengthen filtering time and create chemical interference in sample analyses.

Commercially manufactured well screens are preferred for monitoring wells so long as the screen slot size is appropriate. Sawed or torch-cut casing may be satisfactory in deposits where medium to coarse sand or gravel predominate. In formations where fine sand, silt, and clay predominate, sawed or torch-cut slots will be too large to retain the aquifer materials, and the well may clog or fill with sediment. The practice of sawing slots in PVC pipe should be avoided in monitoring situations where organic chemicals are of concern, because this procedure exposes fresh surfaces of PVC, increasing the possibility of releasing compound ingredients or reaction products.

It may be helpful to have several slot-sized well screens on site so that the proper manufactured screen and slots can be placed in the hole after the aquifer materials have been inspected. Gravel pack of a size compatible with the selected screen slot size will further help retain the finer fractions of material and allow free entry of water into the well by creating a zone of higher permeability around the well screen.

For natural-packed wells, where relatively homogeneous, coarse materials predominate, a slot size should be selected based on the effective size and uniformity coefficient of the formation materials. The effective size is equivalent to the sieve size that will retain 90 percent (or passes 10 percent) of the formation material; the uniformity coefficient is the ratio of the sieve size that will retain 40 percent (or pass 60 percent) of the formation material to the effective size (Aller and others 1989). If an artificial pack is used, a uniform gravel-pack size that is from three to five times the 50 percent retained size of the formation and a screen size that will retain at least 90 percent of the pack material should be selected (Walker, 1974). The gravel-pack should be composed of clean, uniform quartz sand.

The gravel-pack should be placed carefully to avoid bridging in the hole and to allow uniform settling around the screen. A tremie pipe can be used to guide the sand to the bottom of the hole and around the screen. The pipe should be lifted slowly as the annulus between the screen and borehole as the borehole fills. If the depth of water standing in the annulus is not great, the sand can be simply poured from the surface. The volume of sand required to fill the annulus to the desired depth (usually about 1 foot above the top of the screen) should be calculated. Field measurements should be taken to confirm that the pack has reached this level before backfilling or sealing procedures start.

Screen Length and Depth of Placement

The length of screen and the depth at which it is placed depend, to a large degree, on the behavior of the contaminant as it moves through the unsaturated and saturated zones, and on the goal of the monitoring program. When monitoring an aquifer used as a water supply, the entire thickness of the water-bearing formation could be screened (just as a production well might be). In regional aquifer studies, production wells commonly are used for sampling. Such samples would provide water integrated over the entire thickness of the water-bearing zone(s), and would be similar in quality to what would be found in a drinking-water supply.

When specific depth intervals must be sampled at one location, vertical nesting of wells is common. This technique is often necessary when the saturated zone is too thick to adequately monitor with one long-screened section (which would dilute the collected sample). Since contaminants tend to stratify within the saturated zone, collecting a sample integrated over a thick zone will provide little or no information on the depth and concentration that a contaminant may have reached. Furthermore, nested wells provide information on the water level or potential that exists at each well screen. These data are essential to an understanding of the vertical component of flow.

Screen lengths of 1 to 2 feet are common in detailed plume geometry investigations. Thick aquifers would require that several wells be completed at different depth intervals. In such situations (and depending on

the magnitude of the aquifer saturated thickness), screen lengths of no more than 5 to 10 feet should be used. Monitoring wells can be constructed in separate holes placed closely together or in one larger diameter hole, as shown in Figure 1-3. Vertical movement of contaminants in the well bore before and after well completion may be difficult to prevent since it is difficult to seal several wells in one hole.Thus, multiple holes may need to be drilled to ensure well integrity. Specially constructed installations have been developed to sample a large number of points vertically over short intervals (Morrison and Brewer, 1981; Pickens, 1981; and Torstensson, 1984; Figures 1-4 and 1-5).

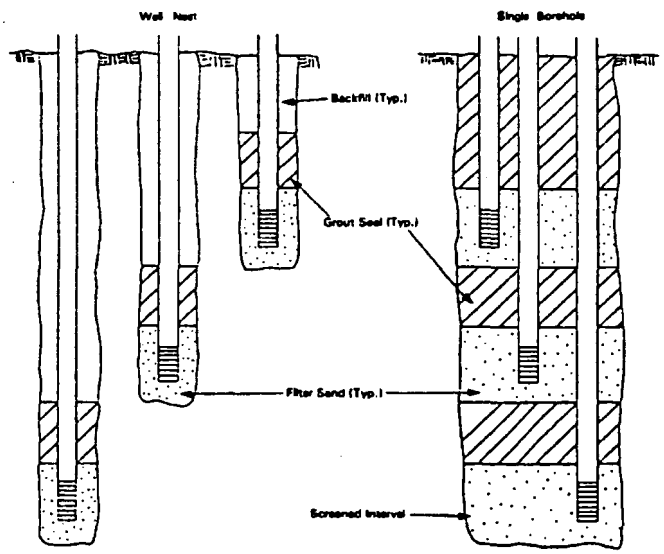


Figure 1-3. Typical Multiwell Installations (from Johnson, 1983)

In other situations, only the first water-bearing zone encountered will require monitoring (for example, when monitoring near a potential contaminant source in a relatively impermeable glacial till). The "aquifer" or zone of interest in such an instance may be only several inches to a few feet thick. Screen length should be limited to 1 to 2 feet in these cases to minimize siltation problems from surrounding fine-grained materials and possible dilution effects from water contributed by uncontaminated zones.

Because of the chemical reactions that occur when ground water contacts the atmosphere, particularly when dealing with volatile compounds, the screened section should not be aerated. Generally, well depth should assure that the screened section is always fully submerged. The design should consider fluctuations in the elevation of the top of the saturated zone caused by seasonal variations or human-induced changes.

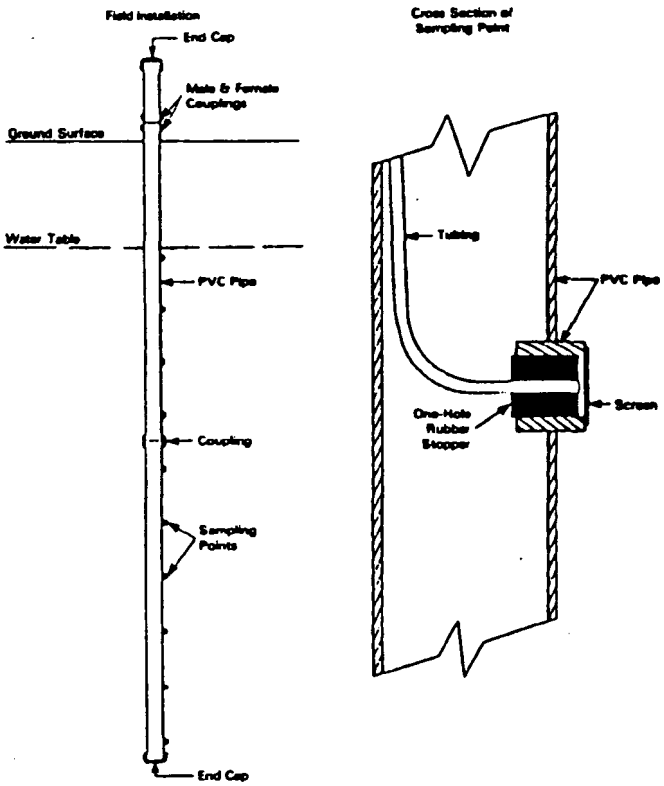


Figure 1-4. Schematic Diagram of a Multilevel Sampling Device (from Pickens and others, 1981)

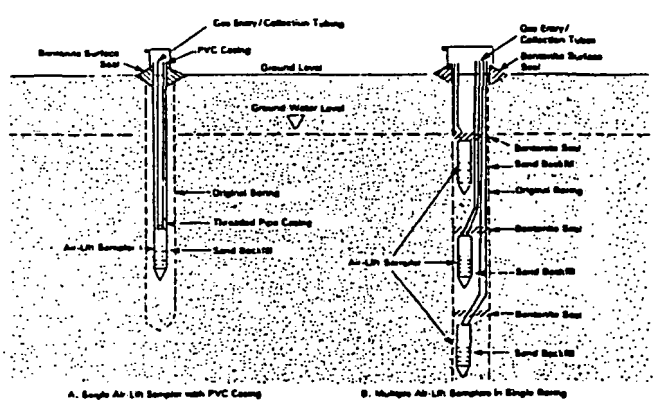


Figure 1-5. Single (a) and Multiple (b) Installation Configurations for an Air-Lift Sampler (from Morrison and Brewer, 1981)

Monitoring for contaminants with densities different than water demands special attention. In particular, low density organic compounds, such as gasoline, will float on the ground-water surface (Gillham and others, 1983). Monitoring wells constructed to detect floating contaminants should contain screens that extend above the zone of saturation so that these lighter substances can enter the well. The screen length and position must

accommodate the magnitude and depth of variations in water-table elevation. However, the thickness of floating products in the well does not necessarily indicate the thickness of the product in the aquifer.

Sealing Materials and Procedures

It is critical that the screened part of each monitoring well access ground water from a specific depth interval. Vertical movement of ground water in the vicinity of the well can greatly influence sample quality (Keith and others, 1982). Rainwater can infiltrate backfill, potentially diluting or contaminating samples; vertical seepage of leachate along the well casing will also produce unrepresentative samples (particularly important in multilevel installations such as in Figures 1-3, 1-4, and 1-5). Even more importantly, the creation of a conduit in the annulus of the monitoring well that could contribute to or hasten the spread of contamination is to be strictly avoided. Several methods have been employed successfully to isolate contaminated zones during the drilling process (Burkland and Raber, 1983; Perry and Hart, 1985).

Monitoring wells are usually sealed with neat cement grout, dry bentonite (powdered, granulated, or pelletized), or bentonite slurry. Well seals usually are installed at two places within the annulus: one just above the screened interval and the other at the ground surface to inhibit downward leakage of surface contaminants.

Bentonite traditionally has been considered to provide a much better seal than cement. However, recent investigations on the use of clay liners for hazardous waste disposal have shown that some organic compounds migrate through bentonite with little or no attenuation (Brown, and others, 1983). Therefore, cement may offer some benefits over bentonite.

Bentonite most often is used as a down-hole seal to prevent vertical migration within the well annulus. When bentonite must be placed below the water table (or where water has risen in the borehole), it is recommended that a bentonite slurry be tremied down the annulus to fill the hole from the bottom upward. In collapsible material conditions, where the borehole has collapsed to a point just above the water table, dry bentonite (granulated or pelletized works best) can be poured down the hole.

Bentonite clay has appreciable ion-exchange capacity, which may interfere with the chemistry of collected samples when the seal is adjacent to the screen or well intake. When improperly placed, cement grout has been known to seriously affect the pH of sampled water. Therefore, special attention and care should be exercised during placement of a down-hole seal. Approximately 1 foot (at a minimum) of gravel-pack or naturally collapsed material should extend above the top of the well screen to ensure that the sealing materials do not migrate downward into the well screen. If the sealing material is too watery before being placed down the hole, sealing materials may settle or migrate into the gravel-pack or screened area, and the fine materials in the seal may penetrate the natural or artificial pack.

While a neat cement (sand and cement, no gravel) grout is often recommended, especially for surface sealing, shrinkage and cracking of the cement upon curing and weathering can create an improper seal. Shrink-resistant cement (such as Type K Expansive Cement) and mixtures of small amounts of bentonite with neat cement have been used successfully to help prevent cracking.

Development

Development is a facet of monitoring well installation that often is overlooked. During the drilling process, fine-grained materials smear on the sides of the borehole, forming a mud "cake" that reduces the hydraulic conductivity of the materials opposite the screened part of the well. To facilitate entry of water into the monitoring well (a particularly important factor for low-yielding geologic materials), this mud cake must be broken down and the fine-grained materials removed from the well or well bore. Development also removes fluids, primarily water, which are introduced to the water-bearing formations during the drilling process.

Additionally, monitoring wells must be developed to provide water free of suspended solids for sampling. When sampling for metal ions and other inorganic constituents, water samples must be filtered and preserved at the well site at the time of sample collection. Improperly developed monitoring wells will produce samples containing suspended sediments that will both bias the chemical analysis of the collected samples and frequently cause clogging of the field filtering mechanisms.

The time and money spent for this important procedure will expedite sample filtration and result in samples more representative of water contained in the formation being monitored. The time saved in field filtration alone will more than offset the cost of development.

Successful development methods include bailing, surging, and flushing with air or water. The basic principle behind each method is to create reversals of flow in and out of the well (and/or borehole), which tend to break down the mud cake and draw the finer materials into the hole for removal. This process also aids in removing the finer fraction of materials in proximity to the borehole,

leaving behind a "natural" pack of coarser-grained materials.

Years ago, small-diameter well development most commonly was achieved through use of a bailer. The bailer was about the only "instrument" that had been developed for use in such wells. Rapidly dropping and retrieving the bailer in and out of the water caused a back-and-forth action of water in the well, moving some of the more loosely bound fine-grained materials into the well where they could be removed.

Depending on the depth of water in the well, the length of the well screen, and the volume of water the bailer could displace, this method was not always very efficient. "Surge blocks," which could fit inside 2-inch diameter wells, provided some improvement over bailing techniques. Such devices are simply plungers that, when moved vigorously up and down, transfer that energy to an in-and-out action on the water near the well screen. Surge blocks have the potential to move larger quantities of water with higher velocities, but they pose some risk to the well casing and screen if the surge block fits too tightly or if the up-and-down action becomes too vigorous. Improved surge block design has been the subject of some recent investigation (Schalla and Landick, 1985).

In more productive aquifers, "overpumping" has been a popular method for well development. With this method, a pump is alternately turned on (usually at a slightly higher rate than the well can sustain) and off to simulate a surging action in the well. A problem with this method is that overpumping does not create as pronounced an outward movement of water as does surging. Overpumping may tend to bridge the fine and coarse materials, limiting the movement of the fine materials into the well and thereby limiting the effectiveness of the method.

Pumping with air also has been used effectively (Figure 1-6). Better development has been accomplished by attaching differently shaped devices to the end of an airline to force the air out into the formation. Figure 1-7 shows an example. Such a device causes a much more vigorous action on the movement of material in proximity to the well screen while also pushing water to the ground surface.

Air development techniques may expose field crews to hazardous constituents when highly contaminated ground water is present. The technique also may cause chemical reactions with species present in the ground water, especially volatile organic compounds. Care also must be taken to filter the injected air to prevent contamination of the well environment with oil and other lubricants present in the compressor and airlines.

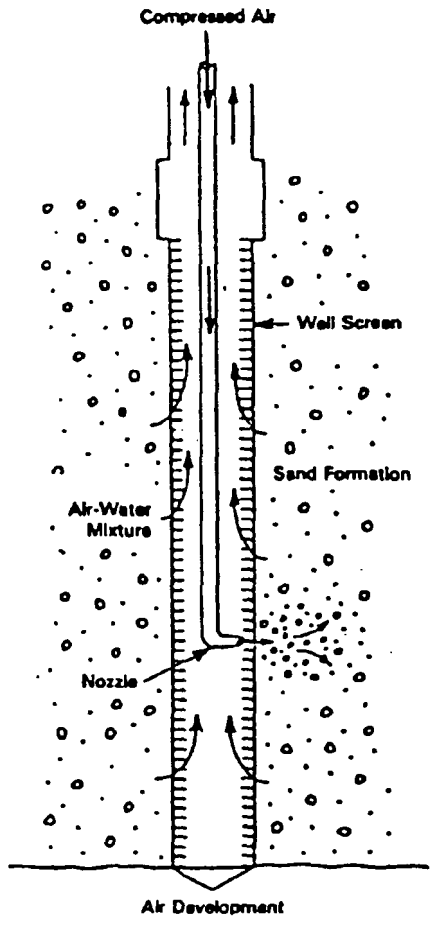


Figure 1-6. Well Developments with Compressed Air

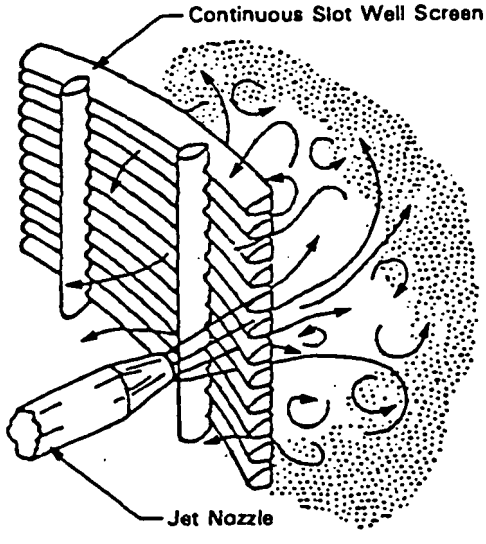


Figure 1-7. The Effects of High-Velocity Jetting Used for Well Development through Openings in a Continuous-Slot Well Screen

Development procedures for monitoring wells in relatively unproductive geologic materials are somewhat limited. Due to the low hydraulic conductivity of the materials, surging of water in and out of the well casing is extremely difficult. Also, when the well is pumped, the entry rate of the water is inadequate to effectively remove fines from the well bore and the gravel-pack material outside the well screen.

Where an open borehole can be sustained in this type of geologic setting, clean water can be circulated down the well casing, out through the screen, and back up the borehole (Figure 1-8). Relatively high water velocities can be maintained and the mud cake from the borehole wall can be broken down effectively and removed. Because of the low hydraulic conductivity of the geologic materials outside the well, only a small amount of water will penetrate the formation being monitored. This development procedure can be done before and after placement of a gravel-pack but must be conducted before a well has been sealed. After the gravel-pack has been installed, water should not be circulated too quickly or the gravel-pack will be lifted out of the borehole. Immediately following development, the well should be sealed, backfilled, and pumped for a short period to stabilize the formation around the outside of the screen and to ensure that the well will produce fairly clear water.

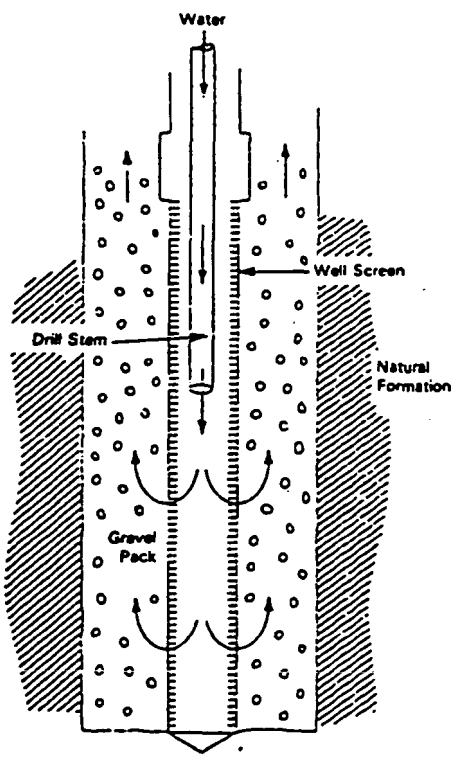


Figure 1-8. Well Development by Back-Flushing with Water

Security

For most monitoring well installations, some precautions must be exercised to protect the surface portions of the well from damage. In many instances, inadvertent vehicular accidents do occur; also, monitoring well installations seem particularly vulnerable to grass mowers. Vandalism is often a major concern, from spontaneous "hunters" looking for a likely target to premeditated destruction of property associated with an unpopular operation. Several simple solutions can be employed to help minimize the damage due to accidental collisions. However, outwitting the determined vandal may be an impossible undertaking and certainly an expensive one.

The basic problem in maintaining the physical condition of any monitoring well is anticipating the hazards that might befall that particular installation. Some situations may call for making the well highly visible whereas others may require keeping the well inconspicuous.

Where the most likely problem is one of vehicular contact, be it mowers, construction traffic, or other types of two-, three-, or four-wheeled traffic, the first thing that can be done is to make the top of the well easy to see. It should extend far enough above ground to be visible above grass, weeds, or small shrubs. If that is not practical, use a "flag" that extends above the well casing. A flag is also helpful for periods when leaves or snow have buried low-lying objects.

The well casing should also be painted a bright color (orange and yellow are the most visible). This not only makes the well more visible but also protects metal casing material from rusting. Care should be taken to prevent paint from getting inside the well casing or in threaded fittings that may contact sampling equipment.

The owners/operators of the site being monitored should also know the location of each installation. They should receive maps clearly and exactly indicating the position of the wells, and their employees should be informed of the importance of those installations, the cost associated with them, and the difficulty involved in replacement.

The segment of the well that extends above the ground also can be reinforced, particularly if the well is constructed of PVC or Teflon^R. The well could be constructed such that only the portion of the well above the water table is metal. In this manner, the integrity of the sample is maintained as ground water contacts only inert material, and the physical condition of the well is maintained as the upper metal portion is better able to withstand impact.

There are two arguments to consider when constructing

a well in this manner. The arguments focus on the weak point in the well construction: at or near the juncture of the metal and nonmetal casings. One argument suggests that a longer section of metal casing is superior because its additional length in the ground provides more strength. Thus, a break is less likely to occur (although the casing is likely to be bent). The other argument suggests that should a break in the casing occur, a shorter length of metal casing is superior because a break nearer to ground surface is easier to repair. Each argument has its merits; only experience with site conditions is likely to produce the best solution.

The use of "well protectors" is another popular solution that involves the use of a larger diameter steel casing placed around the monitoring well at the ground surface and extending several feet below ground (Figure 1-9). The protectors are usually seated in the cement surface seal to a depth below the frost line.

Commonly, well protectors are equipped with a locking cap, which ensures against tampering with the inside of the well. Dropping objects down the well may clog the well screen or prohibit the sampling device from reaching water, and the quality of the ground water may be altered, particularly where small quantities (perhaps drops) of an organic liquid may be sufficient to completely contaminate the well.

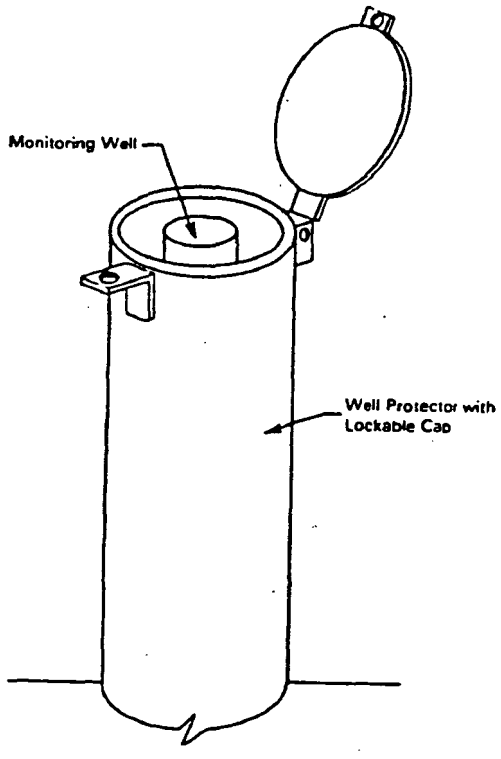


Figure 1-9. Typical Well Protector Installation

Problems associated with vandalism run from simple curiosity to outright wanton destruction. Obviously, sites within secured, fenced areas are less likely to be vandalized. However, there is probably no sure way to deter the determined vandal, short of posting a 24-hour guard. In such situations, well protectors are a must. The wells should be kept as inconspicuous as possible. However, the benefits of "hiding" monitoring wells must be weighed against the costs of delays in finding them for sampling and the potential costs for repairs or maintenance on untried security designs.

In some situations, it might be a good policy to notify the public of the need for the monitoring wells. Properly asserting that each well serves an environmental monitoring purpose and that the wells have been constructed to ensure public well-being may create a civic conscience that would help to minimize vandalism.

As with all the previously mentioned monitoring well components, no single solution will best meet every monitoring situation. Knowledge of the social, political, and economic conditions of the geographic area and circumstances surrounding the need for ground-water monitoring will dictate, to a large degree, the type of well protection needed.

Monitoring Well Drilling Methods

As might be expected, different drilling techniques can influence the quality of a ground-water sample. This applies to the drilling method employed (e.g., augered, driven, or rotary), as well as the driller. There is no substitute for a conscientious driller willing to take the extra time and care necessary to complete a good monitoring well installation.

Among the criteria used to select an appropriate drilling method are the following factors, listed in order of importance:

1. Hydrologic information
   a. type of formation
   b. depth of drilling
   c. depth of desired screen setting below the top of the zone of saturation
2. Types of pollutants expected
3. Location of drilling site, i.e., accessibility
4. Design of monitoring well
5. Availability of drilling equipment

Table 1-1 summarizes several different drilling methods, and their advantages and disadvantages when used for monitoring well construction. Several excellent publications are referenced for detailed discussions (Campbell and Lehr, 1973; Fenn and others, 1977;

Method	Drilling Principle	Advantages	Disadvantages
Drive Point	1.25 to 2 inch ID casing with pointed screen mechanically driven to depth.	Inexpensive. Easy to install, by hand if necessary. Water samples can be collected as driving proceeds. Depending on overburden, a good seal between casing and formation can be achieved.	Difficult to sample from smaller diameter drive points if water level is below suction lift. Bailing possible. No formation samples can be collected. Limited to fairly soft materials. Hard to penetrate compact, gravelly materials. Hard to develop. Screen may become clogged if thick clays are penetrated. PVC and Teflon casing and screen are not strong enough to be driven. Must use metal construction materials which may influence some water quality determinations.
Auger, Hollow- and Solid-stem	Successive 5-foot flights of spiral-shaped drill stem are rotated into the ground to create a hole. Cuttings are brought to the surface by the turning action of the auger.	Inexpensive. Fairly simple operation. Small rigs can get to difficult-to-reach areas. Quick set-up time. Can quickly construct shallow wells in firm, noncavey materials. No drilling fluid required. Use of hollow-stem augers greatly facilitates collection of split-spoon samples. Small-diameter wells can be built inside hollow-stem flights when geologic materisl are cavey.	Depth of penetration limited, especially in cavedy materials. Maximum depths 150 feet. Cannot be used in rock or well-cemented formations. Difficult to drill in cobbles/boulders. Log of well is difficult to interpret without collection of split spoons due to the lag time for cuttrings to reach ground surface. Vertical leakage of water through borehole during driling is likely to occur. Solid-stem limited to fine grained, unconsolidated materials that will not collapse when unsupported. With hollow-stem flights, heaving materials can present a problem. May need to add water down auger to control heaving or wash materials from auger before completing well.
Jetting	Washing action of water forced out of the bottom of the drill rod clears hole to allow penetration. Cuttings brought to surface by water flowing up the outside of the drill rod.	Inexpensive. Driller often not needed for shallow holes. In firm, noncavey deposits where hole will stand open, well construction fairly simple.	Somewhat slow, especially with increasing depth. Extremely difficult to use in very coarse materials, i.e., cobbles/boulders. A water supply is needed that is under enough pressure to penetrate the geologic materials present. Difficult to interpret sequence of geologic materisl from cuttings. Maximum depth 150 feet, depending on geology and water pressure capabilities.
Cable-tool (Percussion)	Hole created by dropping a heavy "string" of drill tools into well bore, crushing materials at bottom. Cuttings are removed occasionally by bailer. Generally, casing is driven just ahead of the the bottom of the hole; a hole greater than 6 inches in diameter is usually made.	Can be used in rock formations as well as unconsolidated formations. Fairly accurate logs can be prepared from cuttings if collected often enough. Driving a casing ahead of hole minimizes cross-contamination by vertical leakage of formation waters. Core samples can be obtained easily.	Requires an experienced driller. Heavy steel drive pipe used to keep hole open and drilling "tools" can limit accessibility. Cannot run some geophysical logs due to presence of drive pipe. Relatively slow drilling method.

Table 1-1. Advantages and Disadvantages of Selected Drilling Methods for Monitoring Well Construction

Method	Drilling Principle	Advantages	Disadvantages
Hydraulic Rotary	Rotating bit breaks formation; cuttings are brought to the surface by a circulating fluid (mud). Mud is forced down the bit, and up the annulus between the drill stem and hole wall. Cuttings are removed by settling in a "mud pit" at the ground surface and the mud is circulated back down the drill stem.	Drilling is fairly quick in all types of geologic materials. Borehole will stay open from formation of a mud wall on sides of borehole by the circulating drilling mud. Eases geophysical logging and well construction. Geologic cores can be collected. Virtually unlimited depths possible.	Expensive, requires experienced driller and fair amount of peripheral equipment. Completed well may be difficult to develop, especially small-diameter wells, because of mud wall on borehole. Geologic logging by visual inspection of cuttings is fair due to persence of drilling mud. Thin beds of sand, gravel, or clay may be missed. Presence of drilling mud can contaminate water samples, especially the organic, biodegradable muds. Circulation of drilling fluid through a contaminated zone can create a hazard at the ground surface with the mud pit and cross-contaminate clean zones during circulation.
Reverse Rotary	Similar to Hydraulic Rotary method except the drilling fluid is circulated down the borehole outside the drill stem and is pumped up the inside, just the reverse of the normal rotary method. Water is used as the drilling fluid, rather than a mud, and the hole is kept open by the hydrostatic pressure of the water standing in the borehole.	Creates a very "clean" hole, not dirtied with drilling mud. Can be used in all geoloic formations. Very deep penetrations possible. Split-spoon sampling possible.	A large water supply is needed to maintain hydrostatic pressure in deep holes and when highly conductive formations are encountered. Expensive-experienced driller and much peripheral equipment required. Hole diameters are usually large, commonly 18 inches or greater. Cross-contamination from circulating water likely. Geologic samples brought to surface are generally poor, circulating water will "wash" finer materials from sample.
Air Rotary	Very similar to Hydraulic Roatary, the main difference being that air is used as the primary drilling fluid as opposed to mud or water.	Can be used in all geologic formations; most successful in highly fractured environments. Useful at any depth. Fairly quick. Drilling mud or water not required.	Relatively expensive. Cross-contamination from vertical communication possible. Air will be mixed with water in the hole and that which is blown from the hole, potentially creating unwanted reactions with contaminants; may affect "representative" samples. Cuttings and water blown from the hole can pose a hazard to crew and surrounding environment if toxic compounds encountered. Organic foam additives to aid cuttings removal may contaminate samples.
Air-Percussion Rotary or Downhole-Hammer	Air Rotary with a reciprocating hammer connected to the bit to fracture rock.	Very fast penetrations. Useful in all geoloogic formations. Only small amounts of water needed for dust and bit temperature control. Cross contamination potential can be reduced by driving casing.	Relatively expensive. As with most hydraulic rotary methods, the rig is fairly heavy, limiting accessibility. Vertical mixing of water and air creates cross-contamination potential. Hazard posed to surface environment if toxic compounds encountered. Organic foam additives for cuttings' removal may contaminate samples.

Table 1-1. Continued

Johnson, Inc., 1972; and Scalf and others, 1981). The table also gives a concept of the advantages and disadvantages that need to be considered when choosing a drilling technique for different site and monitoring situations (see, also, Lewis, 1982; Luhdorff and Scalmanini, 1982; Minning, 1982; and Voytek, 1983).

Hollow-stem augering is one of the most desirable drilling methods for constructing monitoring wells. No drilling fluids are used and disturbance of the geologic materials penetrated is minimal. Depths are usually limited to no more than 150 feet. Typically, auger rigs are not used when consolidated rock must be penetrated.

In formations where the borehole will not stand open, the monitoring well can be constructed inside the hollow-stem auger prior to its removal from the hole. Generally, this limits the diameter of the well that can be built to 4 inches. The hollow-stem has an added advantage in offering the ability to collect continuous in situ geologic samples without removal of the auger sections.

The solid-stem auger is most useful in fine-grained, unconsolidated materials that will not collapse when unsupported. The method is similar to the hollow-stem except that the auger flights must be removed from the hole to allow the insertion of the well casing and screen. Cores cannot be collected when using a solid-stem. Therefore, geologic sampling must rely on cuttings that come to the surface, which is an unreliable method because the depth from which the cuttings are derived is not precisely known.

Cable-tool drilling is one of the oldest methods used in the water well industry. Even though the rate of penetration is rather slow, this method offers many advantages for monitoring well construction. With the cable-tool, excellent formation samples can be collected and the presence of thin permeable zones can be detected. As drilling progresses, a casing is normally driven and this provides an ideal temporary casing within which to construct the monitoring well.

In air-rotary drilling, air is forced down the drill stem and back up the borehole to remove the cuttings. This technique has been found to be particularly well suited to drilling in fractured rock. If the monitoring is intended for organic compounds, the air must be filtered to ensure that oil from the air compressor is not introduced into the formation to be monitored. Air-rotary should not be used in highly contaminated environments because the water and cuttings blown out of the hole are difficult to control and can pose a hazard to the drill crew and observers. Where volatile compounds are of interest, air-rotary can volatilize them and cause water samples to be unrepresentative of in situ conditions. The use of foam additives to aid cuttings' removal presents the opportunity for organic contamination of the monitoring well.

Air-rotary with percussion hammer increases the effectiveness of air-rotary for materials likely to cave and highly creviced formations. Addition of the percussion hammer gives air-rotary the ability to drive casing, which reduces the loss of air circulation in fractured rock and aids in maintaining an open hole in soft formations. The capability to construct monitoring wells inside the driven casing, prior to its being pulled, adds to the appeal of air-percussion. However, the problems with contamination and crew safety must be considered.

Reverse circulation rotary drilling has limited application for monitoring well construction. Reverse circulation rotary requires that large quantities of water be circulated down the borehole and up the drill stem to remove cuttings. If permeable formations are encountered, significant quantities of water can move into the formation to be monitored, thus altering the quality of the water to be sampled.

Hydraulic or "mud" rotary is probably the most popular method used in the water well industry. Hydraulic rotary, however, presents some disadvantages for monitoring well construction. In hydraulic rotary technique, a drilling mud (usually bentonite) is circulated down the drill stem and up the borehole to remove cuttings. The mud creates a wall on the side of the borehole that must be removed from the screened area by development procedures. With small diameter wells, the drilling mud is not always completely removed. The ion-exchange potential of most drilling muds is high and may effectively reduce the concentration of trace metals in water entering the well. In addition, the use of biodegradable, organic drilling muds, rather than bentonite, can introduce organic components to water sampled from the well.

Most ground-water monitoring wells will be completed in glacial or unconsolidated materials, and generally will be less than 75 feet in depth. In these applications, hollow-stem augering usually will be the method of choice. Solid-stem auger, cable-tool, and air-percussion also offer advantages depending on the geology and contaminant of interest.

Geologic Samples

Permit applications for disposal of waste materials often require that geologic samples be collected at the disposal site. Investigations of ground-water movement and contaminant transport also should include the collection of geologic samples for physical inspection and testing.

Stratigraphic samples are best collected during monitoring well drilling.

Samples can be collected continuously, at each change in stratigraphic unit, or, in homogeneous materials, at regular intervals. These samples may later be classified, tested, and analyzed for physical properties, such as particle-size distribution, textural classification, and hydraulic conductivity, and for chemical analyses, such as ion-exchange capacity, chemical composition, and specific parameter leachability.

Probably the most common method of material sampling is a "split-spoon" sampler. This device consists of a hollow cylinder, 2 inches in diameter, that is 12 or 18 inches long, and split in half lengthwise. The halves are held together with threaded couplings at each end; the top end attaches to the drill rod, and the bottom end is a drive shoe (Figure 1-10a). The sampler is lowered to the bottom of the hole and driven ahead of the hole with a weighted "hammer" striking an anvil at the upper end of the drill rod. The sample is forced up the inside of the hollow tube and is held in place with a basket trap or flap valve. The trap or valve allows the sample to enter the tube but not exit, although retention of noncohesive, sandy material in the tube is often difficult. After the sampler is withdrawn from the hole, the sample is removed by unscrewing the couplings and separating the collection tube.

Another common sampler is the thin wall tube or "Shelby" tube. These tubes are usually 2 to 5-1/2 inches in diameter and about 24 inches long. The cutting edge of the tube is sharpened and the upper end is attached to a coupling head by means of cap screws or a retaining pin (Figure 1-10b). A Shelby tube has a minimum ratio of wall area to sample area and creates the least disturbance to the sample of any drive-type sampler in current use (for hydraulic conductivity tests, minimal disturbance is critical). After retraction, the tube is disconnected from the head and the sample is forced from the tube with a jack or press. If sample preservation is a major concern, the tube can be sealed and shipped to the laboratory.

Apart from permit requirements, material samples are very helpful for deciding at what depth to complete a monitoring well. Unexpected changes encountered during drilling can alter preconceived ideas concerning the local ground-water flow regime. In many instances, the driller will be able to detect a variation in the formation by a change in penetration rate, sound, or "feel" of the drilling rig. However, due to the lag time for cuttings to come to the surface and the amount of mixing the cuttings may undergo as they come up the borehole, the only way to exactly determine the character of the subsurface is to stop drilling and collect a sample.

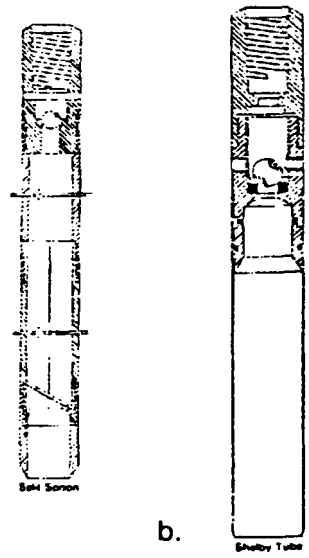


Figure 1-10. Cross-Sectional Views of (a) Split Spoon and (b) Shelby Tube Samplers (from Mobile Drilling Co., 1972)

Case History

Several different types of monitoring wells were constructed during the investigation of a volatile organic contaminant plume in northern Illinois (Wehrmann, 1984). A brief summary of the types of wells employed and the reasons for their use helps illustrate how an actual ground-water quality monitoring problem was approached.

During the final weeks of a 1-year study of nitrate in ground water in north-central Illinois, the presence of several organic compounds was detected in the drinking water of all five homes sampled within a large rural residential subdivision. The principal compound, trichloroethylene (TCE), was present in concentrations ranging from 50 to 1,000 micrograms per liter (µg/L). All the homes in the subdivision used private wells, 65 to 75 feet deep, that tapped a surficial sand and gravel deposit. Figure 1-11 shows a geologic cross section of the study area.

Two immediate concerns needed to be addressed. First, how many other water wells were affected and, second, what was the contaminant source? Early thoughts connected the TCE to the contamination potential of the large number of septic systems in the subdivision. Earlier work (Wehrmann, 1983) had established a south-southwest direction of ground-

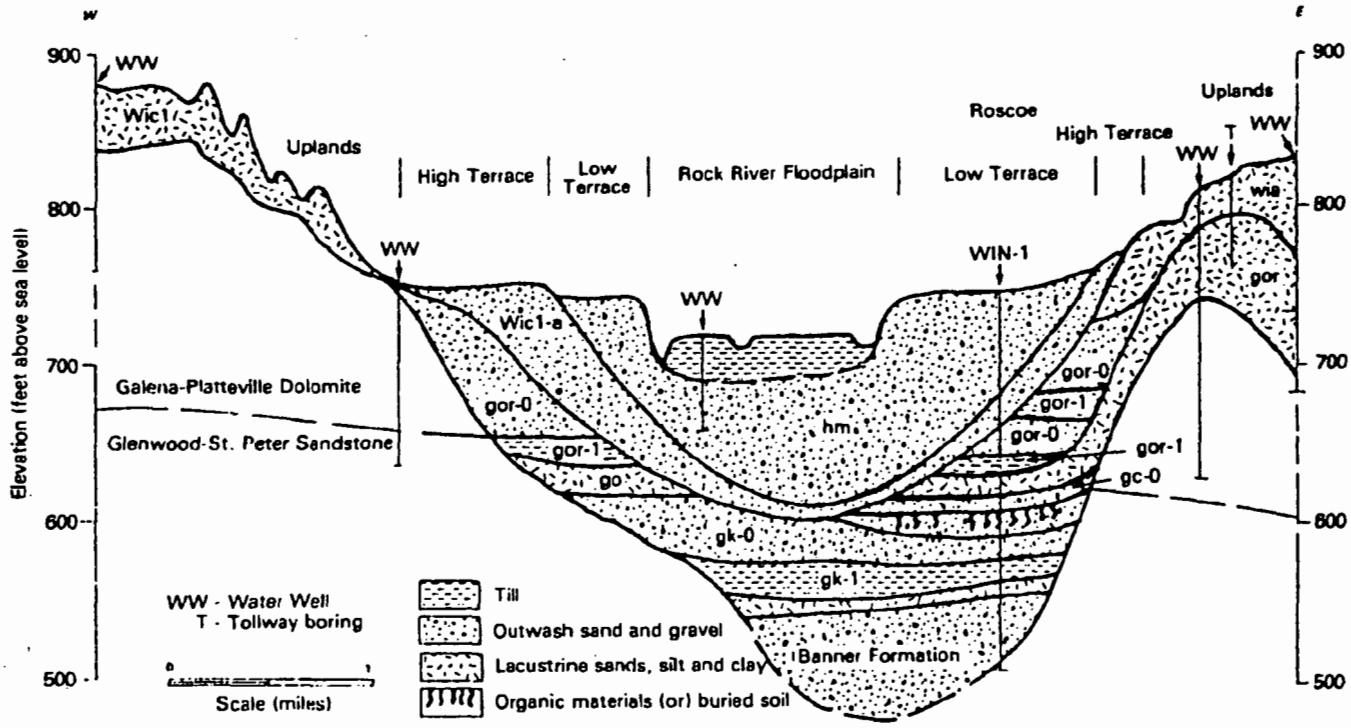


Figure 1-11. East-West Cross Section Across Rock River Valley at Roscoe (from Berg and others, 1981)

water flow beneath the subdivision. Because the area upgradient of the subdivision was primarily farmland, several monitoring wells were placed in that area to help confirm or deny the possibility that the septic systems were the source of the TCE.

Five "temporary" monitoring wells were constructed upgradient of the affected subdivision. Original plans called for driving a 2-inch diameter sandpoint to depths from 40 to 70 feet. Water samples would be collected at 10-foot intervals as the point was driven. Once 70 feet was reached, the sandpoint would be pulled, the hole properly abandoned, and the point driven at a new sampling location. The first hole was to be placed north (upgradient) of a domestic well found to be highly contaminated, and additional holes were to be placed successively in an upgradient direction across the field. In this manner, ground-water samples could be collected quickly at many depths and locations, the well materials recovered, and the field left relatively undisturbed.

Once drilling commenced, however, it became clear that driving sandpoints into the coarse sand and gravel was not possible. Consequently, an air-percussion rig was brought on site and a new approach was established. A 4-inch diameter screen, 2 feet long and with a drive shoe, was welded to a 4-inch diameter steel casing. This assembly was driven by air hammer to the desired sampling depth. The bottom of the drive shoe, being open, forced the penetrated geologic materials into the casing and screen. These materials were then removed by air rotary once the desired depth was reached. All well construction materials were steam-cleaned prior to use to avoid cross-contamination. Figure 1-12 shows the locations of the temporary well sites and the analytical results for TCE from samples bailed at depths of 40 and 50 feet.

The temporary sampling program revealed that the contaminant source was outside of the subdivision. Due to the construction and sampling methods employed for these wells, emphasis was not placed on the quantitative aspects of the sampling results; however, important qualitative conclusions were made. The temporary wells confirmed the presence of VOCs directly upgradient of the subdivision and provided information for the

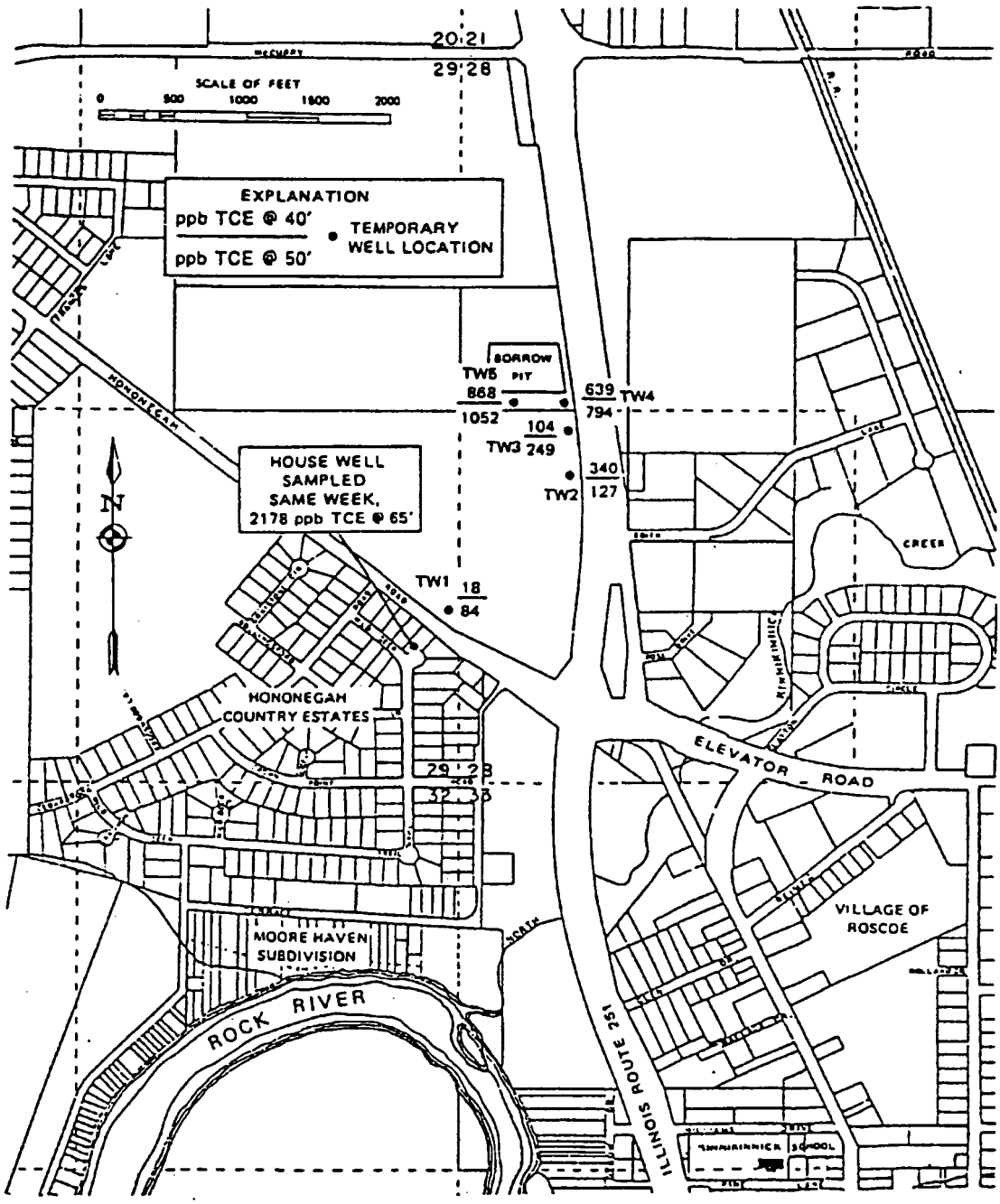


Figure 1-12. Locations and TCE Concentrations for Temporary Monitoring Wells at Roscoe, Illinois (from Wehrmann, 1984)

subsequent location and depth of nine permanent monitoring wells.

Due to the problems associated with organic compound leachability and adsorption from PVC casing and screen, flush-threaded stainless steel casing and screen, 2 inches in diameter, were used for the permanent monitoring wells. The screens were 2 feet long with 0.01-inch wire-wound slot openings. All materials associated with the monitoring well construction, including the drill rig, were steam-cleaned prior to the commencement of drilling to avoid organic contamination from cutting oils and grease. Prior to use, the casing and screen materials were kept off site in a covered, protected area. To ensure that the sandy materials would not collapse after drilling, casing lengths and the screen were joined aboveground and placed inside of the augers before the auger flights were pulled out of the

hole. The sand and gravel below the water table collapsed around the screen and casing as the augers were removed. To prevent vertical movement of water down along the casing, about 3 feet of a wet bentonite/cement mixture was placed in the annulus just above the water table. Cuttings (principally clean, fine to medium sand) were backfilled above the bentonite/cement seal to within 4 feet of land surface. Another bentonite/cement mixture was placed to form a seal at ground surface, further preventing movement of water down along the well casing. A 4-inch diameter steel protective cover with locking cap was placed over the casing and into the surface seal to protect against vandalism.

The nine wells were drilled at four locations with paired wells at three sites and a nest of three wells at one site (Figure 1-13). The locations were based on the analytical results of the samples taken from the temporary wells and basic knowledge of the ground-water flow direction.

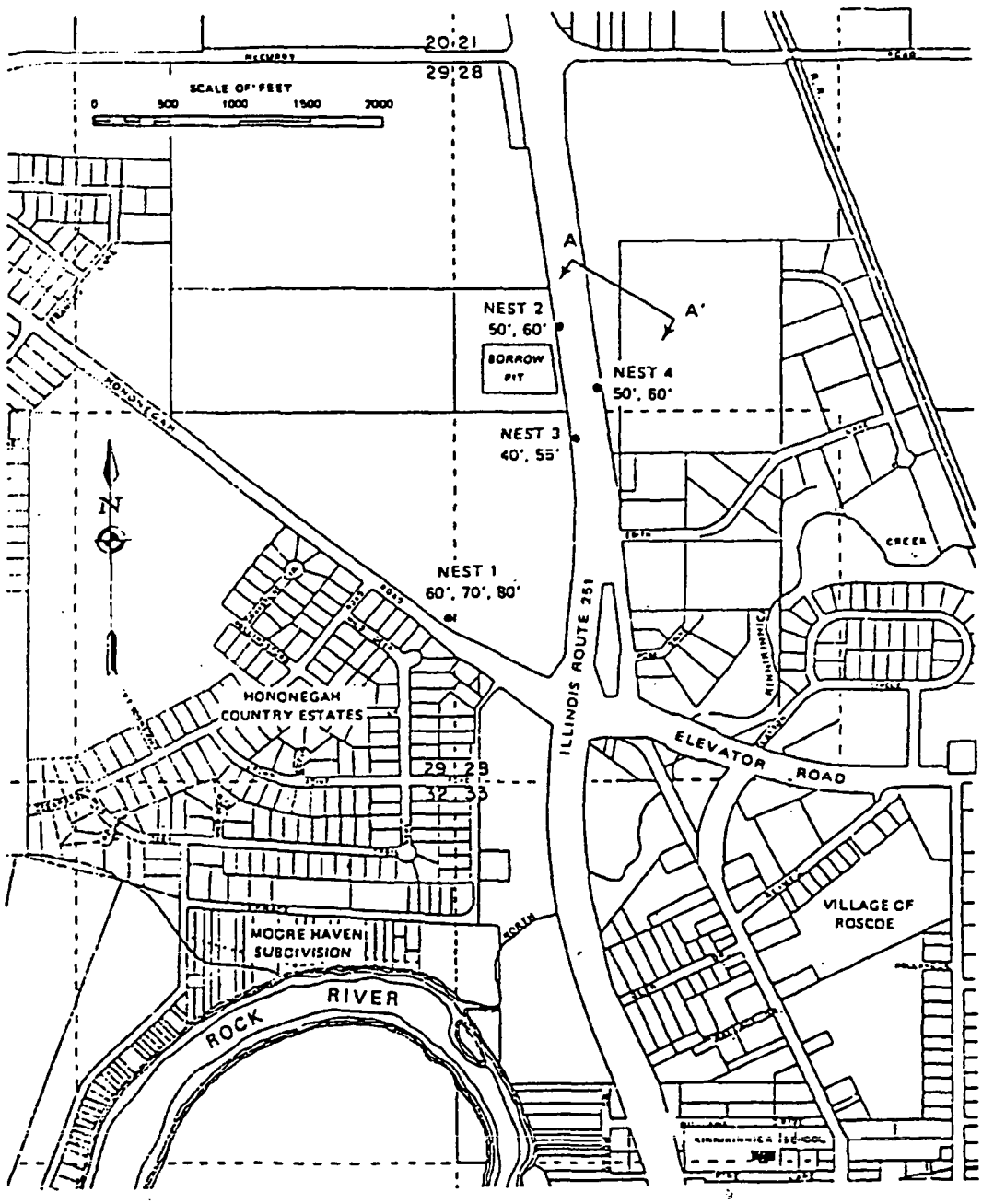


Figure 1-13. Location of Monitoring Well Nests and Cross Section A-A' at Roscoe, Illinois (from Wehrmann, 1984)

Locations were numbered as nests 1 through 4 in order of their construction. Nest 1, located immediately north of the affected subdivision, consists of three wells completed at depths of approximately 60, 70, and 80 feet below ground surface. Nest 2 consists of two wells, one 50 feet deep, and the other 60 feet deep. Nest 3 consists of two wells 40 and 55 feet deep, while nest 4 consists of two wells 50 and 60 feet deep.

Prior to completing the monitoring wells, it was felt an additional well, 100 feet deep and adjacent to nest 1, was needed to further define the vertical extent of the contaminant plume. Because the hollow-stem auger rig was no longer available, arrangements were made to use a cable-tool rig. The well was constructed over a period of 2 days, which was somewhat slower than any of the other methods previously used (but typical of cable-tool speeds). A 6-inch casing was driven several feet, a bit was used to break up the materials inside the casing, and then the materials were removed from the casing with a dart-valve bailer. This procedure was repeated until 100 feet was reached; then the well casing and screen were screwed together and lowered down the hole. The 6-inch casing was then pulled back, which allowed the hole to collapse about the well, which was constructed of stainless steel exactly like the other nine monitoring wells. All drilling equipment and well construction materials were steam-cleaned prior to use.

Appraisal of the sampling results of the monitoring wells and the domestic wells in the area produced the pictorial representations shown in Figures 1-14 and 1-15. Figure 1-14 conceptually illustrates a downgradient cross section of the TCE plume in the vicinity of monitoring nests 2, 3, and 4. Figure 1-15 shows the likely extent of the VOC contaminant plume. This map includes a limited amount of data from privately owned monitoring wells located on industrial property just upgradient of monitoring nests 2 and 4. The dashed lines indicate the probable extent of the contaminant plume based on the dimensions of the plume where it passes beneath the developed area along the Rock River.

This monitoring situation clearly indicates the role different drilling and construction techniques can play in a ground-water sampling strategy. In each instance, much consideration was given to the effect the methods used for construction and sampling would have on the resultant chemical data. Where quantitative results for a fairly "quick" preliminary investigation were not necessary and driving sandpoints was too difficult, air-percussion rotary methods were deemed acceptable. For the placement of the permanent monitoring wells, wells that may become crucial for contaminant source identification and possibly for litigation, the hollow-stem auger was the technique of choice. Finally, when the hollow-stem auger was not available, a cable-tool rig was chosen. Since only one hole was to be drilled, the relative slowness of the method became less important. Also, the depth of completion (100 feet) in the cavey sand and gravel made the cable-tool preferable over the hollow-stem. In addition, each method chosen was capable of maintaining an open hole without the use of drilling mud, which could have affected the results of the chemical analyses of the ground water.

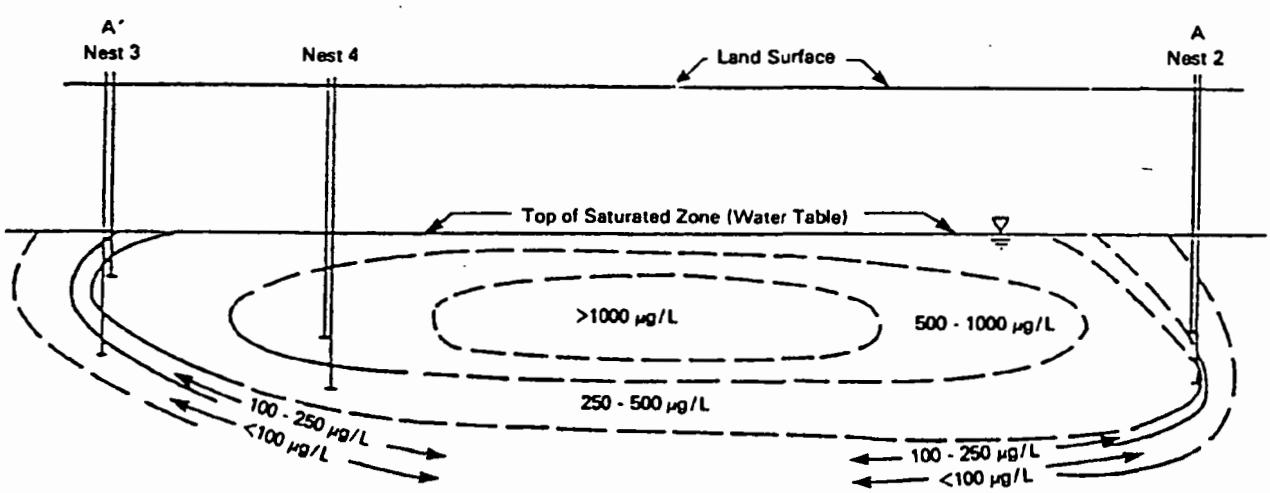


Figure 1-14. Cross Section A-A' through Monitoring nests 2, 3, and 4, Looking in the Direction of Ground-Water Flow (from Wehrmann, 1984)

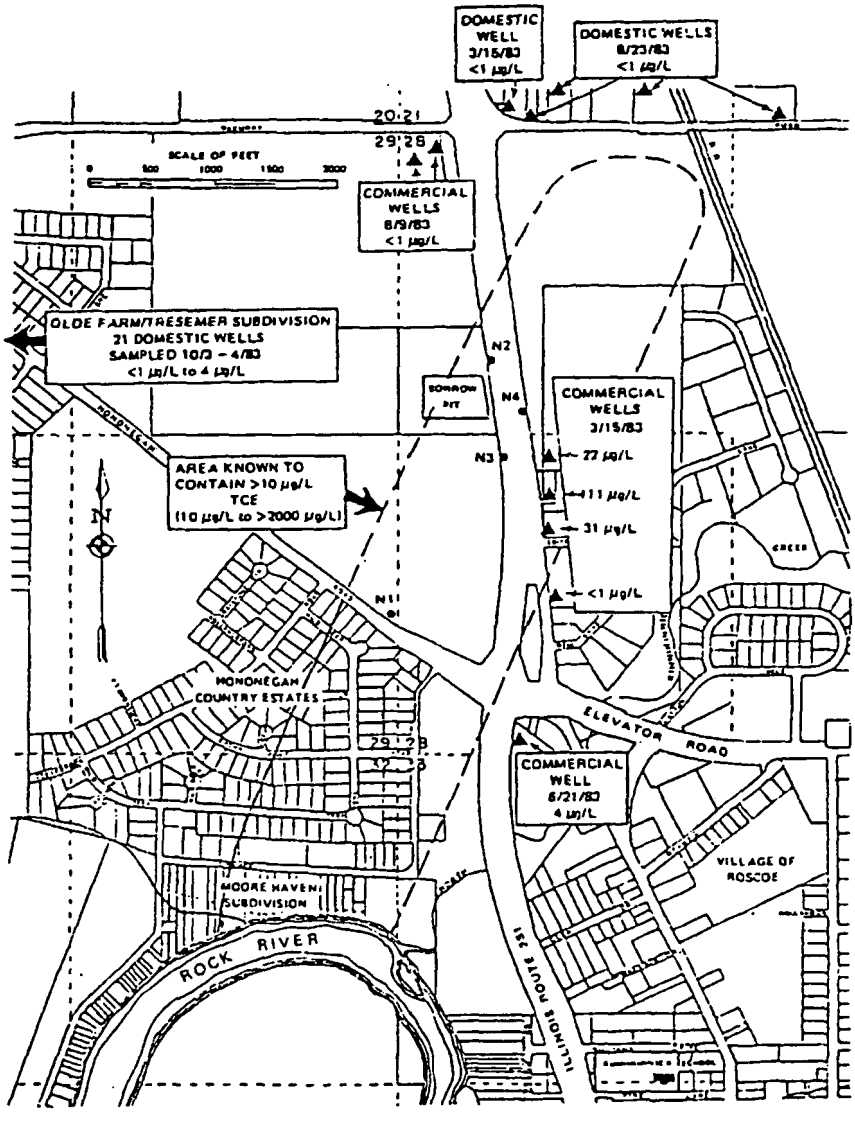


Figure 1-15. General Area of Known TCE Contamination (from Wehrmann, 1984)

Summary

Critical considerations for the design of ground-water quality monitoring networks include alternatives for well design and drilling techniques. With a knowledge of the principal chemical constituents of interest and the local hydrogeology, and appreciation of subsurface geochemistry, appropriate materials for well design and drilling techniques can be selected. Whenever possible, physical disturbance and the amount of foreign material introduced into the subsurface should be minimized.

The choices of drilling methods and well construction materials are very important in every type of ground-water monitoring program. Details of network construction can introduce significant bias into monitoring data, which frequently may be corrected only by repeating the process of well siting, installation, completion, and development. This can be quite costly in time, effort, money, and loss of information. Undue expense is avoidable if planning decisions are made cautiously with an eye to the future.

The expanding scientific literature on effective ground-water monitoring techniques should be read and evaluated on a continuing basis. This information will help supplement guidelines, such as this, for applications to specific monitoring efforts.

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Chapter 2

GROUND-WATER SAMPLING

Introduction

Background

Ground-water sampling is conducted to provide information on the condition of subsurface water resources. Whether the goal of the monitoring effort is detection or assessment of contamination, the information gathered during sampling efforts must be of known quality and be well documented. The most efficient way to accomplish these goals is by developing a sampling protocol, which is tailored to the information needs of the program and the hydrogeology of the site or region under investigation. This sampling protocol incorporates detailed descriptions of sampling procedures and other techniques that, of themselves, are not sufficient to document data quality or reliability. Sampling protocols are central parts of networks or investigatory strategies.

The need for reliable ground-water sampling procedures has been recognized for years by a variety of professional, regulatory, public, and private groups. The technical basis for the use of selected sampling procedures for environmental chemistry studies has been developed for surface-water applications over the last four decades. Ground-water quality monitoring programs, however, have unique needs and goals that are fundamentally different from previous investigative activities. The reliable detection and assessment of subsurface contamination require minimal disturbance of geochemical and hydrogeologic conditions during sampling.

At this time, proven well construction, sampling, and analytical protocols for ground-water sampling have been developed for many of the more problematic chemical constituents of interest. However, the acceptance of these procedures and protocols must await more careful documentation and firm regulatory guidelines for monitoring program execution. The time and expense of characterizing actual subsurface conditions place severe restraints on the methods that can be employed. Since the technical basis for documented, reliable drilling, sample collection, and handling procedures is in the early stages of development, conscientious efforts to document method performance under real conditions should be a part of any ground-water investigation (Barcelona and others, 1985; Scalf and others, 1981).

Information Sources

Much of the literature on routine ground-water monitoring methodology has been published in the last 10 years. The bulk of this work has emphasized ambient resource or contaminant resource monitoring (detection and assessment), rather than case preparation or enforcement efforts. General references that are useful to the design and execution of sampling efforts are the U.S. Geological Survey (1977), Wood (1976), the U.S. Environmental Protection Agency (Brass and others, 1977; Dunlap and others, 1977; Fenn and others, 1977; Sisk, 1981) and others (National Council of the Paper Industry, 1982; Tinlin, 1976). In large part, these works treat sampling in the context of overall monitoring programs, providing descriptions of available sampling mechanisms, sample collection techniques, and sample handling procedures. The impact of specific methodologies on the usefulness or reliability of the resulting data has received little discussion (Gibb and others, 1981).

High-quality chemical data collection is essential in ground-water monitoring programs. The technical difficulties involved in "representative" samplings have been recognized only recently (Gibb and others, 1981; Grisak and others, 1978). The long-term collection of high-quality ground-water chemistry data is more involved than merely selecting a sampling mechanism and agreeing on sample handling procedures. Efforts to detect and assess contamination can be unrewarding without accurate (i.e., unbiased) and precise (i.e., comparable and complete) concentration data on

ground-water chemical constituents. Also, the expense of data collection and management argue for documentation of data quality.

Gillham and others (1983) published a very useful reference on the principal sources of bias and imprecision in ground-water monitoring results. Their treatment is extensive and stresses the minimization of random error, which can enter into well construction, sample collection, and sample handling operations. They further stress the importance of collecting precise data over time to maximize the effectiveness of trend analysis, particularly for regulatory purposes. Accuracy also is very important, since the ultimate reliability of statistical comparisons of results from different wells (e.g., upgradient versus downgradient samples) may depend on differences between mean values for selected constituents from relatively small replicate sample sets. Therefore, systematic error must be controlled by selecting proven methods for establishing sampling points and sample collection to ensure known levels of accuracy.

The Subsurface Environment

The subsurface environment may be categorized broadly into two zones, the unsaturated or vadose zone and the saturated zone. The use of the term "vadose" is more accurate because isolated saturated areas may exist in the unsaturated zone above the water table of unconfined aquifers.

Investigators have discovered recently that the subsurface is neither devoid of oxygen (Winograd and Robertson, 1982) nor sterile (Wilson and McNabb, 1983; Wilson and others, 1983). These facts may significantly influence the mobility and persistence of chemical species, as well as the transformations of the original components of contaminant mixtures (Schwarzenbach and others, 1985) that have been released to the subsurface.

The subsurface environment also is quite different from surface water systems in that vertical gradients in pressure and dissolved gas content have been observed within the usual depth ranges of monitoring interest (i.e., 1 to 150 m [3 to 500 ft]). In some cases, these gradients can be linked to well-defined hydrologic or geochemical processes. However, reports of apparently anomalous geochemical processes have increased in recent years, particularly at contaminated sites (Barcelona and Garske, 1983; Heaton and Vogel, 1981; Schwarzenbach and others, 1985; Winograd and Robertson, 1982; Wood and Petraitis, 1984).

The subsurface environment is not as readily accessible as surface water systems, and some disturbance is necessary to collect samples of earth materials or ground water. Therefore, "representative" (i.e., artifact or error free) sampling is really a function of the degree of detail needed to characterize subsurface hydrologic and geochemical conditions and the care taken to minimize disturbance of these conditions in the process (Claasen, 1982). Each well or boring represents a potential conduit for short-circuited contaminant migration or ground-water flow, which must be considered a potential liability to investigative activities.

The subsurface environment is dynamic over extended time frames and the processes of recharge and ground-water flow are very important to a thorough understanding of the system. Detailed descriptions of contaminant distribution, transport, and transformation necessarily rely on the understanding of basic flow and fluid transport processes. Short-term investigations may only provide a snapshot of contaminant levels or distributions. Since water-quality monitoring data are normally collected on discrete dates, it is very important that reliable collection methods are used to assure high data quality over the course of the investigation. The reliability of the methods should be investigated thoroughly during the preliminary phase of monitoring network implementation.

Although the scope of this discussion is on sampling ground water for chemical analysis, the same data quality requirements apply to water-level measurements and to hydraulic conductivity testing. These hydrologic determinations form the basis for interpreting chemical constituent data and may well limit the validity of fluid or solute transport model applications. Hydrologic measurements must be included in the development of the quality assurance/quality control (QA/QC) program for ground-water quality monitoring networks.

The Sampling Problem and Parameter Selection

Cost-effective water-quality sampling is difficult in ground-water systems because proven field procedures have not been extensively documented. Regulations that call for "representative sampling" alone are not sufficient to ensure high-quality data collection. The most appropriate monitoring and sampling procedures for a ground-water quality network will depend on the specific purpose of the program. Resource evaluation, contaminant detection, remedial action assessments, and litigation studies are purposes for which effective networks can be designed once the information needs have been identified. Due to the time, personnel needs, and cost of most water-quality monitoring programs, the optimal network design should be phased so as to make the most of the available information as it is collected. This approach allows for the gradual refinement of program goals as the network is implemented.

Two fundamental considerations are common to most ground-water quality monitoring programs: establishing individual sampling points (i.e., in space and time) and determining the elements of the water sampling protocol that will be sufficient to meet the information needs of the overall program. The placement and number of sampling points can be phased to gradually increase the scale of the monitoring program. Similarly, the chemical constituents of initial interest should provide background ground-water quality data from which a list of likely contaminants may be prepared as the program progresses. Table 2-1 shows candidate chemical and hydrologic parameters for both detective and assessment monitoring activities. Special care should be taken to account for possible subsurface transformation of the principal pollutant species. Ground-water transport of contaminants can produce chemical distributions that vary substantially over time and space. In particular, transformation of organic compounds can change substantially the identity of the original contaminant mixture (Mackay and others, 1985; Schwarzenbach and others, 1985).

Detective Monitoring
Chemical Parameters*
pH, Ω^{-1}, TOC, TOX, Alkalinity, TDS, Eh, Cl^-, NO_3^-, $SO_4^=$, $PO_4^{\equiv}$, SiO_2, Na^+, K^+, Ca^{++}, Mg^{++}, NH_4^+, Fe, Mn
Hydrologic Parameters
Water Level, Hydraulic Conductivity
Assessment Monitoring
Chemical Parameters*
pH, Ω^{-1}, TOC, TOX, Alkalinity, TDS, Eh, Cl^-, NO_3^-, $SO_4^=$, $PO_4^{\equiv}$, SiO_2, B, Na^+, K^+, Ca^{++}, Mg^{++}, NH_4^+, Fe, Mn, Zn, Cd, Cu, Pb, Cr, Ni, Ag, Hg, As, Sb, Se, Be
Hydrologic Parameters
Water Level, Hydraulic Conductivity

*Ω^{-1} = specific conductance, a measure of the charged species in solution.

Table 2-1. Suggested Measurements for Ground-Water Monitoring Programs

Contaminant detection is generally the most important aspect of a water-quality program, and must be assured in network design. False negative contaminant readings due to the loss of chemical constituents or the introduction of interfering substances that mask the presence of the contaminants in water samples can be very serious. Such errors may delay needed remedial action and expose either the public or the environment to an unreasonably high risk. False positive observations of contaminants may call for costly remedial actions or more intensive study, which are not warranted by the actual situation. Thus, reliable sample collection and data interpretation procedures are central to an optimized network design.

In this respect, monitoring in the vadose zone is attractive because it should provide an element of "early" detection capability. The methodologies available for this type of monitoring have been under development for some time. There are distinct limitations, however, to many of the available monitoring devices (Everett and McMillion, 1985; Everett and others, 1982; Wilson, 1981; Wilson, 1983), and it is frequently difficult to relate observed vadose zone concentrations quantitatively to actual contaminant distributions in ground water (Everett and others, 1984; Lindau and Spalding, 1984). Soil gas sampling techniques and underground storage tank monitors have been commercially developed that can be extremely useful for source scouting. Given the complexity of vadose zone monitoring procedures and the need for additional investigation (Robbins and Gemmell, 1985), implementing these techniques in routine ground-water monitoring networks may be difficult.

This chapter addresses water-quality sampling in the saturated zone, reflecting the advanced state of monitoring technology appropriate for this compartment of the subsurface. There are a number of useful reference materials for the development of effective ground-water sampling protocols, which include information on the types of drilling methods, well construction materials, sampling mechanisms, and sample handling methods currently available (Barcelona and others, 1985; Barcelona and others, 1983; Gillham and others, 1983; Scalf and others, 1981; Todd and others, 1976). To collect sensitive, high-quality contaminant concentration data, investigators must identify the type and magnitude of errors that may arise in ground-water sampling. Figure 2-1 presents a generalized diagram of the steps involved in sampling and the principal sources of error.

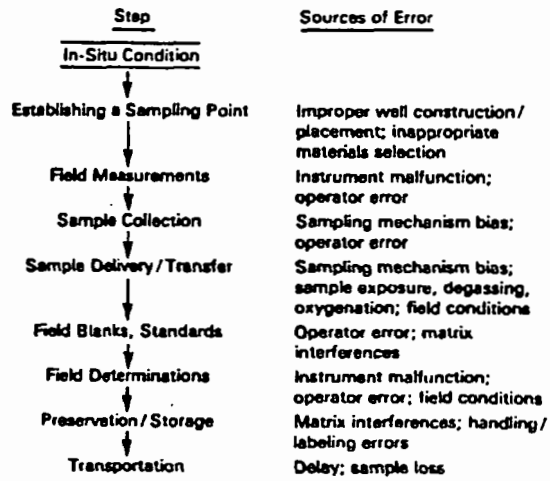

Figure 2-1. Steps and Sources of Error in Ground-Water Sampling

Strict error control at each step is necessary for the collection of high-quality data representative of in situ conditions.

There are two major obstacles to controlling ground-water sampling errors. First, field blanks, standards, and split samples used in data quality assurance programs cannot account for changes that may occur in the integrity of samples prior to sample delivery to the land surface. Second, most of the sources of error that may affect sample integrity prior to delivery are not well documented in the literature for many of the contaminants of current interest. Among these sources of error are the contamination of the subsurface by drilling fluids, grouts, or sealing materials; the sorptive or leaching effects on water samples due to well casing; pump or sampling tubing materials' exposures; and the effects on the solution chemistry due to oxygenation, depressurization, or gas exchange caused by the sampling mechanism. These sources of error have been investigated to some extent for volatile organic contaminants under laboratory conditions. However, to achieve confidence in field monitoring and sampling instrumentation for routine applications, common sense and a "research" approach to regulatory monitoring may be needed. Two of the most critical elements of a monitoring program are establishing both reliable sampling points and simple, efficient sampling protocols that will yield data of known quality.

Establishing a Sampling Point

Taking adequate care in selecting drilling methods, well construction materials, and well development techniques should allow the approximation of representative ground-water sampling from a monitoring well. The representative nature of the water samples can be maintained consistently with a trained sampling staff and good field-laboratory communication. Also, important hydrologic measurements, such as water level and hydraulic conductivity, can be made from the same sampling point. A representative water sample may then be defined as a minimally disturbed sample taken after proper well purging, which will allow the determination of the chemical constituents of interest at predetermined levels of accuracy and precision. Sophisticated monitoring technology and sampling instrumentation are poor substitutes for an experienced sampling team that can follow a proven sampling protocol.

This section details some of the considerations in establishing a reliable sampling point. There are a number of alternative approaches for selecting a sampling point in monitoring network design, including deploying arrays of either nested monitoring wells or multilevel devices (Barvenik and Cadwgan, 1983; Pickens and others, 1978) at various sites within the area of interest. Different approaches have their individual merits, based on the ease of verifying sampling point isolation, durability, cost, ease of installation, and site-specific factors.

The most effective option for specific programs should be chosen with representative sampling criteria in mind. The sampling points must be durable, inert towards the chemical constituents of interest, allow for purging of stagnant water, provide sufficient water for analytical work with minimum disturbance, and permit the evaluation of the hydrologic characteristics of the formation of interest. Monitoring wells can be constructed to meet these criteria because a variety of drilling methods, materials, sampling mechanisms, and pumping regimes for sampling and hydrologic measurements can be selected to meet the current needs of most monitoring programs.

The placement and number of wells will depend on the complexity of the hydrologic setting and the degree of spatial and temporal detail needed to meet the goals of the program. Both the directions and approximate rates of ground-water movement must be known in order to satisfactorily interpret the chemical data. With this knowledge, it also may be possible to estimate the nature and location of pollutant sources (Gorelick and others, 1983). Subsurface geophysical techniques can be very helpful in determining the optimum placement of monitoring wells under appropriate conditions and when sufficient hydrogeologic information is available (Evans and Schweitzer, 1984). Well placement should be viewed as an evolutionary activity that may expand or contract as the needs of the program dictate.

Well Design and Construction

Effective monitoring well design and construction require considerable care and at least some understanding of the hydrogeology and subsurface geochemistry of the site. Preliminary borings, well drilling experience, and the details of the operational history of a site can be very helpful. Monitoring well design criteria include depth, screen size, gravel-pack specifications, and yield potential. These considerations differ substantially from those applied to production wells. The simplest, small diameter well completions that will permit development, accommodate the sampling gear, and minimize the need to purge large volumes of potentially contaminated water are preferred for effective routine monitoring activities. Helpful references include Barcelona and others (1983), Scalf and others (1981), and Wehrmann (1983).

Well Drilling

The selection of a particular drilling technique should depend on the geology of the site, the expected depths of the wells, and the suitability of drilling equipment for the contaminants of interest (see Chapter 1). Regardless of the technique used, every effort should be made to minimize subsurface disturbance. For critical applications, the drilling rig and tools should be steam-cleaned to minimize the potential for cross-contamination between formations or successive borings. The use of drilling muds can be a liability for trace chemical constituent investigations because foreign organic matter will be introduced into the penetrated formations. Even "clay" muds without polymeric additives contain some organic matter, which is added to stabilize the clay suspension and may interfere with some analytical determinations. Table 2-2 contains information on the total and soluble organic carbon contents of some common drilling and grouting materials (Wood, 1976). The effects of drilling muds on ground-water solution chemistry have not been investigated in detail.

However, existing reports indicate that the organic carbon introduced during drilling can cause false water-quality observations for long periods of time (Barcelona, 1984; Brobst, 1984). The fact that these interferences are observable for gross indicators of levels of organic carbon compounds (i.e., TOC) and reduced substances (i.e., COD) strongly suggests that drilling aids are a potential source of serious error. Special situations may call for innovative drilling techniques (Yare, 1975).

Well Development, Hydraulic Performance, and Purging Strategy

Once a well is completed, it is necessary to prepare the sampling point for water sampling and begin to evaluate the hydraulic characteristics of the producing zone. These steps provide a basis for maintaining reliable sampling points over the duration of a ground-water monitoring program.

Well Development. The proper development of monitoring wells is essential to the collection of "representative" water samples. During the drilling process, fine particles are forced through the sides of the borehole into the formation, forming a mud cake that reduces the hydraulic conductivity of the materials in the immediate area of the well bore. To allow water from the formation being monitored to freely enter the monitoring well, this mud cake must be broken down opposite the well screen and the fine material removed from the well. This process also enhances the yield potential of the well, which is a critical factor when constructing monitoring wells in low-yielding geologic materials.

More importantly, monitoring wells must be developed to provide water free of suspended solids. When sampling for metal ions and other dissolved inorganic constituents, water samples must be filtered and preserved at the well site at the time of sample collection. Improperly developed monitoring wells will produce samples containing suspended sediments that may both bias the chemical analysis of the collected samples and cause frequent clogging of field filtering mechanisms. The additional time and money spent for well development will expedite sample filtration and result in samples that are more representative of water chemistry in the formation being monitored.

Development procedures used for monitoring wells are similar to those used for production wells. The first step in development involves the movement of water at alternately high and low velocity into and out of the well

	Ash (% by wt)	Organic Content (% by wt)	Soluble Carbon (% by wt)	Soluble Carbon in Total Organic Content (% by wt)
"Bentonite" muds/grouts				
Volclay[a] (~90% montmorillonite)	98.2	1.8	<0.001	94.4
Benseal[b]	88.5	11.5	<0.001	3.7
"Organic" muds/drilling aids				
Ez-Mud[c] (acrylamide-sodium acrylate copolymer dispersed in food-grade oil [normally used in 0.25% dilution])	11.5	21.5	17.9	2.1
Revert[d] (guar bean starch-based mixture)	1.6	98.4	33.8	85.6

[a]All percentages determined on a moisture-free basis.
[b]Trademark of American Colloid Co.
[c]Trademark of NL Baroid/NL Industries Inc.
[d]Trademark of Johnson Division, UOP Inc.
Source: Wood, 1976.

Table 2-2. Composition of Selected Sealing and Drilling Muds

screen and gravel-pack to break down the mud cake on the well bore and loosen fine particles in the borehole. This step is followed by pumping to remove these materials from the well and the immediate area outside the well screen. This procedure should be continued until the water pumped from the well is visually free of suspended materials or sediments.

Hydraulic Performance of Monitoring Wells. The importance of understanding the hydraulics of the geologic materials at a site cannot be overemphasized. Collection of accurate water-level data from properly located and constructed wells provides information on the direction of ground-water flow. The success of a monitoring program also depends on knowledge of the rates of travel of both the ground water and solutes. The response of a monitoring well to pumping also must be known to determine the proper rate and length of time of pumping prior to collecting a water sample.

Hydraulic conductivity measurements provide a basis for judging the hydraulic connection of the monitoring well and adjacent screened formation to the hydrogeologic setting. These measurements also allow an experienced hydrologist to estimate an optimal sampling frequency for the monitoring program (Barcelona and others, 1985).

Traditionally, hydraulic conductivity testing has been achieved by collecting drill samples, which were then taken to the laboratory for testing. Several techniques involving laboratory permeameters are routinely used. Falling head or constant head permeameter tests on recompacted samples in fixed wall or triaxial test cells are among the most common. The relative applicability of these techniques depends on both operator skill and methodology since calibration standards are not available. The major problem with laboratory test procedures is that the determined values are based on recompacted geologic samples rather than undisturbed geologic materials. Only limited work has been done to date on performing laboratory tests on "undisturbed" samples to improve the field applicability of laboratory hydraulic conductivity results. Melby (1989) reported that laboratory-determined values of hydraulic conductivity for cores of unconsolidated, fine-grained material from Oklahoma were three to six orders of magnitude smaller than values determined by aquifer testing. Considerable care must be exercised when evaluating laboratory-derived hydraulic conductivity coefficients.

Hydraulic conductivity is most effectively determined under field conditions by aquifer testing methods, such as pumping or slug testing (see Chapter 4). The water-level drawdown can be measured during pumping. Alternatively, water levels can be measured after the static water level is depressed by application of gas pressure or elevated by the introduction of a slug of water. These procedures are rather straightforward for wells that have been properly developed.

Well Purging Strategies. The number of well volumes to be removed from a monitoring well prior to collecting a water sample must be tailored to the hydraulic properties of the geologic materials being monitored, the well construction parameters, the desired pumping rate, and the sampling methodology to be employed. No single number of well volumes to be pumped fits all situations. The goal in establishing a well purging strategy is to obtain water from the geologic materials being monitored while minimizing the disturbance of the regional flow system and the collected sample. To accomplish this goal, a basic understanding of well hydraulics and the effects of pumping on the quality of water samples is essential. Water that has remained in the well casing more than about 2 hours has had the opportunity to exchange gases with the atmosphere and to interact with the well casing material. Therefore, the chemistry of water stored in the well casing is not representative of that in the aquifer and should not be collected for analysis. Purge volumes and pumping rates should be evaluated on a case-by-case basis.

Gibb (1981) has shown how the measurements of hydraulic conductivity can be used to estimate the well-purging requirement. Figures 2-2a and 2-2b show an example of this procedure. In practice, it may be necessary to test the hydraulic conductivity of several wells within a network. The calculated purging requirement should then be verified by measurements of pH and specific conductance during pumping to signal equilibration of the water being collected.

The selection of purging rates and volumes of water to be pumped prior to sample collection also can be influenced by the anticipated water quality. In hazardous environments where purged water must be contained and disposed of in a permitted facility, it is desirable to minimize this amount. This can be accomplished by pumping the wells at very low pumping rates (100 mL/min) to minimize the drawdown in the well and maximize the percentage of aquifer water delivered to the surface in the shortest period of time. Pumping at low rates, in effect, isolates the column of stagnant water in the well bore and negates the need for its removal. This approach is only valid in cases where the pump intake is placed at the top of, or in, the well screen.

In summary, well purging strategies should be established by (1) determining the hydraulic performance of the well; (2) calculating reasonable purging

requirements, pumping rates, and volumes based on hydraulic conductivity data, well construction data, site hydrologic conditions, and anticipated water quality; (3) measuring the well purging parameters to verify chemical "equilibrated" conditions; and (4) documenting the entire effort (actual pumping rate, volumes pumped, and purging parameter measurements before and after sample collection).

Given:
48-foot deep, 2-inch diameter well
2-foot long screen
3-foot thick aquifer
static water level about 15 feet below land surface
hydraulic conductivity = 10^{-3} cm/sec

Assumptions:
A desired purge rate of 500 mL/min and sampling rate of 100 mL/min will be used.

Calculations:
One well volume = (48 ft - 15 ft) x 613 mL/ft (2-inch diameter well)
= 20.2 liters
Aquifer Transmissivity = hydraulic conductivity x aquifer thickness
= 10^{-3} m/sec x 1 meter
= 10^{-3} m^2/sec or 8.64 m^2/day
From Figure 2-2b:
At 5 minutes: 95% aquifer water and
(5 min x 0.5 L/min)/20.2 L
= 0.12 well volumes
At 10 minutes: 100% aquifer water and
(10 min x 0.5 L/min)/20.2 L
= 0.24 well volumes

It appears that a high percentage of aquifer water can be obtained within a relatively short time of pumping at 500 mL•min^{-1}. This pumping rate is below that used during well development to prevent well damange or further development.

Figure 2-2a. Example of Well Purging Requirement Estimating Procedure (Barcelona and others, 1985)

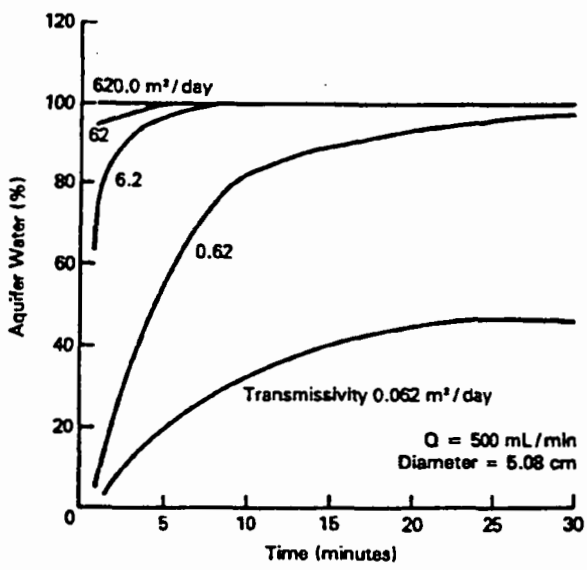

Figure 2-2b. Percentage of Aquifer Water Versus Time for Different Transmissivities

Sampling Materials and Mechanisms. In many monitoring situations, it is not possible to predict the requirements that either materials for well casings, pumps, and tubing, or pumping mechanisms must meet in order to provide error-free samples of ground water. Ideally, these components of the system should be durable and inert relative to the chemical properties of samples or the subsurface so as to neither contaminate nor remove chemical constituents from the water samples. Due to the long duration of regulatory program requirements, well casing materials, in particular, must be sufficiently durable and nonreactive to last several decades. It is generally much easier to substitute more appropriate sampling pumps or pump/tubing materials as knowledge of subsurface conditions improves than to drill additional wells to replace inadequate well casing or screen materials. Also, there is no simple way to account for errors that occur prior to handling a sample at the land surface. Therefore, it is good practice to carefully choose the components of the sampling system that make up the rigid materials in well casing/screens or pumps, and the flexible materials used in sample delivery tubing.

Rigid Materials. An experienced hydrologist can base well construction details mainly on hydrogeologic criteria, even in challenging situations where a separate contaminant phase may be present (Villaume, 1985). However, the best material for a specific monitoring application must be selected by considering subsurface geochemistry and the likely contaminants of interest. Therefore, strength, durability, and inertness should be balanced with cost considerations in the choice of rigid materials for well casing, screens, pumps, etc. (see Chapter 1).

Common well casing materials include TFE (Teflon^R), PVC (polyvinyl chloride), stainless steel, and other ferrous materials. The strength, durability, and potential for sorptive or leaching interferences with chemical constituents have been reviewed in detail for these materials (Barcelona and others, 1985; Barcelona and others, 1983). Unfortunately, there is very little documentation of the severity or magnitude of well casing interferences from actual field investigations. This is the point at which optimized monitoring network design takes on an element of "research," as the components of the monitoring installation will need to be systematically evaluated.

Polymeric materials have the potential to absorb dissolved chemical constituents and leach either previously sorbed substances or components of the polymer formulations. Similarly, ferrous materials may adsorb dissolved chemical constituents and leach metal

ions or corrosion products, which may introduce errors into the results of chemical analysis. This potential in both cases is real, yet not completely understood. The recommendations in the references noted above can be summarized as follows:

Teflon[R] is the well casing material least likely to cause significant error in ground-water monitoring programs focused on either organic or inorganic chemical constituents. It has sufficient strength for most applications at shallow depth (i.e., < 100 m) and is among the most inert materials ever made. For deeper installations, it can be linked to another material above the highest seasonal water level.

Stainless steel (either 316 or 304 type) well casing, under noncorrosive conditions, is the second least likely material to cause significant error for organic chemical constituent monitoring investigations. Fe, Mn, or Cr may be released, under corrosive conditions. Organic constituent sorption effects also may provide significant sources of error after corrosion processes have altered the virgin surface.

Rigid PVC well casing material that has National Sanitation Foundation approval should be used in monitoring well applications when noncemented or threaded joints are used, and organic chemical constituents are not expected to be of either present or future interest. Significant losses of strength, durability and inertness (i.e., sorption or leaching) may be expected under conditions where organic contaminants are present in high concentration. PVC should, however, perform adequately in inorganic chemical constituent studies when concentrations of organic constituents are not high and tin or antimony species are not being targeted.

Monitoring wells made of appropriate materials and screened over discrete sections of the saturated thickness of geologic formations can yield a wealth of chemical and hydrologic information. Whether or not this level of performance is achieved frequently may depend on the care taken in evaluating the hydraulic performance of the sampling point.

Flexible Materials. Pump components and sample delivery tubing may contact a water sample more intimately than other components of a sampling system, including storage vessels and well casing. Similar considerations of inertness and noncontaminating properties apply to tubing, bladder, gasket and seal materials. Experimental evidence (Barcelona and others, 1985) has supported earlier recommendations drawn from manufacturers' specifications (Barcelona and others, 1983). A summary is provided in Table 2-3. Again, the care taken in materials' selection for the specific needs of the sampling program can pay real dividends and provides greater assurance of error-free sampling.

Sample Mechanisms. It is important to remember that sampling mechanisms themselves are not protocols. The sampling protocol for a particular monitoring network

Materials	Recommendations
Polytetrafluoroethylene (Teflon*)	Recommended for most monitoring work, particularly for detailed organic analytical schemes. The material least likely to introduce significant sampling bias or imprecision. The easiest material to clean in order to prevent cross-contamination.
Polypropylene Polyethylene (linear)	Strongly recommended for corrosive high dissolved solids solutions. Less likely to introduce significant bias into analytical results than polymer formulations (PVC) or other flexible materials with the exception of Teflon*.
PVC (flexible)	Not recommended for detailed organic analytical schemes. Plasticizers and stabilizers make up a sizable percentage of the material by weight as long as it remains flexible. Documented interferences are likely with several priority pollutant classes.
Viton* Silicone (medical grade only) Neoprene	Flexible elastomeric materials for gaskets, O-rings, bladder, and tubing applications. Performance expected to be a function of exposure type and the order of chemical resistance as shown. Recommended only when a more suitable material is not available for the specific use. Actual controlled exposure trials may be useful in assessing the potential for analytical bias.

*Trademark of DuPont, Inc.

Table 2-3. Recommendations for Flexible Materials in Sampling Applications

is basically a step-by-step written description of the procedures used for well purging, delivering samples to the surface, and handling samples in the field. Once the protocol has been developed and used in a particular investigation, it provides a basis for modifying the program, if the extent or type of contamination requires more intensive work. An appropriate sampling mechanism is, however, an important part of any protocol. Ideally, the pumping mechanism should be capable of purging the well of stagnant water at rates of liters or gallons per minute and also of delivering ground water to the surface so that sample bottles may be filled at low flow rates (i.e., about 100 mL/min^{-1}) to minimize turbulence and degassing of the sample. In this way the criteria for representative sampling can be met while keeping the purging and sample collection steps simple. Nielsen and Yeates (1985) reviewed the types of sample collection mechanisms commercially available (Anonymous, 1985). This review supports the results, of research studies of their performance (Barcelona and others, 1984; Stoltzenburg and Nichols, 1985). Figure 2-3 shows examples of types of pumps or other samplers, which are fully described in a number of references (Barcelona and others, 1985; Gillham and others, 1983; Scalf and others, 1981). Given all of the varied hydrogeologic settings and potential chemical constituents of interest, several types of pumps or sampling mechanisms may be suitable for specific applications. Figure 2-4 contains some recommendations for reliable sampling mechanisms relative to the sensitivity of the sample to error. The main criteria for sampling pumps are the capabilities to purge stagnant water from the well and to deliver the water samples to the surface with minimal loss of sample integrity. Clearly, a mechanism that is shown to provide accurate and precise samples for volatile organic compound determinations should be suitable for most chemical constituents of interest.

After establishing a sampling point and the means to collect a sample, the next step is the development of the detailed sampling protocol.

Elements of the Sampling Protocol

There are few aspects of this subject that generate more controversy than the sampling steps, which make up the sampling protocol. Efforts to develop reliable protocols and optimize sampling procedures require particular attention to sampling mechanism effects on the integrity of ground-water samples (Barcelona and others, 1984; Stolzenburg and Nichols, 1985), as well as to the potential errors involved in well purging, delivery tubing exposures (Barcelona and others, 1985; Ho, 1983), sample handling, and the impact of sampling frequency on both the sensitivity and reliability of chemical constituent monitoring results. Quality assurance measures, including field blanks, standards, and split control samples, cannot account for errors in these steps of the sampling protocol. Actually, the sampling protocol is the focus of the overall study network design (Nacht, 1983), and it should be prepared flexibly so that it can be refined as information on site improves.

Each step within the protocol has a bearing on the quality and completeness of the information being collected. This is perhaps best shown by the progression of steps depicted in Figure 2-5. Corresponding to each step is a goal and recommendation for achieving that goal. The principal utility of this description is that it provides an outlined agenda for high-quality chemical and water-quality data.

To ensure maximum utility of the sampling effort and resulting data, it is essential to document the sampling protocol as performed in the field. In addition to noting the obvious information (i.e., persons conducting the sampling, equipment used, weather conditions, adherence to the protocol, and unusual observations), three basic elements of the sampling protocol should be recorded: (1) water-level measurements made prior to sampling, (2) the volume and rate at which water is removed from the well prior to sample collection (well purging), and (3) the actual sample collection, including measurement of well-purging parameters, sample preservation, sample handling, and chain of custody.

Water-Level Measurement

Prior to well purging or sample collection, it is extremely important to measure and record the water level in the well. These measurements are needed to estimate the amount of water to be purged prior to sample collection. Likewise, this information can be useful when interpreting monitoring results. Low water levels may reflect the influence of the cone of depression surrounding a nearby production well. High water levels, compared to measurements made at other times of the year, may be indicative of recent recharge events. In relatively shallow settings, high water levels from recent natural recharge events may result in the increase of certain constituents leached from the unsaturated zone or in the dilution of the dissolved solids content in the collected sample.

Documenting the nonpumping water levels for all wells at a site will provide historical information on the hydraulic conditions at the site. Analysis of this information may reveal changes in flow paths and serve as a check on the effectiveness of the wells to monitor changing hydrologic conditions. It is very useful to develop an

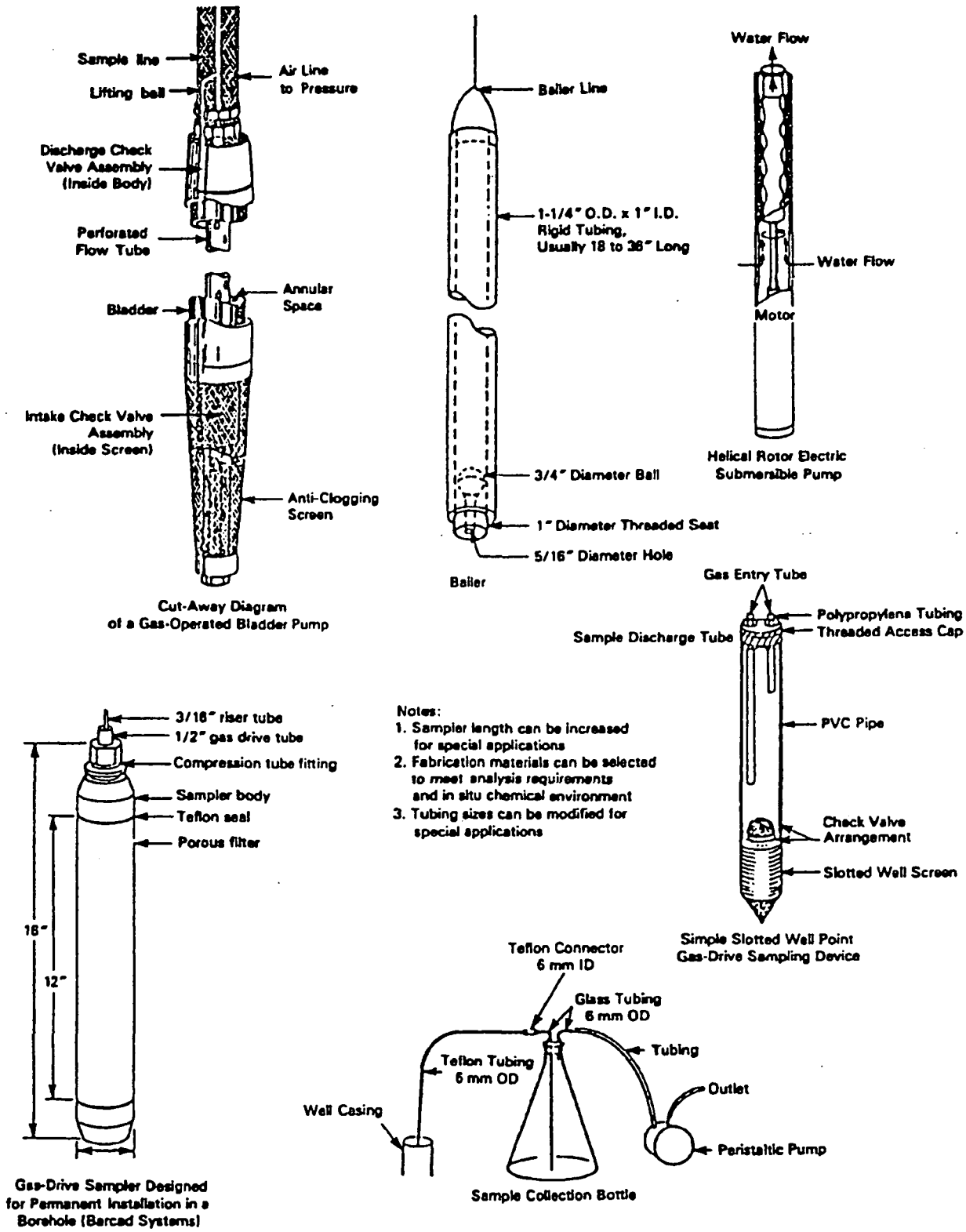

Figure 2-3. Schematic Diagrams of Common Ground-Water Sampling Devices (Nelisen and Yeates, 1985)

Type of constituent	Example of constituent	Positive displacement bladder pumps	Thief, in situ or dual check valve bailers	Mechanical positive displacement pumps	Gas-drive devices	Suction mechanisms
		INCREASING RELIABILITY OF SAMPLING MECHANISMS ←				
Volatile Organic Compounds Organometallics	Chloroform TOX CH_3Hg	Superior performance for most applications	May be adequate if well purging is assured	May be adequate if design and operation are controlled	Not recommended	Not recommended
Dissolved Gases Well-purging Parameters	O_2, CO_2 pH, Ω^{-1} Eh	Superior performance for most applications	May be adequate if well purging is assured	May be adequate if design and operation are controlled	Not recommended	Not recommended
Trace Inorganic Metal Species Reduced Species	Fe, Cu NO_2^-, S^-	Superior performance for most applications	May be adequate if well purging is assured	Adequate	May be adequate	May be adequate if materials are appropriate
Major Cations & Anions	Na^+, K^+, Ca^{++} Mg^{++} Cl^-, SO_4^-	Superior performance for most applications	Adequate May be adequate if well purging is assured	Adequate	Adequate	Adequate

INCREASING SAMPLE SENSITIVITY ↑

Figure 2-4. Matrix of Sensitive Chemical Constituents and Various Sampling Mechanisms

Step	Goal	Recommendations
Hydrologic Measurements	Establish nonpumping water level.	Measure the water level to ±0.3 cm (±0.01 ft).
Well Purging	Remove or isolate stagnant H_2O which would otherwise bias representative sample.	Pump water until well purging parameters (e.g., pH, T, Ω^{-1}, Eh) stabilize to ±10% over at least two successive well volumes pumped.
Sample Collection	Collect samples at land surface or in well-bore with minimal disturbance of sample chemistry.	Pumping rates should be limited to ~100 mL/min for volatile organics and gas-sensitive parameters.
Filtration/Preservation	Filtration permits determination of soluble constituents and is a form of preservation. It should be done in the field as soon as possible after collection.	Filter: Trace metals, inorganic anions/cations, alkalinity. Do not filter: TOC, TOX, volatile organic compound samples; other organic compound samples only when required.
Field Determinations	Field analyses of samples will effectively avoid bias in determining parameters/constituents which do not store well; e.g., gases, alkalinity, pH.	Samples for determining gases, alkalinity and pH should be analyzed in the field if at all possible.
Field Blanks/Standards	These blanks and standards will permit the correction of analytical results for changes which may occur after sample collection: preservation, storage, and transport.	At least one blank and one standard for each sensitive parameter should be made up in the field on each day of sampling. Spiked samples are also recommended for good QA/QC.
Sample Storage/Transport	Refrigerate and protect samples to minimize their chemical alteration prior to analysis.	Observe maximum sample holding or storage periods recommended by the Agency. Documentation of actual holding periods should be carefully performed.

Figure 2-5. Generalized Ground-Water Sampling Protocol

understanding of the seasonal changes in water levels and associated chemical concentration variability at the monitored site.

Purging

The volume of stagnant water that should be removed from the monitoring well should be calculated from the analysis of field hydraulic conductivity measurements. Rule-of-thumb guidelines for the volume of water to be purged can cause time delays and unnecessary pumping of excess contaminated water. These rules (i.e., 3-, 5- or 10-well volumes) largely ignore the hydraulic characteristics of individual wells and geologic settings. One advantage of using the same pump to both purge stagnant water and collect samples is the ability to measure pH and specific conductance in an in-line flow

cell. These parameters aid in verifying the purging efficiency and also provide a consistent basis for comparing samples from a single well or wells at a particular site. Since pH is a standard variable for aqueous solutions that is affected by degassing and depressurization (i.e., loss of CO_2), in-line measurements provide more accurate and precise determinations than discrete samples collected by grab sampling mechanisms.

The following example illustrates some of the other advantages of verifying the purge requirement for monitoring wells.

Documentation of the actual well purging process employed should be a part of a standard field sampling protocol. The calculated well purging requirement (e.g., >90 percent aquifer water) calls for the removal of five well volumes prior to sample collection. Field measurements of the well purging parameters have historically confirmed this recommended procedure. During a subsequent sampling effort, 12 well volumes were pumped before stabilized well purging parameter readings were obtained. Several possible causes could be explored: (1) a limited plume of contaminants may have been present at the well at the beginning of sampling and inadvertently discarded while pumping in an attempt to obtain stabilized indicator parameter readings; (2) the hydraulic properties of the well may have changed due to silting or encrustation of the screen, indicating the need for well rehabilitation or maintenance; (3) the flow-through device used for measuring the indicator parameters may have been malfunctioning; or (4) the well may have been tampered with by the introduction of a contaminant or relatively clean water in an attempt to bias the sample results.

Sample Collection and Handling

Water samples should be collected when the solution chemistry of the ground water being pumped has stabilized as indicated by pH, Eh, specific conductance, and temperature readings.

In practice, stable sample chemistry is indicated when the purging parameter measurements have stabilized over two successive well volumes. First, samples for volatile constituents, TOC, TOX, and those constituents that require field filtration or field determination should be collected. Then large-volume samples for extractable organic compounds, total metals, or nutrient anion determinations should be collected.

All samples should be collected as close as possible to the well head. A "tee" fitting placed ahead of the in-line device for measuring the well purging parameters makes this more convenient. Regardless of the sample mechanism in use or the components of the sampling train, wells that are located upgradient of a site, and therefore are expected to be representative of background quality, should be sampled first to minimize the potential for cross-contamination. Laboratory detergent solutions and distilled water should be used to clean the sampling train between samples. An acid rinse (0.1 N HCl) or solvent rinse (i.e., hexane or methanol) may be used to supplement these cleaning steps, if necessary. Cleaning procedures should be followed by distilled water rinses, which may be saved to check cleaning efficiency.

The order in which samples are taken for specific types of chemical analyses should be decided by the sensitivity of the samples to handling (i.e., most sensitive first) and the need for specific information. For example, the flowchart shown in Figure 2-6 depicts a priority order for a generalized sample collection effort. The samples for organic chemical constituent determinations are taken in decreasing order of sensitivity to handling errors, while the inorganic chemical constituents, which may require filtration, are taken afterwards.

Instances arise, even with properly developed monitoring wells, that call for the filtration of water samples. It should be evident, however, that adequate well development procedures, which require 2 to 3 hours of bailing, swabbing, pumping, or air purging at each well, may save many hours in sample filtration. Well development may have to be repeated at periodic intervals to minimize the collection of turbid samples. In this respect, it is important to minimize the disturbance of fines that accumulate in the well bore. This can be achieved by careful placement of the sampling pump intake at the top of the screened interval, low pumping rates, and avoiding the use of bailing techniques that disturb sediment accumulations.

It is advisable to refrain from filtering TOC, TOX, or other organic compound samples because the increased handling required may result in the loss of chemical constituents of interest. Allowing any fine material to settle prior to analysis, followed by decanting the sample, is preferable to filtration in these instances. If filtration is necessary for the determination of extractable organic compounds, it should be performed in the laboratory using nitrogen pressure. When samples must be filtered, it may be necessary to run parallel sets of filtered and unfiltered samples with standards to establish the recovery of hydrophobic compounds. All of the materials' precautions used in the construction of the sampling train should be observed for filtration apparatus. Vacuum filtration of ground-water samples is not recommended.

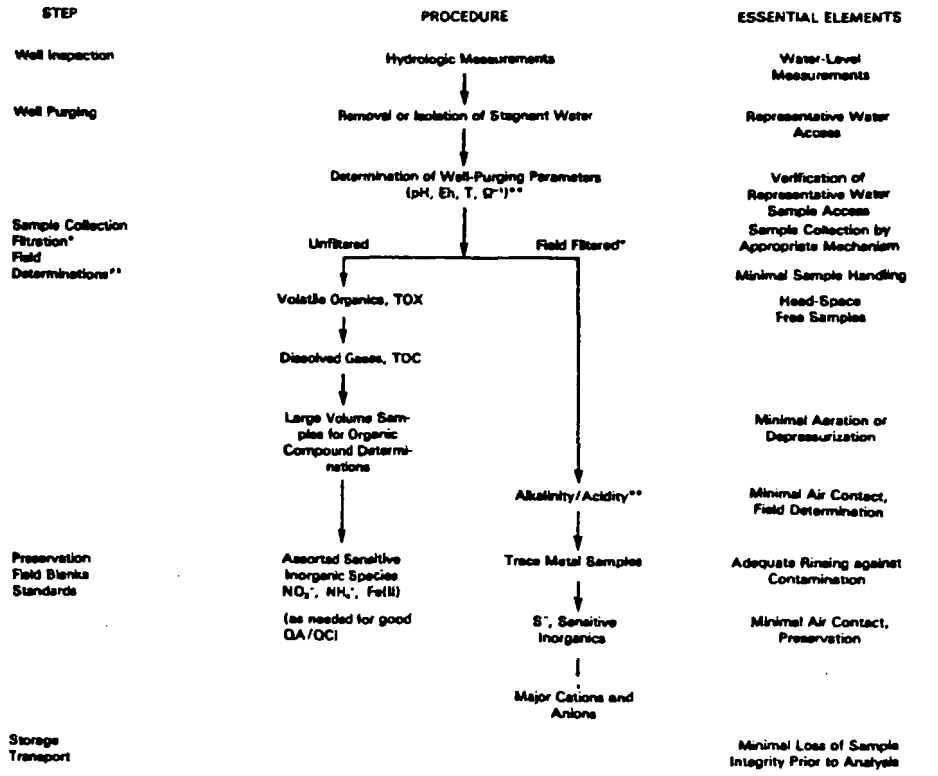

Figure 2-6. Generalized Flow Diagram of Ground-Water Sampling Steps (Barcelona and others, 1985)

Water samples for dissolved inorganic chemical constituents (e.g., metals, alkalinity, and anionic species) should be filtered in the field. The preferred arrangement is an in-line filtration module, which utilizes sampling pump pressure for its operation. These modules have tubing connectors on the inlet and outlet parts and range in diameter from 2.5 to 15 cm. Large diameter filter holders, which can be rapidly disassembled for filter pad replacement, are the most convenient and efficient designs (Kennedy and others, 1976; Skougstad and Scarbo, 1968).

Representative sampling results from the execution of a carefully planned sampling protocol. An important consideration for maintaining sample integrity after collection is to minimize sample handling, which may bias subsequent determinations of chemical constituents. Since opportunities to collect high-quality data for the characterization of site conditions may be limited by time, it is prudent to conduct sample collection as carefully as possible from the beginning of the sampling period. It is preferable to risk error on the conservative side when doubt exists as to the sensitivity of specific chemical constituents to sampling or handling errors. Repeat sampling or analysis cannot make up for lost data collection opportunities.

For samples collected for specific chemical constituents, recommended sample handling and analysis procedures may need to be modified. Samples that contain several chemicals and have undergone extended storage periods can cause significant problems. It is frequently more effective to perform a rapid field determination of specific inorganic constituents (e.g., alkalinity, pH, ferrous iron, sulfide, nitrite, or ammonium) than to attempt sample preservation followed by laboratory analysis of these samples.

Many samples can be held for the U.S. EPA recommended maximum holding times after proper preservation (Table 2-4).

Quality Assurance/Quality Control

Planning for valid water-quality data collection depends upon both the knowledge of the system and continued refinement of all sample handling/collection procedures.

Parameters (Type)	Volume Required (mL) 1 Sample*	Container (Material)	Preservation Method	Maximum Holding Period
Well Purging				
pH (grab)	50	T,S,P,G	None; field det.	<1 hr**
Ω^{-1} (grab)	100	T,S,P,G	None; field det.	<1 hr**
T (grab)	1000	T,S,P,G	None; field det.	None
Eh (grab)	1000	T,S,P,G	None; field det.	None
Contamination Indicators				
pH, Ω^{-1} (grab)	As above	As above	As above	As above
TOC	40	G,T	Dark, 4°C	24 hr
TOX	500	G,T	Dark, 4°C	5 days
Water Quality				
Dissolved gases (O_2, CH_4, CO_2)	10 mL minimum	G,S	Dark, 4°C	<24 hr
Alkalinity/Acidity	100	T,G,P	4°C/None	<6 hr**/ <24 hr
	Filtered under pressure with appropriate media			
(Fe, Mn, Na^+, K^+, Ca^{++}, Mg^{++})	All filtered 1000 mL	T,P	Field acidified to pH <2 with HNO_3	6 months***
(PO_4^-, Cl^-, Silicate)	@50	(T,P,G glass only)	4°C	24 hr/ 7 days; 7 days
NO_3^-	100	T,P,G	4°C	24 hr
SO_4^-	50	T,P,G	4°C	7 days
NH_4^+	400	T,P,G	4°C/H_2SO_4 to pH <2	24 hr/ 7 days
Phenols	500	T,G	4°C/H_3PO_4 to pH <4	24 hr
Drinking Water Suitability				
As, Ba, Cd, Cr, Pb, Hg, Se, Ag	Same as above for water quality cations (Fe, Mn, etc.)	Same as above	Same as above	6 months
F^-	Same as chloride above	Same as above	Same as above	7 days
Remaining Organic Parameters	As for TOX/TOC, except where analytical method calls for acidification of sample			24 hr

*It is assumed that at each site, for each sampling date, replicates, a field blank and standards must be taken at equal volume to those of the samples.

**Temperature correction must be made for reliable reporting. Variations greater than ± 10% may result from longer holding period.

***In the event that HNO_3 cannot be used because of shipping restrictions, the sample should be refrigerated to 4°C, shipped immediately, and acidified on receipt at the laboratory. Container should be rinsed with 1:1 HNO_3 and included with sample.

Note: T = Teflon; S = stainless steel; P = PVC, polypropylene, polyethylene; G = borosilicate glass.

From Scalf et al., 1981.

Table 2-4. Recommended Sample Handling and Preservation Procedures for a Detective Monitoring Program

As discussed earlier, the need to begin QA/QC planning with the installation of the sampling point cannot be overemphasized.

The use of field blanks, standards, and spiked samples for field QA/QC performance is analogous to the use of laboratory blanks, standards, and procedural or validation standards. The fundamental goal of field QC is to ensure that the sample protocol is being executed faithfully and that situations that might lead to error are recognized before they seriously impact the data. The use of field blanks, standards, and spiked samples can account for changes in samples that occur during sample collection.

Field blanks and standards enable quantitative correction for bias (i.e., systematic errors), which arise due to handling, storage, transportation, and laboratory procedures. Spiked samples and blind controls provide the means to correct combined sampling and analytical accuracy or recoveries for the actual conditions to which the samples have been exposed.

All QC measures should be performed for at least the most sensitive chemical constituents for each sampling date. Examples of sensitive constituents would be benzene or trichloroethylene as volatile organic compounds and lead or iron as metals. It is difficult to use laboratory blanks alone for determining the limits of detection or quantitation. Laboratory distilled water may contain apparently higher levels of volatile organic compounds (e.g., methylene chloride) than would uncontaminated ground-water samples. The field blanks and spiked samples should be used for this purpose, conserving the results of lab blanks as checks on elevated laboratory background levels.

Whether or not the ground water is contaminated with interfering compounds, spiked samples provide a basis for both identifying the constituents of interest and correcting their recovery (or accuracy) based on the recovery of the spiked standard compounds. For example, if trichloroethylene in a spiked sample is recovered at a mean level of 80 percent (-20 percent bias), the concentrations of trichloroethylene determined in the samples for this sampling date may be corrected by a factor of 1.2 for low recovery. Similarly, if 50 percent recovery (-50 percent bias) is reported for the spiked standard, it is likely that sample handling or analytical procedures are out of control and corrective measures should be taken at once. It is important to know if the laboratory has performed these corrections or taken corrective action when it reports the results of analyses. It should be further noted that many regulatory agencies require evidence of QC and analytical performance but do not generally accept data that have been corrected.

Field blanks, standards, and blind control samples provide independent checks on handling and storage, as well as the performance of the analytical laboratory. Ground-water analytical data are incomplete unless the analytical performance data (e.g., accuracy, precision, detection, and quantitation limits) are reported with each set of results. Discussions of whether ground-water quality has changed significantly must be tempered by the accuracy and precision performance for specific chemical constituents.

Table 2-5 is a useful guide to the preparation of field standards and spiking solutions for split samples. It is important that the field blanks and standards are made on the day of sampling and are subjected to all conditions to which the samples are exposed. Field spiked samples or blind controls should be prepared by spiking with concentrated stock standards in an appropriate

				Stock Solution for Field Spike of Split Samples		
Sample Type	Volume	Composition	Field Standard (Concentration)	Solvent	Concentration of Components	Field Spike Volume
Alkalinity	50 mL	Na^+, HCO_3^-	10.0; 25 (ppm)	H_2O	10,000; 25,000 (ppm)	(50 µL)
Anions	1 L	K^+, Na^+, Cl^-, SO_4^- F^-, NO_3^-, $PO_4^{\equiv}$, SI	25, 50 (ppm)	H_2O	25,000; 50,000 (ppm)	(1 mL)
Cations	1 L	Na^+, K^+ Ca^{++}, Mg^{++}, Cl^-, NO_3^-	5.0; 10.0 (ppm)	H_2O, H^+ (acid)	5,000; 10,000 (ppm)	(1 mL)
Trace Metals	1 L	Cd^{++}, Cu^{++}, Pb^{++} Cr^{+++}, Ni^{++}, Ag^+ Fe^{+++}, Mn^{++}	10.0; 25.0 (ppm)	H_2O, H^+ (acid)	10,000; 25,000 (ppm)	(1 mL)
TOC	40 mL	Acetone KHP	0.2; 0.5 (ppm-C) 1.8; 4.5 (ppm-C)	H_2O	200; 500 (ppm-C) 1,800; 4,500 (ppm-C)	(40 µL)
TOX	50 mL	Chloroform 2,4,6 Trichlorophenol	12.5; 25 (ppb) 12.5; 25 (ppb)	H_2O/poly* (ethylene glycol)	12,500; 25 (ppm) 12,500; 25 (ppm)	(500 µL)
Volatiles	40 mL	Dichlorobutane, Toluene Dibromopropane, Xylene	25; 50 (ppb)	H_2O/poly* (ethylene glycol)	25; 50 (ppm)	(40 µL)
Extractables A	1 L	Phenol Standards	25; 50 (ppb)	Methanol**	25; 50 (ppm)	(1 mL)
Extractables B	1 L	Polynuclear Aromatic Standards	25; 50 (ppb)	Methanol	25; 50 (ppm)	(1 mL)
Extractables C	1 L	Standards as Required	25; 50 (ppb)	Methanol	25; 50 (ppm)	(1 mL)

*75:25 water/polyethylene glycol (400 amu) mixture.

**Glass distilled methanol.

Table 2-5. Field Standard and Sample Spiking Solutions

background solution prior to the collection of any actual samples. Additional precautions should be taken against the depressurization of samples during air transport and the effects of undue exposure to light during sample handling and storage. All of the QC measures noted above will provide both a basis for high-quality data reporting and a known degree of confidence in data interpretation. Well-planned overall quality control programs also will minimize the uncertainty in long-term trends when different personnel have been involved in collection and analysis.

Sample Storage and Transport

The storage and transport of ground-water samples often are the most neglected elements of the sampling protocol. Due care must be taken in sample collection, field determinations, and handling. Transport should be planned so as not to exceed sample holding time before laboratory analysis. Every effort should be made to inform the laboratory staff of the approximate time of arrival so that the most critical analytical determinations can be made within recommended storage periods. This may require that sampling schedules be adjusted so that the samples arrive at the laboratory during working hours.

The documentation of actual sample storage and treatment may be handled by chain of custody procedures. Figure 2-7 shows an example of a chain of custody form. Briefly, the chain of custody record should

CHAIN OF CUSTODY RECORD

Sampling Date _____ Site Name _____

Well or Sampling Points: _____

_____

Sample Sets for Each: Inorganic, Organic, Both

Inclusive Sample Numbers:

Company's Name _____ Telephone (___) _____

Address _____
number street city state zip

Collector's Name _____ Telephone (___) _____

Date Sampled _____ Time Started _____ Time Completed _____

Field Information (Precautions, Number of Samples, Number of Sample Boxes, Etc.):

1. _____
name organization location
2. _____
name organization location

Chain of Possession (After samples are transported off-site or to laboratory):

1. _____ _____ ____ ____ (IN)
signature title
_____ _____ ____ ____(OUT)
name (printed) date/time of receipt
2. _____ _____ ____ ____ (IN)
signature title
_____ _____ ____ ____(OUT)
name (printed) date/time of receipt

Analysis Information:

Aliquot	Analysis Begun (date/time)	Initials	Analysis Completed (date/time)	Initials
1.				
2.				
3.				
4.				
5.				

Figure 2-7. Sample Chain of Custody Form

contain the dates and times of collection, receipt, and completion of all the analyses on a particular set of samples. Frequently, it is the only record that exists of the actual storage period prior to the reporting of analytical results. The sampling staff members who initiate the chain of custody should require that a copy of the form be returned to them with the analytical report. Otherwise, verification of sample storage and handling will be incomplete.

Shipping should be arranged to ensure that samples are neither lost nor damaged enroute to the laboratory. Several commercial suppliers of sampling kits permit refrigeration by freezer packs and include proper packing. It may be useful to include special labels or distinctive storage vessels for acid-preserved samples to accommodate shipping restrictions.

Summary

Ground-water sampling is conducted for a variety of reasons, ranging from detection or assessment of the extent of a contaminant release to evaluations of trends in regional water quality. Reliable sampling of the subsurface is inherently more difficult than either air or surface water sampling because of the inevitable disturbances that well drilling or pumping can cause and the inaccessibility of the sampling zone. Therefore, "representative" sampling generally requires minimal disturbance of the subsurface environment and the properties of a representative sample are scale dependent. For any particular case, the applicable criteria should be set at the beginning of the effort to judge representativeness.

Reliable sampling protocols are based on the hydrogeologic setting of the study site and the degree of analytical detail required by the monitoring program. Quality control begins with the evaluation of the hydraulic performance of the sampling point or well and the proper selection of mechanisms and materials for well purging and sample collection. All other elements of the program and variables that affect data validity may be accounted for by field blanks, standards, and control samples.

Although research is needed on a host of topics involved in ground-water sampling, defensible sampling protocols can be developed to ensure the collection of data of known quality for many types of programs. If properly planned and developed, long-term sampling efforts can benefit from the refinements that research progress will bring. Careful documentation will provide the key to this opportunity.

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Chapter 3

TRANSPORT AND FATE OF CONTAMINANTS IN THE SUBSURFACE

Introduction

Protection and remediation of ground-water resources require an understanding of processes that affect fate and transport of contaminants in the subsurface environment. This understanding allows: (1) prediction of the time of arrival and concentration of contaminants at a receptor, such as a monitoring well, a water supply well, or a body of surface water; (2) design of cost-effective and safe waste management facilities; (3) installation of effective monitoring systems; and (4) development of efficient and cost-effective strategies for remediation of contaminated aquifers (Palmer and Johnson, 1989a).

Contaminants in ground water will move primarily in a horizontal direction that is determined by the hydraulic gradient. The contaminants will decrease in concentration because of such processes as dispersion (molecular and hydrodynamic), filtration, sorption, various chemical processes, microbial degradation, time rate release of contaminants, and distance of travel (U.S. Environmental Protection Agency, 1985). Processes such as hydrodynamic dispersion affect all contaminants equally, while sorption, chemical processes, and degradation may affect various contaminants at different rates. The complex factors that control the movement of contaminants in ground water and the resulting behavior of contaminant plumes are commonly difficult to assess because of the interaction of the many factors that affect the extent and rate of contaminant movement. Predictions of movement and behavior can be used only as estimates, and modeling is often a useful tool to integrate the various factors.

The U.S. Environmental Protection Agency (EPA) sponsored a series of technology transfer seminars between October 1987 and February 1988 that provided an overview of the physical, chemical, and biological processes that govern the transport and fate of contaminants in the subsurface. The following discussion is a summary of the workshops, and is based on the seminar publication, Transport and Fate of Contaminants in the Subsurface (U.S. Environmental Protection Agency, 1989).

Physical Processes Controlling the Transport of Contaminants in the Aqueous Phase in the Subsurface

Advection-Dispersion Theory

The study of advection and dispersion processes is useful for predicting the time when an action limit, i.e., a concentration limit used in regulations such as drinking water standards, will be reached. Knowledge of advection-dispersion also can be used to select technically accurate and cost-effective remedial technologies for contaminated aquifers.

If concentrations of a contaminant were measured in a monitoring well that was located between a contaminant source and a receptor such as a water supply well, a graph of concentrations versus time would show a breakthrough curve, i.e., the concentrations do not increase in a step-function (i.e., plug flow), but rather in an S-shaped curve (Figure 3-1). In a one-dimensional, homogeneous system, the arrival of the center of the mass is due to advection, while the spread of the breakthrough curve is the result of dispersion (Palmer and Johnson, 1989a).

Advection

Advection is defined by the transport of a non-reactive, conservative tracer at an average ground-water velocity (Palmer and Johnson, 1989a). The average linear velocity is dependent on (1) the hydraulic conductivity of the subsurface geologic formation in the direction of ground-water flow, (2) the porosity of the formation and (3) the hydraulic gradient in the direction of ground-water flow. For waste contaminants that react through precipitation/dissolution, adsorption, and/or partitioning reactions within the subsurface formation, the velocity can be different from the average ground-water velocity.

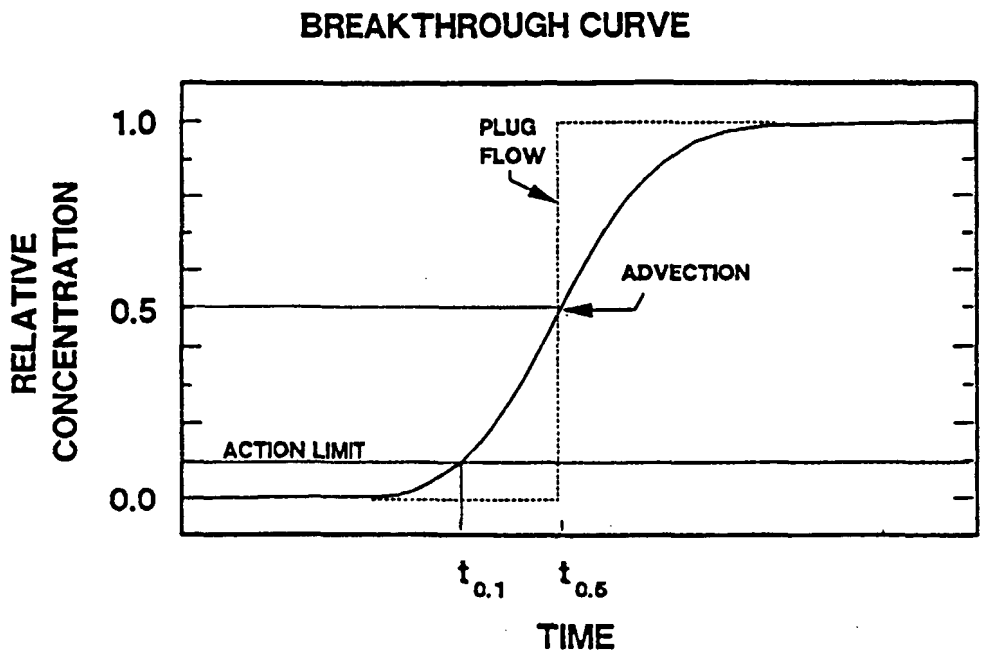


Figure 3-1. Breakthrough Curve for a Contaminant, as Measured in a Monitoring Well (Palmer and Johnson, 1989a)

Dispersion

Dispersion of waste contaminants in an aquifer causes the concentration of contaminants to decrease with increasing length of flow (U.S. Environmental Protection Agency, 1985). Dispersion is caused by: (1) molecular diffusion (important only at very low velocities) and (2) hydrodynamic mixing (occurring at higher velocities in laminar flow through porous media). Contaminants traveling through porous media have different velocities and flow paths with different lengths. Contaminants moving along a shorter flow path or at a higher velocity, therefore, arrive at a specific point sooner than contaminants following a longer path or traveling at a lower velocity, resulting in hydrodynamic dispersion.

Figure 3-2 shows that dispersion can occur in both longitudinal (in the direction of ground-water flow) and transverse (perpendicular to ground-water flow) directions, resulting in the formation of a conic waste plume downstream from a continuous pollution source (U.S. Environmental Protection Agency, 1985). The concentration of waste contaminants is less at the margins of the plume and increases towards the source. A plume will increase in size with more rapid flow within a time period, because dispersion is directly related to ground-water velocity.

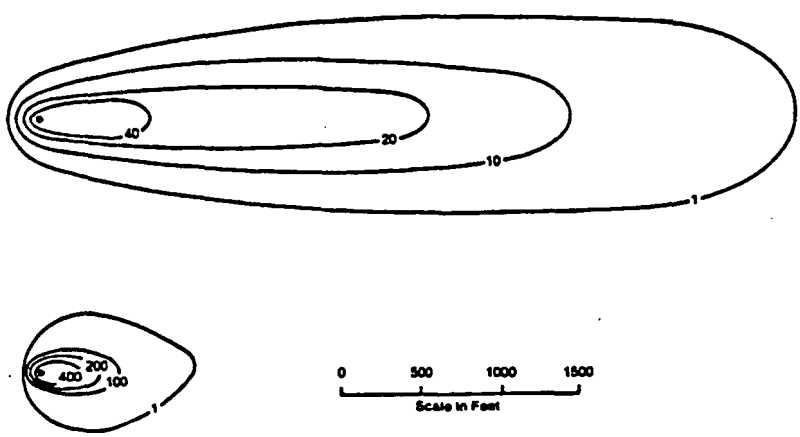


Figure 3-2. The Effects of Ground-Water Velocity on Plume Shape. Upper Plume Velocity: 1.5 ft/day and Lower Plume Velocity: 0.5 ft/day (U.S. Environmental Protection Agency, 1985).

The dispersion coefficient varies with ground-water velocity. At low velocity, the dispersion coefficient is relatively constant, but increases linearly with velocity as ground-water velocity increases. Based on these observations, investigators proposed that the dispersion coefficient can be expressed as a sum of an effective molecular diffusion coefficient and a mechanical dispersion coefficient (Palmer and Johnson, 1989a).

The effective molecular diffusion coefficient is a function of the solution diffusion coefficient and the tortuosity of the medium. Tortuosity accounts for the increased distance a diffusing ion must travel around sand grains. The mechanical dispersion coefficient is proportional to velocity. Specifically, mechanical dispersion is a result of: (1) velocity variations within a pore, (2) different pore geometries, and (3) divergence of flow lines around sand grains present in a porous medium (Gillham and Cherry, 1982).

The term dispersivity is often confused with dispersion. Dispersivity does not include velocity, so to convert dispersivity to dispersion requires multiplication by velocity. Since dispersion is dependent on site-specific velocity parameters and configuration of pore spaces within an aquifer, a dispersion coefficient should be determined experimentally or empirically for a specific aquifer. The selection of appropriate dispersion coefficients that adequately reflect existing aquifer conditions is critical to the success of chemical transport modeling (U.S. Environmental Protection Agency, 1985).

Advection-Dispersion Equation

An advection-dispersion equation is used to express the mass balance of a waste contaminant within an aquifer as a result of dispersion, advection, and change in storage. The mass balance is a function of the dispersion coefficient, the ground-water velocity, concentration of the contaminant, distance, and time (Palmer and Johnson, 1989a). An advection-dispersion equation can be applied to the description of three-dimensional transport of waste contaminants in an aquifer, using three dispersion coefficients (one longitudinal and two transverse). Mathematically detailed descriptions of the advection-dispersion equation are presented in Bear (1969, 1979).

Discrepancies between results generated from advection-dispersion equations and laboratory and field experiments have been found. These discrepancies have been attributed to: (1) immobile zones of water within the aquifer, (2) solution-solid interface processes, (3) anion exclusion, and (4) diffusion in and out of aggregates (Palmer and Johnson, 1989a).

Field observations using field tracer studies also have shown that longitudinal dispersivity values are usually much larger than transverse dispersivity measurements (Palmer and Johnson, 1989a). Figure 3-3 shows three-dimensional field monitoring that has corroborated these observations by identifying long, thin contaminant plumes rather than plumes spread over the thickness of an aquifer. (Kimmel and Braids, 1980; MacFarlane and others, 1983). The large longitudinal dispersion coefficients are thought to result from aquifer heterogeneity. In an ideally stratified aquifer with layers of sediment of different hydraulic conductivities, contaminants move rapidly along layers with higher permeabilities and more slowly along the lower permeability layers (Figure 3-4) (Palmer and Johnson, 1989a). Sample concentration of a contaminant is an integration of the concentrations of each layer, if water is sampled from monitoring wells that are screened

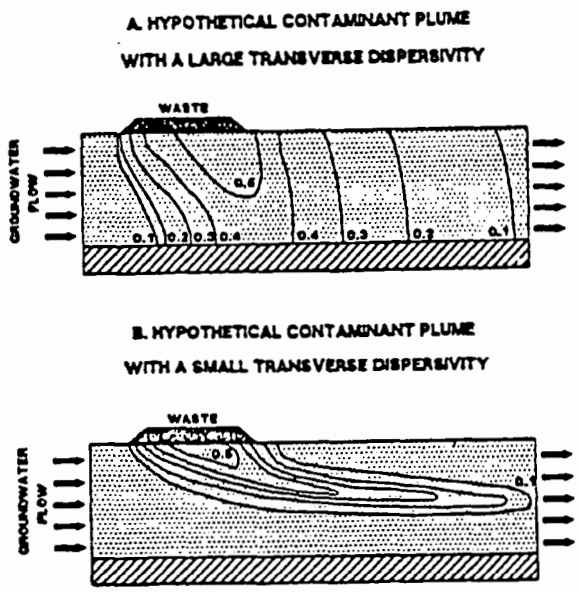


Figure 3-3. Hypothetical Contaminant Plumes for Large (A) and Small (B) Dispersivities (Palmer and Johnson, 1989a)

through the various layers. Results from plotting concentration versus distance show a curve with large differences in concentrations, even though only advection is considered. This dispersion is the result of aquifer heterogeneity and not pore-scale processes.

However, defining hydraulic conductivities in the subsurface is difficult, since not all geologic formations are perfectly stratified, but may contain cross-stratification or graded bedding (Palmer and Johnson, 1989a). To quantify heterogeneity in an aquifer, hydraulic conductivity is considered to be random, and statistical characteristics, such as mean, variance, and autocorrelation function, are determined.

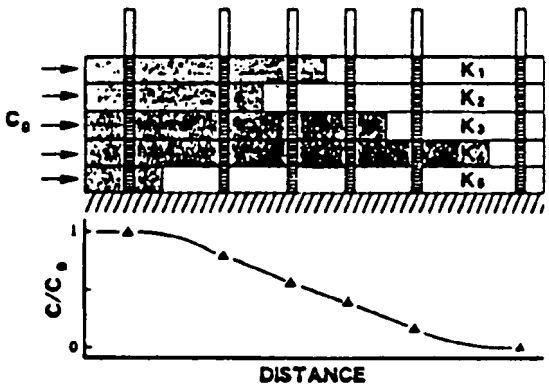


Figure 3-4. Contaminant Distributions and Concentrations in an Ideally Stratified Aquifer (after Gillham and Cherry, 1982, by Palmer and Johnson, 1989a)

In addition to aquifer heterogeneity, other processes contributing to the spread of contaminants include: (1) diverging flow lines resulting in the spread of contaminants by advection over a larger cross section of the aquifer, (2) temporal variations in the water table resulting in change of direction of ground-water flow and lateral spread of contamination, and (3) variations in concentration of contaminants at the source resulting in apparent dispersion in the longitudinal direction (Frind and Hokkanen, 1987; Palmer and Johnson, 1989a).

Ground-water sampling methods also may result in detection of apparent spreading of contaminant plumes (Palmer and Johnson, 1989a). An underestimation of contaminant concentrations at specific locations in an aquifer may be due to insufficient well-purging. Monitoring wells with different screen lengths that integrate ground water from different sections of the aquifer may yield dissimilar contaminant concentrations.

Diffusive Transport through Low Permeability Materials

In materials with low hydraulic conductivities (e.g., unfractured clays and rocks with conductivities less than 10 to 9 m/s), diffusive transport of waste contaminants is large compared to advective transport (Neuzil, 1986; Palmer and Johnson, 1989a). Contaminants can diffuse across natural aquitards or clay liners with low hydraulic conductivities, resulting in aquifer contamination. The extent of movement is dependent on diffusive flux, rate of ground-water flow in the aquifer, and the length of the source area in the direction of ground-water flow.

Effects of Density on Transport of Contaminants

The density of a contaminant plume may contribute to the direction of solute transport if dissolved concentrations of contaminants are large enough (Palmer and Johnson, 1989a). For example, assume that the density of ground water within an aquifer is 1.00, the natural horizontal gradient is 0.005, and the natural vertical gradient is 0.000. If the density of the contaminant plume is equal to the density of the ground water, the plume moves horizontally with the naturally existing hydraulic gradient. If the density of the contaminated water is 1.005 (a concentration of approximately 7,000 mg/L total dissolved solids), then the driving force in the vertical direction is the same as the driving force in the horizontal direction. If the aquifer is isotropic, then the resulting vector of these two forces descends at 45 degrees into the aquifer. The contaminant plume moves deeply into the aquifer and may not be detected with shallow monitoring systems installed under the assumption of horizontal flow.

Retardation of Contaminants

If contaminants undergo chemical reactions while being transported through an aquifer, their movement rate may be less than the average ground-water flow rate (Palmer and Johnson, 1989a). Such chemical reactions that slow movement of contaminants in an aquifer include precipitation, adsorption, ion exchange, and partitioning into organic matter or organic solvents. Chemical reactions affect contaminant breakthrough, as shown in Figure 3-5. If the retardation factor, R (calculated from equations for contaminant transport that include retardation), is equal to 1.0, the solute is

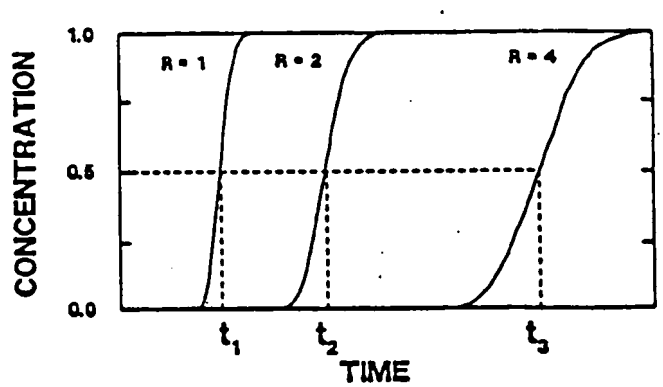


Figure 3-5. Time Required for Movement of Contaminants at Different Retardation Factors (Palmer and Johnson, 1989a)

nonreactive and moves with the ground water. If R is greater than 1.0, the average velocity of the solute is less than the velocity of the ground water, and the dispersion of the solute is reduced. If a monitoring well is located a distance from a contaminant source such that a nonreactive solute requires time, t1, to travel from the source to the well, a contaminant with a retardation factor of 2 will require 2t1 to reach the well, and 4t1 will be required for a contaminant with a retardation factor of 4.

Contaminants with lower retardation factors are transported greater distances over a given time than contaminants with larger retardation factors (Figure 3-6) (Palmer and Johnson, 1989a). A monitoring well network has a greater chance of detecting contaminants with lower retardation factors because they are found in a greater volume of the aquifer. Estimates of the total mass of a contaminant with a retardation factor of 1.0 in an aquifer may be more accurate than estimates for contaminants with greater amounts of retardation. Therefore, estimates of time required to remove nonreactive contaminants may be more accurate than time estimates for retarded contaminants. The slow movement of retarded contaminants may control the time and costs required to remediate a contaminated aquifer.

Transport through Fractured Media

Because fractured rock has both primary and secondary porosity, models used to describe solute transport in porous media, such as aquifers in recent alluvial deposits or glacial sediments, may not be appropriate for use at sites on fractured rock (Palmer and Johnson, 1989a). Primary porosity is the pore space formed at the time of deposition and formation of the rock mass, and secondary porosity is the pore space formed as the result of fracture of the rock.

Transport mechanisms in fractured media are advection and dispersion, the same as in porous media (Figure 3-7) (Palmer and Johnson, 1989a). In fractured media, however, contaminants are transported by advection only along fractures. Dispersion in fractured media is due to: (1) mixing at fracture intersections, (2) variations in opening widths across the width of the fracture, (3) variations in opening widths along stream lines, (4) molecular diffusion into microfractures penetrating the interfracture blocks and (5) molecular diffusion into interfracture porous matrix blocks (more important in fractured porous rock than in fractured crystalline rock).

Transport of contaminants through fractured media is described by one of four general models: continuum,

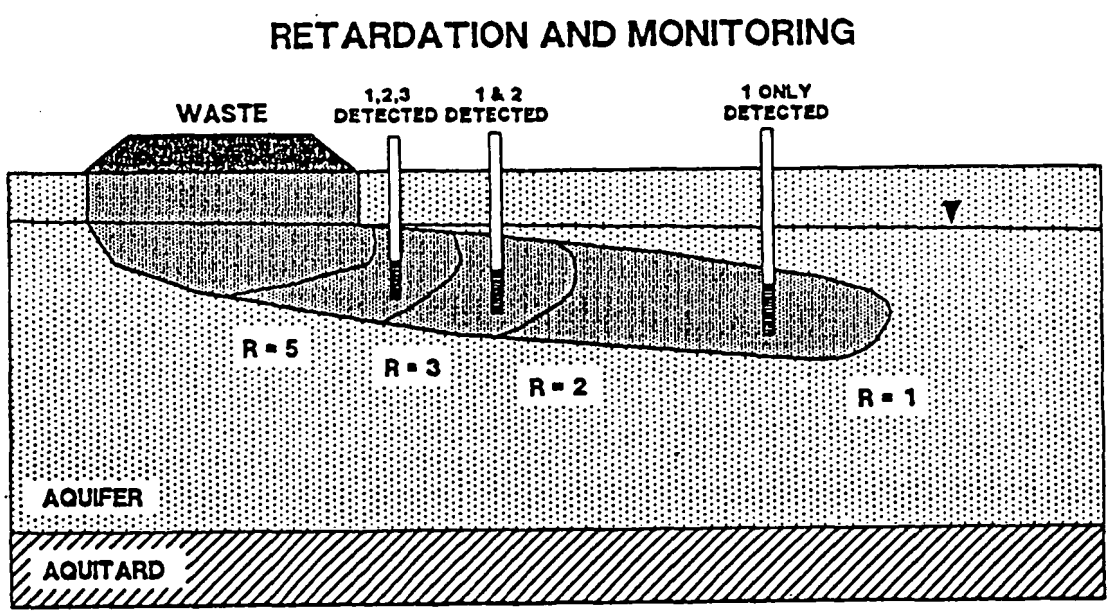


Figure 3-6. Transport of Contaminants with Varying Retardation Factors at a Waste Site (Palmer and Johnson, 1989a)

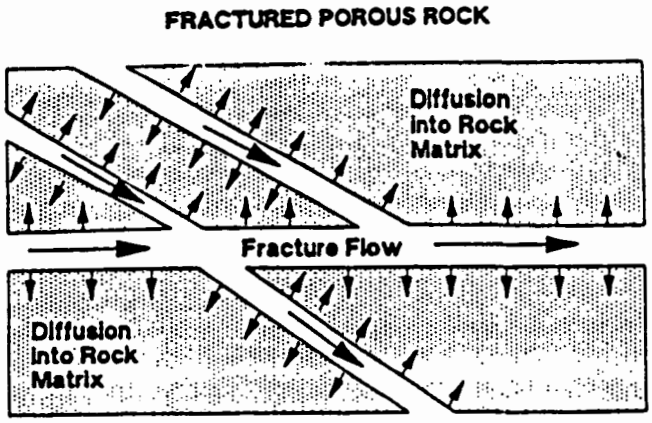


Figure 3-7. Transport In Fractured Porous Rock (Palmer and Johnson, 1989a)

discrete fracture, hybrid, and channel (Palmer and Johnson, 1989a).

In continuum models, individual fractures are ignored and the entire medium is considered to act as an equivalent porous medium. Single porosity continuum models are applicable where the only porosity of the rock mass is the fracture porosity, such as in fractured granite or basalt. Double porosity models are applicable to media in which there is both primary and secondary porosity such as sandstones and shales.

Discrete fracture models try to describe flow and transport in individual fractures.Becqause it can be difficult to obtain information about each fracture in the rock mass, stochastic models usually are required. These models use statistical information about distribution of fracture properties such as orientation and aperture widths to describe flow and transport.

Hybrid models are combinations of discrete fracture and continuum models, while channel models describe solute transport as small fingers or channels rather than as a uniform front along the width of a fracture.

Particle Transport through Porous Media

In addition to solute transport through porous media, the transport of particles (including bacteria, viruses, inorganic precipitates, natural organic matter, asbestos fibers, or clays) also may be important in investigations of contaminant transport. Particles can be removed from solution by surface filtration, straining, and physical-chemical processes (Figure 3-8) (Palmer and Johnson, 1989a).

The effectiveness of each process is dependent on the size of the specific particles present (Palmer and Johnson, 1989a). If particles are larger than the largest

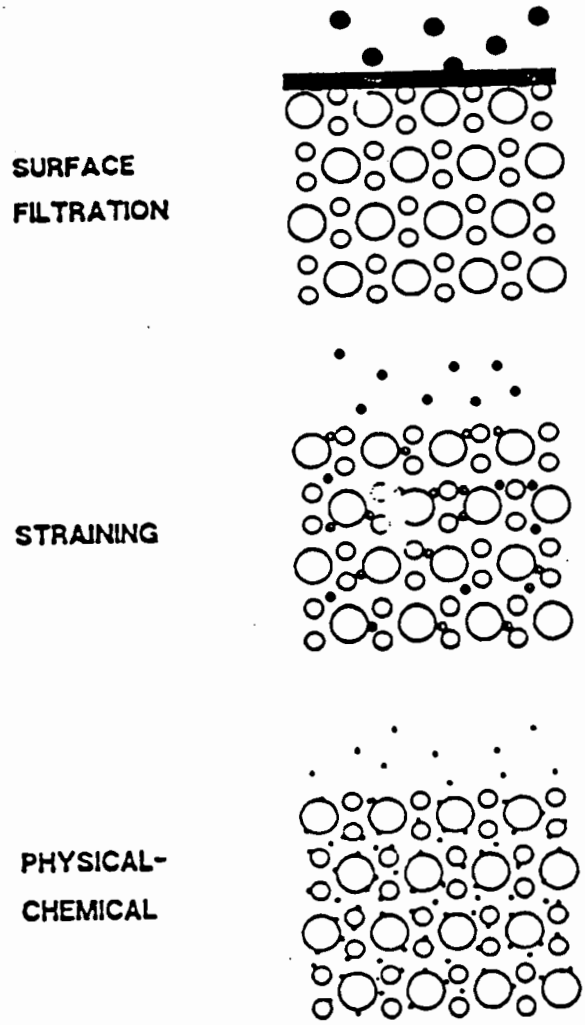


Figure 3-8. Mechanisms of Filtration (Palmer and Johnson, 1989a)

pore diameters, they cannot penetrate into the porous medium and are filtered at the surface of the medium. If particles are smaller than the largest pores but larger than the smallest, the particles are transported through the larger pore channels, but eventually encounter a pore channel with a smaller diameter and are removed by straining. If particles are smaller than the smallest pore openings, the particles can be transported long distances through the porous medium.

The rate at which particles move through the porous medium depends on several physical-chemical processes (Palmer and Johnson, 1989a). Particles may undergo random collisions with sand grains, and in a percentage of those collisions particles will adhere to the solid matrix by interception. Chemical conditions may affect particle transport; e.g., such processes as aggregation formation due to pH changes may change particle surface properties. These larger aggregates

can then be strained or filtered from the water.

Microorganism movement through geologic materials is limited by many processes (Palmer and Johnson, 1989a). Some bacteria are large enough to be strained from the water. Although viruses, which are smaller than bacteria, can pass through the pores, they may adsorb to geologic materials because their surfaces are charged. Microorganisms, like chemical constituents, can be transported by diffusion, or if they are motile, can move in response to changes in environmental conditions and chemical concentrations. Since microorganism live and die, the rates of these processes should be included in the description of their transport in the subsurface.

Physical Processes Controlling the Transport of Non-Aqueous Phase Liquids (NAPLs) in the Subsurface

Transport and Dissolution of NAPLs

Non-aqueous phase liquids (NAPLs) are those liquids that do not readily dissolve in water and can exist as a separate fluid phase. (Palmer and Johnson, 1989b). NAPLs are divided into two classes: those that are lighter than water (LNAPLs) and those with a density greater than water (DNAPLs). LNAPLs include hydrocarbon fuels, such as gasoline, heating oil, kerosene, jet fuel, and aviation gas. DNAPLs include the chlorinated hydrocarbons, such as 1,1,1-trichloroethane, carbon tetrachloride, chlorophenols, chlorobenzenes, tetrachloroethylene, and polychlorinated biphenyls (PCBs).

As NAPLs move through geologic media, they displace water and air (Palmer and Johnson, 1989b). Water is the wetting phase relative to both air and NAPLs and tends to line edges of pores and cover sand grains. NAPLs are the non-wetting phase and tend to move through the center of pore spaces. Neither the water nor the NAPL phase occupies the entire pore, so the permeability of the medium with respect to these fluids is different than when the pore space is entirely occupied by a single phase. This reduction in permeability depends upon the specific medium and can be described in terms of relative permeability, i.e., permeability at a certain fraction of pore space occupied by the NAPL compared to the permeability of the medium at saturation with the NAPL. Relative permeability ranges from 1.0 at 100 percent saturation to 0.0 at 0 percent saturation.

Figure 3-9 shows permeability of a NAPL in a hypothetical medium during multiphase flow. (Palmer and Johnson, 1989b). At 100 percent water saturation, the relative permeabilities of the water and NAPL are 1.0 and 0.0, respectively. As the fraction of the pore space occupied by NAPL increases, a corresponding decrease occurs

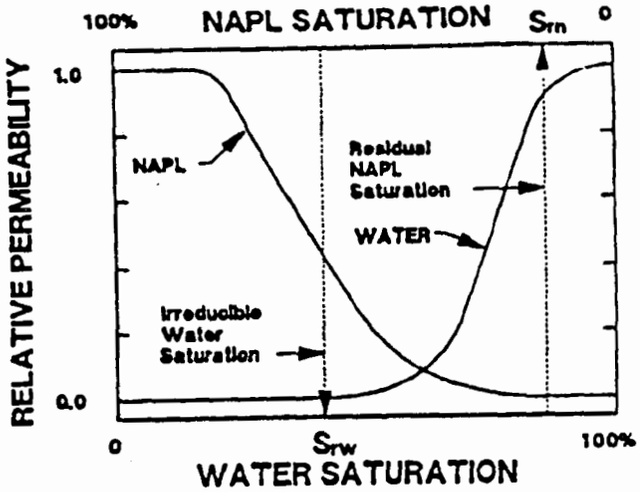


Figure 3-9. Relative Permeability as a Function of Saturation (Palmer and Johnson, 1989b)

in the fraction of water within the pore space. As the water fraction decreases, the relative permeability with respect to the water phase decreases to zero. Zero relative permeability is not obtained when the fraction of water within the pore space equals zero, but at the irreducible water saturation (S_{rw}), i.e., the level of water saturation at which the water phase is effectively immobile and there is no significant flow of water. The relative permeability of NAPL is similar. At 100 percent NAPL saturation, the relative permeability for the NAPL is equal to 1.0, but as the NAPL saturation decreases, the relative permeability of the NAPL decreases. At the residual NAPL saturation (S_{rn}), the relative permeability for the NAPL is effectively zero, and the NAPL is considered immobile. These immobile fractions of NAPL cannot be easily removed from pores except by dissolution by flowing water.

Transport of Light NAPLs

If small volumes of a spilled LNAPL enter the unsaturated zone (i.e., vadose zone), the LNAPL will flow through the central portion of the unsaturated pores until residual saturation is reached (Figure 3-10a) (Palmer and Johnson, 1989b). A three-phase system consisting of water, LNAPL, and air is formed within the vadose zone. Infiltrating water dissolves the components within the LNAPL (e.g., benzene, xylene, and toluene) and transports them to the water table. These dissolved contaminants form a contaminated plume radiating from the area of the residual product. Many components found in LNAPLs are volatile and can partition into soil air and be transported by molecular diffusion to other parts of the aquifer. As these vapors diffuse into adjoining soil areas, they may partition back into the water phase and transfer contamination over wider areas. If the soil

surface is relatively impermeable, vapors will not diffuse across the surface boundary and concentrations of contaminants in the soil atmosphere may build up to equilibrium conditions. However, if the surface is not covered with an impermeable material, vapors may diffuse into the atmosphere.

If large volumes of LNAPL are spilled (Figure 3-10b), the LNAPL flows through the pore space to the top of the capillary fringe of the water table. Dissolved components of the LNAPL precede the less soluble components and may change the wetting properties of the water, causing a reduction in the residual water content and a decrease in the height of the capillary fringe.

Since LNAPLs are lighter than water, they will float on top of the capillary fringe. As the head formed by the infiltrating LNAPLs increases, the water table is depressed and the LNAPLs accumulate in the depression. If the source of the spilled LNAPLs is removed or contained, LNAPLs within the vadose zone continue to flow under the force of gravity until reaching residual saturation. As the LNAPLs continue to enter the water table depression, they spread laterally on top of the capillary fringe (Figure 3-10c). The draining of the upper portions of the vadose zone reduces the total head at the interface between the LNAPLs and the ground water, causing the water table to rebound slightly. The rebounding water displaces only a portion of the LNAPLs because the LNAPLs remain at residual saturation. Ground water passing through the area of residual saturation dissolves constituents of the residual LNAPLs, forming a contaminant plume. Water infiltrating from the surface also can dissolve the residual LNAPLs and add to the contaminant load of the aquifer.

Decrease in the water table level from seasonal variations or ground-water pumping also causes dropping of the pool of LNAPLs. If the water table rises again, part of the LNAPLs may be pushed up, but a portion remains at residual saturation below the new water table. Variations in the water table height, therefore, can spread LNAPLs over a greater thickness of the aquifer, causing larger volumes of aquifer materials to be contaminated. Selection of a remedial technology for LNAPLs in the ground water should not include techniques that move LNAPLs into uncontaminated areas where more LNAPLs can be held at residual saturation.

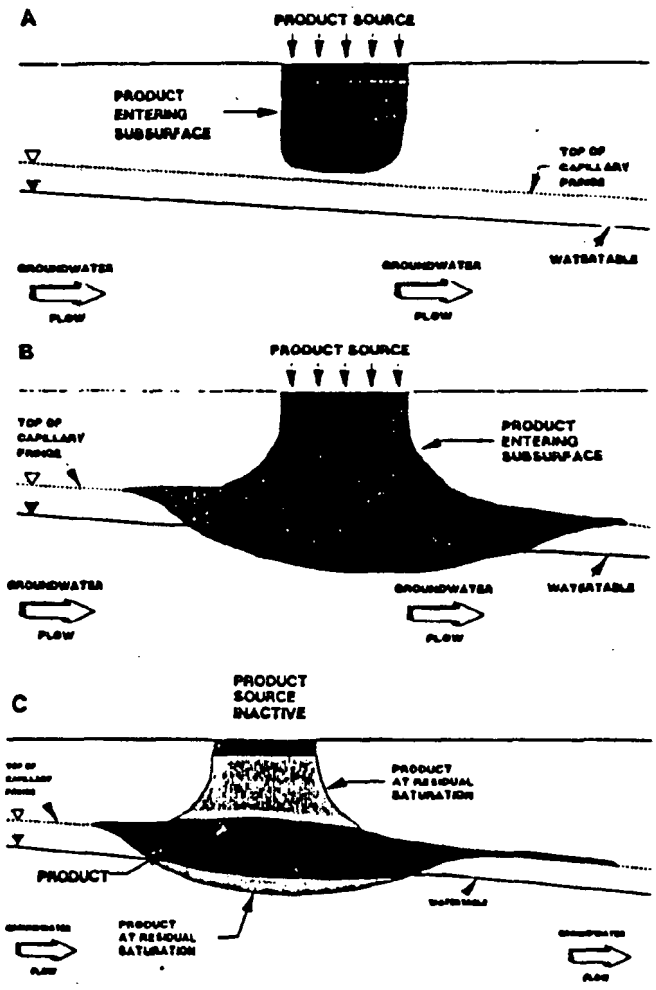


Figure 3-10. Movement of LNAPLs into the Subsurface: (A) Distribution of LNAPLs after Small Volume has Been Spilled; (B) Depression of the Capillary Fringe and Water Table; (C) Rebounding of the Water Table as LNAPLs Drain From Overlying Pore Space (Palmer and Johnson, 1989b)

Transport of Dense NAPLs

DNAPLs are very mobile in the subsurface because of their relatively low solubility, high density, and low viscosity (Palmer and Johnson, 1989b). The low solubility means that DNAPLs do not readily mix with water and remain as separate phases. Their high density provides a driving force that can carry them deep into aquifers. The combination of high density and low viscosity results in the displacement of the lower density, higher viscosity fluid, i.e., water, by DNAPLs, causing "unstable" flow and viscous fingering (Saffman and Taylor, 1958; Chouke and others, 1959; Homsy, 1987; Kueper and Frind, 1988).

If a small amount of DNAPL is spilled (Figure 3-11a), the DNAPL will flow through the unsaturated zone under the influence of gravity toward the water table, flowing until reaching residual saturation in the unsaturated zone (Palmer and Johnson, 1989b). If water is present in the vadose zone, viscous fingering

of the DNAPLs will be observed during infiltration. No viscous fingering will be exhibited if the unsaturated zone is dry. The DNAPLs can partition into the vapor phase, with the dense vapors sinking to the capillary fringe. Residual DNAPLs or vapors can be dissolved by infiltrating water and be transported to the water table, resulting in a contaminant plume within the aquifer.

If a greater amount of DNAPL is spilled (Figure 3-11b), the DNAPLs flow until they reach the capillary fringe and begin to penetrate the aquifer. To move through the capillary fringe, the DNAPLs must overcome the capillary forces between the water and the medium. A critical height of DNAPLs is required to overcome these forces. Larger critical heights are required for DNAPLs to move through unfractured, saturated clays and silts; thus these types of materials may be effective barriers to the movement of DNAPLs if the critical heights are not exceeded.

After penetrating the aquifer, DNAPLs continue to move through the saturated zone until they reach residual saturation. DNAPLs are then dissolved by ground water passing through the contaminated area, resulting in a contaminant plume that can extend over a large thickness of the aquifer. If finer-grained strata are contained within the aquifer, infiltrating DNAPLs accumulate on top of the strata, creating a pool. At the interface between the ground water and the DNAPL pool, the solvent dissolves into the water and spreads vertically by molecular diffusion. As water flows by the DNAPL pool, the concentration of the contaminants in the ground water increases until saturation is achieved or the downgradient edge of the pool is reached. DNAPLs, therefore, often exist in fingers or pools in the subsurface, rather than in continuous distributions. The density of pools and fingers of DNAPLs within an aquifer are important for controlling the concentrations of dissolved contaminants originating from DNAPLs.

If even larger amounts of DNAPLs are spilled (Figure 3-11c), DNAPLs can penetrate to the bottom of the aquifer, forming pools in depressions. If the impermeable lower boundary is sloping, DNAPLs flow down the dip of the boundary. This direction can be upgradient from the original spill area if the impermeable boundary slopes in that direction. DNAPLs also can flow along bedrock troughs, which may be oriented differently from the direction of ground-water flow. Flow along impermeable boundaries can spread contamination in directions that would not be predicted based on hydraulics.

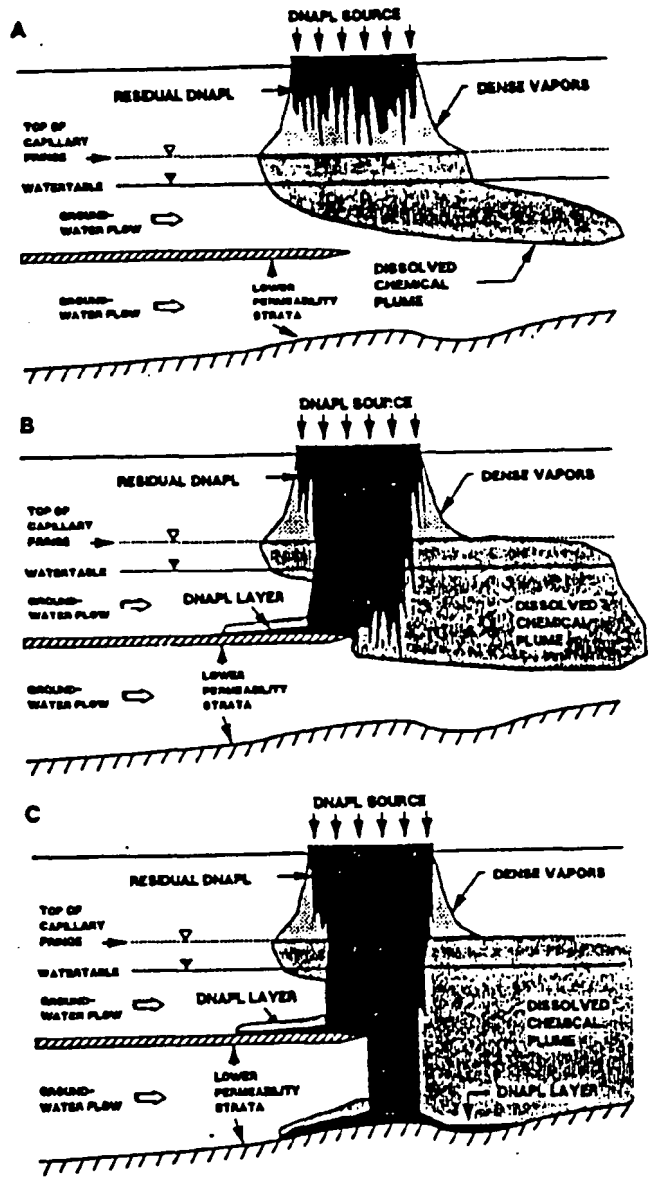


Figure 3-11. Movement of DNAPLs into the Subsurface (A) Distribution of DNAPLs after Small Volume has Been Spilled; (B) Distribution of DNAPLs after Moderate Volume has Been Spilled; (C) Distribution of DNAPLs after Large Volume has Been Spilled (after Feenstra and Cherry, 1988, by Palmer and Johnson, 1989b)

Chemical Processes Controlling the Transport of Contaminants in the Subsurface

Introduction

Subsurface transport of contaminants often is controlled by complex interactions between physical, chemical, and biological processes. The advection-dispersion equation used to quantitatively describe and predict contaminant movement in the subsurface also must contain reaction terms added to the basic equation to account for chemical and biological processes important in controlling contaminant transport and fate (Johnson and others, 1989).

Chemical Reactions of Organic Compounds
Chemical reactions may transform one compound into another, change the state of the compound, or cause a compound to combine with other organic or inorganic chemicals (Johnson and others, 1989). For use in the advection-dispersion equation, these reactions represent changes in the distribution of mass within the specified volume through which the movement of the chemicals is modeled.

Chemical reactions in the subsurface often are characterized kinetically as equilibrium, zero, or first order, depending on how the rate is affected by the concentrations of the reactants. A zero-order reaction is one that proceeds at a rate independent of the concentration of the reactant(s). In a first-order process, the rate of the reactions is directly dependent on the concentration of one of the reactants. The use of zero or first-order rate expressions may oversimplify the description of a process, but higher order expressions, which may be more realistic, are often difficult to measure and/or model in complex environmental systems. Also first-order reactions are easy to incorporate into transport models (Johnson and others, 1989).

Sorption. Sorption is probably the most important chemical process affecting the transport of organic contaminants in the subsurface environment. Sorption of non-polar organics is usually considered an equilibrium-partitioning process between the aqueous phase and the porous medium (Chiou and others, 1979). When solute concentrations are low (i.e., either $\leq 10^{-5}$ Molar, or less than half the solubility, whichever is lower), partitioning often is described using a linear Freundlich isotherm, where the sorbed concentration is a function of the aqueous concentration and the partition coefficient (K_p) (Karickhoff and others, 1979; Karickhoff, 1984). K_p usually is measured in laboratory batch equilibrium tests, and the data are plotted as the concentration in the aqueous phase versus the amount sorbed onto the solid phase (Figure 3-12) (Chiou and others, 1979).

Under conditions of linear equilibrium partitioning, the sorption process is represented in the advection-dispersion equation as a "retardation factor," R (Johnson and others, 1989). The retardation factor is dependent on the partition coefficient K_p, bulk density of aquifer materials, and porosity.

The primary mechanism of organic sorption is the formation of hydrophobic bonding between a contaminant and the natural organic matter associated with aquifers (Tanford, 1973; Karickhoff and others, 1979; Karickhoff, 1984; Chiou and others, 1985; MacKay and Powers, 1987). Therefore, the extent of sorption of

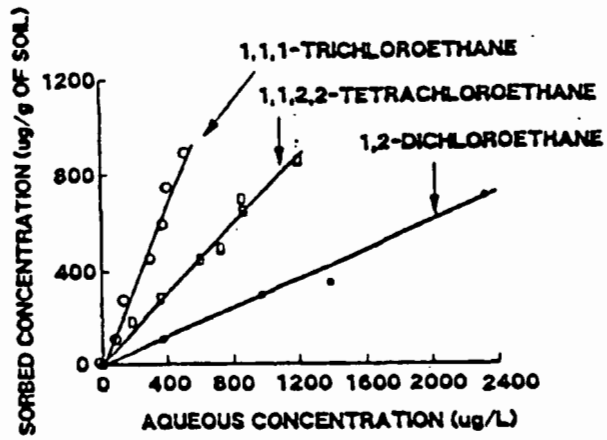


Figure 3-12. Batch Equilibrium Data for 1,1,1-TCA, 1,1,2,2,-TeCA and 1,2-DCA (adapted from Chiou and others, 1979, by Johnson and others, 1989)

a specific chemical can be estimated from the organic carbon content of the aquifer materials (f_{oc}) and a proportionality constant characteristic of the chemical (K_{oc}), if the organic content is sufficiently high (i.e., fraction organic carbon content (f_{oc}) > 0.001) (Karickhoff and others, 1979; Karickhoff, 1984). K_{oc} values for many compounds are not known, so correlation equations relating K_{oc} to more easily available chemical properties, such as solubility or octanol-water partition coefficients (Kenaga and Goring, 1980; Karickhoff, 1981; Schwarzenbach and Westall, 1981; Chiou and others, 1982, 1983), have been developed. Within a compound class, K_{oc} values derived from correlation expressions often can provide reasonable estimates of sorption. However, if correlations were developed covering a broad range of compounds, errors associated with the use of Koc estimates can be large (Johnson and others, 1989).

This method of estimation of sorption, using K_{oc} and f_{oc} values, is less expensive than the use of batch equilibrium tests. However, in soils with lower carbon content, sorption of neutral organic compounds onto the mineral phase can cause significant errors in the estimate of the partition coefficient (Chiou and others, 1985).

Hydrolysis. Hydrolysis, an important abiotic degradation process in ground water for certain classes of compounds, is the direct reaction of dissolved compounds with water molecules (Mabey and Mill, 1978). Hydrolysis of chlorinated compounds, which are often resistant to biodegradation (Siegrist and McCarty, 1987), forms an alcohol or alkene (Figure 3-13).

Most information concerning rates hydrolysis is obtained from laboratory studies, since competing reactions and

$$RX + HOH \longrightarrow ROH + HX$$

$$\overset{H}{\underset{|}{C}}-\overset{X}{\underset{|}{C}} \xrightarrow[OH^-]{H^+ \text{ or}} C=C + HX$$

Figure 3-13. Schematic of Hydrolysis Reactions for Halogenated Organic Compounds (Johnson and others, 1989)

slow degradation rates make hydrolysis difficult to measure in the field. (Johnson and others, 1989). Often data for hydrolysis are fitted as a first-order reaction, and a hydrolysis rate constant, K, is obtained. The rate constant multiplied by the concentration of the contaminant is added to the advection-dispersion equation to account for hydrolysis of the contaminant.

Cosolvation and Ionization. Cosolvation and ionization are processes that may decrease sorption and thereby increase transport velocity (Johnson and others, 1989). The presence of cosolvents decreases entropic forces that favor sorption of hydrophobic organic contaminants by increasing interactions between the solute and the solvent (Nkedi-Kizza and others, 1985; Zachara and others, 1988). If biologically derived or anthropogenic solvent compounds are present at levels of 20 percent or more by volume, the solubility of hydrophobic organic contaminants can be increased by an order of magnitude or more (Nkedi-Kizza and others, 1985). In Figure 3-14, decrease in sorption of anthracene in three soils, as described by the sorption coefficient K_p, is illustrated, with methanol as the cosolvent. Since cosolvent concentration must be large for solute velocity to be increased substantially, cosolvation is important primarily near sources of ground-water contamination.

In the process of ionization, acidic compounds, such as phenols or organic acids, can lose a proton in solution to form anions that, because of their charge, tend to be water-soluble (Zachara and others, 1986). For example, the K_{oc} of 2,4,5-trichlorophenol can decrease from 2,330 for the phenol, to almost zero for the phenolate (Figures 3-15 and 3-16) (Johnson and others, 1989). Acidic compounds tend to ionize more as the pH increases. However, for many compounds, such as the chlorophenols, substantial ionization can occur at neutral pH values.

Volatilization and Dissolution. Two important pathways for the movement of volatile organic compounds in the subsurface are volatilization into the unsaturated zone and dissolution into the ground water

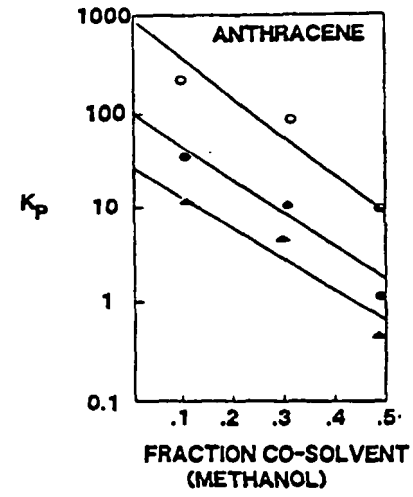


Figure 3-14. Effect of Methanol as a Cosolvent on Anthracene Sorption for Three Soils (Adapted from Nkedi-Kizza and others, 1985, by Johnson and others, 1989)

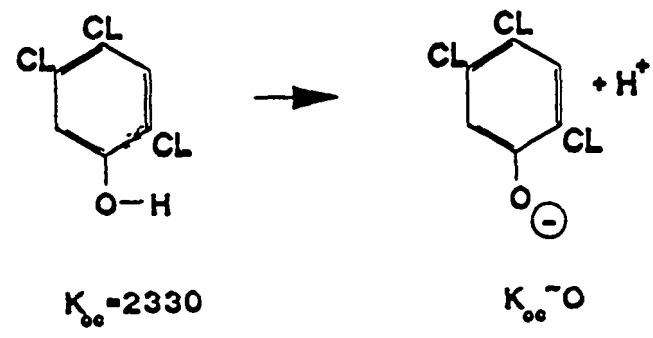


Figure 3-15. K_{oc} values for 2,4,5-trichlorophenol and 2,4,5-trichlorophenolate (Johnson and others, 1989)

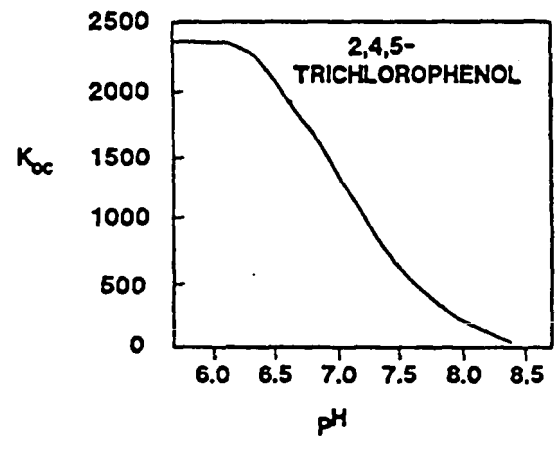


Figure 3-16. K_{oc} versus pH for 2,4,5-trichlorophenol (Johnson and others, 1989)

(Johnson and others, 1989). Contaminants in the aqueous and vapor phases are also more amenable to degradation.

The degree of volatilization of a contaminant is determined by: (1) the area of contact between the contaminated area and the unsaturated zone, which is affected by the nature of the medium (e.g., grain size, depth to water, water content) and the contaminant (e.g., surface tension and liquid density); (2) the vapor pressures of the contaminants; and (3) the rate at which the compound diffuses in the subsurface (Johnson and others, 1989).

The residual saturation remaining when immiscible liquids move downward through unsaturated porous media provides a large surface area for volatilization (Johnson and others, 1989). Vapor concentrations in the vicinity of the residual are often at saturation concentrations. Movement of vapor away from the residual saturation is usually controlled by molecular diffusion, which is affected by the tortuosity of the path through which the vapors move. Tortuosity also is affected by the air-filled porosity of the medium, so diffusion is reduced in porous media with a high water content.

Diffusion also is reduced by the partitioning of the vapors out of the gas phase and into the solid or aqueous phases (Johnson and others, 1989). The retardation factor developed for partitioning between the aqueous and solid phases can be modified with a term to describe partitioning between the vapor and aqueous phases.

When immiscible fluids reach the capillary fringe, their further movement is determined by the density of the fluids relative to water (Scheigg, 1984; Schwille, 1988). The LNAPLs pool on top of the water table while the DNAPLs penetrate into the ground water. Floating pools of LNAPL can provide substantial surface area for volatilization, with diffusion controlling the mass transfer of organic contaminants into the vapor phase.

The transport and fate of DNAPLs that penetrate into the ground water is controlled by dissolution. Experiments have shown that saturation concentration values can be maintained even with high ground-water velocities (e.g., 1 m/day) through a zone of contamination (Anderson and others, 1987). During remedial activities, such as pump-and-treat, ground-water velocities may be high, but the dissolution process should still be effective.

Chemical Reactions of Inorganic Compounds

In studies of organic contamination, the most important chracteristic is the total concentration of a contaminant in a certain phase (e.g., in water versus aquifer solid materials). However, studies of inorganic contamination are often more difficult because inorganic materials can occur in many chemical forms, and knowledge of these forms (i.e, species) is required to predict their behavior in ground water (Morel, 1983; Sposito, 1986).

In ground water, an inorganic contaminant may occur as: (1) "free ions" (i.e., surrounded only by water molecules); (2) insoluble species; (3) metal/ligand complexes; (4) adsorbed species; (5) species held on a surface by ion exchange; or (6) species differing by oxidation state (e.g., manganese (II) and (IV) or chromium (III) and (VI)) (Johnson and others, 1989).

The total concentration of an inorganic compound may not provide sufficient information to describe the fate and behavior of that compound in ground water. Mobility, reactivity, biological availability, and toxicity of metals and other inorganic compounds depend upon their speciation (Johnson and others, 1989). The primary reactions affecting the speciation of inorganic compounds are solubility and dissolution, complexation reactions, adsorption and surface chemistry, ion exchange, and redox chemistry.

Solubility, Dissolution, and Precipitation. Dissolution and weathering of minerals determine the natural composition of ground water (Johnson and others, 1989). Dissolution is the dissolving of all components within a mineral, while weathering is a partial dissolution process in which certain elements leach out of a mineral, leaving others behind.

Mineral dissolution is the source of most inorganic ions in ground water. In principle a mineral can dissolve up to the limits of its solubility, but in many cases, reactions occur at such a slow rate that true equilibrium is never attained (Morgan, 1967).

The contribution of ions from one mineral may affect the solubility of other minerals containing the same ion (i.e., the "common ion effect"). Computer programs such as MINTEQ (Felmy and others, 1984), MINEQL (Westall and others, 1976), and WATEQ2 (Ball and others, 1980) may be used to predict the equilibrium distribution of chemical species in ground water and indicate if the water is undersaturated, supersaturated, or at equilibrium with various mineral phases. Some of these programs also may be used to predict the ionic composition of ground water in equilibrium with assumed mineral phases (Jennings and others, 1982).

The weathering of silicate minerals contributes cations, such as calcium, magnesium, sodium, potassium, and

silica, to water and forms secondary weathering products such as kaolinite and montmorillonite clays (Johnson and others, 1989). This weathering increases the alkalinity of ground water to a level greater than its rainwater origins.

Weathering and dissolution also can be a source of contaminants. Leachates from mine tailings can yield arsenate, toxic metals, and strong mineral acids (Hem, 1970), while leachates from fly-ash piles can contribute selenium, arsenate, lithium, and toxic metals (Stumm and Morgan, 1981; Honeyman and others, 1982; Murarka and MacIntosh, 1987).

The opposite of dissolution reactions is precipitation of minerals or contaminants from an aqueous solution (Johnson and others, 1989). During precipitation, the least-soluble mineral at a given pH level is removed from solution. An element is removed by precipitation when its solution concentration saturates the solubility of one of its solid compounds. If the solution concentration later drops below the solubility limit, the solid will begin to dissolve until the solubility level is attained again. Contaminants may initially precipitate, then slowly dissolve later after a remedial effort has reduced the solution concentration; thus complete remediation of the aquifer may require years.

A contaminant initially may be soluble but later precipitate after mixing with other waters or after contact with other minerals (Drever, 1982; Williams, 1985; Palmer, 1989). For example, pumping water from an aquifer may mobilize lead until it converges and mixes with waters high in carbonates from a different formation and precipitates as a lead carbonate solid.

Complexation Reactions. In complexation reactions, a metal ion reacts with an anion that functions as a ligand (Johnson and others, 1989). The metal and the ligand bind together to form a new soluble species called a complex. Transition metals form the strongest complexes (Stumm and Morgan, 1981); alkaline earth metals form only weak complexes, while alkali metals do not form complexes (Dempsey and O'Melia, 1983). The approximate order of complexing strength of metals is:

Fe(III)> Hg> Cu> Pb> Ni> Zn> Cd> Fe(II)> Mn> Ca> Mg

Common inorganic ligands that bind with metals include: OH^-, Cl^-, $SO_4^=$, $CO_3^=$, $S^=$, F^-, NH_3, PO_4, CN^-, and polyphosphates. Their binding strength depends primarily on the metal ion with which they are complexing (Johnson and others, 1989). Inorganic ligands are usually in excess compared to the "trace" metals with which they bind, and, therefore, they affect the fate of the metals in the environmental system, rather than vice versa (Morel, 1983).

Organic ligands generally form stronger complexes with metals than inorganic ligands (Johnson and others, 1989). Organic ligands include: (1) synthetic compounds from wastes, such as amines, pyridines, phenols, and other organic bases and weak acids; and (2) natural organic materials, primarily humic materials (Schnitzer, 1969; Hayes and Swift, 1978; Stevenson, 1982, 1985; Johnson and others, 1989). Humic materials are complex structures, and their complexation behavior is difficult to predict (Perdue and Lytle, 1983; Sposito, 1984; Perdue, 1985; Dzombak and others, 1986; Fish and others, 1986). Generally, humic materials are found in significant concentrations only in shallow aquifers. In these aquifers, however, they may be the primary influence on the behavior of metals (Thurman, 1985).

Equilibrium among reactants and complexes for a given reaction is predicted by an equilibrium (or "stability") constant, K, which defines a mass-law relationship among the species (Johnson and others, 1989). For given total ion concentrations (measured analytically), stability constants can be used to predict the concentration of all possible species (Martell and Smith, 1974, 1977; Smith and Martell, 1975).

Because complexes decrease the amount of free ions in solution, less metal may sorb onto aquifer solid materials or participate in precipitation reactions (Johnson and others, 1989). The metal is more soluble because it is primarily bound up in the soluble complex. Research has demonstrated that a metal undergoing complexation may be less toxic to aquifer microorganisms (Reuter and others, 1979).

Sorption and Surface Chemistry. Surface sorption, in many cases, is the most important process affecting toxic metal transport in the subsurface (Johnson and others, 1989). Changes in metal concentration, as well as pH, can have a significant effect on the extent of sorption (Figure 3-17).

Approaches to predicting behavior of metal ions based on sorption processes include using isotherms (indicating that data were collected at a fixed temperature) to graphically and mathematically represent sorption data (Johnson and others, 1989). Two types of isotherms are commonly used: the Freundlich isotherm and the Langmuir isotherm (Figure 3-18). The Freundlich isotherm is empirical, and sorbed (S) and aqueous (C) concentration data are fitted by adjusting two parameters (K and a). The Langmuir

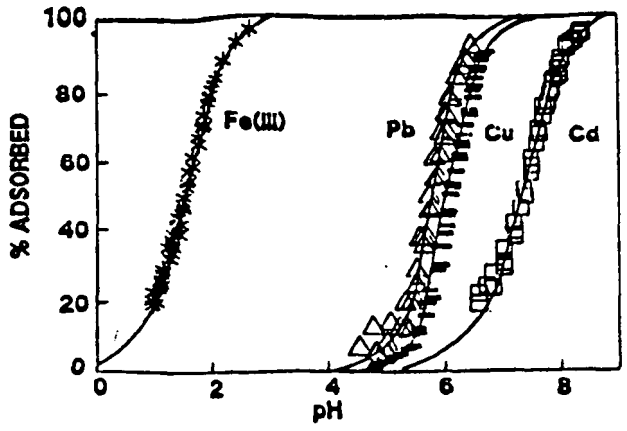


Figure 3-17. Adsorption of Metal Ions on Amorphous Silica as a Function of pH (adapted from Schindler and others, 1976, by Johnson and others, 1989)

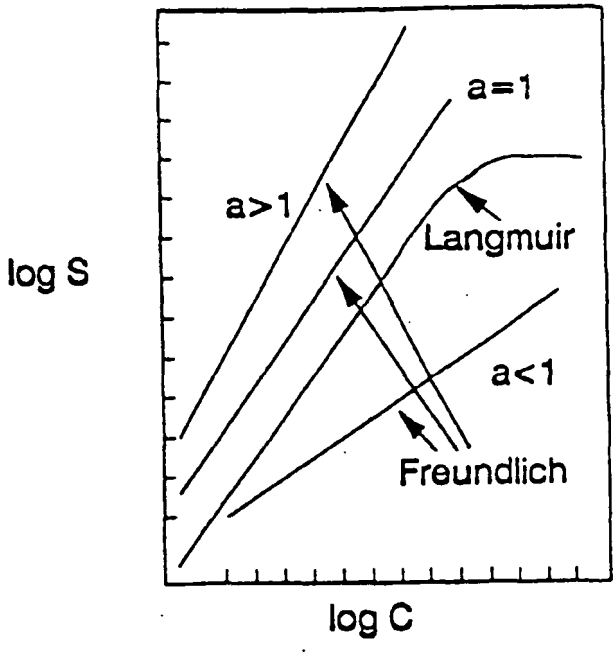


Figure 3-18. Schematic Representation of Freundlich and Langmuir Isotherm Shapes for Batch Equilibrium Tests (Johnson and others, 1989)

isotherm is based on the theory of surface complexation, using a parameter corresponding to the maximum amount that can be sorbed and the partition coefficient, K (Morel, 1983).

Another method to describe sorption is to use surface complexation models that represent sorption as ions binding to specific chemical functional groups on a reactive surface (Johnson and others, 1989). All surface sites may be identical or may be grouped into different classes of sites (Benjamin and Leckie, 1981). Each type of site has a set of specific sorbing constants, one for each sorbing compound. Electrostatic forces at the surface also contribute to the overall sorption constant (Davis and others, 1978). Binding of ions to the surface is calculated from constants using mass-law equations similar to those used to calculate complex formation (Schindler and others, 1976; Stumm and others, 1976; Dzombak and Morel, 1986). However, the parameters used in surface complexation models are data-fitting parameters, which fit a specified set of data to a particular model, but have no thermodynamic meaning and no generality beyond the calibrating data set (Westall and others, 1980).

Ion-Exchange Reactions. Ion-exchange reactions are similar to sorption. However, sorption is coordination bonding of metals (or anions) to specific surface sites and is considered to be two-dimensional, while an ion-exchanger is a three-dimensional, porous matrix containing fixed charges (Helfferich, 1962; Johnson and others, 1989). Ions are held by electrostatic forces rather than by coordination bonding. Ion-exchange "selectivity coefficients" are empirical and vary with the amount of ion present (Reichenburg, 1966). Ion exchange is used to describe the binding of alkali metals, alkaline earths, and some anions to clays and humic materials (Helfferich, 1962; Sposito, 1984). Knowledge of ion exchange is used to understand the behavior of major natural ions in aquifers and also is useful for understanding behavior of contaminant ions at low levels. In addition, ion exchange models are used to represent competition among metals for surface binding (Sposito, 1984).

Redox Chemistry. Reduction-oxidation (redox) reactions involve a change in the oxidation state of elements (Johnson and others, 1989). The amount of change is determined by the number of electrons transferred during the reaction (Stumm and Morgan, 1981). The oxidation status of an element can be important in determining the potential for transport of that element. For example, in slightly acidic to alkaline environments, Fe(III) precipitates as a highly sorptive phase (ferric hydroxide), while Fe(II) is soluble and does not retain other metals. The reduction of Fe(III) to Fe(II) releases not only Fe^{+2} to the water, but also other contaminants sorbed to the ferric hydroxide surfaces (Evans and others, 1983; Sholkovitz, 1985).

Chromium (Cr) (VI) is a toxic, relatively mobile anion, while Cr (III) is immobile, relatively insoluble, and strongly sorbs to surfaces. Selenate (Se) (VI) is mobile but less toxic, while selenite Se(IV) is more toxic but less mobile (Johnson and others, 1989).

The redox state of an aquifer is usually closely related to microbial activity and the type of substrates available to the microorganisms (Johnson and others, 1989). As organic contaminants are oxidized in an aquifer, oxygen is depleted and chemically reducing (anaerobic) conditions form. The redox reactions that occur depend on the dominant electron potential, which is defined by the primary redox-active species. The combination of Fe(II)/Fe(III) defines a narrow range of electron potentials, while (S)(sulfur)(+IV)/S(-II) defines a broader range. Pairs of chemical species are called redox couples.

After oxygen is depleted from ground water, the most easily reduced materials begin to react and, along with the reduced product, determine the dominant potential. After that material is reduced, the next most easily reduced material begins to react. These series of reactions continue, usually catalyzed by microorganisms. An aquifer may be described as "mildly reducing" or "strongly reducing," depending on where it is in the chemical series (Stumm and Morgan, 1981).

The electron potential of water may be measured in volts, as Eh, or expressed by the "pe," which is the negative logarithm of the electron activity in the water (Johnson and others, 1989). A set of redox reactions is often summarized on a pH-pe (or pH-Eh) diagram, which shows the predominant redox species at any specified pH and pe (or Eh). In this theoretical approach, only one redox couple should define the redox potential of the system at equilibrium. However, in an aquifer, many redox couples not in equilibrium can be observed simultaneously (Lindberg and Runnels, 1984). Therefore, redox behavior of chemicals in aquifers is difficult to predict. However, the redox status of an aquifer is important because of its effects on the mobility of elements and the potential effects on biodegradation of organic contaminants. Anaerobic (reducing) conditions are not favorable for hydrocarbon degradation, but reducing conditions favor dehalogenation of chlorinated and other halogenated compounds (Johnson and others, 1989).

Biological Processes Controlling the Transport of Contaminants in the Subsurface

Introduction

Historically, ground water was thought to be a safe water source because it was protected by a metabolically diverse "living filter" of microorganisms in the soil root zone that converted organic contaminants to innocuous end-products (Suflita, 1989a). Aquifers were considered to be abiotic environments, based on studies that showed that microbial numbers decreased with soil depth (Waksman, 1916) and that indicated that most microorganisms were attached to soil particles (Balkwill and others, 1977). In addition, by estimating the time required for surface water to vertically penetrate subsurface formations, researchers felt that microorganisms travelling with water would utilize available nutrients and rapidly die off. Therefore, since aquifers were considered to be sterile, they could not be biologically remediated if contaminated with organic contaminants. However, microscopic, cultivation, metabolic, and biochemical investigations, using aseptically obtained aquifer materials, have shown that there are high numbers of metabolically diverse procaryotic and eucaryotic organisms present in the terrestrial subsurface environment (Suflita, 1989a).

Evidence of Subsurface Microorganisms

Microbiological investigations have detected high numbers of microorganisms (up to 50 x 106 total cells/mL) in both contaminated and uncontaminated aquifers at various depths and geological composition (Suflita, 1989a). Even deep geological formations may be suitable habitats for microorganisms (Kuznetsov and others, 1963; Updegraff, 1982). The microorganisms that have been detected in the subsurface are small, capable of response to addition of nutrients, and are primarily attached to solid surfaces. Eucaryotic organisms are present in the subsurface but are few in numbers and are probably of minor significance, existing as inert resting structures (Suflita, 1989a).

Suitable sampling technology was developed to demonstrate the existence of subsurface microorganisms (Suflita, 1989a). Samples must not be contaminated with nonindigenous microorganisms originating from drilling machinery, surface soil layers, drilling muds, and water used to make up drilling muds. Since most subsurface microorganisms are associated with aquifer solid materials, current sampling efforts use core recovery and dissection to remove microbiologically contaminated portions of the cores (McNabb and Mallard, 1984). This dissection is performed in the field, to prevent nonindigenous organisms from penetrating to the inner portions of the core, or in the laboratory if it is nearby. The outer few centimeters and the top and bottom portions of the aquifer cores are removed because of possible contamination by nonindigenous bacteria, and the center portions of the cores are used for microbiological analysis. An alcohol-sterilized paring device is used in the dissection process. The paring device has an inner diameter that is smaller than the diameter of the core itself. As the aquifer material is extruded out of the sampling core barrel and over the paring device, the potentially contaminated material is stripped away. For anaerobic aquifers, this field paring dissection is performed inside plastic anaerobic glove bags while the latter is purged with nitrogen to minimize

exposure of the microorganisms to oxygen (Beeman and Suflita, 1987). Samples obtained by this technique are considered to be aseptically acquired and are suitable for microbiological analyses.

Evidence of Activity of Subsurface Microorganisms
Although direct and conclusive evidence had been obtained about the existence of microorganisms in the subsurface, questions remained about their significance in ground water. Such questions included: (1) whether or not the indigenous microorganisms were metabolically active, (2) what was the diversity of the metabolic activities, (3) what factors served to limit and/or stimulate the growth and metabolism of these organisms, and (4) could the inherent metabolic versatility of aquifer microorganisms be utilized to remediate contaminated aquifers (Suflita, 1989a).

Microbial subsurface activity was studied, and the following metabolic processes were identified in the subsurface environment: (1) biodegradation of organic pollutants, including petroleum hydrocarbons, alkylpyridines, creosote chemicals, coal gasification products, sewage effluent, halogenated organic compounds, nitriloacetate (NTA), and pesticides; (2) nitrification; (3) denitrification; (4) sulfur oxidation and reduction; (5) iron oxidation and reduction; (6) manganese oxidation; and (7) methanogenesis (Suflita, 1989a). These metabolic processes include aerobic and anaerobic carbon transformations, many of which are important in aquifer contaminant biodegradation. The other processes are those required for the cycling of nitrogen, sulfur, iron, and manganese in microbial communities.

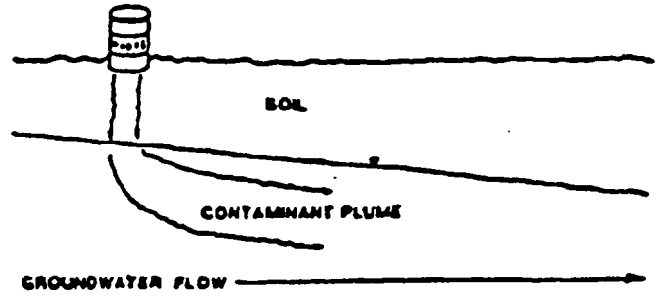
Biodegradation may refer to complete mineralization of organic contaminants (i.e., the parent compounds), to carbon dioxide, water, inorganic compounds, and cell protein (Sims and others, 1990). The ultimate products of aerobic metabolism are carbon dioxide and water, while under anaerobic conditions, metabolic activities also result in the formation of incompletely oxidized simple organic substances such as organic acids and other products such as methane or hydrogen gas.

Since contaminant biodegradation in the natural environment is frequently a stepwise process involving many enzymes and many species of organisms, a contaminant may not be completely degraded. Instead, it may be transformed to intermediate product(s) that may be less, equally, or more hazardous than the parent compound, and more or less mobile in the environment (Sims and others, 1990). The loss of a chemical, therefore, may or may not be a desirable consequence of the biodegradation process if biodegradation results in the production of undesirable metabolites with their own environmental impact and persistence characteristics (Suflita, 1989b). For example, the reductive removal of tetrachloroethylene (TeCE) under anaerobic conditions results in a series of dehalogenated intermediates. TeCE's halogens are removed and replaced by protons in a series of sequential steps. However, the rate of reductive dehalogenation decreases as fewer and fewer halogens remain. Consequently, highly toxic vinyl chloride accumulates and, from a regulatory standpoint, causes greater concern than the parent contaminant. Bioremedial technologies should be selected with knowledge of metabolic processes of the specific contaminants at the site.

Biodegradation of most organic compounds in aquifer systems may be evaluated by monitoring their disappearance from the aquifer through time. Disappearance, or rate of degradation, is often expressed as a function of the concentration of one or more of the contaminants being degraded (Sims and others, 1990). Biodegradation in natural systems often can be modeled as a first-order chemical reaction (Johnson and others, 1989). Both laboratory and field data suggest that this is true when none of the reactants are in limited supply. A useful term to describe reaction kinetics is the half-life, t1/2, which is the time required to transform 50 percent of the initial constituent.

As decomposable organic matter enters an oxygenated aquifer (Figure 3-19), microbial metabolism will likely begin to degrade the contaminating substrate; i.e., the indigenous microorganisms utilize the contaminant as an electron donor for heterotrophic microbial respiration (Suflita, 1989a). The aquifer microorganisms use oxygen as a co-substrate and as an electron acceptor to support their respiration. This oxygen demand may deplete oxygen and establish anaerobic conditions. When oxygen becomes limiting, aerobic respiration slows, and other microorganisms become active and continue to degrade the organic contaminants. Under conditions of anoxia, anaerobic bacteria use organic chemicals or certain inorganic anions as alternate electron acceptors.

Nitrate present in ground water is not rapidly depleted until oxygen is utilized. Organic matter is still metabolized, but, instead of oxygen, nitrate becomes the terminal electron acceptor during denitrification. Sulfate becomes a terminal electron acceptor when nitrate is limiting. When this occurs, hydrogen sulfide, an odorous gas, can often be detected in the ground water as a metabolic end-product. When very highly reducing conditions are present in an aquifer, carbon dioxide becomes an electron acceptor and methane is formed. Sometimes a spatial separation of dominant metabolic processes can occur in an aquifer, depending on the availability of



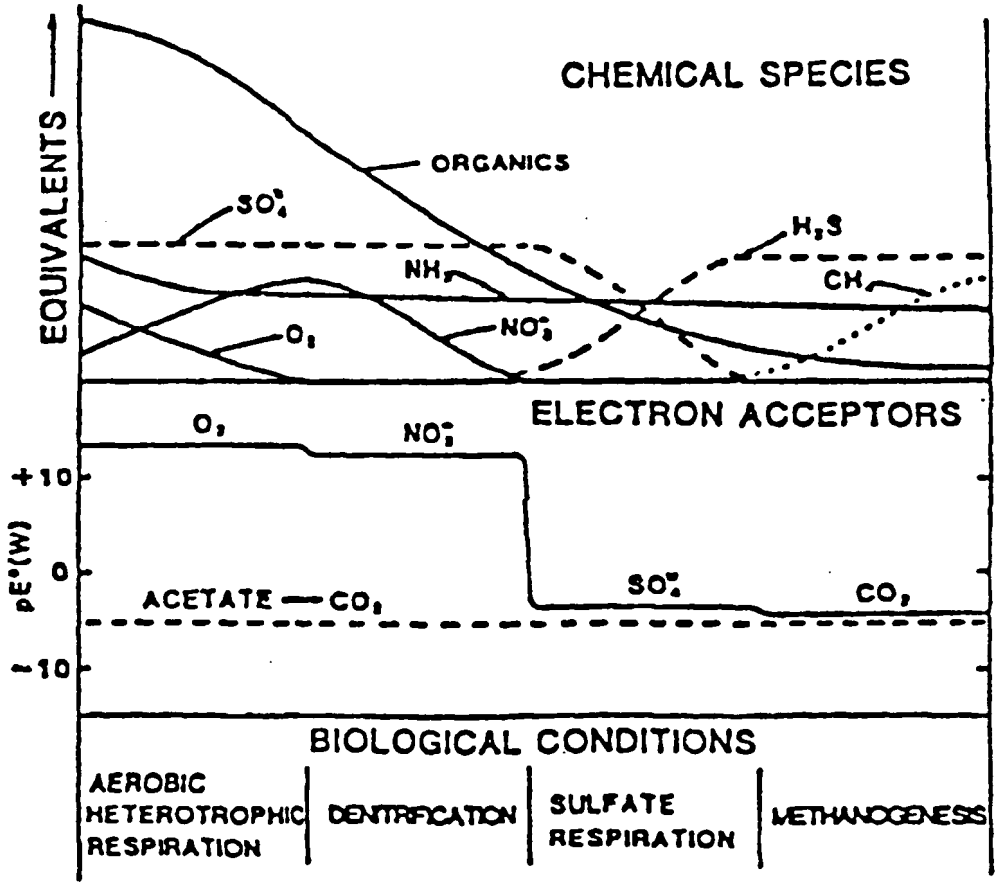


Figure 3-19. Microbially Mediated Changes in Chemical Species, Redox Conditions, and Spatial Regions Favoring Different Types of Metabolic Processes Along the Flow Path of a Contaminant Plume (adapted from Bouwer and McCarty, 1984, by Suflita, 1989a)

electron acceptors, the presence of suitable microorganisms, and the energy benefit of the metabolic process to the specific microbial communities. As organic matter is transported in a contaminant plume, a series of redox zones can be established that range from highly oxidized to highly reduced conditions. The biodegradation potential and the expected rates of metabolism will be different in each zone (Suflita, 1989a). For many contaminants, aerobic decomposition is relatively fast, especially compared to methanogenic conditions. However, some contaminants, such as certain halogenated aliphatic compounds and 2,4,5-T, degrade faster when anaerobic conditions exist (Bouwer and others, 1981; Bouwer and McCarty, 1984; Gibson and Suflita, 1986).

Environmental Factors Affecting Biodegradation
Microorganisms need a suitable physical and chemical environment to grow and actively metabolize organic contaminants, (Suflita, 1989a). Extremes of temperature, pH, salinity, osmotic or hydrostatic pressures, radiation, free water limitations, contaminant concentration, and/or the presence of toxic metals or other toxicant materials can limit the rate of microbial growth and/or substrate utilization. Often, two or more environmental factors interact to limit microbial decomposition processes. Selected critical environmental factors are presented in Table 3-1.

Limitations in the ability to alter environmental factors in the subsurface environment are important in selecting and implementing aquifer bioremedial technologies (Suflita, 1989a). For example, the temperature of aquifers probably cannot be significantly altered to stimulate in situ microbial growth and metabolism, but temperatures could be changed in a surface biological treatment reactor.

Physiological Factors Affecting Biodegradation
In addition to environmental conditions, microbial physiological factors also influence organic contaminant biodegradation (Suflita, 1989a). The supply of carbon and energy contained in organic contaminants must be sufficient for heterotrophic microbial growth. Too high a substrate concentration can limit microbial metabolism due to the toxicity of the substrate to microorganisms. If concentrations are too low, microbial response may be inhibited, or the substrates may not be suitable for

Environmental Factor	Optimum Levels
Available soil water	25-85% of water holding capacity; -0.01 MPa
Oxygen	Aerobic metabolism: Greater than 0.2 mg/l dissolved oxygen, minimum air-filled pore space of 10% by volume; Anaerobic metabolism: O_2 concentrations less than 1% by volume
Redox potential	Aerobes & facultative anaerobes: greater than 50 millivolts; Anaerobes: less than 50 millivolts
pH	pH values of 5.5 - 8.5
Nutrients	Sufficient nitrogen, phosphorus, and other nutrients so as to not limit microbial growth (Suggested C:N:P ratio of 120:10:1)
Temperature	15 - 45° C (Mesophiles)

Table 3-1. Critical Environmental Factors for Microbial Activity (Sims and others, 1984; Huddleston and others, 1986; Paul and Clark, 1989)

growth. Growth and energy sources do not have to be supplied by the same carbon substrate. Growth and metabolism of microorganisms can be stimulated by providing a non-toxic primary carbon substrate so that the rate and extent of contaminant degradation can be increased (McCarty and others,1981; McCarty, 1985; McCarty and others, 1984).

A contaminant also will be poorly metabolized if it is unable to enter microbial cells and gain access to intracellular metabolic enzymes, which may occur with larger molecular weight compounds (Suflita, 1989a). A substrate also will persist if it fails to de-repress the enzymes required for its degradation. Appropriate enzymes sometimes can be induced by an alternate chemical compound. Sometimes initial biochemical reactions result in metabolites that tend to inhibit degradation of the parent molecule.

The absence of other necessary microorganisms can limit contaminant degradation, since often several microbial groups are required for complete degradation (Suflita, 1989a). Microbial consortia are especially important in anaerobic mineralization of contaminants (McInemey and Bryant, 1981); if any individual members of a consortium are absent, biodegradation of the parent material effectively ceases.

Chemical Factors Affecting Biodegradation

One of the most important factors affecting contaminant biodegradation in aquifers is the structure of the contaminant, which determines its physical state (i.e., soluble, sorbed) and its tendency to biodegrade (Suflita, 1989a). Aquifer contaminants may contain chemical linkages that tend to favor or hinder microbial degradation. The number, type, and position of substituents on a contaminant molecule should be considered when evaluating its metabolic fate in an aquifer.

Usually the closer a contaminant structurally resembles a naturally occurring compound, the better the possibility that the contaminant will be able to enter a microbial cell, de-repress the synthesis of metabolic enzymes, and be converted by those enzymes to metabolic intermediates (Suflita, 1989a). Biodegradation is less likely (though not precluded) for those molecules having unusual structural features infrequently encountered in the natural environment. Therefore, xenobiotic compounds tend to persist in the natural environment because microorganisms have not evolved necessary metabolic pathways to degrade those compounds. However, microorganisms are nutritionally versatile, have the potential to grow rapidly, and possess only a single copy of DNA. Therefore, any genetic mutation or recombination is immediately expressed. If the alteration is of adaptive significance, new species of microorganisms can be formed and grow. Contaminated environments supply selection pressure for the evolution of organisms with new metabolic potential that can grow utilizing the contaminating substance.

Aquifer Bioremediation

If an aquifer contaminant is determined to be susceptible to biodegradation, the goal of bioremediation is to utilize the metabolic capabilities of the indigenous microorganisms to eliminate that contaminant (Suflita, 1989a). This practice generally does not include the inoculation of the aquifer with foreign bacteria.

Bioremedial technologies attempt to impose particular conditions in an aquifer to encourage microbial growth and the presence of desirable microorganisms. Bioremediation is based on knowledge of the chemical and physical needs of the microorganisms and the predominant metabolic pathways (Suflita, 1989a). Most often, microbial activity is stimulated by supplying nutrients necessary for microbial growth. Bioremediation can take place either above ground or in situ. In situ systems are especially appropriate for contaminants that sorb to aquifer materials, since many decades of pumping may be required to reduce the contaminants to sufficiently low levels.

Successful implementation of aquifer bioremediation depends on determining site-specific hydrogeological variables, such as type and composition of an aquifer, permeability, thickness, interconnection to other aquifers, location of discharge areas, magnitude of water table fluctuations, and ground-water flow rates (Suflita, 1989b). Generally, bioremediation is utilized in more permeable aquifer systems where movement of ground water can be more successfully controlled.

Removal of free product also is important for the success of bioremediation. Many substances that serve as suitable nutrients for microbial growth when present at low concentrations are inhibitory at high concentrations (Suflita, 1989b).

Modeling Transport and Fate of Contaminants in an Aquifer

Introduction

Models are simplified representations of real-world processes and events, and their creation and use require many judgments based on observation of simulations of specific natural processes. Models may be used to simulate the response of specific problems to a variety of possible solutions (Keely, 1989b).

Physical models, including sand-filled tanks used to simulate aquifers and laboratory columns used to study contaminant flow through aquifer materials, often are used to obtain information on contaminant movement (Keely, 1989b). Analog models also are physically based, but are only similar to actual processes. An example is the electric analog model, where capacitors and resistors are used to replicate the effects of the rate of water release from storage in aquifers. The main disadvantage of physical models is the time and effort required to generate a meaningful amount of data.

Mathematical models are non-physical and rely on quantification of relationships between specific parameters and variables to simulate the effects of natural processes (Keely, 1989b, Weaver and others, 1989). Because mathematical models are abstract, they often do not provide an intuitive knowledge of real-world situations. However, mathematical models can provide insights into the functional dependencies between causes and effects in an actual aquifer. Large amounts of data can be generated quickly, and experimental modifications made easily, making possible for many situations to be studied in detail for a given problem.

Use and Categories of Mathematical Models

The application of mathematical models is subject to error in real-world situations when appropriate field determinations of natural process parameters are lacking. This source of error is not addressed adequately by sensitivity analyses or by the application of stochastic techniques for estimating uncertainty. The high degree of hydrogeological, chemical, and microbiological complexity typically present in field situations requires the use of site-specific characterization of the influences of various natural processes by detailed field and laboratory investigations (Keely, 1989b).

Mathematical models have been categorized by their technical bases and capabilities as: (1) parameter identification models; (2) prediction models; (3) resource management models; and (4) data manipulation codes. (Bachmat and others, 1978; van der Heidje and others, 1985).

Parameter identification models are used to estimate aquifer coefficients that determine fluid flow and contaminant transport characteristics (e.g., annual recharge, coefficients of permeability and storage, and dispersivity (Shelton, 1982; Guven and others, 1984; Puri, 1984; Khan, 1986a, b; Strecker and Chu, 1986)). Prediction models are the most numerous type because they are the primary tools used for testing hypotheses (Mercer and Faust, 1981; Anderson and others, 1984; Krabbenhoft and Anderson, 1986).

Resource management models are combinations of predictive models, constraining functions (e.g., total pumpage allowed), and optimization routines for objective functions (e.g., scheduling wellfield operations for minimum cost or minimum drawdown/pumping lift). Few of these types of models are developed well enough or supported to the degree that they are useful (van der Heidje, 1984a and b; van der Heidje and others, 1985).

Data manipulations codes are used to simplify data entry to other kinds of models and facilitate the productions of graphic displays of model outputs (van der Heidje and Srinivasan, 1983; Srinivasan, 1984; Moses and Herman, 1986).

Quality Control Measures

Quality control measures are required to assess the soundness and utility of a mathematical model and to evaluate its application to a specific problem. Huyakorn et al. (1984) and Keely (1989b) have suggested the following quality control measures:

1. Validation of the model's mathematical basis by comparing its output with known analytical solutions to specific problems.

2. Verification of the model's application to various problem categories by successful simulation of observed field data.

3. Benchmarking the problem-solving efficiency of a model by comparison with the performance of other models.

4. Critical review of the problem conceptualization to ensure that the modeling considers all physical, chemical, and biological processes that may affect the problem.

5. Evaluation of the specifics of the model's application, e.g., appropriateness of the boundary conditions, grid design, time steps.

6. Appraisal of the match between the mathematical sophistication of the model and the temporal and spatial resolution of the data.

Summary

Transport and fate assessments require interdisciplinary analyses and interpretations because processes are interdependent (Keely 1989a). Each transport process should be studied from interdisciplinary viewpoints, and interactions among processes identified and understood. In addition to a sound conceptual understanding of

transport processes, the integration of information on geologic, hydrologic, chemical, and biological processes into an effective contaminant transport evaluation requires data that are accurate, precise, and appropriate at the intended problem scale and that attempt to account for spatial and temporal variations.

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Chapter 4

GROUND-WATER TRACERS

In hydrogeology, "tracer" is a distinguishable matter or energy in ground water that carries information on the ground-water system. A tracer can be entirely natural, such as the heat carried by hot-spring waters; accidentally introduced, such as fuel oil from a ruptured storage tank; or intentionally introduced, such as dyes placed in water flowing within limestone caves.

Types and Uses of Tracer Tests

The variety of tracer tests is almost infinite, considering the various combinations of tracer types, local hydrologic conditions, injection methods, sampling methods, and geological settings. Tracer tests mainly are used (1) to measure one or more hydrogeologic parameters of an aquifer; and (2) to identify sources, velocity, and direction of movement of contaminants. Tracer tests also can be broadly classified according to whether they rely on natural gradient flow or an induced flow from pumping or some other means. Quinlan and others (1988) discuss how to recognize falsely negative or positive tracer results.

Measurement of Hydrogeologic Parameters

Tracers can be used to measure or estimate a wide variety of hydrogeologic parameters, most commonly direction and velocity of flow and dispersion. Depending on the type of test and the hydrogeologic conditions, other parameters such as hydraulic conductivity, porosity, chemical distribution coefficients, source of recharge, and age of ground water can be measured.

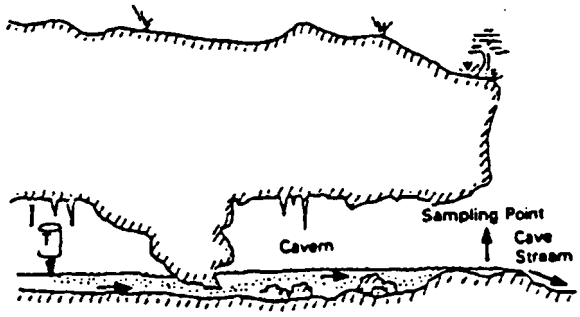
Figure 4-1 shows six examples of tracer measurement of hydrogeologic characteristics by natural gradient flow. Figure 4-1a shows flow velocity in a cave system, and Figure 4-1b shows subsurface flow patterns in a karst area with sinking and rising streams. Figure 4-1c shows the velocity of movement of dissolved material between two wells. Both velocity and direction of flow can be measured in a single well as shown in Figure 4-1d and by using multiple downgradient sampling wells as shown in Figure 4-1e. Finally, hydrodynamic dispersion can be measured by multiwell, multilevel sampling down gradient (Figure 4-1f).

Figure 4-2 shows four examples of tracer measurement of hydrogeologic parameters using induced flow. A tracer in surface water combined with pumping from a nearby well can verify a connection, as shown in Figure 4-2a. Interconnections between fractures can be mapped using tracers and inflatable packers in two uncased wells, as shown in Figure 4-2b. Figure 4-2c shows the measurement of a number of aquifer parameters using a pair of wells with forced circulation between wells. Figure 4-2d shows the evaluation of geochemical interactions between multiple tracers and aquifer material by alternating injection and pumping.

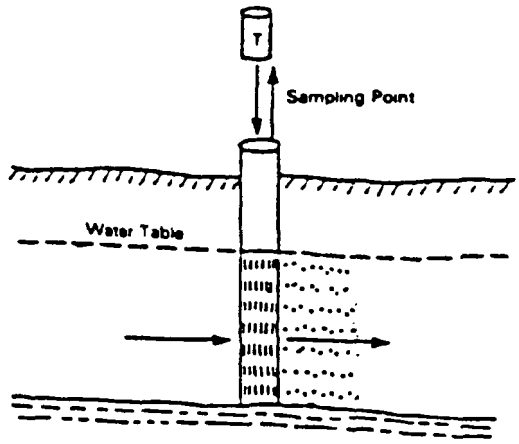
Tracers also can be used to determine ground-water recharge using environmental isotopes (Ferronsky and Polyakov, 1982; Moser and Rauert, 1985; Vogel and others, 1974), and to date ground water (Davis and Bentley, 1982).

Delineation of Contaminant Plumes

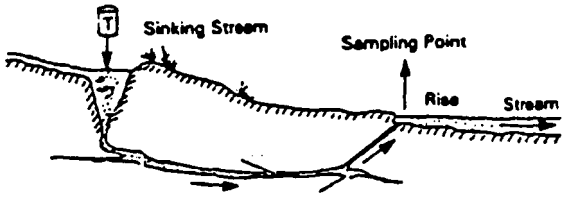
Any contaminant that moves in ground water acts as a tracer; thus the contaminant itself may be mapped, or other tracers may be added to map the velocity and direction of the flow. Contaminant plumes are not tracers in the sense used in this chapter and are not discussed further here. However, Figure 4-3 shows three examples of noncontaminant tracers used to identify contaminant sources and flow patterns. Figure 4-3a shows the use of a tracer in a sinkhole to determine if trash at a particular location is contributing to contamination of a spring. Similarly, Figure 4-3b shows that by flushing a dye tracer down a toilet one can determine whether septic seepage is causing contamination of a well or surface water. Figure 4-3c shows the use of multiple tracers at multiple sources of potential contamination to pinpoint the actual source.



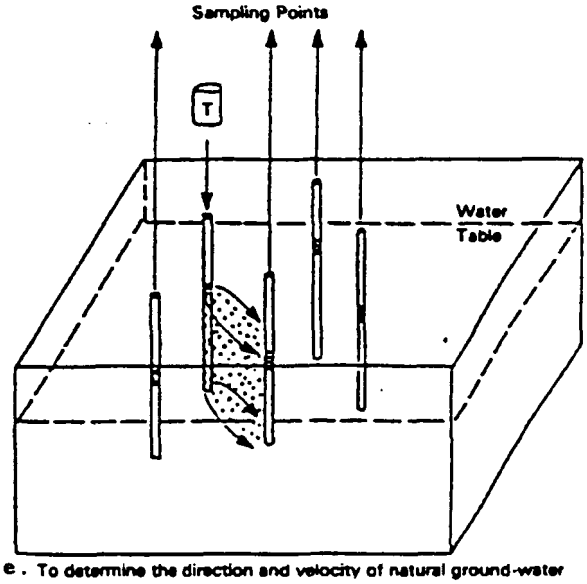
a. To measure velocity of water in cave stream.



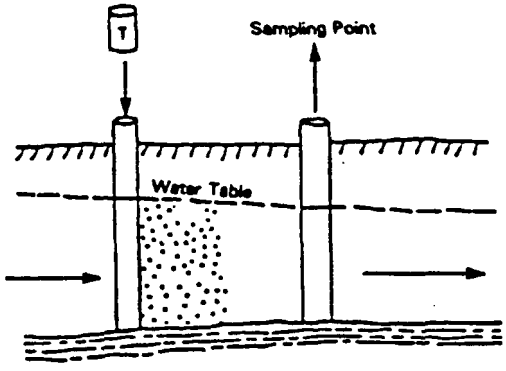
d. To determine velocity and direction of ground-water flow under natural conditions. Injection followed by sampling from same well



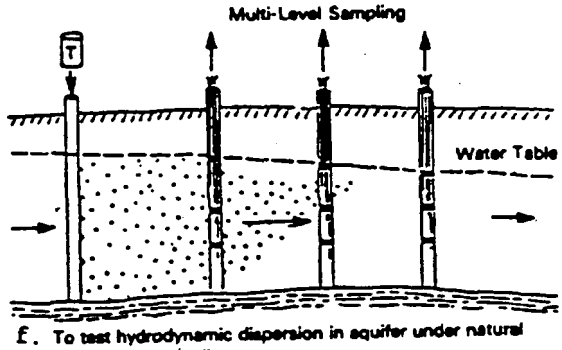
b. To check source of water at rise in stream bed.



e. To determine the direction and velocity of natural ground-water flow by drilling an array of sampling wells around a tracer injection well.

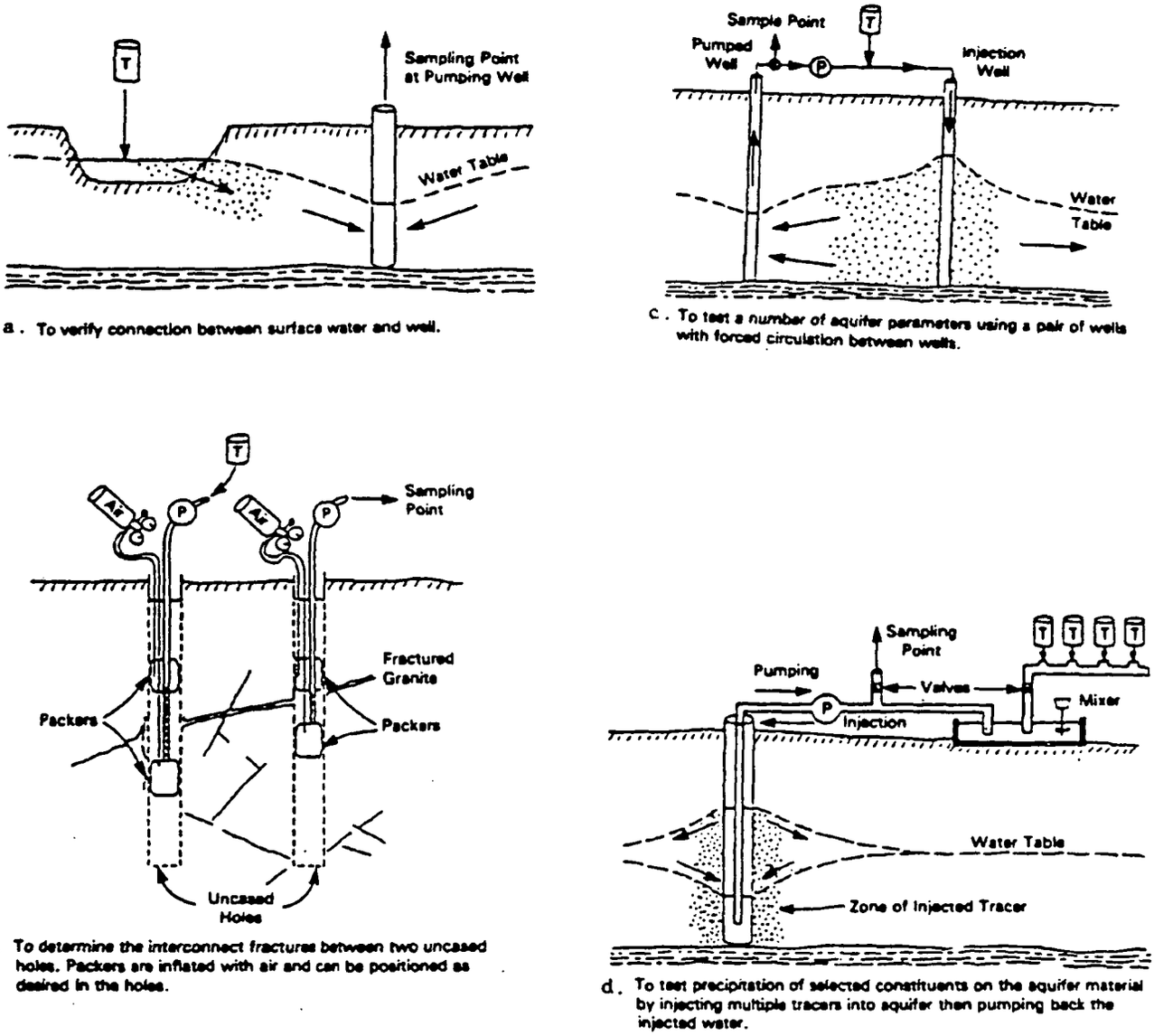


c. To test velocity of movement of dissolved material under natural ground-water gradients.



f. To test hydrodynamic dispersion in aquifer under natural ground-water gradients.

Figure 4-1. Common Configurations for Use of Tracer to Measure Hydrogeologic Parameters Using Natural Gradient Flow (from Davis and others, 1985)



a. To verify connection between surface water and well.

c. To test a number of aquifer parameters using a pair of wells with forced circulation between wells.

b. To determine the interconnect fractures between two uncased holes. Packers are inflated with air and can be positioned as desired in the holes.

d. To test precipitation of selected constituents on the aquifer material by injecting multiple tracers into aquifer then pumping back the injected water.

Figure 4-2. Common Configurations for Use of Tracers to Measure Hydrogeologic Parameters Using Induced Flow (from Davis and others, 1985)

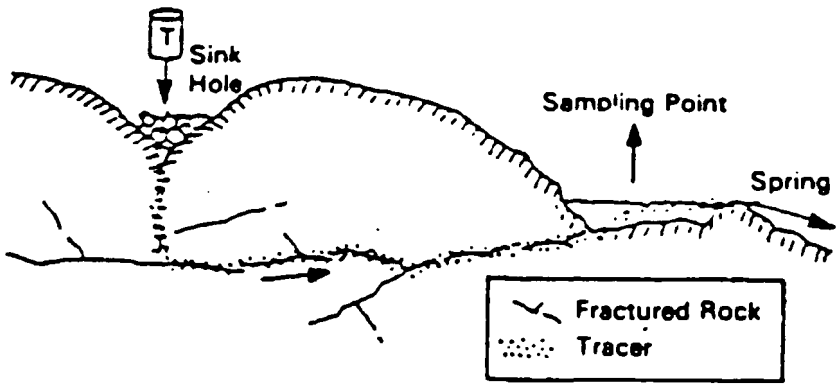
Tracer Selection

Overview of Types of Tracers

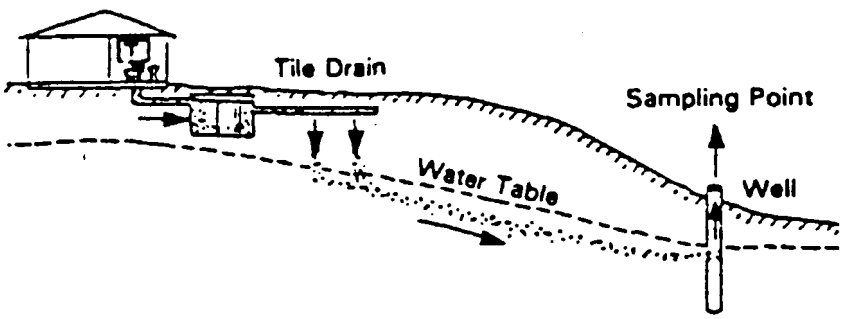
Ground-water tracers can be broadly classified as natural (environmental) tracers and injected tracers. Table 4-1 lists 14 natural tracers and 30 injected tracers. Table 4-2 lists review papers, reports, and bibliographies that are good sources for general information on ground-water tracing.

The potential chemical and physical behavior of the tracer in ground water is the most important selection criterion. Conservative tracers, used for most purposes, travel with the same velocity and direction as the water and do not interact with solid material. Nonconservative tracers, which tend to be slowed by interactions with the solid matrix, are used to measure distribution coefficients and preferential flow zones in the vadose zone. For most uses, a tracer should be nontoxic, inexpensive, and easily detected to a low concentration with widely available and simple technology. If the tracer occurs naturally in ground water, it should be present in concentrations well above background concentrations. Finally, the tracer itself should not modify the hydraulic conductivity or other properties of the medium being studied.

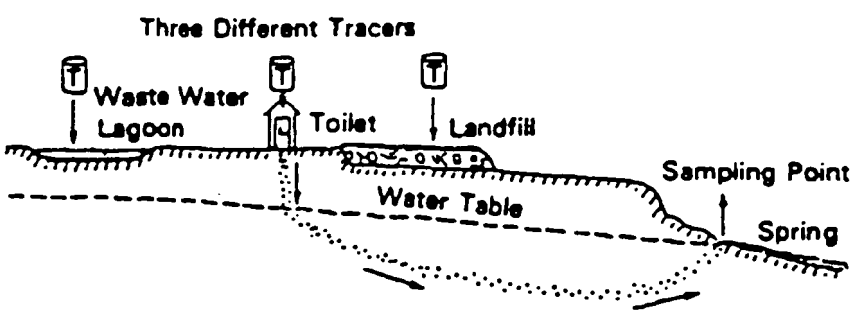
No one ideal tracer has been found. Because natural systems are so complex and the requirements for the tracers themselves are so numerous, the selection and



a. To determine if trash in sinkhole contributes to contamination of spring.



b. To determine if tile drain from septic tank contributes to contamination of well.



c. To determine source of pollution from three possibilities.

Figure 4-3. Common Configurations for Use of Tracers to Identify Contaminant Sources Using Natural Gradient Flow (from Davis and others, 1985)

NATURAL TRACERS				INJECTED TRACERS			
Stable Isotopes		Radioactive		Activatable		Inactive	
						Ionized Substances	Drift Material
Deuterium	^{2}H	Tritium	^{3}H	Bromine	^{35}Br	Salts: $Na^{+}Cl^{-}$	Lycopodium Spores
Oxygen-18	^{18}O	Sodium-24	^{24}Na	Indium	^{49}In	$K^{+}Cl^{-}$	Bacteria
Carbon-13	^{13}C	Chromium-51	^{51}Cr	Manganese	^{25}Mn	$Li^{+}Cl^{-}$	Viruses
Nitrogen-15	^{15}N	Cobalt-58	^{58}Co	Lanthanum	^{57}La	$Na^{+}I^{-}$	Fungi
Strontium-88	^{88}Sr	Cobalt-60	^{60}Co	Dysprosium	^{66}Dy	$K^{+}Br^{-}$	Sawdust
Radioactive Isotopes						Fluorescent Dyes:	
Tritium-3	^{3}H	Gold-198	^{198}Au			Optical Brighteners	
Carbon-14	^{14}C	Iodine-131	^{131}I			Tinopal 5Bm6x(FDA 22)	
Silicon-32	^{32}Si	Phosphorus-32	^{32}P			Direct Yellow 96	
Chlorine-36	^{36}Cl					Fluorescein	
Argon-37	^{37}Ar					Acid Yellow 7	
Argon-39	^{39}Ar					Rhodamine WT	
Krypton-81	^{81}Kr					Eosin (Acid Red 87)	
Krypton-85	^{85}Kr					Amidorhodamine 6 (Acid Red 50)	
						Physical Characteristics	
						Water Temperature	
						Flood Pulse	

Source: Modified from Jones, 1984.

Table 4-1. Survey of Ground-Water Tracers

Reference	Description
Atkinson and Smart (1981)	Review paper on uses of artificial tracers in hydrology.
Davis and others (1980)	Review paper on ground-water tracers.
Davis and others (1985)	Introductory EPA report on ground-water tracing. See also discussion by Quinlan (1986) and reply by Davis (1986)
Drew and Smith (1969)	Focuses on fluorescent dyes and lycopodium spores, but also contains annotated bibliography on other tracers.
Gaspar (1987)	Compilation of papers on modern trends in tracer hydrology.
Grisak and others (1983)	Report evaluating ground-water tracers for nuclear fuel waste management studies.
Kaufman and Orlob (1956)	Early review paper on use of radioactive and chemical tracers in porous media.
Keswick et al. (1982)	Review paper on use of microorganisms as ground-water tracers.
Knuttson (1968)	Review paper on use of tracers for ground-water investigations.
McLaughlin (1982)	Review paper on use of dyes as soil water tracers.
Molz and others (1986)	Focuses on aquifer tracer tests in porous media and use in contaminant transport modeling.
Smart and Laidlaw (1977)	Classic paper on the use of fluorescent dyes for water tracers.
Taylor and Dey (1985)	Bibliography on borehole geophysics as applied to ground-water hydrology containing 42 references on tracers.
Turner Design's Ground Water Tracing Series	A series of annotated bibliographies concerning solute movement in aquifers and use of dyes as tracers. Smart et al. (1988) review 57 papers that compare dyes with other tracers. See also Edwards and Smart (1988a, b).
van der Leeden (1987)	The section in this bibliography on tracers and ground-water dating contains 69 references.

Table 4-2. Sources of Information on General Ground-Water Tracing

use of tracers is almost as much an art as a science. The following sections discuss factors that should be considered when selecting a tracer.

Hydrogeologic Considerations

The initial step in determining the physical feasibility of a tracer test is to collect as much hydrogeologic information about the field area as possible. The logs of the wells at the site to be tested, or logs of the wells closest to the proposed site, will give some idea of the homogeneity of the aquifer, layers present, fracture patterns, porosity, and boundaries of the flow system. Local or regional piezometric maps, or any published reports on the hydrology of the area (including results of aquifer tests), are valuable, as they may give an indication of the hydraulic gradient and hydraulic conductivity.

Major hydrogeologic factors that should be considered when selecting a tracer include:

Lithology. Fine-grained materials, particularly clays, have higher sorptive capacities than coarse-grained material. The sorptive capacity must be considered when evaluating the potential mobility of a tracer.

Flow Regime. Whether flow is predominantly through porous media (alluvium, sandstone, soil), solution features (karst limestone), or fractures will influence the choice of tracer. For example, fluorescent dyes work well in karst settings, but because of sorption effects are less effective than ground-water tracers in porous media.

Direction of Flow. For tracer studies using two or more wells, the general direction of ground-water movement must be known.

Travel Time. The equation for estimating travel time was discussed previously. In two-well tracer tests, travel time must be known to estimate spacing for wells.

Dispersion. Tracer tests often are used to measure dispersion. In two-well tests, some preliminary estimates may be required to estimate the quantity of tracer to inject so that concentrations will be high enough to detect.

Tracer Characteristics

Tracers have a wide range of physical, chemical, and biological characteristics. These properties, as they relate to hydrogeologic and other factors will determine the most suitable tracer for the purposes desired.

Detectability. Injected tracers should have no, or very low, natural background levels. Lower detection limit is for instruments (ppm, ppb, ppt), are better. The degree of dilution is a function of type of injection, distance, dispersion, porosity, and hydraulic conductivity. Too much dilution may result in failure to observe the tracer when it reaches a sampling point because concentrations are below the detection limit. Possible interferences from other tracers and natural water chemistry may have the same effect.

Mobility. Conservative tracers used to measure aquifer parameters such as flow direction and velocity should be (1) stable (i.e., not subject to transformation by biodegradation or nonbiological processes during the length of the test and analysis); (2) soluble in water; (3) of a similar density and viscosity; and (4) not subject to adsorption or precipitation. Nonconservative, nontoxic tracers used to simulate transport of contaminants should have adsorptive and other chemical properties similar to the contaminant of concern.

Toxicity. Nontoxic tracers should be used if at all possible. If a tracer may be toxic at certain concentrations, maximum permissible levels as determined by federal, state, or county agencies must be considered in relation to expected dilution and proximity to drinking water sources. Most agencies have set no limits, partly because the commonly used tracers are nontoxic in concentrations usually employed, and partly because they never considered tracers to be a problem demanding regulation.

Other Considerations

A tracer may be suitable for the test's purpose and the hydrogeologic setting, yet still not be suitable for reasons of economics, technological availability or sophistication, or public health.

Economics. The tracer or the instrumentation to analyze samples may be expensive. In this situation, another less-expensive tracer with somewhat less favorable characteristics may suffice.

Technology. Some tracers may be difficult to obtain, or may require more complicated sampling methods. Gases, for example, will escape easily from poorly sealed containers. Similarly, instrumentation for gas or isotope analyses may not be available; e.g., only one or two laboratories in the world can perform analyses of 36Cl.

Public Health. Tracer injections must involve a careful consideration of possible health implications. Some local or state health agencies insist on review

authority prior to use of artificially introduced tracers, but most do not. Local citizens must be informed of the tracer injections, and usually the results should be made available to the public. Under some circumstances, analytical work for tracer studies must be performed in appropriately certified laboratories. These are job-specific decisions.

Tracing In Karst vs. Porous Media

Ground-water flow in karst terranes is characterized by conduit flow and diffuse flow through often complex subsurface channel systems. Ground-water contaminants tend to move rapidly in karst and resurge at the surface in locations that cannot be readily predicted from the morphology of surface drainage patterns. In contrast, ground-water flow in porous media is characterized by slow travel times and more generally predictable flow directions. These differences require substantially different approaches to conducting tracer tests, as discussed in karst and porous media sections in this document.

Types of Tracers

Considering the full range of organic ground-water contaminants, hundreds, and possibly thousands, of substances have been used as tracers in ground water. The most commonly used tracers can be grouped into six categories: (1) water temperature, (2) particulates (called drift material in Table 4-1), (3) ions, (4) dyes, (5) gases, and (6) isotopes. These categories are not mutually exclusive (i.e., isotopes may take the form of ions or gases). Selected tracers in each category in relation to applicability in different hydrologic settings, field methods, and type of detection used, are discussed in the following sections .

Water Temperature

The temperature of water changes slowly as it migrates through the subsurface, because water has a high specific heat capacity compared to most natural materials. For example, temperature anomalies associated with the spreading of warm wastewater in the Hanford Reservation in south central Washington have been detected more than 8 km (5 mi) from the source.

Water-temperature tracing is potentially useful, although it has not been used frequently. The method may be applicable in granular media, fractured rock, or karst regions. Keys and Brown (1978) traced thermal pulses from the artificial recharge of playa lake water into the Ogallala formation in Texas. They described the use of temperature logs (temperature measurements at intervals in cased holes) to detect hydraulic conductivity differences in an aquifer. Temperature logs also have been used to determine vertical movement of water in a borehole (Keys and MacCary, 1971; Sorey, 1971).

Changes in water temperature are accompanied by changes in water density and viscosity, which in turn alter the velocity and direction of flow. For example, injected ground water with a temperature of 40°C will travel more than twice as fast in the same aquifer under the same hydraulic gradient as water at 5°C. Because the warm water has a slightly lower density than cold water, buoyant forces give rise to flow that "floats" on top of the cold water. To minimize temperature-induced-convection problems, accurately measured small temperature differences should be used if hot or cold water is in the introduced tracer.

Davis and others (1985) used temperature as a tracer for small-scale field tests, in shallow drive-point wells 2 feet apart in an alluvial aquifer. The transit time of the peak temperature was about 107 min, while the resistivity data indicated a travel time of about 120 min (Figure 4-4). The injected water had a temperature of 38°C, while the ground-water temperature was 20°C; the peak temperature obtained in the observation well was 27°C. In these tests, temperature indicated breakthrough of

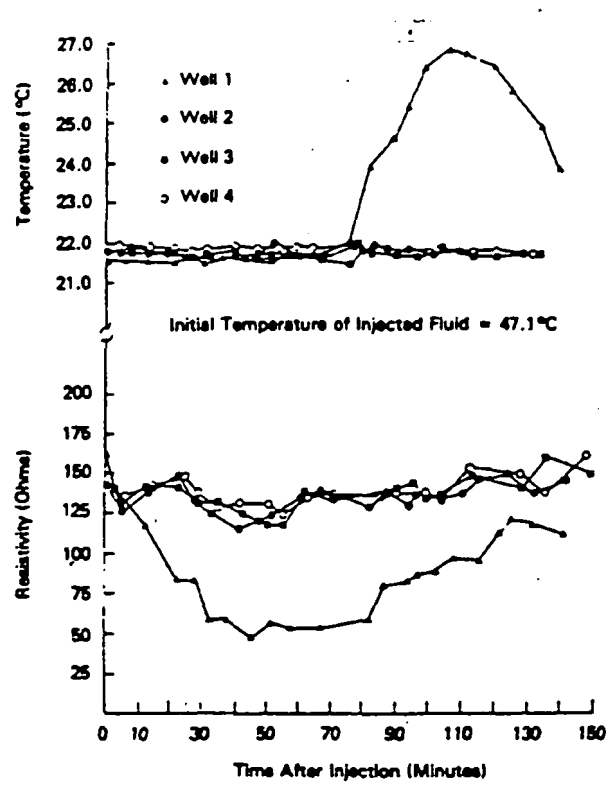


Figure 4-4. Results of Field Test Using a Hot Water Tracer (from Davis and others, 1985)

the chemical tracers, aiding in the timing of sampling. It also was useful as a simple, inexpensive tracer for determining the correct placement of sampling wells.

Water-temperature tracing also can be used to detect river recharge in an aquifer. Most rivers have large seasonal water temperature fluctuations. If the river is recharging an aquifer, the seasonal fluctuations can be detected in the ground water relative to the river (Rorabaugh, 1956).

Particulates

Solid material in suspension, such as spores, can be a useful tracer in areas where water flows in large conduits such as in some basalt, limestone, or dolomite aquifers. Seismic methods at the surface have been used to detect the location of time-delayed explosives floating through a cave system (Arandjelovic, 1969). Small particulate tracers, such as bacteria, can travel through any porous media such as soils and fractured bedrock where the pore size is larger than the size of the microorganism. Microorganisms are probably the most commonly used particulate tracers. Table 4-3 compares characteristics of microbial tracers.

Yeast. Wood and Ehrlich (1978) reported the use of baker's yeast (Saccharomyces cerevisiae) as a ground-water tracer in a sand and gravel aquifer. Yeast is a single-celled fungus that is ovoid in shape. The diameter of a yeast cell is 2 to 3 µm, which closely approximates the size of pathogenic bacterial cells. This tracer probably provides most information about the potential movement of bacteria.

Wood and Ehrlich (1978) found that the yeast penetrated more than 7 m into a sand and gravel aquifer in less than 48 hours after injection. This tracer is very inexpensive, as is analysis. Another advantage is the lack of environmental concerns.

Bacteria. Bacteria are the most commonly used microbial tracers, because they grow well and are easily detected. Keswick and others (1982) reviewed over 20 case studies of bacteria tracers. Some bacteria that have been used successfully are Escherichia coliform (E. coli), Streptococcus faecalis, Bacillus stearothermophilus, Serratia marcescens, and Serratia indica. These bacteria range in size from 1 to 10 µm and have been used in a variety of applications.

A fecal coliform, E. coli, has been used to indicate fecal pollution at pit latrines, septic fields, and sewage disposal

Tracer	Size (µm)	Time Required for Assay (days)	Essential Equipment Required
Bacteria	1-10	1-2	Incubator*
Spores	25-33	1/2	Microscope Plankton nets
Yeast	2-3	1-2	Incubator*
Viruses: Animal (enteric)	0.2-0.8	3-5	Incubator Tissue Culture Laboratory
Bacterial	0.2-1.0	1/2-1	Incubator*

*Many may be assayed at room temperature

Source: Keswick and others (1982)

Table 4-3. Comparison of Microbial Tracers

sites. A "marker" such as antibiotic resistance or H_2S production is used to distinguish the tracer from background organisms.

The greatest health concern in using these tracers is that the bacteria must be nonpathogenic to humans. Even E. coli has strains that can be pathogenic. Davis and others (1970) and Wilkowske and others (1970) have reported that Serratia marcescens may be life-threatening to patients who are hospitalized with other illnesses. Antibiotic-resistant strains are another concern, as the antibiotic resistance can be transferred to potential human pathogens. This problem can be avoided by using bacteria that cannot transfer this genetic information. As is true with most other injected tracers, permission to use bacterial tracers should be obtained from the proper federal, state, and local health authorities.

Viruses. Animal, plant, and bacterial viruses also have been used as ground-water tracers. Viruses are generally much smaller than bacteria, ranging from 0.2 to 1.0 µm (see Table 4-1). In general, human enteric viruses cannot be used because of disease potential. Certain vaccine strains, however, such as a type of polio virus, have been used but are considered risky. Most animal enteric viruses are considered safer as they are not known to infect humans (Keswick and others, 1982). Neither human nor most animal viruses, however, are generally considered suitable tracers for field work because of their potential to infect humans.

Spores. Lycopodium spores have been widely used as tracers in karst hydrogeologic systems in Europe since the early 1950s, and less frequently used in the United States since the 1970s. Much of the literature on the use of spores, however, is in obscure European and American speleological journals. More readily accessible references on the use of spores include Atkinson and others (1973), Gardner and Gray (1976), and Smart and Smith (1976).

Lycopodium is a clubmoss that has spores nearly spherical in shape, with a mean diameter of 33 µm. It is composed of cellulose and is slightly denser than water, so that some turbulence is required to keep the material in suspension. Some advantages of using lycopodium spores as a tracer are:

- The spores are relatively small.
- They are not affected by water chemistry or adsorbed by clay or silt.
- They travel at approximately the same velocity as the surrounding water.
- The injection concentration can be very high (e.g., 8×10^6 spores per cm^3).
- They pose no health threat.
- The spores are easily detectable under the microscope.
- At least five dye colors may be used, allowing five tracings to be conducted simultaneously in a karst system.

Some disadvantages associated with lycopodium spores include the large amount of time required for their preparation and analysis, and the filtration of spores by sand or gravel if flow is not sufficiently turbulent.

The basic procedure involves adding a few kilograms of dyed spores to a cave or sinking stream. The movement of the tracer is monitored by sampling downstream in the cave or with plankton nets installed in the stream bed at a spring. The sediment caught in the net is concentrated and treated to remove organic matter. The spores are then examined under the microscope.

Tracing by lycopodium spores is most useful in open joints or solution channels (karst terrane) where there is minimal suspended sediment. It is not useful in wells or boreholes unless the water is pumped continuously to the surface and filtered. The spores survive well in polluted water, but do not perform well in slow flow or in water with a high sediment concentration. A velocity of a few miles per hour has been found sufficient to keep the spores in suspension. According to Smart and Smith (1976), lycopodium is preferable to dyes for use in large-scale water resource reconnaissance studies in karst areas. Skilled personnel should be available to sample and analyze the spores and a relatively small number of sampling sites should be used.

Ions

Inorganic ionic compounds such as common salts have been used extensively as ground-water tracers. This category of tracers includes those compounds that undergo ionization in water, resulting in their separation into charged species possessing a positive charge (cations) or a negative charge (anions). The charge on an ion affects its movement through aquifers by numerous mechanisms.

Ionic tracers have been used as tools to determine flow paths and residence times and measure aquifer properties. Slichter (1902, 1905) was probably the first to use ionic tracers to study ground water in the United States. Specific characteristics of individual ions or

ionic groups may approach those of an ideal tracer, particularly dilute concentrations of certain anions.

In most situations, anions (negatively charged ions) are not affected by the aquifer medium. Mattson (1929), however, showed that the capacity of clay minerals for holding anions increases with decreasing pH. Under conditions of low pH, anions in the presence of clay, other minerals, or organic detritus may undergo anion exchange. Other possible effects include anion exclusion and precipitation/dissolution reactions. Cations (positively charged ions) react much more frequently with clay minerals through the process of cation exchange, which displaces other cations such as sodium and calcium into solution. Because of their interaction with the aquifer media, little work has been done with cations. Natural variations in Ca and Mg concentrations, however, have been used to separate baseflow and stormflow components in a karst aquifer (Dreiss, 1989).

One advantage of simple ionic tracers is that they do not decompose and, therefore, are not lost from the system. However, a large number of ions (including Cl- and NO_3^-) have high natural background concentrations; thus requiring the injection of a highly concentrated tracer. More importantly, several hundred pounds of chloride or nitrate may have an adverse effect on water quality and biota, thus becoming a pollutant. This also may result in density separation and gravity segregation during the tracer test (Grisak and Pickens, 1980b). Density differences will alter flow patterns, the degree of ion exchange, and secondary chemical precipitation, all of which may change the aquifer permeability. Comparisons of tracer mobilities under laboratory and field conditions by Everts and others (1989) found bromide (BR-) to be only slightly less mobile than nitrate. The generally low background concentrations of bromide often make it the ion of choice when a conservative tracer is desired.

Various applications of ionic tracers have been described in the literature. Murray and others (1981) used lithium bromide (LiBr) in carbonate terrane to establish hydraulic connection between a landfill and a freshwater spring, where use of Rhodamine WT dye tracer proved inappropriate. Mather and others (1969) used sodium chloride (NaCl) to investigate the influence of mining subsidence on the pattern of ground-water flow. Tennyson and Settergren (1980) used bromide (Br-) to evaluate pathways and transit time of recharge through soil at a proposed sewage effluent irrigation site. Schmotzer and others (1973) used post-sampling neutron activation to detect a Br- tracer. Chloride (Cl-) and calcium (Ca+) were used by Grisak and Pickens (1980b) to study solute transport mechanisms in fractures. Potassium (K+) was used to determine leachate migration and the extent of dilution by receiving waters located by a waste disposal site (Ellis, 1980).

Non-ionic organic compounds that are not dyes (see below) have received little attention as injected tracers. The ubiquitousness of trace levels of organic contaminants such as methylene chloride creates some problems in evaluating the integrity of clay liners at waste disposal sites. Ilgenfritz and others (1988) have suggested using fluorobenzene as a field monitoring tracer because it would not be likely to occur in normal industrial and commercial activities.

Dyes

Dyes are relatively inexpensive, simple to use, and effective. Either fluorescent or nonfluorescent dyes may be useful in studies of water movement in soil if the soil material that has absorbed the dye is excavated and visually inspected. Fluorescent dyes are preferable to nonfluorescent varieties in ground-water tracer studies because they are easier to detect. Dole (1906) was the first recommended use of dyes to study ground water in the United States by reporting the results of fluorescein and other dyes used in France beginning around 1882. Stiles and others (1927) conducted early experiments using uranine (fluorescein) to demonstrate pollution of wells in a sandy aquifer, and Meinzer (1932) described use of fluorescein as a ground-water tracer. However, extensive use of fluorescent dyes for ground-water tracing did not begin until after 1960. Quinlan (1986) provides a concise, but comprehensive, guide to the literature on dye tracing.

The advantages of using fluorescent dyes include very high detectability, rapid field analysis, and relatively low cost and low toxicity. Smart and Laidlaw (1977) classified commonly used fluorescent dyes by color: orange (Rhodamine B, Rhodamine WT, and Sulforhodamine B); green (fluorescein, Lissamine FF, and pyranine); and blue—also called optical brighteners. Aley and others (in press) classify dyes according to the detector (also called bug) used to recover them: dyes recovered on cotton include optical brighteners (such as Tinopal 5BM GX, and Phorwhite BBH) and Direct Yellow 96; and dyes recovered on activated charcoal (fluorescein and Rhodamine WT).

The literature on fluorescent dye use is plagued by a lack of consistency in dye nomenclature (Quinlan, 1986). The standard reference to dyes is the Colour Index (CI) (SDC & AATCC, 1971-1982). Most dyes are classified according to the CI generic name (related to method of dyeing) and chemical structure (the CI constitution number). Abrahart (1968, pp. 15-43) provides a concise guide to dye nomenclature. Dyes also are classified according to their use in foods, drugs and cosmetics

(Marmion, 1984). There are numerous commercial names for most dyes. Consequently reported results of dye tracing experiments should always specify (1) the CI generic name or CI constitution number, and (2) the manufacturer and the manufacturer's commercial name. The full name of the dye should be mentioned at least once to distinguish it from other dyes with the same or similar names. For example, in 1985, four structurally different kinds of Rhodamine were sold in the United States under 11 different names by five manufacturers, and there are more than 180 kinds of Direct Yellow dye (Quinlan, 1986).

The first part of the commerical name of a dye should not be confused with the dye itself. For example, Tinopal and Phorwhite are trade names used for whole series of chemically unrelated dyes made by a single company and should be capitalized. Seven chemically different Tinopals and 20 different Phorwhites are currently sold in the United States as optical brighteners (Aley and others, in press).

A particularly confusing point of dye nomenclature is that there are two fluorescein dyes with the same CI name and number, although they do have different (Drug and Cosmetic) D&C designations: fluorescein (C20H12O5)—D&C Yellow 7—and fluorescein sodium (C20H12O5Na2)—D&C Yellow 8. Only D&C Yellow 8 is soluble in water and, therefore, suitable for groundwater tracing. In the American and British literature this is referred to as fluorescein, whereas in the European literature it is called uranine (Quinlan, 1986).

Although fluorescent dyes exhibit many of the properties of an ideal tracer, a number of factors interfere with concentration measurement. Fluorescence is used to measure dye concentration, but the amount of fluorescence may vary with suspended sediment load, temperature, pH, $CaCO_3$ content, salinity, etc. Other variables that affect tracer test results are "quenching" (some emitted fluorescent light is reabsorbed by other molecules), adsorption, and photochemical and biological decay. A disadvantage of fluorescent dyes in tropical climates is poor performance because of chemical reactions with dissolved carbon dioxide (Smart and Smith, 1976).

Fluorescence intensity is inversely proportional to temperature. Smart and Laidlaw (1977) described the numerical relationship and provided temperature correction curves. Low pH tends to reduce fluorescence. Figure 4-5 shows that the fluorescence of Rhodamine WT decreases rapidly at increasingly acidic pHs below about 6.0. An increase in the suspended sediment concentration also generally causes a decrease in fluorescence.

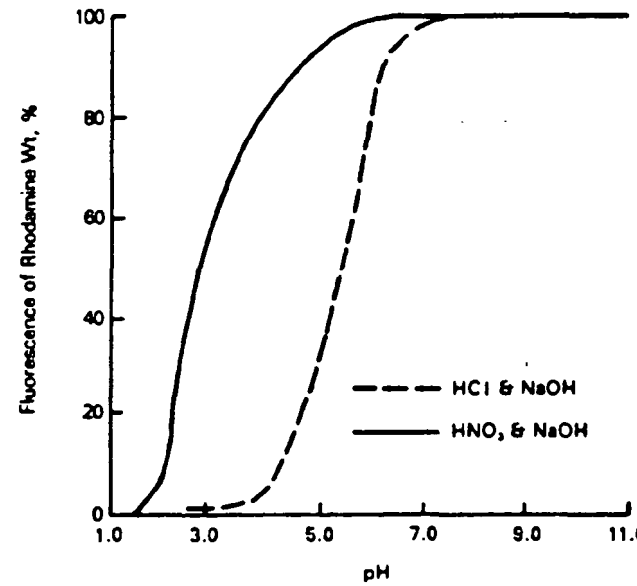


Figure 4-5. The Effect of pH on Rhodamine WT (adapted from Smart and Laidlaw, 1977)

Dyes travel slower than water due to adsorption, and are generally not as conservative as radioactive tracers or some of the ionic tracers. Adsorption can occur on organic matter, clays (bentonite, kaolinite, etc.), sandstone, limestone, plants, plankton, and even glass sample bottles. However, the detected fluorescence may decrease or actually increase due to adsorption. Adsorption on kaolinite caused a decrease in the measured fluorescence of several dyes, as measured by Smart and Laidlaw (1977). If dye is adsorbed onto suspended solids, and the fluorescence measurements are taken without separating the water samples from the sediment, the dye concentration is a measure of sediment content rather than water flow.

These possible adsorption effects are a strong incentive to choose a dye that is nonsorptive for the type of medium tested. Different dyes vary greatly in amount of sorption on specific materials. For example, Repogle and others (1966) measured sorption of three orange dyes on bentonite clay with the following results: Rhodamine WT, 28%; Rhodamine B, 65%; and Sulforhodamine B, 96%.

In a review of the toxicity of 12 fluorescent dyes Smart (1984) identified only three tracers (Tinopal CBS-X, Fluorescein, and Rhodamine WT) with no demonstrated carcinogenic or mutagenic hazard. Use of Rhodamine B was not recommended because it is a known

carcinogen. Use of the other dyes was considered acceptable provided normal precautions are observed during dye handling. Aulenbach and others (1978) concluded that Rhodamine B should not be used as a ground-water tracer simply on the basis of sorption losses.

Currently, the U.S. Geological Survey has a policy of limiting the maximum concentration of fluorescent dyes at water-user withdrawal points to 0.01 ppm (Hubbard and others, 1982). This is a conservative, non-obligatory limit, and Field and others (1990) recommend that tracer concentrations not exceed 1 ppm for a period in excess of 24 hours in ground water. Dyes should probably not be used where water supplies are chlorinated because dye molecules may react with chlorine to form chlorophenols (Smart and Laidlaw, 1977). Field and others (1990) recommend careful evaluation of a tracer before use in a sensitive or unique ecosystem.

General references on fluorescent dye use are three U.S. Geological Survey publications (Hubbard and others, 1982; Kilpatrick and Cobb, 1985; Wilson and others, 1986), reviews by Smart and Laidlaw (1977) and Jones (1984), and two reports prepared for EPA (Mull, 1988; Quinlan, 1989). Aley and Fletcher (1976) remains a classic but outdated text on practical aspects of dye tracing; it will be replaced by The Joy of Dyeing (Aley and others, in press) when that compendium is published.

Fluorescein, also known as uranine, sodium fluorescein, and other names, has been one of the most widely used green dyes. Like all green dyes, its use is commonly complicated by high natural background fluorescence, which lowers sensitivity of analyses and makes interpretation of results more difficult. Feuerstein and Selleck (1963) recommend that fluorescein be restricted to short-term studies of only the highest quality water.

Lewis and others (1966) used fluorescein in a fractured rock study. Mather and others (1969) recorded its use in a mining subsidence investigation in South Wales. Tester and others (1982) used fluorescein to determine fracture volumes and diagnose flow behavior in a fractured granitic geothermal reservoir. They found no measurable adsorption or decomposition of the dye during the 24-hr exposures to rocks at 392°F. At the other extreme, Rahe and others (1978) did not recover any injected dye in their hillslope studies, even at a distance of 2.5 m downslope from the injection point. The same experiment used bacterial tracers successfully.

Another green fluorescent dye, pyranine, has a stronger fluorescent signal than does fluorescein, but is much more expensive. It has been used in several soil studies. Reynolds (1966) found pyranine to be the most stable dye for use in an acidic, sandy soil. Drew and Smith (1969) stated that pyranine is not as easily detectable as fluorescein, but is more resistant to decoloration and adsorption. Pyranine has a very high photochemical decay rate, and is strongly affected by pH in the range found in most natural waters (McLaughlin, 1982).

Rhodamine WT has been considered one of the most useful tracers for quantitative studies, based on minimum detectability, photochemical and biological decay rates, and adsorption (Knuttson, 1968; Smart and Laidlaw, 1977; Wilson and others, 1986). Rhodamine WT is the most conservative dye available for stream tracing (Hubbard and others, 1982). Fluorescein is the most common dye used for tracing ground water in karst.

Aulenbach and others (1978) compared Rhodamine B, Rhodamine WT, and tritium as tracers in effluent from a sewage treatment plant that was applied to natural delta sand beds. The Rhodamine B was highly adsorbed, while the Rhodamine WT and tritium yielded similar breakthrough curves. Aulenbach and Clesceri (1980) found Rhodamine WT very successful in a sandy medium. Gann and Harvey (1975) used Rhodamine WT for karst tracing in a limestone and dolomite system in Missouri.

Rhodamine B and Sulforhodamine B are poor tracers for use in ground water and most surface waters; it could be said the "B" stands for "bad." Amidorhodamine G is a significantly better tracer; similarly, it can be said that the "G" stands for "good" (personal communication, James Quinlan, ATEC Environmental Consultants, Nashville, TN, July, 1990).

Blue fluorescent dyes, or optical brighteners, have been used in increasing amounts in the past decade in textiles, paper, and other materials to enhance their white appearance. Water that has been contaminated by domestic waste entering septic tank soil absorption fields can be used as a "natural" tracer if it contains detectable amounts of the brighteners. Glover (1972) was the first to describe the use of optical brighteners as tracers in karst environments. Since then, they have been extensively used in the United States (Quinlan, 1986). The tracer Amino G acid is a dye intermediate used in the manufacture of dyes that is sometimes mistakenly classified as an optical brightener (Quinlan, 1986). Amino G acid is now recognized as a carcinogenic and should not be used in water that might be used for drinking (personal communication, James Quinlan, ATEC Environmental Consultants, Nashville, TN, July,

1990). Smart and Laidlaw (1977) provide detailed information on the characteristics of the optical brightener Photine CU and Amino G acid.

Gases

Numerous natural and artificially produced gases have been found in ground water. Some of the naturally produced gases can be used as tracers, and gas also can be injected into ground water where it dissolves and can be used as a tracer. Only a few examples of gases being used as ground-water tracers are found in the literature, however. Table 4-4 lists possible gases to use in hydrogeologic studies. Gases are useful tracers in the saturated zone. They are less reliable in the unsaturated zone because bleeding into the atmosphere can give falsely negative results.

Inert Natural Gases. Because of their nonreactive and nontoxic nature, noble gases are potentially useful tracers. Helium is used widely as a tracer in industrial processes. Carter and others (1959) studied the feasibility of using helium as a tracer in ground water and found that it traveled at a slightly lower velocity than chloride. Advantages of using helium as a tracer are its (1) safety, (2) low cost, (3) relative ease of analysis, (4) low concentrations required, and (5) chemical inertness. Disadvantages identified by Carter and others (1959) include (1) relatively large errors in analysis, (2) difficulties in maintaining a constant recharge rate, (3) time required to develop equilibrium in unconfined aquifers, and (4) possible loss to the atmosphere in unconfined aquifers. Neon, krypton, and xenon are other possible candidates for injected tracers because their natural concentrations are very low (Table 4-4). Although the gases do not undergo chemical reactions and do not participate in ion exchange, the heavier noble gases (krypton and xenon) do sorb to some extent on clay and organic material. The solubility of the noble gases decreases with increases in temperature. Therefore, the natural concentrations of these gases in ground water are an indication of surface temperatures at the time of water infiltration. This property has been used to reconstruct palaeoclimatic trends in a sandstone aquifer in England using argon and krypton for age estimates (Andrews and Lee, 1979). Sugisaki (1969) and Mazor (1972) also have used natural inert gases in this way.

Anthropogenic Gases. Numerous artificial gases have been manufactured during the past decade, and several of them have been released in sufficient volumes to produce measurable concentrations in the atmosphere on a worldwide scale. One of the most interesting groups of these gases is the fluorocarbons. These gases generally pose a very low biological hazard, are generally stable for periods measured in years, do not react chemically with other materials, can be detected in very low concentrations, and sorb only slightly on most minerals. They do sorb strongly, however, on organic matter.

Fluorocarbons have two primary applications. First, because large amounts of fluorocarbons were not

	Approximate Natural Background Assuming Equilibrium with Atmosphere at 20°C (mg gas/L water)	Maximum Amount in Solution Assuming 100% Gas at Pressure of 1 atm at 20°C (mg gas/L water)
Argon	0.57	60.6
Neon	1.7×10^{-4}	9.5
Helium	8.2×10^{-6}	1.5
Krypton	2.7×10^{-4}	234
Xenon	5.7×10^{-5}	658
Carbon monoxide	6.0×10^{-6}	28
Nitrous oxide	3.3×10^{-4}	1,100

Source: Davis and others (1985)

Table 4-4. Gases of Potential Use as Tracers

released into the atmosphere until the later 1940s and early 1950s, the presence of fluorocarbons in ground water indicates that the water was in contact with the atmosphere within the past 30 to 40 years (Thompson and Hayes, 1979). The second application of fluorocarbon compounds is as injected tracers (Thompson and others, 1974). Because detection limits are so low, large volumes of water can be labeled with the tracers at a rather modest cost. Despite the problem of sorption on natural material and especially on organics, initial tests have been quite encouraging.

Isotopes

An isotope is any of two or more forms of the same element having the same atomic number and nearly the same chemical properties but with different atomic weights and different numbers of neutrons in the nuclei. Isotopes may be stable (they do not emit radiation) or radioactive (they emit alpha, beta, and/or gamma rays). There are over 280 isotopic forms of stable elements and 40 or so radioactive isotopes (Glasstone, 1967). A wide variety of stable and radioactive isotopes have been used in ground-water tracer studies. There is an extensive literature on the use of isotopes in ground-water investigations; Table 4-5 lists 15 general sources of information. Isotopes have beenused mainly in porous media to study regional ground-water flow regimes and measure aquifer parameters. Back and Zoetl (1975) and LaMoreaux and others (1984) review use of isotopes in karst hydrologic systems. Lack of familiarity with techniques to analyze environmental isotopes has limited their use by practicing field

Reference	Description
Back and Cherry (1976)	Contains a brief review of use of environmental isotopes in ground-water studies.
Csallany (1966)	Early review paper on use of radioisotopes in water resources research.
Davis and Bentley (1982)	Review paper on ground-water dating techniques.
Ferronsky and Polyakov (1982)	Text on use of environmental isotopes in the study of water.
Fritz and Fontes (1980, 1986)	Handbook on environmental isotope geochemistry (two volumes).
Gaspar and Oncescu (1972)	Text on use of radioactive tracers in hydrology (14 chapters).
IAEA (1963)	Symposium on radioisotopes in hydrology.
IAEA (1966)	Symposium on isotopes in hydrology with 21 papers on subsurface hydrology.
IAEA (1967)	Symposium on radioisotope tracers in industry and geophysics contains a number of papers related to ground-water applications.
IAEA (1970)	Symposium on isotopes in hydrology with 25 papers on subsurface hydrology.
IAEA (1974)	Symposium on isotopes in ground-water hydrology with 51 papers.
IAEA (1978)	Symposium on isotopes in hydrology with 41 papers on subsurface hydrology.
Moser and Rauert (1985)	Review paper on use of environmental isotopes for determining ground-water movement.
Wiebenga and others (1967)	Review paper on use of radioisotopes in ground-water tracing.

Table 4-5. Sources of Information on Uses of Isotopes in Ground-Water Tracing

hydrogeologists ground-water contamination studies. Hendry (1988) recommends the use of hydrogen and oxygen isotopes as a relatively inexpensive way to estimate the age of near-surface ground-water samples.

Stable Isotopes. Stable isotopes are rarely used for artificially injected tracer studies in the field because (1) it is difficult to detect small artificial variations of most isotopes against the natural background, (2) their analysis is costly, and (3) preparing isotopically enriched tracers is expensive. The average stable isotope composition of deuterium (2H) and 180 in precipitation changes with elevation, latitude, distance from the coast, and temperature. Consequently, measurement of these isotopes in ground water can be used to trace the large-scale movement of ground water and to locate areas of recharge (Gat, 1971; Ferronsky and Polyakov, 1982).

The two abundant isotopes of nitrogen (14N and 15N) can vary significantly in nature. Ammonia (NH_4) escaping as vapor from decomposing animal wastes, for example, will tend to remove the lighter (14N) nitrogen and will leave behind a residue rich in heavy nitrogen. In contrast, many fertilizers with an ammonia base will be isotopically light. Natural soil nitrate will be somewhat between these two extremes. As a consequence, nitrogen isotopes have been used to determine the origin of unusually high amounts of nitrate in ground water. Also, the presence of more than about 5 mg/L of nitrate is commonly an indirect indication of contamination from chemical fertilizers and sewage.

The stable sulfur isotopes (32S, 34S, and 36S) have been used to distinguish between sulfate originating from natural dissolution of gypsum ($CaSO_4 \cdot 2H_2O$) and sulfate originating from an industrial spill of sulfuric acid (H_2SO_4).

Two stable isotopes of carbon (12C and 13C) and one radioisotope (14C) are used in hydrogeologic studies. Although not as commonly studied as 14C, the ratio of the stable isotopes, 13C/12C, is potentially useful in sorting out the origins of certain contaminants found in water. For example, methane (CH_4) originating from some deep geologic deposits is isotopically heavier then methane originating from near-surface sources. This contrast forms the basis for identifying aquifers contaminated with methane from pipelines and from subsurface storage tanks.

Isotopes of other elements such as chlorine, strontium, and boron are used to determine regional directions of ground-water flow rather than to identify sources of contamination.

Radionuclides. Radioactive isotopes of various elements are collectively referred to as radionuclides. In the early 1950s there was great enthusiasm for using radionuclides both as natural "environmental" tracers and as injected artificial tracers. The use of artificially injected radionuclides has all but ceased in many countries, including the United States, however, because of concerns about possible adverse health effects (Davis and others, 1985). Artificially introduced radioactive tracing mostly is confined to carefully controlled laboratory experiments or to deep petroleum production zones that are devoid of potable water. Table 4-6 lists eight radionuclides commonly used as injected tracers, their half-lives, and the chemical form in which they are typically used.

Radionuclide	Half-Life y = year d = day h = hour	Chemical Compound
2H	12.3y	H_2O
32p	14.3d	Ha_2HPO_4
51Cr	27.8d	EDTA-Cr and CrC13
60Co	.25y	EDTA-Co and $K_3Co(CN_6)$
82Br	33.4h	NH_4Br, NaBr,LiBr
85Kr	10.7y	Kr (gas)
131I	8.1d	I and KI
98Au	2.7d	$AuCl_3$

Source: Davis and others (1985)

Table 4-6. Commonly Used Radioactive Tracers for Ground-Water Studies

The use of natural environmental tracers has expanded so that they are now a major component of many hydrochemical studies. A number of radionuclides are present in the atmosphere from natural and artificial sources, and many of these are carried into the subsurface by rain water. The most common hydrogeologic use of these radionuclides is to estimate the average length of time ground water has been isolated from the atmosphere. This measurement is complicated by dispersion in the aquifer and mixing in wells that sample several hydrologic zones. Nevertheless, the age of water in an aquiferusually can be established as being older than some given limiting value. For example, detection of atmospheric radionuclides might indicate that ground water was recharged more than 1,000 years ago or that, in another region, all the ground water in a given shallow aquifer is younger than 30 years.

Since the 1950s, atmospheric tritium, the radioactive isotope of hydrogen (3H) with a half life of 12.3 years, has been dominated by tritium from the detonation of thermonuclear devices. Thermonuclear explosions increased the concentration of tritium in local rainfall to more than 1,000 tritium units (TU) in the northern hemisphere by the early 1960s (Figure 4-6). As a result, ground water in the northern hemisphere with more than about 5 TU is generally less than 30 years old. Very small amounts of tritium, 0.05 to 0.5 TU, can be produced by natural subsurface processes, so the presence of these low levels does not necessarily indicate a recent age.

The radioactive isotope of carbon, 14C (with a half-life of 5,730 years), is also widely studied in ground water. In practice, the use of 14C is rarely simple. Sources of old carbon, primarily from limestone and dolomite, will dilute the sample, and a number of processes, such as the formation of CH_4 gas or the precipitation of carbonate minerals, will fractionate the isotopes and alter the apparent age. Interpreting 14C "ages" of water is so complex that it should be attempted only by hydrochemists specializing in isotope hydrology. Despite the complicated nature of 14C studies, they are highly useful in determining the approximate residence time of old water (500 to 30,000 years) in aquifers. In certain circumstances, this information cannot be obtained in any other way.

Inert Radioactive Gases. Chemically inert but radioactive 133Xe and 85Kr appear to be suitable for many injected tracer applications (Robertson, 1969; Wagner, 1977), provided legal restrictions can be overcome. 222Rn, one of the daughter products from the spontaneous fission of 238U, is the most abundant of the natural inert radioactive gases. Radon is present

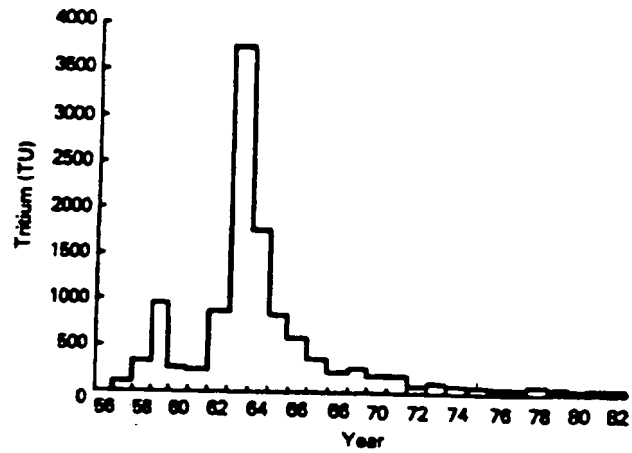


Figure 4-6. Average Annual Tritium Concentration of Rainfall and Snow for Arizona, Colorado, New Mexico, and Utah (from Davis and others, 1985, after Vuataz and others, 1984)

in the subsurface, but owing to the short half-life (3.8 d) of 222Rn, and the absence of parent uranium nuclides in the atmosphere, radon is virtually absent in surface water that has reached equilibrium with the atmosphere. Surveys of radon in surface streams and lakes have, therefore, been useful in detecting locations where ground water enters surface waters (Rogers, 1958). Hoehn and von Gunten (1989) measured dilution of radon in ground water to assess infiltration from surface waters to an aquifer.

Tracer Tests in Karst

Probably no hydrogeologic system has been more extensively studied by a more diverse group of people with such a plethora of tracing techniques as karst limestone terranes. Geese (Aley and Fletcher, 1976), tagged eels (Bögli, 1980), computer punch-card confetti (Davis and others, 1985), and time bombs (Arandjelovich, 1969) are among the more exotic tracers that have been used in karst.

There is an extensive international literature on karst tracing. Table 4-7 describes 18 major sources of general information on this topic. There is a substantial English-language literature in American caving journals, such as Cave Notes/Caves and Karst (which ceased publication in 1973), Missouri Speleology, and the National Speleological Society Bulletin, and similar British periodicals, such as Transactions of the Cave Research Group (now Cave Science), and the Proceedings of the University of Bristol Speleological Society. The international symposia on underground

water tracing (SUWT— see Table 4-7) provide the best systematic compilations of international research on this topic. Probably the easiest way to monitor the international literature on dye-tracing in karst terranes and other karst and speleological literature is the annual Speleological Abstracts published by the Union Internationale de Spèléologie in Switzerland.

Table 4-8 summarizes information on the most commonly used water tracers in North American karst studies. Dyes are almost ideal tracers because the adsorption is usually not a problem in karst hydrogeologic systems. Smart (1985) lists four applications of fluorescent dye tracers in evaluating existing or potential contamination in carbonate rocks: (1) confirmation of leachate contamination, (2) determination of on site hydrology, (3) determination of hydraulic properties of landfill materials, and (4) prediction of leachate contamination and dilution.

Fluorescein, Rhodamine WT, optical brighteners (Tinopal 5BM GX), and Direct Yellow 96 are the most commonly used dyes. The amount of dye injected depends on whether qualitative or quantitative analysis is planned. Qualitative tests involve simple visual detection of dye in flowing water or captured by a

Reference	Description
Aley and Fletcher (1976)	Classic guide to use of tracers in karst. Should be replaced by Aley et al. (in press) when it is published.
Aley and others (in press)	Compendium of techniques for ground-water tracing focusing on karst terranes.
Back and Zoetl (1975)	Review of the use of geochemical, isotopic, dye, spore, and artificial radioisotopes as tracers in karst systems.
Bögli (1980)	Pages 138-143 review use of tracers in karst hydrology.
Brown (1972)	Chapter III reviews tracer methods in karst hydrologic systems.
Gospodaric and Habic (1976)	Pages 217-230 contain reviews of the applicability of dyes, salts, radionuclides, drifting materials, and other tracers in karst.
Gunn (1982)	Review paper on karst water tracing in Ireland.
Jones (1984)	Review paper on use of dye tracers in karst.
LaMoreaux and others	Pages 196-210 of the 1984 annotated bibliography focus (1984, 1989) on isotope techniques for water tracing in carbonate rocks. The 1989 annotated bibliography contains a section reviewing pollution assessment in carbonate terranes.
Milanovia (1981)	Pages 263-309 focus on karst water tracing.
Mull and others (1988)	EPA report on dye-tracing techniques in karst terranes.
Quinlan (1989)	EPA report with recommended dye-tracing protocols for ground-water tracing in karst terranes.
Sweeting (1973)	Pages 218-251 focus on karst water and karst water tracing.
SUWT (1966, 1970, 1976, 1981, and 1986)	Publications related to the various international symposia on underground water tracing (SUWT) contain numerous papers on ground-water tracing techniques, mostly focusing on karst.
Thrailkill and others (1983)	Report focusing on karst dye-tracing techniques.

Table 4-7. Sources of Information on Ground-Water Tracing in Karst Systems

Tracer & Color	Passive Detector	Test (elutriant)	Maximum Excitatin & Emission nm	Detectable Conc.	Advantages	Disadvantages	Remarks
Fluorescein Sodium $C_{20}H_{10}Na_2O_5$ Yellow-Green Xanthene	Activated coconut charcoal 6-14 mesh	Ethyl alcohol and 5% KOH. Visual test or fluorometer & 2A-47B; 2A-12, 65A filters	485 515	0.1 µg/1 Dependent on background levels. "Controls" must be used to determine background	1) Does not require constant monitoring or any special equipment. 2) Inexpensive.	1) Dye is photochemically unstable. 2) Moderate sorption on clay. 3) pH sensitive.	This is the most popular method used in the USA. Carbon detectors first suggested by Dunn, 1957.
Rhodamine WT $C_{29}H_{26}N_2O_5Cl$ Red-Purple Xanthene	Activated coconut charcoal 6-14 mesh	Ethyl alcohol & 5% KOH or 1-Propanol + NH_4OH. Solution tested using fluorometer and 546-590 filters.	550 580	.01 µg/1. Dependent on background levels and fluctuation.	1) Dye is photochemically stable. 2) Dye may be used in low pH waters.	1) Requires the use of a flurometer. 2) Moderate clay sorption.	Rhodamine has been used extensively in Canada & USA. This is not a suitable method for amateurs without access to a flourometer.
Lycopodium Spores *Lycopodium Calvitum*	Plankton netting nylon-25 micron	Spores & sediment are washed from the nets. Microscopic examination is used to identify spores.	N/A	Dependent on background levels. Several kilograms of spores are usually used.	1) Several simultaneous tests may be conducted using different colored spores 2) No coloring of water occurs.	1) Spores may be prematurely filtered out. 2) Field collection system elaborate. 3) System is generally more expensive.	1) Spores have not been used in North America.
Optical Brighteners Colorless normal light	Unbleached cotton	Visual examination of detectors under UV light or 7-37; 2A + 47B Filters.	360 435	Dependent on background levels, but generally at least .1 µg/1.	1) Inexpensive. 2) No coloring of water occurs	1) Background readings may be excessively high. 2)Adsorbed onto organics.	May be used simultaneously with a green & red dye using fluorometric separation.
Direct Yellow (DY 96) Low Visibility Stilbene derivative	Unbleached cotton	Visual examination of detectors under UV light or 7-37; 2A + 47B Filters	N/A	1.0 µg/1 on cotton, and with fluorometric analysis.	1) Little natural background. 2) Good stability and low sorption. 3) No coloring of water.	1) Moderate cost. 2) Sensitive to pH.	Has been used extensively in Kentucky.
Salt NaCl Colorless	Recording specific conductance meter or regular sampling	Either a direct test for an increase in chloride, or a substantial increase in specific conductance	N/A	Dependent on background levels. Several hundred kilograms may be needed for larger tests.	1) Generally considered safe for use on public water systems. 2) Useful where fluorescent background conditions exclude other methods	1) Large quantities usually needed. 2) Background specific conductance is often high.	Salt is occasionally used by the US Geological Survey for tests dealing with public water supplies.

[1] G.K. Turner Filters for Turner 111 Filter Fluorometer.
[2] Dye is usually most visible in clear water, deep pools, and in bright sunlight. These figures are not exact.
[3] Very dilute dye solutions may be concentrated upon the detector over a period of time.

Source: Jones, 1984.

Table 4-8. Evaluation of Principal Water Tracers Used in North American Karst Studies

detector (see discussion below). Semi-quantitative results can be obtained by using a fluorometer or spectrofluorometer to detect amounts of dye captured by detectors such as activated charcoal that may not be discernible to the eye. Interpretation of values from such measurements is limited due to lack of precise information on the variation in ground water flow and dye concentration between collection of detectors. Quantitative tests involve precise measurement of dye concentrations in grab samples of water. If the exact amount of injected dye is known, and flow measurements are taken along with each sample, a mass-balance analysis allows estimation of how much dye has been distributed through different parts of the subsurface flow system.

In qualitative tests, enough dye must be injected for visual detection; quantitative tests using a fluorometer or spectrofluorometer generally require one-tenth to one hundredth as much dye. Determination of the

correct quantity to inject is as much an art as a science, and this should be determined by, or with the assistance of, someone with experience in karst tracer tests.

Dye is recovered with detectors called bugs (cotton or activated charcoal, depending on the tracer), that are typically suspended in streams and springs on hydrodynamically stable stands called gumdrops. Detectors are placed at springs or in streams where flow from the point of injection is suspected of reaching the surface. At chosen time intervals related to the distance from the source of injection, detectors are collected and replaced with fresh detectors. Detectors are usually collected frequently during the first few days after injection to pinpoint the most rapid dye arrival time, and then typically on a daily basis for several weeks. Background tests always must be run before injection, especially with optical brighteners because sewage effluent from individual septic tank absorption fields may increase background levels substantially.

Qualitative tracer tests in which two dyes are injected into two different locations are readily done by combining a fluorescent dye and an optical brightener, which use different detectors. Quantitative techniques are available (developed originally in Europe) for separating mixtures of fluorescent dyes (Quinlan, 1986). A 5-dye tracer test has recently been conducted using these techniques (personal communication, James Quinlan, January 1990). Perhaps the most comprehensive karst tracing experiments in a single location were carried out in Slovenia, Yugoslavia, in the early 1970s where five dyes, lycopodium spores, lithium chloride, potassium chloride, chromium-51, and detergents all were used (Gospodaric and Habic, 1976).

Reports prepared for EPA by Mull and others (1988) and Quinlan (1989) are the most comprehensive references currently available on procedures for dye-tracing in karst terranes. Aley and others (in press) should be obtained when it becomes available. Smoot and others (1987), and Smart (1988a) describe quantitative dye-tracing techniques in karst, and Smart (1988b) describes an approach to the structural interpretation of ground-water tracers in karst terrane.

Tracer Tests In Porous Media

Tracer tests in porous media are used primarily to characterize aquifer parameters such as regional velocity (Leap, 1985), hydraulic conductivity distributions (Molz and others, 1988), anisotropy (Kenoyer, 1988), dispersivity (Bumb and others, 1985), and distribution coefficient or retardation (Pickens and others, 1981; Rainwater and others, 1987). Smart and others (1988) have prepared an annotated bibliography on ground-water tracing that focuses on use of tracers in porous media.

The purpose and practical constraints of a tracer test must be clearly understood prior to actual planning. Following are a few of the questions that need to be addressed:

- Is only the direction of water flow to be determined?
- Are other parameters such as travel time, porosity, and hydraulic conductivity of interest?
- How much time is available for the test?
- How much money is available for the test?

If results must be obtained within a few weeks, then certain kinds of tracer tests would normally be out of the question. Those using only the natural hydraulic gradient between two wells that are more than about 20 m apart typically require long time periods for the tracer to flow between the wells. Another primary consideration is budget. Costs for tests that involve drilling several deep holes, setting packers to control sampling or injection, and analyzing hundreds of samples in an EPA-certified laboratory could easily exceed $1 million. In contrast, some short-term tracer tests may cost less than $1,000.

Choice of a tracer will depend partially on which analytical techniques are easily available and which background constituents might interfere with these analyses. The chemist or technician who will analyze the samples can advise whether background constituents might interfere with the analytical techniques to be used. Bacteria, isotopes, and ions are the most frequently used types of tracers in porous media. Fluorescent dyes are less commonly used as tracers because they tend to adsorb. A more common use of dyes in porous media is to locate zones of preferential flow in the vadose zone. In this application, adsorption on soil particles is desirable because it allows visual inspection of flow patterns when the soil is excavated.

Estimating the Amount of Tracer to Inject

The amount of tracer to inject is based on the natural background concentrations, the detection limit for the tracer, the dilution expected, and experience. Adsorption, ion exchange, and dispersion will decrease the amount of tracer arriving at the observation well, but recovery of the injected mass is usually not less than 20 percent for two-hole tests using a forced recirculation system and conservative tracers. The concentration should not be increased so much that density effects become a problem. Lenda and Zuber (1970) presented graphs that can be used to estimate the approximate

quantity of tracer needed. These values are based on estimates of the porosity and dispersion coefficient of the aquifer.

Single-Well Techniques

Two techniques, injection/withdrawal and borehole dilution, produce parameter values from a single well that are valid at a local scale. Advantages of single-well techniques are:

- Less tracer is required than for two-well tests.
- The assumption of radial flow is generally valid, so natural aquifer velocity can be ignored, making solutions easier.
- Knowledge of the exact direction of flow is not necessary.

Molz and others (1985) describe design and performance of single-well tracer tests conducted at the Mobile site.

Injection/Withdrawal. The single-well injection/withdrawal (or pulse) technique can be used to obtain a pore velocity value and a longitudinal dispersion coefficient. The method assumes that porosity is known or can be estimated with reasonable accuracy. In this procedure, a given quantity of tracer is instantaneously added to the borehole, the tracer is mixed, and then two to three borehole volumes of freshwater are pumped in to force the tracer to penetrate the aquifer. Only a small quantity is injected so as not to disturb natural flow.

After a certain time, the borehole is pumped out at a constant rate large enough to overcome the natural ground-water flow. Tracer concentration is measured with time or pumped volume. If the concentration is measured at various depths with point samplers, the relative permeability of layers can be determined. The dispersion coefficient is obtained by matching experimental breakthrough curves with theoretical curves based on the general dispersion equation. A finite difference method is used to simulate the theoretical curves (Fried, 1975).

Fried concluded that this method is useful for local information (2-to 4-m radius) and for detecting the most permeable strata. A possible advantage of this test is that nearly all of the tracer is removed from the aquifer at the end of the test.

Borehole Dilution. This technique, also called point dilution, can be used to measure the magnitude and direction of horizontal tracer velocity and vertical flow (Fried, 1975; Gaspar and Oncescu, 1972; Klotz and others, 1978).

The procedure introduces a known quantity of tracer instantaneously into the borehole, mixes it well, and then measures the concentration decrease with time. The tracer is generally introduced into an isolated volume of the borehole using packers. Radioactive tracers have been most commonly used for borehole dilution tests, but other tracers can be used.

Factors to consider when conducting a point dilution test include the homogeneity of the aquifer, effects of drilling (mudcake, etc.), homogeneity of the mixture of tracer and well water, degree of tracer diffusion, and density effects.

Ideally, the test should be conducted using a borehole with no screen or gravel pack. If a screen is used, it should be next to the borehole because dead space alters the results. Samples should be very small in volume so that flow is not disturbed by their removal.

A variant of the point dilution method allows measurement of the direction of ground-water flow. In this procedure, a section of the borehole is usually isolated by packers, and a tracer (often radioactive) is introduced slowly and without mixing. Then, after some time, a compartmental sampler (four to eight compartments) within the borehole is opened. The direction of minimum concentration corresponds to the flow direction. A similar method is to introduce a radioactive tracer and subsequently measure its adsorption on the borehole or well screen walls by means of a counting device in the hole. Gaspar and Oncescu (1972) describe the method in more detail.

Another common strategy is to inject and subsequently remove the water containing a conservative tracer from a single well. If injection is rapid and immediately followed by pumping to remove the tracer, then almost all of the injected conservative tracer can be recovered. If the pumping is delayed, the injected tracer will drift downgradient with the general flow of the ground water and the percentage of tracer recovery will decrease with time. Successive tests with increasingly longer delay times between injection and pumping can be used to estimate ground-water velocities in permeable aquifers with moderately large hydraulic gradients.

Two-Well Techniques

There are two basic approaches to using tracers with multiple wells: one measures tracer movement in uniform (natural) flow and the other measures movement

by radial (induced) flow. The parameters measured (dispersion coefficient and porosity) are assumed to be the same for both types of flow.

Uniform Flow. This approach involves placing a tracer in one well without disturbing the flow field, and sampling periodically to detect the tracer in observation wells. This test can be used at a local (2 to 5 m) or intermediate (5 to 100 m) scale, but it requires much more time than radial tests. If the direction and magnitude of the velocity are not known, a large number of observation wells are needed. Furthermore, local flow directions may diverge widely from directions predicted on the basis of widely spaced water wells. Failure to intercept a tracer in a well just a few meters away from the injection well is not uncommon under natural-gradient flow conditions.

The quantity of tracer needed to cover a large distance can be expensive. On a regional scale, environmental tracers, including seawater intrusion, radionuclides, or stable isotopes of hydrogen and oxygen, are used. Manmade pollution also has been used. For regional problems, a mathematical model is calibrated with concentration versus time curves from field data, and is used to predict future concentration distributions.

Local- or intermediate-scale uniform flow problems can be solved analytically, semianalytically, or by curve-matching. Layers of different permeability can cause distorted breakthrough curves, which can usually be analyzed using one- or two-dimensional models (Gaspar and Oncescu, 1972). Fried (1975) and Lenda and Zuber (1970) present analytical solutions.

Radial Flow. Radial flow techniques work by altering the flow field of an aquifer through pumping. Solutions are generally easier if radial flow velocity greatly exceeds uniform flow. This method yields values for porosity and the dispersion coefficient, but not natural ground-water velocity. Types of radial flow tests include diverging, converging, and recirculating tests.

A diverging test involves constant injection of water into an aquifer. The tracer is introduced into the injected water as a slug or continuous flow and the tracer is detected at an observation well that is not pumping. Point or integrated samples of small volume are carefully taken at the observation well so that flow is not disturbed. Packers can be used in the injection well to isolate an interval.

In a converging test, the tracer is introduced at an observation well, while another well is pumped. Concentrations are monitored at the pumped well. The tracer often is injected between two packers or below one packer; then two to three well-bore volumes are injected to push the tracer out into the aquifer. At the pumping well, intervals of interest are isolated (particularly in fractured rock), or an integrated sample is obtained.

A recirculating test is similar to a converging test, but the pumped water is injected back into the injection well. This tests a significantly greater part of the formation because the wells inject to and pump from 360 degrees. The flow lines are longer, however, partially canceling out the advantage of a higher gradient. Sauty (1980) provides theoretical curves for recirculating tests.

Design and Construction of Test Wells

In many tracer tests, construction of the test wells is the single greatest expense. Procedures for the proper design and construction of monitoring wells for sampling ground-water quality (discussed in Chapter 3) apply equally to wells used for tracer tests.

Special considerations for designing and constructing test wells for tracer tests include:

- Drilling muds and mud additives tend to have a high capacity for the sorption of most types of tracers and, therefore, should be avoided.
- Drilling methods that alter the hydrologic characteristics of the aquifer being tested (such as clogging of pores) should be avoided.
- Use of packers to isolate the zones being sampled from the rest of the water in the well (see Figure 4-2b) allows the most precise measurements of vertical variations in hydrologic parameters. This approach tends to be more expensive, takes longer, and requires more technical training than whole-well tests.
- If packers are not used, the diameter of the sampling well should be as small as possible so that the amount of "dead" water in the well during sampling is minimized.
- Well casing material should not be reactive with the tracer used.
- Well-screen slot size and gravel pack must be selected and installed with special care when using single-well tests with alternating cycles of injection and pumping large volumes of water into and out of loose fine-grained sand. On the other hand, if the aquifer being tested contains a very permeable coarse gravel and the casing

diameter is small, then numerous holes drilled in the solid casing may be adequate.

As with any monitoring well, tracer test wells should be properly developed to remove silt, clay, drilling mud, and other materials that would prevent free movement of water in and out of the well.

Injection and Sample Collection

Choice of injection equipment depends on the depth of the borehole and the funds available. In very shallow holes, the tracer can be lowered through a tube, placed in an ampule that is lowered into the hole, and broken, or just poured in. Mixing of the tracer with the aquifer water is desirable and important for most types of tests and is simple for very shallow holes. For example, a plunger can be surged up and down in the hole or the tracer can be released through a pipe with many perforations. Flanges on the outer part of the pipe will mix the tracer as the pipe is raised and lowered. For deeper holes, tracers must be injected under pressure and equipment can be quite sophisticated.

Sample collection also can be simple or sophisticated. For tracing thermal pulses, only a thermistor needs to be lowered into the ground water. For chemical tracers, a variety of sampling methods may be used. Some special sampling considerations for tracer tests include:

- Bailers should not be used if mixing of the tracer in the borehole is to be avoided.
- Where purging is required, removal of more than the minimum required to obtain fresh aquifer water may create a gradient towards the well and distort the natural movement of the tracer.
- Use of existing water wells that tap multiple aquifers should be generally avoided in tracer tests except to establish whether a hydrologic connection with the point of injection exists.

Interpretation of Results

This section provides a brief qualitative introduction to the interpretation of tracer test results. More extensive and quantitative treatments are found in the works of Halevy and Nir (1962), Theis (1963), Fried (1975), Sauty (1978), and Grisak and Pickens (1980a,b). Some more recent papers on analysis of tracer tests include Güven and others (1985, 1986), Molz and others (1986, 1987), and Bullivant and O'Sullivan (1989).

The basic plot of the concentration of a tracer as a function of time or water volume passed through the system is called a breakthrough curve. The concentration either is plotted as the actual concentration (Figure 4-7) or, quite commonly, as the ratio of the measured tracer concentration at the sampling point, C, to the input tracer concentration, C_0 (Figure 4-8).

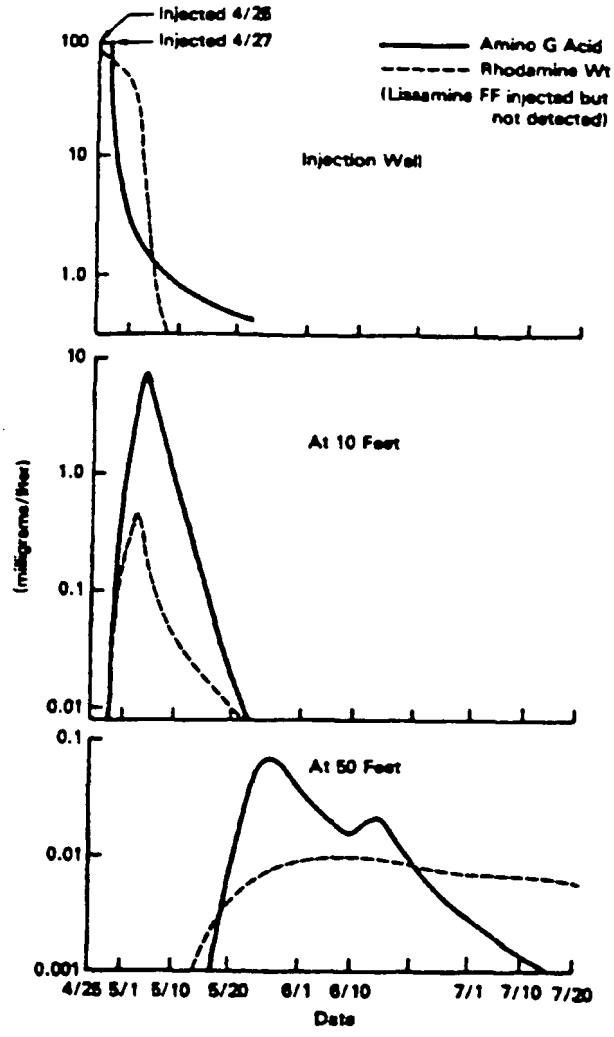


Figure 4-7. Results of Tracer Tests at the Sand Ridge State Forest, Illinois (from Naymik and Sievers, 1983)

The measured quantity that is fundamental for most tracer tests is the first arrival time of the tracer as it goes from an injection point to a sampling point. The first arrival time conveys at least two bits of information. First, it indicates that a connection for ground-water flow actually exists between the two points. For many tracer tests, particularly in karst regions, this is all the information that is desired. Second, if the tracer is conservative, the maximum velocity of ground-water flow between the two points may be estimated.

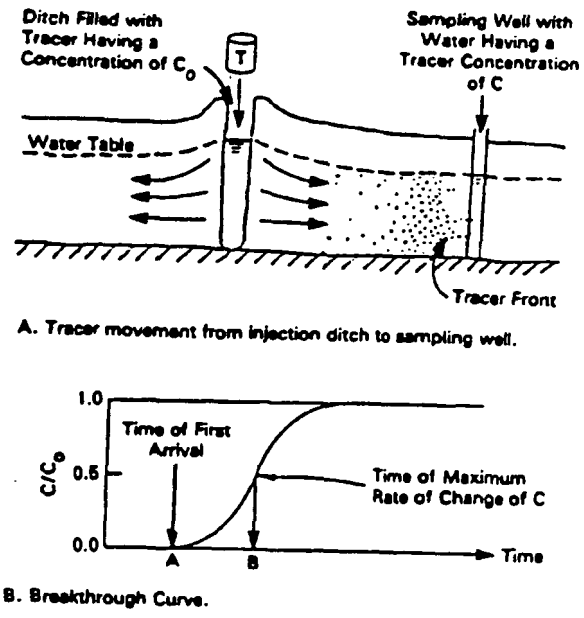


Figure 4-8. Tracer Concentration at Sampling Well, C, Measured Against Tracer Concentration at Input, C_o (from Davis and others, 1985)

Interpretations more elaborate than the two mentioned above depend very much on the type of aquifer being tested, the velocity of ground-water flow, the configuration of the tracer injection and sampling systems, and the type of tracer or mixture of tracers used in the test.

The value of greatest interest after the first arrival time is the arrival time of the peak concentration for a slug injection; or, for a continuous feed of tracers, the time since injection when the concentration of the tracer changes most rapidly as a function of time (Figure 4-8). In general, if conservative tracers are used, this time is close to the theoretical travel time of an average molecule of ground water traveling between the two points.

If a tracer is being introduced continuously into a ditch penetrating an aquifer, as shown in Figure 4-8, then the ratio C/C_o will approach 1.0 after the tracer starts to pass the sampling point. The ratio of 1.0 is rarely approached in most tracer tests in the field, however, because waters are mixed by dispersion and diffusion in the aquifer and because wells used for sampling will commonly intercept far more ground water than has been tagged by tracers (Figure 4-9). Ratios of C/C_o ranging between 10^{-5} and 2×10^{-1} often are reported from field tests.

If a tracer is introduced passively into an aquifer but it is recovered by pumping a separate sampling well, then various mixtures of the tracer and the native ground water will be recovered depending on the amount of water pumped, the transmissivity of the aquifer, the slope of the water table, and the shape of the tracer plume. Keely (1984) has presented this problem graphically with regard to the removal of contaminated water from an aquifer.

With the introduction of a mixture of tracers, possible interactions between the tracers and the solid part of the aquifer may be studied. If interactions take place, they can be detected by comparing breakthrough curves of a conservative tracer with the curves of the other tracers being tested (Figure 4-10). Quantitative analyses of tracer breakthrough curves are generally conducted by curve-matching computer-generated type curves, or by applying analytical methods.

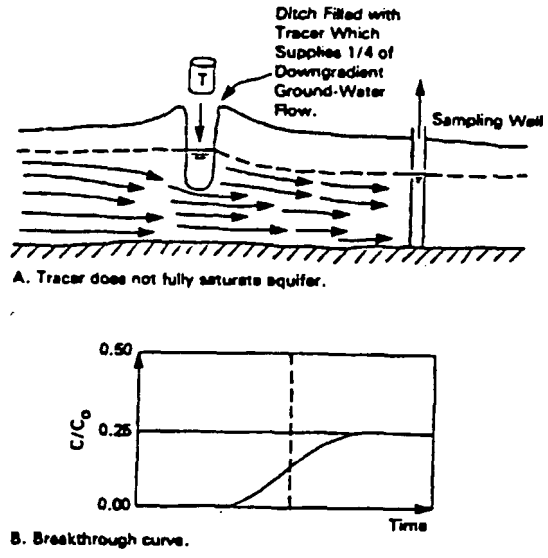


Figure 4-9. Incomplete Saturation of Aquifer with Tracer (from Davis and others, 1985)

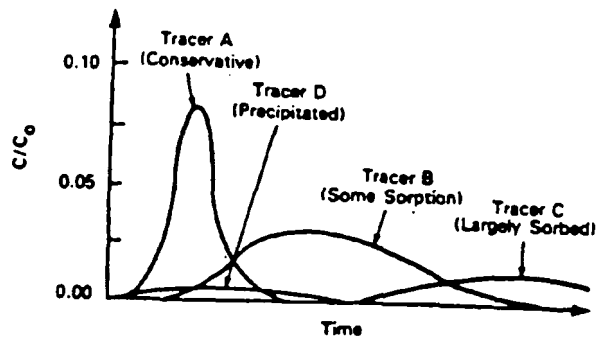


Figure 4-10. Breakthrough Curves for Conservative and Nonconservative Tracers (from Davis and others, 1985)

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Chapter 5

INTRODUCTION TO AQUIFER TEST ANALYSIS

Cone of Depression

Both wells and springs can be ground-water supply sources. However, most springs with yields large enough to meet municipal, industrial, and large commercial and agricultural needs are located only in areas underlain by cavernous limestones and lava flows. Most ground-water needs, therefore, are met by withdrawals from wells.

An understanding of the response of aquifers to withdrawals from wells is important to an understanding of ground-water hydrology. When withdrawals start and water is removed from storage in the well, the water level in the well begins to decline. The head in the well falls below the level in the surrounding aquifer, and water begins to move from the aquifer into the well. As pumping continues, the water level in the well continues to decline, and the rate of flow into the well from the aquifer continues to increase until the rate of inflow equals the rate of withdrawal.

When water moves from an aquifer into a well, a cone of depression is formed (Figure 5-1). Because water must converge on the well from all directions and because the area through which the flow occurs decreases toward the well, the hydraulic gradient must get steeper toward the well.

There are several important differences between cones of depression in confined and unconfined aquifers. Withdrawals from an unconfined aquifer cause drainage of water from the rocks, and the water table declines as the cone of depression forms (Figure 5-1a). Because the storage coefficient of an unconfined aquifer equals the specific yield of the aquifer material, the cone of depression expands very slowly. On the other hand, dewatering of the aquifer results in a decrease in transmissivity, which causes, in turn, an increase in drawdown both in the well and in the aquifer.

Withdrawals from a confined aquifer cause a drawdown in artesian pressure but normally do not cause a dewatering of the aquifer (Figure 5-1b). The water withdrawn from a confined aquifer is derived from expansion of the water and compression of the rock

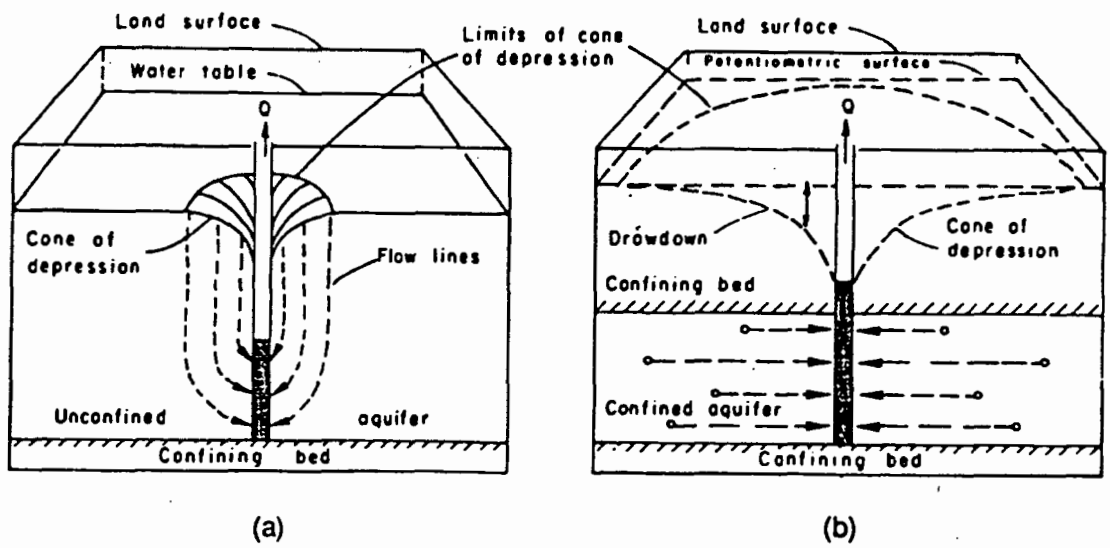

Figure 5-1. Cone of Depression in an Unconfined and a Confined Aquifer

skeleton of the aquifer. The small storage coefficient of confined aquifers results in a rapid expansion of the cone of depression. Consequently, the mutual interference of expanding cones around adjacent wells occurs more rapidly in confined aquifers than it does in unconfined aquifers.

SOURCE OF WATER DERIVED FROM WELLS

Both the economic development and the effective management of any ground-water system require an understanding of the system's response to withdrawals from wells. The first concise description of the hydrologic principles involved in this response was presented by Theis (1940).

Theis pointed out that the aquifer's response to withdrawals from wells depends on:

1. The rate of expansion of the cone of depression caused by the withdrawals, which depends on the transmissivity and the storage coefficient of the aquifer.
2. The distance to areas in which the rate of water discharging from the aquifer can be reduced.
3. The distance to recharge areas in which the rate of recharge can be increased.

Over a sufficiently long period of time and under natural conditions—that is, before the start of withdrawals—the discharge from every ground-water system equals the recharge to it (Figure 5-2a). This property is expressed by the equation:

natural discharge (D) = natural recharge (R)

In the eastern United States and in the more humid areas in the West, the amount and distribution of precipitation are such that the period of time over which discharge and recharge balance may be less than a year or, at most, a few years. In the drier parts of the country—that is, in the areas that generally receive less than about 500 mm of precipitation annually—the period over which discharge and recharge balance may be several years or even centuries. Over shorter periods of time, differences between discharge and recharge involve changes in ground-water storage. When discharge exceeds recharge, ground-water storage (S) is reduced by an amount (ΔS) equal to the difference between discharge and recharge:

$$D = R + \Delta S \quad (1)$$

Conversely, when recharge exceeds discharge, ground-water storage is increased:

$$D = R - \Delta S \quad (2)$$

When withdrawal through a well begins, water is removed from storage in the well's vicinity as the cone of depression develops (Figure 5-2b). Thus, the withdrawal (Q) is balanced by a reduction in ground-water storage:

$$Q = \Delta S \quad (3)$$

As the cone of depression expands outward from the pumping well, it may reach an area where water is discharging from the aquifer. The hydraulic gradient will be reduced toward the discharge area, and the rate of natural discharge will decrease (Figure 5-2c). To the extent that the decrease in natural discharge compensates for the pumpage, the rate at which water is being removed from storage also will decrease, and the rate of expansion of the cone of depression will decline. If and when the reduction in natural discharge (ΔD) equals the rate of withdrawal (Q), a new balance will be established in the aquifer. This balance is represented as:

$$(D-\Delta D)+Q=R \quad (4)$$

Conversely, if the cone of depression expands into a recharge area rather than into a natural discharge area, the hydraulic gradient between the recharge area and the pumping well will increase. If, under natural conditions, more water was available in the recharge area than the aquifer could accept (the condition that Theis referred to as rejected recharge), the increase in the gradient away from the recharge area will permit more recharge to occur, and the rate of growth of the cone of depression will decrease. If the increase in recharge (ΔR) equals the rate of withdrawal (Q), a new balance will be established in the aquifer, and expansion of the cone of depression will cease. The new balance is represented as:

$$D + Q = R + \Delta R \quad (5)$$

In the eastern United States, gaining streams are relatively closely spaced, and areas in which rejected recharge occurs are relatively unimportant. In this region, the growth of cones of depression first commonly causes a reduction in natural discharge. If the pumping wells are near a stream or if the withdrawals are continued long enough, ground-water discharge to a stream may be stopped entirely in the vicinity of the wells, and water may be induced to move from the stream into the aquifer (Figure 5-2d). The tendency in this region is for withdrawals to change discharge areas into recharge areas. This consideration is important where the streams contain brackish or polluted water or where the streamflow is committed or required for other purposes.

In summary, withdrawal of ground water through a well reduces the water in storage in the source aquifer

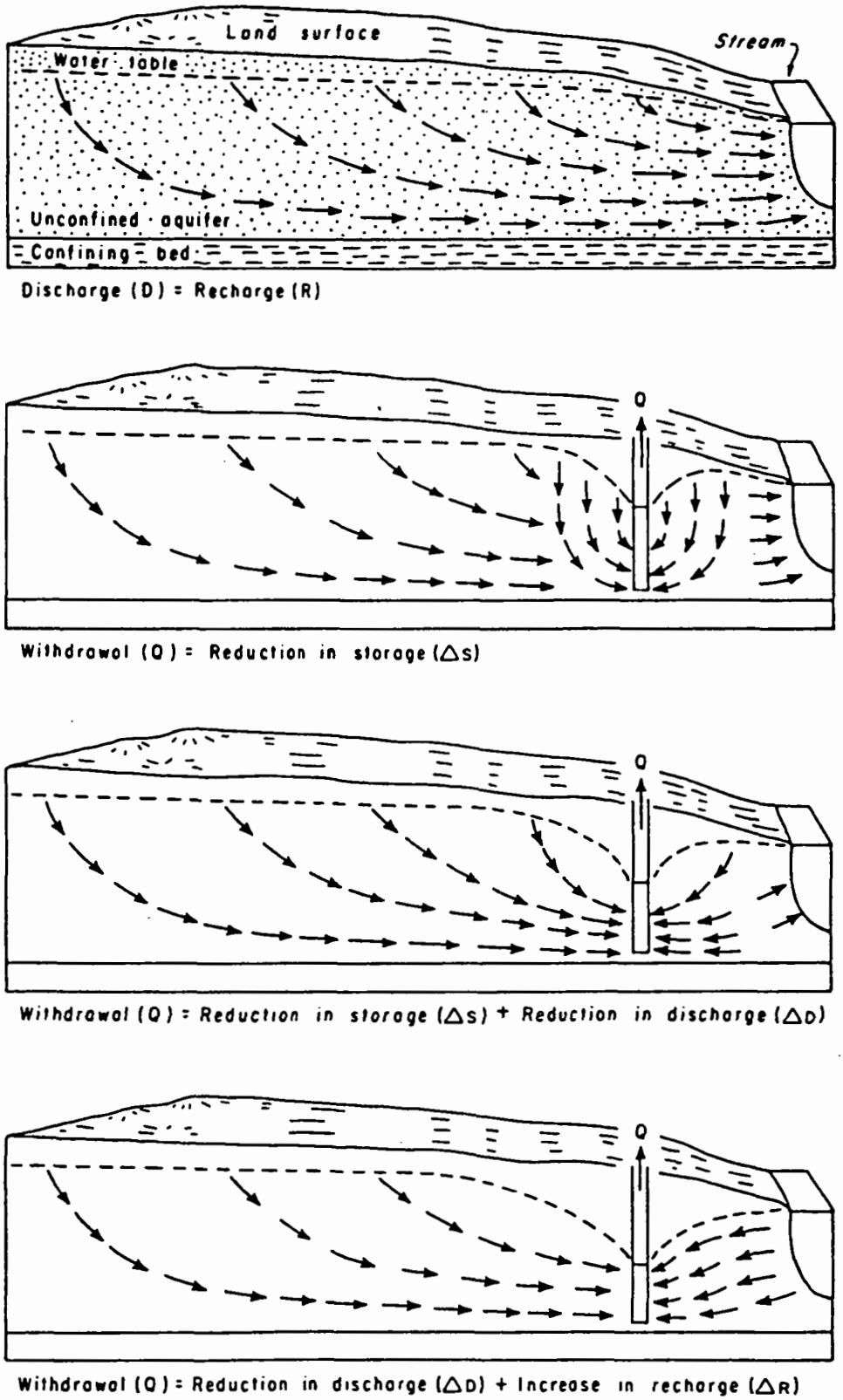

Figure 5-2. Source of Water Derived From Wells

during the growth of the cone of depression. If the cone of depression ceases to expand, the rate of withdrawal is being balanced by a reduction in the rate of natural discharge and (or) by an increase in the rate of recharge. Under this condition,

$$Q = \Delta D + \Delta R \quad (6)$$

AQUIFER TESTS

Determining the yield of ground-water systems and evaluating the movement and fate of ground-water pollutants require, among other information, knowledge of:

1. The position and thickness of aquifers and confining beds.
2. The transmissivity and storage coefficient of the aquifers.
3. The hydraulic characteristics of the confining beds.
4. The position and nature of the aquifer boundaries.
5. The location and amounts of ground-water withdrawals.
6. The locations, kinds, and amounts of pollutants and pollutant practices.

Acquiring knowledge of these factors requires both geologic and hydrologic investigations. One of the most important hydrologic studies involves analyzing the change, with time, in water levels (or total heads) in an aquifer caused by withdrawals through wells. This type of study is referred to as an aquifer test and, in most cases, includes pumping a well at a constant rate for a period ranging from several hours to several days and measuring the change in water level in observation wells located at different distances from the pumped well (Figure 5-3).

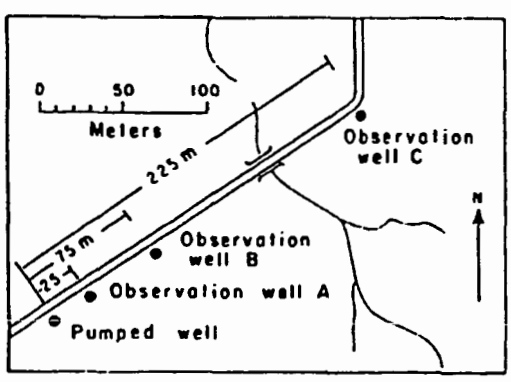

Figure 5-3. Map of Aquifer Test Site

Successful aquifer tests require, among other things:

1. Determination of the prepumping water-level trend (that is, the regional trend).
2. A carefully controlled constant pumping rate.
3. Accurate water-level measurements made at precise times during both the drawdown and the recovery periods.

Drawdown is the difference between the water level at any time during the test and the position at which the water level would have been if withdrawals had not started. Drawdown is very rapid at first. As pumping continues and the cone of depression expands, the rate of drawdown decreases (Figure 5-4).

The recovery of the water level under ideal conditions is a mirror image of the drawdown. The change in water level during the recovery period is the same as if withdrawals had continued at the same rate from the pumped well but, at the moment of pump cutoff, a recharge well had begun recharging water at the same point and at the same rate. Therefore, the recovery of the water level is the difference between the actual measured level and the projected pumping level (Figure 5-4).

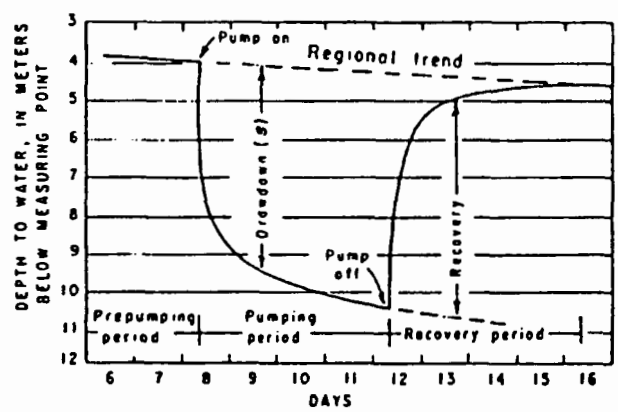

Figure 5-4. Change of Water Level in Well B

In addition to the constant-rate aquifer test mentioned above, analytical methods also have been developed for several other types of aquifer tests. These methods include tests in which the rate of withdrawal is variable and tests that involve leakage of water across confining beds into confined aquifers. The analytical methods available also permit analysis of tests conducted on both vertical wells and horizontal wells or drains.

The most commonly used method of aquifer-test-data

analysis—that for a vertical well pumped at a constant rate from an aquifer not affected by vertical leakage and lateral boundaries—is discussed below. The method of analysis requires the use of a type curve based on the values of W(μ) and 1/μ listed in Table 5-1. Preparation and use of the type curve are covered in the following discussion.

3. The discharging well penetrates the entire thickness of the aquifer, and its diameter is small in comparison with the pumping rate, so that storage in the well is negligible.

These assumptions are most nearly met by confined aquifers at sites remote from their boundaries. However,

1/u	10	7.69	5.88	5.00	4.00	3.33	2.86	2.5	2.22	2.00	1.67	1.43	1.25	1.11
10^{-1}	0.219	0.135	0.075	0.049	0.025	0.013	0.007	0.004	0.002	0.001	0.000	0.000	0.000	0.000
1	1.82	1.59	1.36	1.22	1.04	.91	.79	.70	.63	.56	.45	.37	.31	.26
10	4.04	3.78	3.51	3.35	3.14	2.96	2.81	2.68	2.57	2.47	2.30	2.15	2.03	1.92
10^2	6.33	6.07	5.80	5.64	5.42	5.23	5.08	4.95	4.83	4.73	4.54	4.39	4.26	4.14
10^3	8.63	8.37	8.10	7.94	7.72	7.53	7.38	7.25	7.13	7.02	6.84	6.69	6.55	6.44
10^4	10.94	10.67	10.41	10.24	10.02	9.84	9.68	9.55	9.43	9.33	9.14	8.99	8.86	8.74
10^5	13.24	12.98	12.71	12.55	12.32	12.14	11.99	11.85	11.73	11.63	11.45	11.29	11.16	11.04
10^6	15.54	15.28	15.01	14.85	14.62	14.44	14.29	14.15	14.04	13.93	13.75	13.60	13.46	13.34
10^7	17.84	17.58	17.31	17.15	16.93	16.74	16.59	16.46	16.34	16.23	16.05	15.90	15.76	15.65
10^8	20.15	19.88	19.62	19.45	19.23	19.05	18.89	18.76	18.64	18.54	18.35	18.20	18.07	17.95
10^9	22.45	22.19	21.92	21.76	21.53	21.35	21.20	21.06	20.94	20.84	20.66	20.50	20.37	20.25
10^{10}	24.75	24.49	24.22	24.06	23.83	23.65	23.50	23.36	23.25	23.14	22.96	22.81	22.67	22.55
10^{11}	27.05	26.79	26.52	26.36	26.14	25.96	25.80	25.67	25.55	25.44	25.26	25.11	24.97	24.86
10^{12}	29.36	20.09	28.83	28.66	28.44	28.26	28.10	27.97	27.85	27.75	27.56	27.41	27.28	27.16
10^{13}	31.66	31.40	31.13	30.97	30.74	30.56	30.41	30.27	30.15	30.05	29.87	29.71	29.58	29.46
10^{14}	33.96	33.70	33.43	33.27	33.05	32.86	32.71	32.58	32.46	32.35	32.17	32.02	31.88	31.76

Examples: When $1/u = 10 \times 10^{-1}$, $W(u) = 0.219$; when $1/u = 3.33 \times 10^2$, $W(u) = 5.23$.

Table 5-1. Selected Values of W(u) for values of 1u

Analysis of Aquifer-Test Data

In 1935, C. V. Theis of the New Mexico Water Resources District of the U.S. Geological Survey developed the first equation to include time of pumping as a factor that could be used to analyze the effect of withdrawals from a well. The Theis equation permitted, for the first time, determination of the hydraulic characteristics of an aquifer before the development of new steady-state conditions resulting from pumping. This capacity is important because, under most conditions, a new steady state cannot be developed or, if it can, many months or years may be required.

In the development of the equation, Theis assumed that:

1. The transmissivity of the aquifer tapped by the pumping well is constant during the test to the limits of the cone of depression.
2. The water withdrawn from the aquifer is derived entirely from storage and is discharged instantaneously with the decline in head.

if certain precautions are observed, the equation also can be used to analyze tests of unconfined aquifers. The forms of the Theis equation used to determine the transmissivity and storage coefficient are

$$T=(Q \times W(u))/(4 \times \pi \times s) \quad (7)$$

$$S=(4 \times T \times t \times u)/r^2 \quad (8)$$

where T is transmissivity, S is the storage coefficient, Q is the pumping rate, s is drawdown, t is time, r is the distance from the pumping well to the observation well, W(u) is the well function of u, which equals

$$-.577216 - \log_e u + u - u^2/(2x2!) + u^3/(3x3!) - u^4/(4x4!) + \ldots$$

$$\text{and } u=(r^2S)/(4Tt). \quad (9)$$

The Theis equation is in a form that cannot be solved directly. To overcome this problem, Theis devised a convenient graphic method of solution that uses a type curve (Figure 5-5). To apply this method, a data plot of drawdown versus time (or drawdown versus t/r^2) is

matched to the type curve of W(u) versus 1/u (Figure 5-6). At some convenient point on the overlapping part of the sheets containing the data plot and type curve, values of s, t (or t/r^2), W(u), and 1/u are noted (Figure 5-6). These values are then substituted in the equations, which are solved for T and S, respectively.

A Theis type curve of W(u) versus 1/u can be prepared

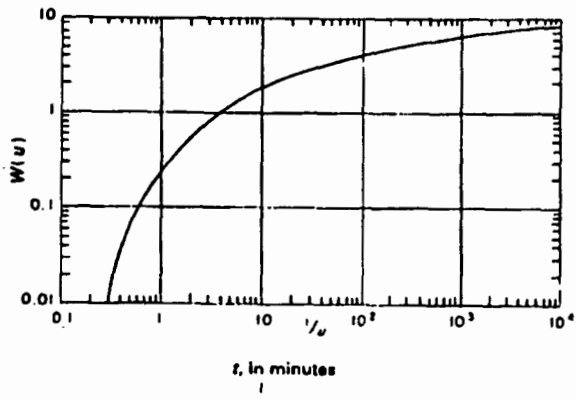

Figure 5-5. Theis Type Curve

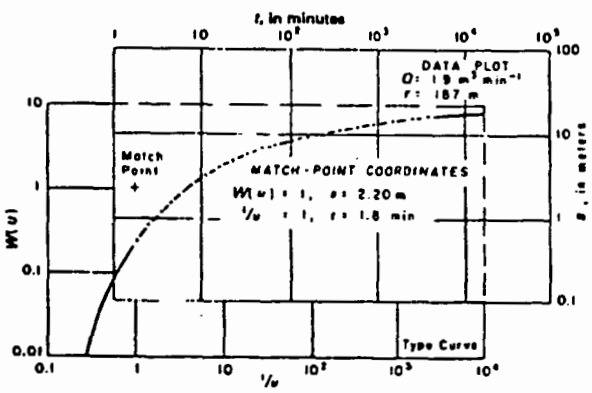

Figure 5-6. Data Plot of Drawdown Versus Time Matched to Theis Type Curve

from the values given in Table 5-1. The data points are plotted on logarithmic graph paper—that is, graph paper having logarithmic divisions in both the x and y directions.

The dimensional units of transmissivity (T) are L^2t, where L is length and t is time in days. Thus, if Q is in cubic meters per day and s is in meters, T will be in square meters per day. Similarly, if T is in square meters per day, t is in days, and r is in meters, S will be dimensionless.

Traditionally, in the United States, T has been expressed in units of gallons per day per foot. The common practice now is to report transmissivity in units of square meters per day or square feet per day. If Q is measured in gallons per minute, as is still normally the case, and drawdown is measured in feet, as is also normally the case, the equation is modified to obtain T in square feet per day as follows:

$$T=(QxW(u))/(4\pi s) = (gal/min) \times (1.440\ min/d) \times (ft^3/7.48\ gal) \times 1/ft \times W(u)/(4x\pi) \qquad (10)$$

or

$$T(in\ ft^2\ d^{-1}) = (15.3 \times Q \times W(u))/s \qquad (11)$$

(when Q is in gallons per minute and s is in feet). To convert square feet per day to square meters per day, divide by 10.76.

The storage coefficient is dimensionless. Therefore, if T is in square feet per day, t is in minutes, and r is in feet, then,

$$S=(4Ttu)/r^2=(4/1) \times ft^2/d \times min/ft^2 \times d/1440\ min \qquad (12)$$

or

$$S=(Ttu)/360r^2) \qquad (13)$$

(when T is in square feet per day, t is in minutes, and r is in feet).

Analysis of aquifer-test data using the Theis equation involves plotting both the type curve and the test data on logarithmic graph paper. If the aquifer and the conditions of the test satisfy Theis' assumptions, the type curve has the same shape as the cone of depression along any line radiating away from the pumping well and the drawdown graph at any point in the cone of depression.

There are two considerations for using the Theis equation for unconfined aquifers. First, if the aquifer is relatively fine grained, water is released slowly over a period of hours or days, not instantaneously with the decline in head. Therefore, the value of S determined from a short-period test may be too small.

Second, if the pumping rate is large and the observation well is near the pumping well, dewatering of the aquifer may be significant, and the assumption that the

transmissivity of the aquifer is constant is not satisfied. The effect of dewatering of the aquifer can be eliminated with the following equation:

$$s'=s-s^2/(2b) \quad (14)$$

where s is the observed drawdown in the unconfined aquifer, b is the aquifer thickness, and s' is the drawdown that would have occurred if the aquifer had been confined (that is, if no dewatering had occurred).

To determine the transmissivity and storage coefficient of an unconfined aquifer, a data plot consisting of s versus t (or t/r^2) is matched with the Theis type curve of W(u) versus 1/u. Both s and b must be in the same units, either feet or meters.

As noted above, Theis assumed in the development of his equation that the discharging well penetrates the entire thickness of the aquifer. However, because it is not always possible, or necessarily desirable, to design a well that fully penetrates the aquifer under development, most discharging wells are open to only a part of the aquifer that they draw from. Such partial penetration creates vertical flow in the vicinity of the discharging well that may affect drawdowns in observation wells located relatively close to the discharging well. Drawdowns in observation wells that are open to the same zone as the discharging well will be larger than the drawdowns in wells at the same distance from the discharging well but open to other zones. The possible effect of partial penetration on drawdowns must be considered in the analysis of aquifer-test data. If aquifer-boundary and other conditions permit, the problem can be avoided by locating observation wells beyond the zone in which vertical flow exists.

Time-Drawdown Analysis

The Theis equation is only one of several methods that have been developed for the analysis of aquifer-test data. Another somewhat more convenient method, was developed from the Theis equation by C. E. Jacob. The greater convenience of the Jacob method derives partly from its use of semilogarithmic graph paper instead of the logarithmic paper used in the Theis method, and from the fact that, under ideal conditions, the data plot along a straight line rather than along a curve.

However, it is essential to note that, whereas the Theis equation applies at all times and places (if the assumptions are met), Jacob's method applies only under certain additional conditions. These conditions also must be satisfied in order to obtain reliable answers. To understand the limitations of Jacob's method, the changes that occur in the cone of depression during an aquifer test must be considered. The changes that are of concern involve both the shape of the cone and the rate of drawdown. As the cone of depression migrates outward from a pumping well, its shape (and, therefore, the hydraulic gradient at different points in the cone) changes. We can refer to this condition as unsteady shape. At the start of withdrawals, the entire cone of depression has an unsteady shape (Figure 5-7a). After a test has been underway for some time, the cone of depression begins to assume a relatively steady shape, first at the pumping well and then gradually to greater and greater distances (Figure 5-7b). If withdrawals continue long enough for increases in recharge and /or reductions in discharge to balance the rate of withdrawal, drawdowns cease, and the cone of depression is said to be in a steady state (Figure 5-7c).

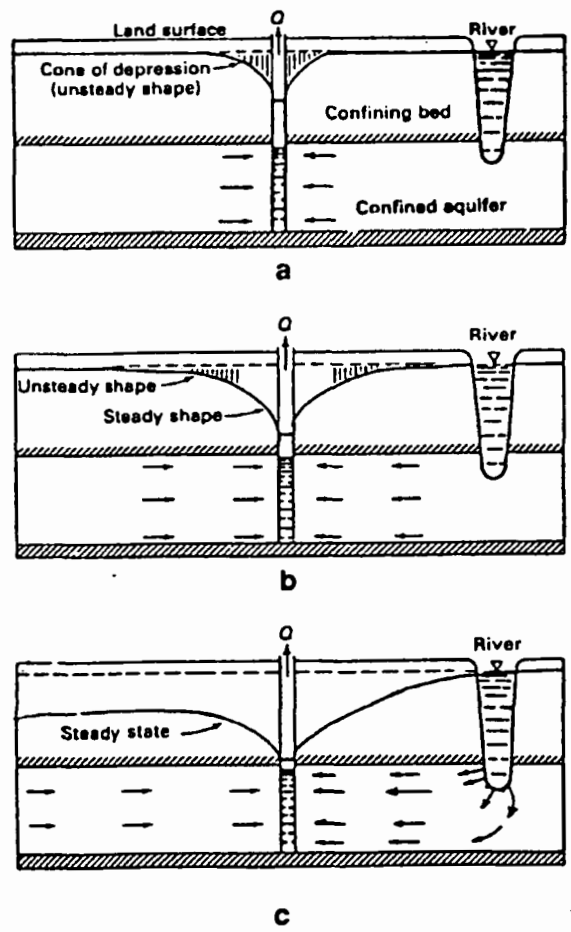

Figure 5-7. Development of Cone of Depression from Start of Pumping to Steady-State

The Jacob method is applicable only to the zone in which steady-shape conditions prevail or to the entire cone only after steady-state conditions have developed. For practical purposes, this condition is met when $u=(r^2S)/(4Tt)$ is equal to or less than about 0.05.

Substituting this value in the equation for u and solving for t, we can determine the time at which steady-shape conditions develop at the outermost observation well. Thus,

$$t_c = (7{,}200\ r^2 S)/T \quad (15)$$

where t_c is the time, in minutes, at which steady-shape conditions develop, r is the distance from the pumping well, in feet (or meters), S is the estimated storage coefficient (dimensionless), and T is the estimated transmissivity, in square feet per day (or square meters per day).

After steady-shape conditions have developed, the drawdowns at an observation well begin to fall along a straight line on semilogarithmic graph paper, as Figure 5-8 shows. Before that time, the drawdowns plot below the extension of the straight line. When a time-drawdown graph is prepared, drawdowns are plotted on the vertical (arithmetic) axis versus time on the horizontal (logarithmic) axis.

one log cycle, t_o is the time at the point where the straight line intersects the zero-drawdown line, and r is the distance from the pumping well to the observation well.

These equations are in consistent units. Thus, if Q is in cubic meters per day and s is in meters, T is in square meters per day. S is dimensionless, so that if r is in square meters per day, then r must be in meters and t_o must be in days.

It is still common practice in the United States to express Q in gallons per minute, s in feet, t in minutes, T in feet, and T in square feet per day. The equations can be modified for direct substitution of these units as follows:

$$T=(2.3Q)/(4\pi\Delta s) = (2.3/4\pi) \times (\text{gal/min}) \times (1{,}440\ \text{min/d}) \times (\text{ft}^3/74.8\ \text{gal}) \times (1/\text{ft}) \quad (18)$$

$$T = (35Q)/s \quad (19)$$

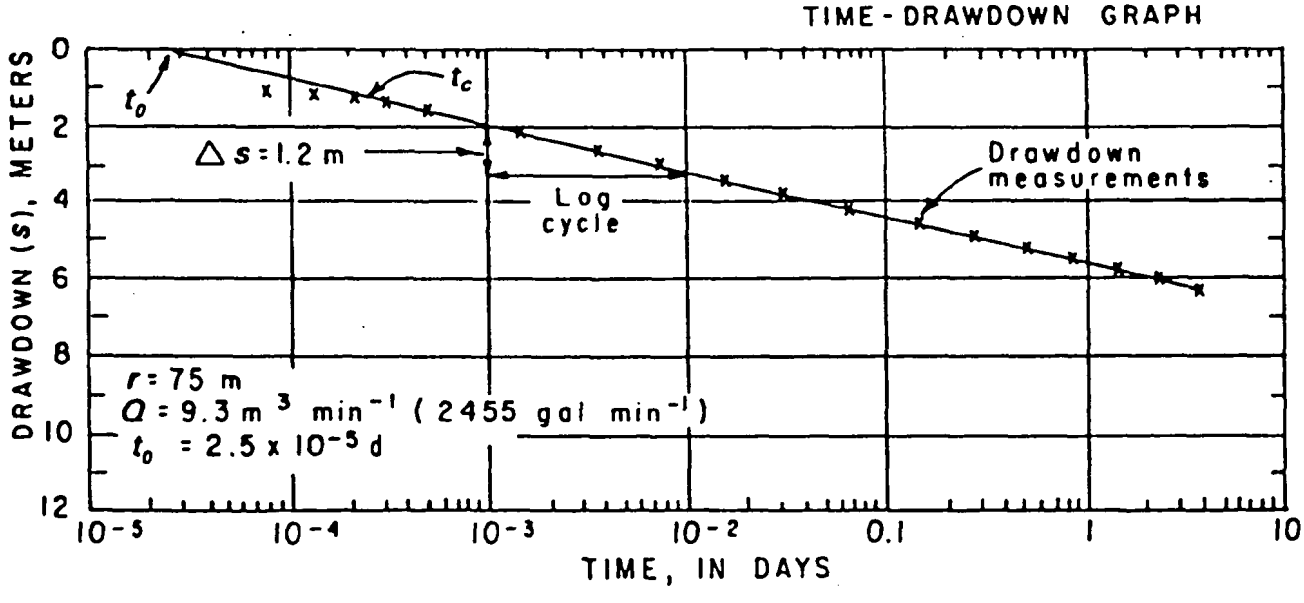

Figure 5-8. Time-Drawdown Graph

The slope of the straight line is proportional to the pumping rate and to the transmissivity. Jacob derived the following equations for determination of transmissivity and storage coefficient from the time-drawdown graphs:

$$T = (2.3Q)/(4\pi\Delta s) \quad (16)$$

$$S = (2.25Tt_o)/r^2 \quad (17)$$

where Q is the pumping rate, Δs is the drawdown across

where T is in square feet per day, Q is in gallons per minute and s is in feet, and

$$S=(2.25Tt_0/r^2) = (2.25/1) \times (\text{ft}^2/\text{d}) \times (\text{min/ft}^2) \times (\text{d}/1{,}440\ \text{min}) \quad (20)$$

$$S=(Tt_0)/(640r^2) \quad (21)$$

where T is in square feet per day, t_o is in minutes, and r is in feet.

Distance-Drawdown Analysis

Aquifer tests should have at least three observation wells located at different distances from the pumping well (Figure 5-9). Drawdowns measured at the same time in these wells can be analyzed with the Theis equation and type curve to determine the aquifer transmissivity and storage coefficient.

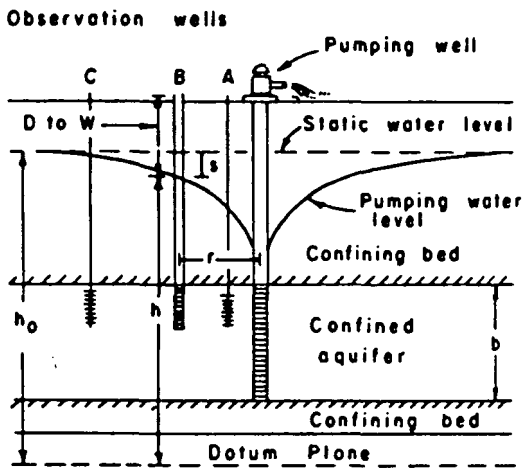

Figure 5-9. Desirable Location for Observation Wells in Aquifer Tests

After the test has been underway long enough, drawdowns in the wells also can be analyzed by the Jacob method, either through the use of a time-drawdown graph using data from individual wells or through the use of a distance-drawdown graph using simultaneous measurements in all of the wells. To determine when sufficient time has elapsed, see the discussion of time-drawdown analysis earlier in this chapter.

In the Jacob distance-drawdown method, drawdowns are plotted on the vertical axis versus distance on the horizontal axis (Figure 5-10). If the aquifer and test conditions satisfy the Theis assumptions and the limitation of the Jacob method, the drawdowns measured at the same time in different wells should plot along a straight line (Figure 5-10).

The slope of the straight line is proportional to the pumping rate and to the transmissivity. Jacob derived the following equations for determination of the transmissivity and storage coefficient from distance-drawdown graphs:

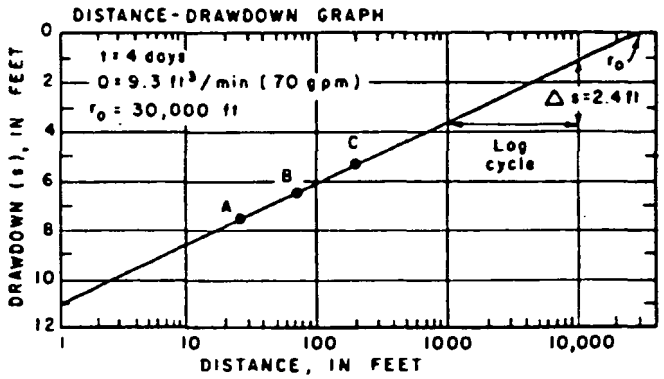

Figure 5-10. Distance-Drawdown Graph

$$T = (2.3Q)/(2\pi\Delta s) \quad (22)$$

$$S = (2.25Tt)/r_o^2 \quad (23)$$

where Q is the pumping rate, Δs is the drawdown across one log cycle, t is the time at which the drawdowns were measured, and r_o is the distance from the pumping well to the point where the straight line intersects the zero-drawdown line.

These equations are in consistent units. For the inconsistent units still in relatively common use in the United States, the equations should be used in the following forms:

$$T = (70Q)/\Delta s \quad (24)$$

where T is in square feet per day, Q is in gallons per minute, and s is in feet and

$$S = (Tt)/(640r_o^2) \quad (25)$$

where T is in square feet per day, t is in minutes, and r_o is in feet.

The distance r_o does not indicate the outer limit of the cone of depression. Because nonsteady-shape conditions exist in the outer part of the cone, before the development of steady-state conditions, the Jacob method does not apply to that part. If the Theis equation were used to calculate drawdowns in the outer part of the cone, it would be found that they would plot below the straight line. In other words, the measurable limit of the cone of depression is beyond the distance r_o.

If the straight line of the distance-drawdown graph is

extended inward to the radius of the pumping well, the drawdown indicated at that point is the drawdown in the aquifer outside of the well. If the drawdown inside the well is found to be greater than the drawdown outside, the difference is attributable to well loss. (See Single-Well Tests.)

The hydraulic conductivities and, therefore, the transmissivities of aquifers may be different in different directions. These differences may cause differences in drawdowns measured at the same time in observation wells located at the same distances but in different directions from the discharging well. Where this condition exists, the distance-drawdown method may yield satisfactory results only where three or more observation wells are located in the same direction but at different distances from the discharging well.

Single-Well Tests

The most useful aquifer tests are multiple-well tests, which are those that include water-level measurements in observation wells. It also is possible to obtain useful data from production wells, even where observation wells are not available. These single-well tests may consist of pumping a well at a single constant rate, or at two or more different but constant rates or, if the well is not equipped with a pump, by instantaneously introducing a known volume of water into the well. The following discussion is limited to tests involving a single constant rate.

In order to analyze the data, the nature of the drawdown in a pumping well must be understood. The total drawdown (s_t) in most, if not all, pumping wells consists of two components (Figure 5-11). One is the drawdown (s_a) in the aquifer, and the other is the drawdown (s_w) that occurs as water moves from the aquifer into the well and up the well bore to the pump intake. Thus, the drawdown in most pumping wells is greater than the drawdown in the aquifer at the radius of the pumping well.

The total drawdown (s_t) in a pumping well can be expressed in the form of the following equations:

$$s_t = s_a + s_w$$

$$s_t = BQ + CQ^2 \quad (26)$$

where s_a is the drawdown in the aquifer at the effective radius of the pumping well, s_w is well loss, Q is the pumping rate, B is a factor related to the hydraulic characteristics of the aquifer and the length of the pumping period, and C is a factor related to the characteristics of the well.

The factor C is normally considered to be constant, so that, in a constant rate test, CQ^2 is also constant. As a result, the well loss (s_w) increases the total drawdown in the pumping well but does not affect the rate of change in the drawdown with time. It is, therefore, possible to analyze drawdowns in the pumping well with

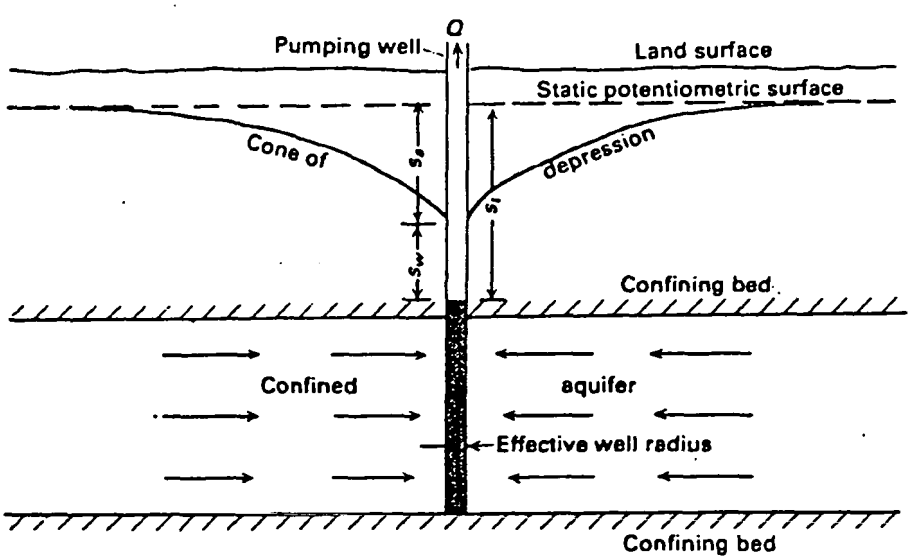

Figure 5-11. Two Components of Total Drawdown in a Pumping Well

the Jacob time-drawdown method using semilogarithmic graph paper. (See "Time-Drawdown Analysis" earlier in this section.) Drawdowns are plotted on the arithmetic scale versus time on the logarithmic scale (Figure 5-12), and transmissivity is determined from the slope of the straight line by using the following equation:

$$T = (2.3Q)/(4\pi\Delta s) \quad (27)$$

Where well loss is present in the pumping well, the storage coefficient cannot be determined by extending the straight line to the line of zero drawdown. Even where well loss is not present, the determination of the storage coefficient from drawdowns in a pumping well likely will be subject to large error because the effective radius of the well may differ significantly from the nominal radius.

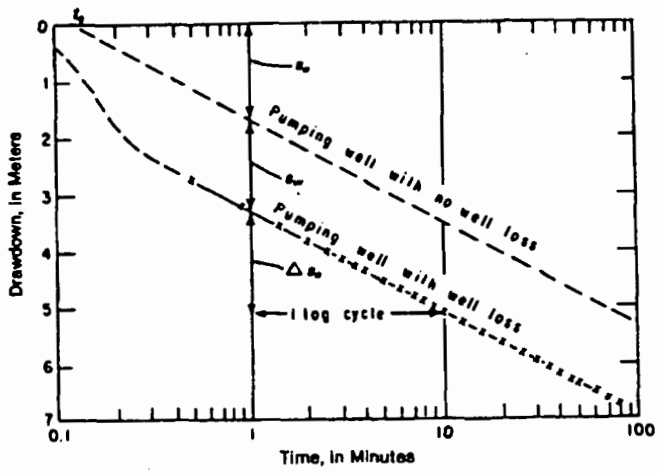

Figure 5-12. Time-Drawdown Plot With and Without Well Loss

In this equation, drawdown in the pumping well is proportional to the pumping rate. The factor B in the aquifer-loss term (BQ) increases with time of pumping as long as water is being derived from storage in the aquifer. The factor C in the well-loss term (CQ^2) is a constant if the characteristics of the well remain unchanged, but, because the pumping rate in the well-loss term is squared, drawdown due to well loss increases rapidly as the pumping rate is increased. The relation between pumping rates and drawdown in a pumping well, if the well was pumped for the same length of time at each rate, is shown in Figure 5-13. The effect of well loss on drawdown in the pumping well is important both for pumping wells data analysis, and supply well design.

Well Interference

Pumping a well causes a drawdown in the ground-water

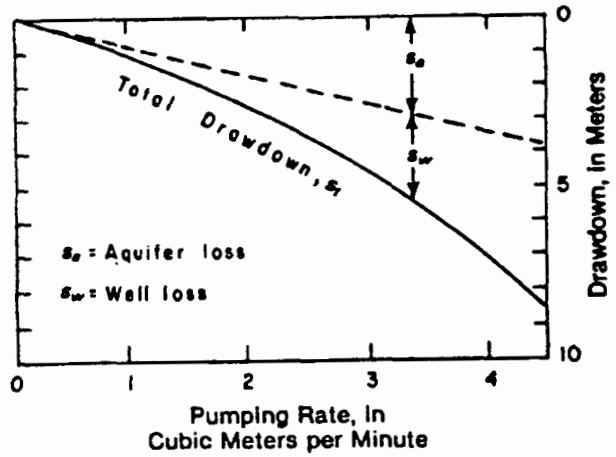

Figure 5-13. Relation of Pumping Rate and Drawdown

level in the surrounding area. The drawdown in water level forms a conical-shaped depression in the water table or potentiometric surface which is referred to as a cone of depression. (See "Cone of Depression" at the beginning of this section.) Similarly a well through which water is injected into an aquifer (that is a recharge or injection well) causes a buildup in ground-water level in the form of a conical-shaped mound.

The drawdown (s) in an aquifer caused by pumping at any point in the aquifer is directly proportional to the pumping rate (Q) and the length of time (t) that pumping has been in progress and is inversely proportional to the transmissivity (T), the storage coefficient (S), and the square of the distance (r^2) between the pumping well and the point. This is represented by the equation:

$$s = (Q,t)/T,S,r^2 \quad (28)$$

Where pumping wells are spaced relatively close together, pumping of one will cause a drawdown in the others. Because drawdowns are additive, the total drawdown in a pumping well is equal to its own drawdown plus the drawdowns caused at its location by other pumping wells (Figures 5-14 and 5-15). The drawdowns in pumping wells caused by withdrawals from other pumping wells are referred to as well interference. As Figure 5-15 shows, a divide forms in the potentiometric surface (or the water table in the case of an unconfined aquifer) between pumping wells.

At any point in an aquifer affected by both a discharging well and a recharging well, the change in water level is

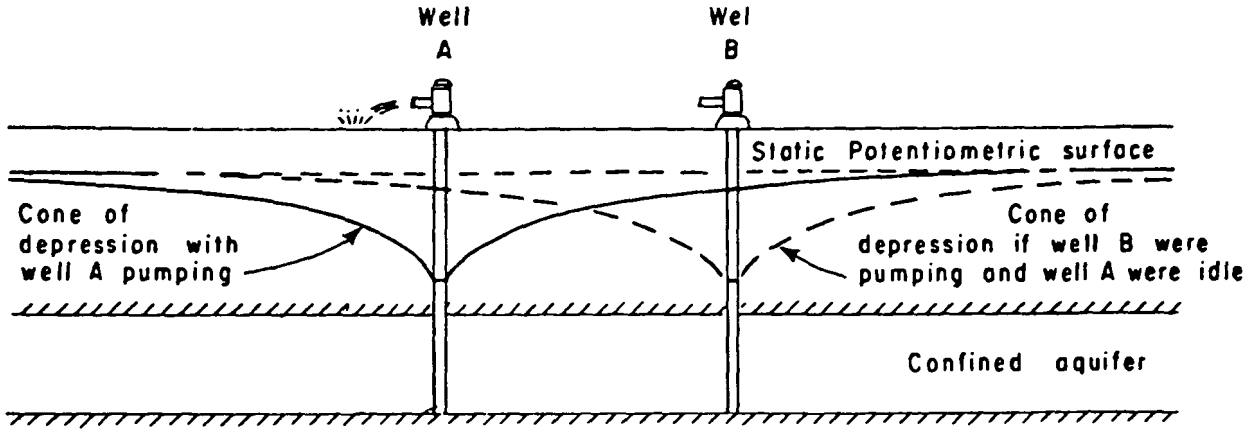

Figure 5-14. Cone of Depression When Well A or B is Pumped

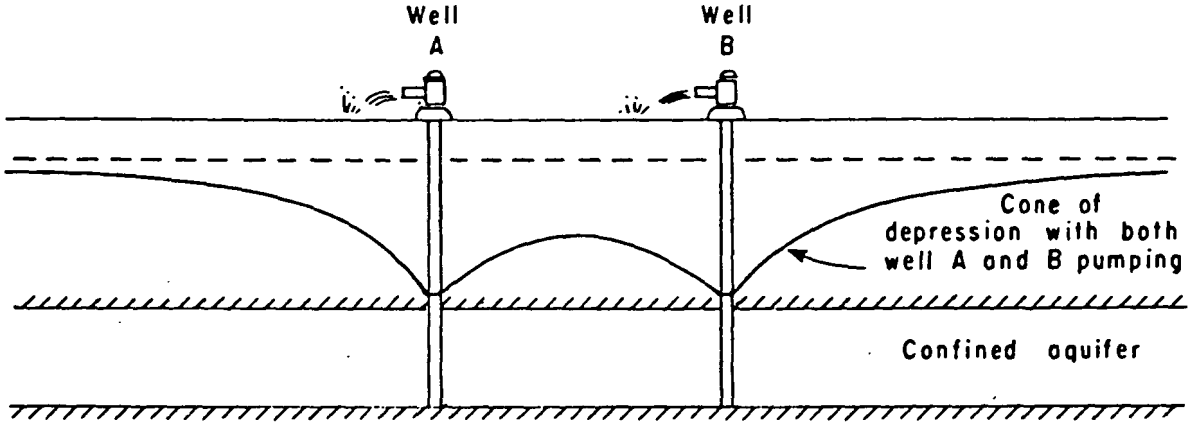

Figure 5-15. Total Drawdown Caused by Overlapping Cones of Depression

equal to the difference between the drawdown and the buildup. If the rates of discharge and recharge are the same and if the wells are operated on the same schedule, the drawdown and the buildup will cancel midway between the wells and the water level at that point will remain unchanged from the static level (Figure 5-16). (See "Aquifer Boundaries" below.)

From the functional equation above, it can be seen that, in the absence of well interference, drawdown in an aquifer at the effective radius of a pumping well is directly proportional to the pumping rate. Conversely, the maximum pumping rate is directly proportional to the available drawdown. For confined aquifers, available drawdown is normally considered to be the distance between the prepumping water level and the top of the aquifer. For unconfined aquifers, available drawdown is normally considered to be about 60 percent of the saturated aquifer thickness.

Where the pumping rate of a well is such that only a part of the available drawdown is utilized, the only effect of well interference is to lower the pumping level and, thereby, increase pumping costs. In the design of a well

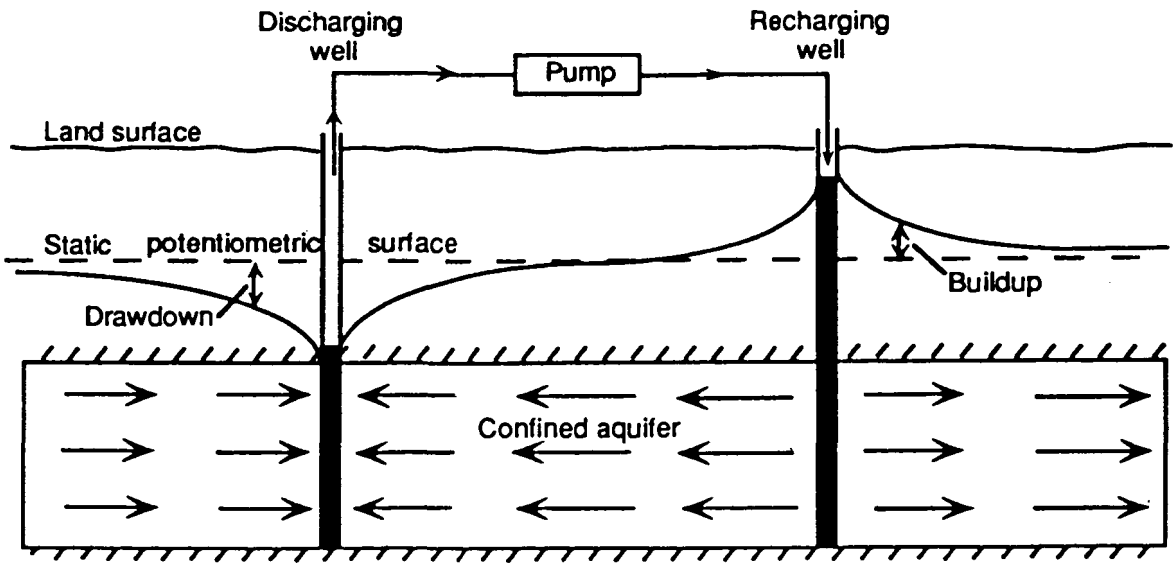

Figure 5-16. Cones of Depression and Buildup Surrounding Discharging and Recharging Wells

field, the increase in pumping cost must be evaluated along with the cost of the additional waterlines and powerlines that must be installed if the spacing of wells is increased to reduce well interference.

Because well interference reduces the available drawdown, it also reduces the maximum yield of a well. Well interference is, therefore, an important matter in the design of well fields where it is desirable for each well to be pumped at the largest possible rate. For a group of wells pumped at the same rate and on the same schedule, the well interference caused by any well on another well in the group is inversely proportional to the square of the distance between the two wells (r^2). Therefore, excessive well interference is avoided by increasing the spacing between wells and by locating the wells along a line rather than in a circle or in a grid pattern.

Aquifer Boundaries

One of the assumptions inherent in the Theis equation (and in most other fundamental ground-water flow equations) is that the aquifer to which it is being applied is infinite in extent. Obviously, no such aquifer exists on Earth. However, many aquifers are areally extensive, and, because pumping will not affect recharge or discharge significantly for many years, most water pumped is from ground-water storage; as a consequence water levels must decline for many years. An excellent example of such an aquifer is that underlying the High Plains from Texas to South Dakota.

All aquifers are vertically and horizontally bounded. Vertical boundaries may include the water table, the plane of contact between each aquifer and each confining bed, and the plane marking the lower limit of the zone of interconnected openings—in other words, the base of the ground-water system.

Hydraulically, aquifer boundaries are of two types: recharge boundaries and impermeable boundaries. A recharge boundary is a boundary along which flow lines originate. Under certain hydraulic conditions, this boundary will serve as a source of recharge to the aquifer. Examples of recharge boundaries include the zones of contact between an aquifer and a perennial stream that completely penetrates the aquifer or the ocean.

An impermeable boundary is a boundary that flow lines do not cross. Such boundaries exist where aquifers terminate against impermeable material. Examples include the contact between an aquifer composed of sand and a laterally adjacent bed composed of clay.

The position and nature of aquifer boundaries are critical to many ground-water problems, including those involved in the movement and fate of pollutants and the response of aquifers to withdrawals. Depending on the direction of the hydraulic gradient, a stream, for example, may be either the source or the destination of a pollutant.

Lateral boundaries within the cone of depression have a profound effect on the response of an aquifer to

withdrawals. To analyze or predict the effect of a lateral boundary, it is necessary to "make" the aquifer appear to be of infinite extent by using imaginary wells and the theory of images. Figures 5-17 and 5-18 show, in both plan view and profile, how image wells are used to compensate hydraulically for the effects of both recharging and impermeable boundaries. (See "Well Interference" earlier in this section.)

The key feature of a recharge boundary is that withdrawals from the aquifer do not produce drawdowns across the boundary. A perennial stream in intimate contact with an aquifer represents a recharge boundary because pumping from the aquifer will induce recharge from the stream. The hydraulic effect of a recharge boundary can be duplicated by assuming that a recharging image well is present on the side of the boundary opposite the real discharging well. Water is injected into the image well at the same rate and on the same schedule that water is withdrawn from the real well. In the plan view in Figure 5-17, flow lines originate at the boundary and equipotential lines parallel the boundary at the closest point to the pumping (real) well.

The key feature of an impermeable boundary is that no water can cross it. Such a boundary, sometimes termed a "no-flow boundary," resembles a divide in the water table or the potentiometric surface of a confined aquifer. The effect of an impermeable boundary can be duplicated by assuming that a discharging image well is present on the side of the boundary opposite the real discharging well. The image well withdraws water at the same rate and on the same schedule as the real well. Flow lines tend to parallel an impermeable boundary and equipotential lines intersect it at a right angle.

The image-well theory is an essential tool in the design of well fields near aquifer boundaries. To minimize lowering water levels, apply the following conditions:

1. Pumping wells should be located parallel to and as close as possible to recharging boundaries.
2. Pumping wells should be located perpendicular to and as far as possible from impermeable boundaries.

Figures 5-17 and 5-18 illustrate the effect of single

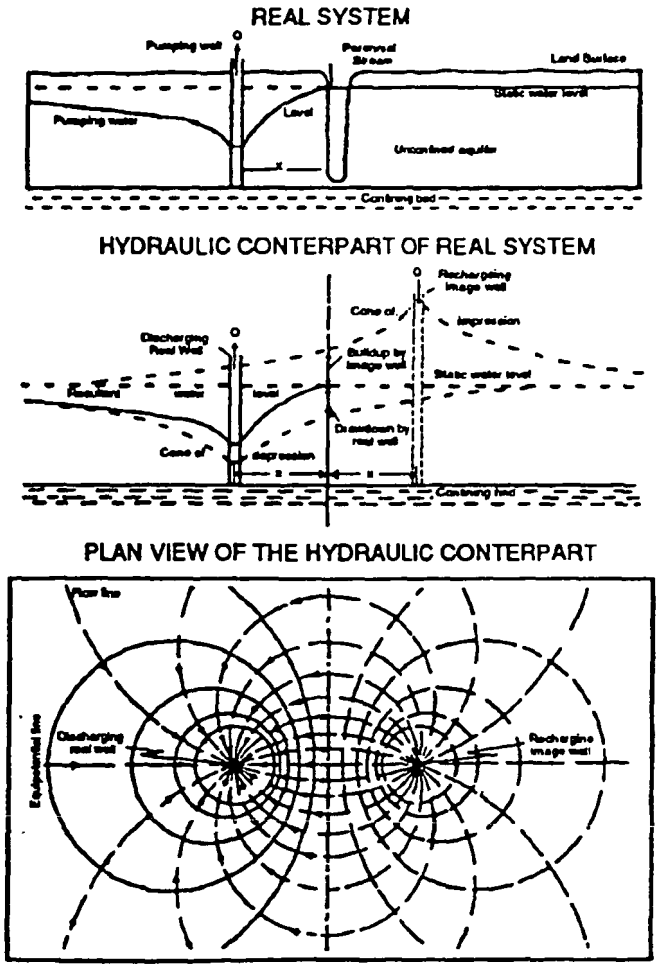

Figure 5-17. Recharge or Positive Boundary

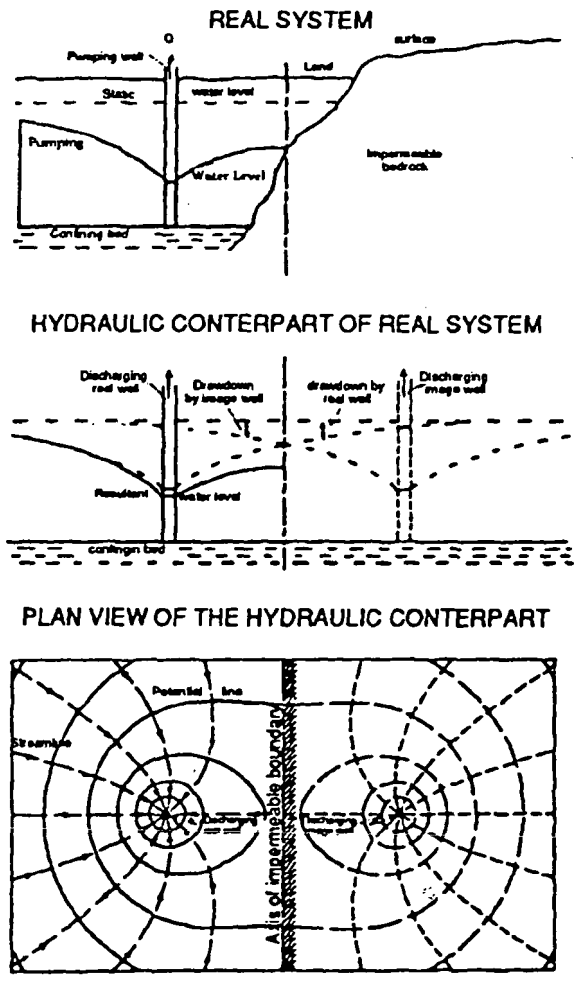

Figure 5-18. Discharge or Negative Boundary

boundaries and show how their hydraulic effect is compensated for through the use of single image wells. It is assumed in these figures that other boundaries are so remote that they have a negligible effect on the areas depicted. At many places, however, pumping wells are affected by two or more boundaries. One example is an alluvial aquifer composed of sand and gravel bordered on one side by a perennial stream (a recharge boundary) and on the other by impermeable bedrock (an impermeable boundary).

Contrary to first impression, these boundary conditions cannot be satisfied with only a recharging image well and a discharging image well. Additional image wells are required, as Figure 5-19 shows, to compensate for the effect of the image wells on the opposite boundaries. Because each additional image well affects the opposite boundary, it is necessary to continue adding image wells until their distances from the boundaries are so great that their effect becomes negligible.

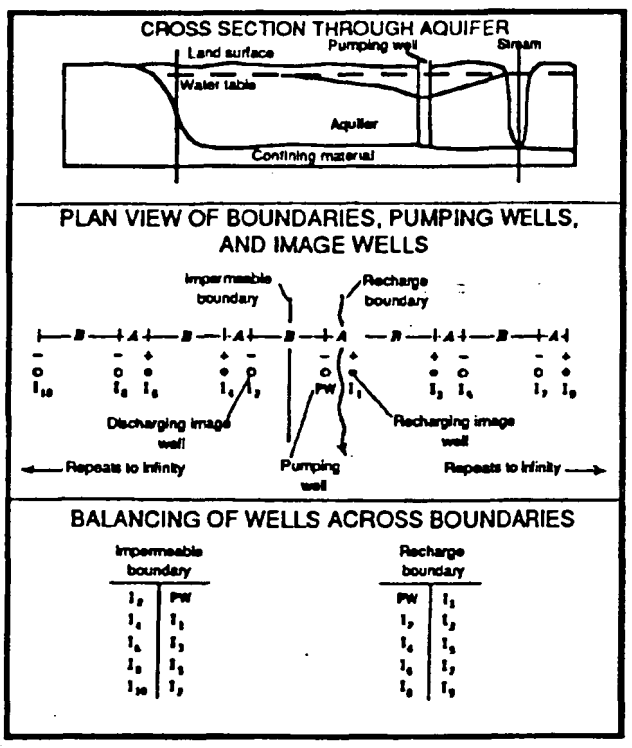

Figure 5-19. A Sequence of Image Wells

Tests Affected By Lateral Boundaries

When an aquifer test is conducted near one of the lateral boundaries of an aquifer, the drawdown data depart from the Theis type curve and from the initial straight line produced by the Jacob method. The hydraulic effect of lateral boundaries is assumed, for analytical convenience, to be due to the presence of other wells. (See "Aquifer Boundaries" earlier in this section.) Thus, a recharge boundary has the same effect on drawdowns as a recharging image well located across the boundary and at the same distance from the boundary as the real well. The image well is assumed to operate on the same schedule and at the same rate as the real well. Similarly, an impermeable boundary has the same effect on drawdowns as a discharging image well.

To analyze aquifer-test data affected by either a recharge boundary or an impermeable boundary, the early drawdown data in the observation wells nearest the pumping well must not be affected by the boundary. These data, then, show only the effect of the real well and can be used to determine the transmissivity (T) and the storage coefficient (S) of the aquifer. (See "Analysis of Aquifer-Test Data" and "Time-Drawdown Analysis" earlier in this section.) In the Theis method, the type curve is matched to the early data and a "match point" is selected to calculate the values of T and S. The position of the type curve in the region where the drawdowns depart from the type curve is traced onto the data plot (Figures 5-20 and 5-21). The trace of the type curve shows where the drawdowns would have plotted if there had been no boundary effect. The differences in drawdown between the data plot and the trace of the type curve show the effect of an aquifer boundary. The direction in which the drawdowns depart from the type curve—that is in the direction of either greater drawdowns or lesser drawdowns—shows the type of boundary.

Drawdowns greater than those defined by the trace of the type curve indicate the presence of an impermeable boundary because, as noted above, the effect of such boundaries can be duplicated with an imaginary discharging well. Conversely, a recharge boundary causes drawdowns to be less than those defined by the trace of the type curve.

In the Jacob method, drawdowns begin to plot along a straight line after the test has been underway for some time (Figures 5-22 and 5-23). The time at which the straight-line plot begins depends on the values of T and S of the aquifer and on the square of the distance between the observation well and the pumping well. (See "Time-Drawdown Analysis" earlier in this section.) Values of T and S are determined from the first straight-line segment defined by the drawdowns after the start of the aquifer test. The slope of this straight line depends on the transmissivity (T) and on the pumping rate (Q). If a boundary is present, the drawdowns will depart from the first straight-line segment and begin to fall along another straight line.

According to image-well theory, the effect of a recharge boundary can be duplicated by assuming that water is

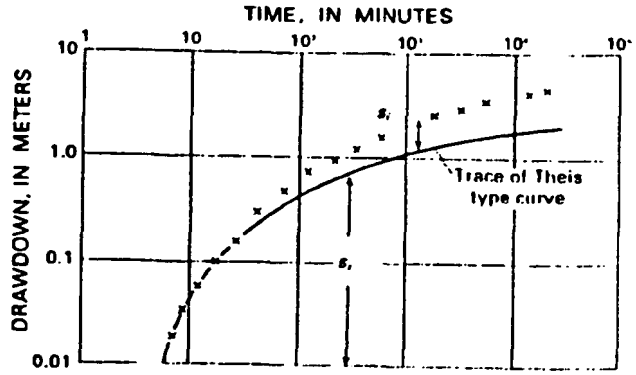

Figure 5-20. Theis Time-Drawdown Plot Showing a Negative Boundary

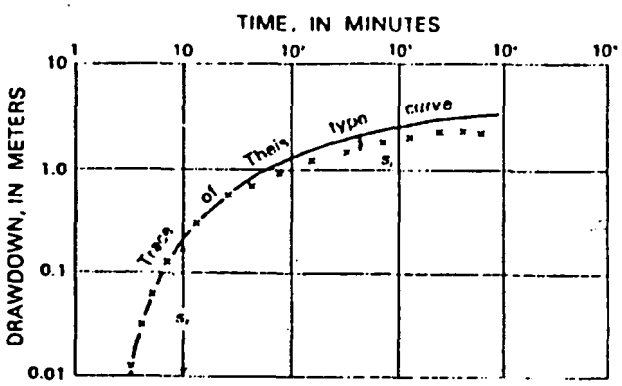

Figure 5-21. Theis Time-Drawdown Plot Showing a Positive Boundary

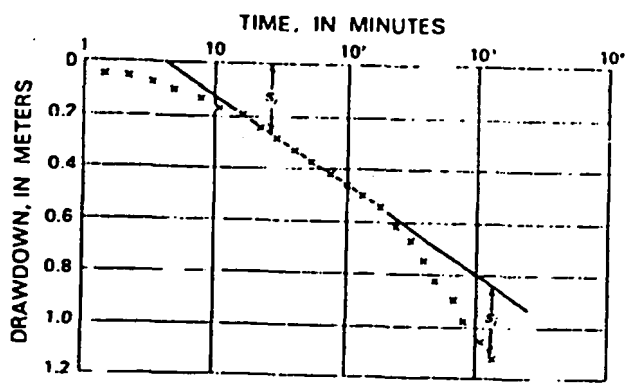

Figure 5-22. Jacob Time-Drawdown Plot Showing a Negative Boundary

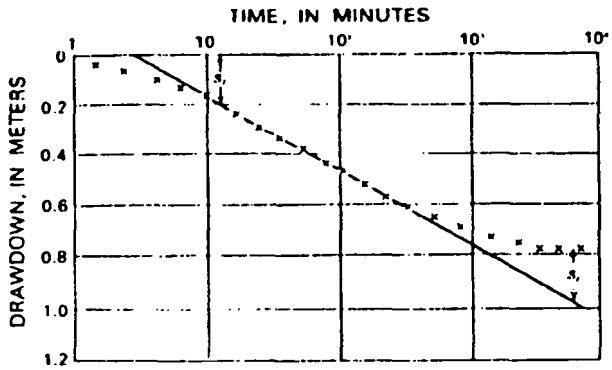

Figure 5-23. Jacob Time-Drawdown Plot Showing a Positive Boundary

injected into the aquifer through a recharging image well at the same rate that water is being withdrawn from the real well. It follows, therefore, that, when the full effect of a recharge boundary is felt at an observation well, there will be no further increase in drawdown and the water level in the well will stabilize. At this point in both the Theis and the Jacob methods, drawdowns plot along a straight line having a constant drawdown. Conversely, an impermeable boundary causes the rate of drawdown to increase. In the Jacob method, as a result, the drawdowns plot along a new straight line having twice the slope as the line drawn through the drawdowns that occurred before the boundary effect was felt.

The Jacob method should be used carefully when it is suspected that an aquifer test may be affected by boundary conditions. In many cases, the boundary begins to affect drawdowns before the method is applicable, the result being that T and S values determined from the data are erroneous and the effect of the boundary is not identified. When it is suspected that an aquifer test may be affected by boundary conditions, the data should, at least initially, be analyzed with the Theis method.

The position and the nature of many boundaries are obvious. For example, the most common recharge boundaries are streams and lakes; possibly, the most common impermeable boundaries are the bedrock walls of alluvial valleys. The hydraulic distance to these boundaries, however, may not be obvious. A stream or lake may penetrate only a short distance into an aquifer and their bottoms may be underlain by fine-grained material that hampers movement of water into the aquifer. Hydraulically, the boundaries formed by these surface-water bodies will appear to be farther from the

pumping well than the near shore. Similarly, if a small amount of water moves across the bedrock wall of a valley, the hydraulic distance to the impermeable boundary will be greater than the distance to the valley wall.

Fortunately, the hydraulic distance to boundaries can be determined from aquifer-test data analysis. According to the Theis equation, for equal drawdowns caused by the real well and the image well (in other words, if $s_r = s_i$), then

$$r_r^2/t_r = r_i^2/t_i \quad (29)$$

where r_r is the distance from the observation well to the real well, r_i is the distance from the observation well to the image well, t is the time at which a drawdown of s is caused by the real well at the observation well, and t_i is the time at which a drawdown of s_i is caused by the image well at the observation well.

Solving this equation for the distance to the image well from the observation well, results in

$$r_i = r_r(t_i/t_r)^{1/2} \quad (30)$$

The image well is located at some point on a circle having a radius of r_i centered on the observation well (Figure 5-24). Because the image well is the same distance from the boundary as the real well, the boundary must be located halfway between the image well and the pumping well (Figure 5-24).

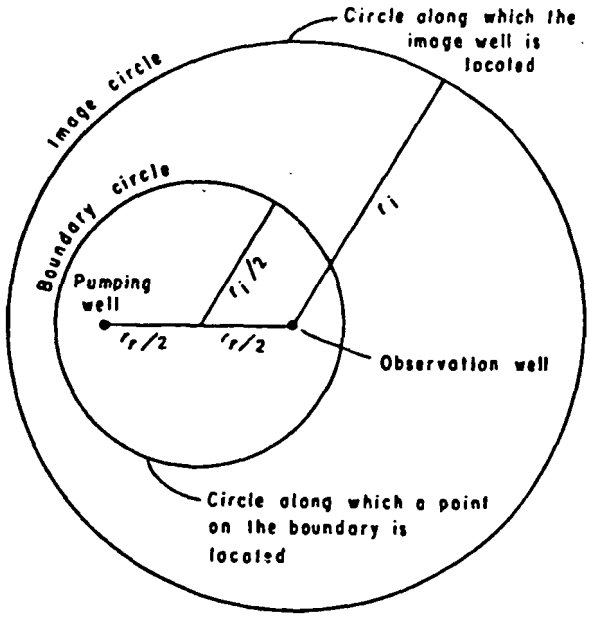

Figure 5-24. Method for Determining Location of Boundary

If the boundary is a stream or valley wall or some other feature whose physical position is obvious, its "hydraulic position" may be determined by using data from a single observation well. If, on the other hand, the boundary is the wall of a buried valley or some other feature not obvious from the land surface, distances to the image well from three observation wells may be needed to identify the position of the boundary.

Tests Affected By Leaky Confining Beds

In the development of the Theis equation for aquifer-test data analysis, it was assumed that all water discharged from the pumping well was derived instantaneously from storage in the aquifer. (See "Analysis of Aquifer-Test Data" earlier in this section.) Therefore, in the case of a confined aquifer, at least during the period of the test, the movement of water into the aquifer across its overlying and underlying confining beds is negligible. This assumption is satisfied by many confined aquifers. Many other aquifers, however, are bounded by leaky confining beds that transmit water into the aquifer in response to the withdrawals and cause drawdowns to differ from those that would be predicted by the Theis equation. The analysis of aquifer tests conducted on these aquifers requires the use of the methods that have been developed for semiconfined aquifers (also referred to in ground-water literature as "leaky aquifers").

Figures 5-25, 5-26, and 5-27 illustrate three different conditions commonly encountered in the field. Figure 5-25 shows a confined aquifer bounded by thick, impermeable confining beds. Water initially pumped from such an aquifer is from storage, and aquifer-test data can be analyzed by using the Theis equation. Figure 5-26 shows an aquifer overlain by a thick, leaky confining bed that, during an aquifer test, yields significant water from storage. The aquifer in this case may properly be referred to as a semiconfined aquifer, and the release of water from storage in the confining bed affects the analysis of aquifer-test data. Figure 5-27 shows an aquifer overlain by a thin confining bed that does not yield significant water from storage but that is sufficiently permeable to transmit water from the overlying unconfined aquifer into the semiconfined aquifer. Methods have been devised largely by Madhi Hantush and C. E. Jacob, 1955, for use in analyzing the leaky conditions illustrated in Figures 5-26 and 5-27.

These methods use matching data plots with type curves, as the Theis method does. The major difference is that, whereas the Theis method uses a single type curve, the methods applicable to semiconfined aquifers involve "families" of type curves, each curve of which reflects different combinations of the hydraulic characteristics of the aquifer and the confining beds.

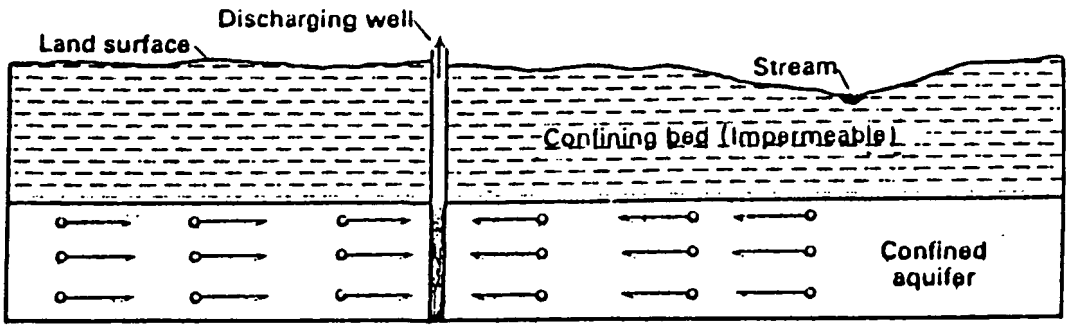

Figure 5-25. Nonleaky Artesian Conditions

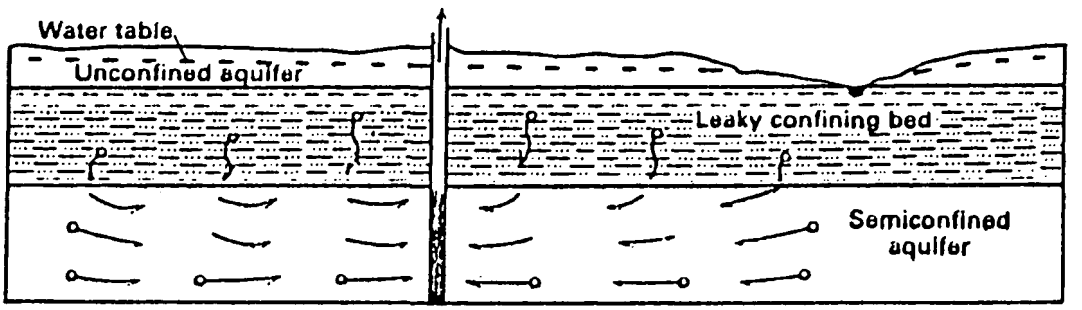

Figure 5-26. Semiconfined Aquifer with Leakage From Confining Bed

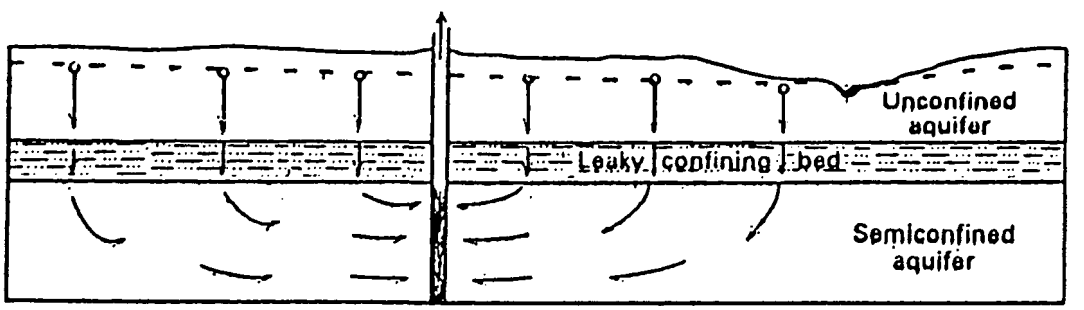

Figure 5-27. Semiconfined Aquifer with Leakage Through a Confining Bed

Data plots of s versus t on logarithmic graph paper for aquifer tests affected by release of water from storage in the confining beds are matched to the family of type curves illustrated in Figure 5-28. For convenience, these curves are referred to as Hantush type. Four match-point coordinates are selected and substituted into the following equations to determine values of T and S:

$$T = (QH(u,\beta))/(4\pi s) \quad (31)$$

$$S = (4Ttu)/r^2 \quad (32)$$

Data plots of s versus t on logarithmic graph paper for aquifer tests affected by leakage of water across confining beds are matched to the family of type curves shown in Figure 5-29. These type curves are based on equations developed by Hantush and Jacob and, for convenience, will be referred to as the Hantush-Jacob curves. The four coordinates of the match point are substituted into the following equations to determine T and S:

$$T=QW(u,r/B)/(4\pi s) \quad (33)$$

$$S = 4Ttu/r^2 \quad (34)$$

In planning and conducting aquifer tests, hydrologists

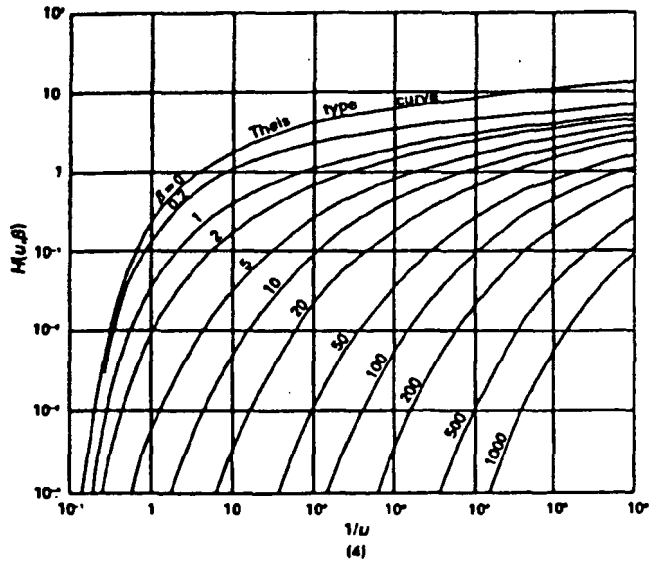

Figure 5-28. Hantush Type Curves for Semiconfined Aquifers that Receive Water From Storage In Confining Beds

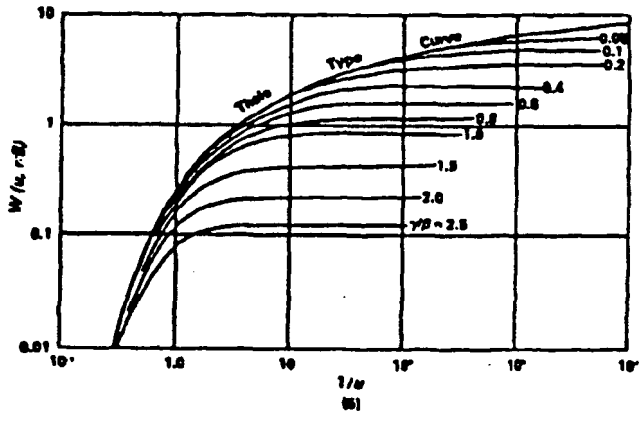

Figure 5-29. Hantush-Jacob Curves for Aquifers Receiving Leakage Across Confining Beds

must give careful consideration to the hydraulic characteristics of the aquifer and to the type of boundary conditions (either recharge or impermeable) that are likely to exist in the vicinity of the test site. Following completion of the test, the next problem is to select the method of analysis that most closely represents the geologic and hydrologic conditions in the area affected by the test. When these conditions are not well known, the common practice is to prepare a data plot of s versus t on logarithmic paper and match it with the Theis type curve. If the data closely match the type curve, the values of T and S determined by using the Theis equation should be reliable. Significant departures of the data from the type curve generally reflect the presence of lateral boundaries or leaky confining beds. Both the geology of the area and the shape of the data plot may provide clues as to which of these conditions most likely exist. It is important to note however that some data plots for tests affected by impermeable boundaries are similar in shape to the Hantush curves.

SPECIFIC CAPACITY AND TRANSMISSIVITY

The specific capacity of a well depends both on the hydraulic characteristics of the aquifer and on the construction and other features of the well. Values of specific capacity available for many supply wells for which aquifer-test data are not available are widely used by hydrologists to estimate transmissivity. Such estimates are used to evaluate regional differences in transmissivity and to prepare transmissivity maps for use in models of ground-water systems.

The factors that affect specific capacity include:

1. The transmissivity of the zone supplying water to the well which, depending on the length of the screen or open hole, may be considerably less than the transmissivity of the aquifer.
2. The storage coefficient of the aquifer.
3. The length of the pumping period.
4. The effective radius of the well, which may be significantly greater than the "nominal" radius.
5. The pumping rate.

The Theis equation can be used to evaluate the effect of the first four factors on specific capacity. The last factor, pumping rate, affects the well loss and can be determined only from a stepped-rate test or an aquifer test in which drawdowns are measured in both the pumping well and observation wells.

The Theis equation modified for the determination of transmissivity from specific capacity is

$$T = W(u)/4\pi \times Q/s \quad (35)$$

where T is transmissivity, Q/s is specific capacity, Q is the pumping rate, s is the drawdown, and W(u) is the well function of u where

$$u = (r^2S)/(4Tt) \quad (36)$$

where r is the effective radius of the well, S is the storage coefficient, and t is the length of the pumping period preceding the determination of specific capacity.

For convenience in using this equation it is desirable to

express $W(u)/4\pi$ as a constant. To do so it is first necessary to determine values for u and, using a table of values of u (or 1/u) and W(u), determine the corresponding values for W(u).

Values of u are determined by substituting in the equation values of T, S, r, and t that are representative of conditions in the area. For example, assume that in an area under investigation and for which a large number of values of specific capacity are available, that:

1. The principal aquifer is confined and aquifer tests indicate that it has a storage coefficient of about 2×10^{-4} and a transmissivity of about 11,000 $ft^2 d^{-1}$.
2. Most supply wells are 8 in. (20 cm) in diameter (radius 0.33 ft).
3. Most values of specific capacity are based on 12-hour well acceptance tests (t = 0.5 d).

Substituting these values, results in

$$u = (r^2S)/(4Tt) = (0.33\ ft)^2 \times (2 \times 10^{-4})/(4 \times (11{,}000\ ft^2\ d^{-1}) \times 0.5\ d \quad (37)$$

$$u = (2.22 \times 10^{-5} ft^2)/(2.2 \times 10^4 ft^2) = 1.01 \times 10^{-9} \quad (38)$$

A table of values of W(u) for values of 1/u is contained in Table 5-1. Therefore the value of u determined above must be converted to 1/u which is 9.91 x 108 and this value is used to determine the value of W(u). Values of W(u) are given for values of 1/u of 7.69×10^8 and 10×10^8 but not for 9.91×10^8. However the value of 10 is close enough to 9.91 for the purpose of estimating transmissivity from specific capacity. From Table 5-1 we determine that, for a value of 1/u of 10×10^8, the value of W(u) is 20.15. Substituting this value we find the constant W(u)/4p to be 1.60.

In using the equation, modified as necessary to fit the conditions in an area, it is important to recognize its limitations. Among the most important factors that affect its use are the accuracy with which the thickness of the zone supplying water to the well can be estimated, the magnitude of the well loss in comparison with drawdown in the aquifer, and the difference between the "nominal" radius of the well and its effective radius.

Relative to these factors the common practice is to assume that the value of transmissivity estimated from specific capacity applies only to the screened zone or to the open hole. To apply this value to the entire aquifer, the transmissivity is divided by the length of the screen or open hole (to determine the hydraulic conductivity per unit of length) and the result is multiplied by the entire thickness of the aquifer. The value of transmissivity determined by this method is too large if the zone supplying water to the well is thicker than the length of the screen or the open hole. Similarly, if the effective radius of the well is larger than the "nominal" radius (assuming that the "nominal" radius is used in the equation), the transmissivity based on specific capacity again will be too large.

On the other hand, if a significant part of the drawdown in the pumping well is due to well loss, the transmissivity based on specific capacity will be too small. Whether the effects of all three of these factors cancel out depends on the characteristics of both the aquifer and the well.

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Chapter 6

MODELS AND COMPUTERS IN GROUND-WATER INVESTIGATIONS

Models, in the broadest sense, are simplified descriptions of an existing physical system. Any ground-water investigation that does more than simply collect and tabulate data involves modeling. A preliminary model, or hypothesis, describing the ground-water system is tested by collecting data. If the data fit the hypothesis, the model is accepted; otherwise, the model must be revised. Models can be (1) qualitative descriptions of how processes operate in a system; (2) simplified physical representations of the system such as "sand tank" physical aquifer models and laboratory batch experiments to measure adsorption isotherms; and (3) mathematical representations of the physical system.

This chapter focuses on models that can be expressed in mathematical form and adapted for use in computer codes. The American Society for Testing and Materials (ASTM) defines model and computer code as follows (ASTM,1984):

A model is an assembly of concepts in the form of a mathematical equation that portrays understanding of a natural phenomenon.

A computer code is the assembly of numerical techniques, bookkeeping, and control languages that represents the model from acceptance of input data and instruction to delivery of output.

Modeling with computers is a specialized field that requires considerable training and experience. In the last few decades, literally hundreds of computer codes for simulating various aspects of ground-water systems have been developed. Refinements to existing codes and development of new codes proceed at a rapid pace. This chapter provides a basic understanding of modeling and data analysis with computers, including (1) their uses; (2) basic hydrogeologic parameters that define their type and capabilities; (3) classification according to mathematical approach and major types of hydrogeologic parameters simulated; (4) special management considerations in their use; and (5) their limitations.

Uses of Models and Computers

The great advantage of the computer is that large amounts of data can be manipulated quickly, and experimental modifications can be made with minimal effort, so that many possible situations for a given problem can be studied in great detail. The danger is that without proper selection, data collection and input, and quality control procedures, the computer's usefulness can be quickly undermined, bringing to bear the adage "garbage in, garbage out."

Computer codes involving ground water can be broadly categorized as (1) predictive, (2) resource optimizing, or (3) manipulative. Predictive codes simulate physical and chemical processes in the subsurface to provide estimates of how far, how fast, and in what directions a contaminant may travel. These are the most widely used codes and are the focus of most of this chapter.

Resource-optimizing codes combine constraining functions (e.g., total pumpage allowed) and optimization routines for objective functions (e.g., optimization of well field operations for minimum cost or minimum drawdown/pumping lift) with predictive codes. The U.S. Forest Service's multiple-objective planning process for management of national forests makes extensive use of resource-optimizing codes (Iverson and Alston, 1986). The availability of such codes for ground-water management is limited and is not a very active area of research and development (van der Heijde and others, 1985).

Manipulative codes primarily process and format data for easier interpretation or to assist in data input into predictive and resource-optimizing codes. A specific computer code may couple one or more of these types

of codes. For example, codes that facilitate data entry (preprocessors) and data output (postprocessors) are becoming an increasingly common feature of predictive codes.

Government Decision-Making

Computers can assist government decisions concerning ground-water evaluation/protection in the areas of (1) policy formulation, (2) rule-making, and (3) regulatory action.

A study by the Holcomb Research Institute (1976) of environmental modeling and decision-making in the United States provides a good overview of modeling for policy formulation, although most of the case studies involve surface water and resources other than ground water. The Office of Technology Assessment (1982) more specifically addresses the use of water resource models for policy formulation.

The U.S. EPA's Underground Injection Control Program regulations on restrictions and requirements for Class I wells exemplify the use of modeling to assist in rule-making (Proposed Rules: 52 Federal Register 32446-32476, August 27, 1987; Final Rules: 53 Federal Register 28118-28157, July 26, 1988). The 10,000-year no-migration standard in 40 CFR 128.20(a)(1) for injected wastes is based, in part, on numerical modeling of contaminant transport in four major hydrogeologic settings by Ward and others (1987). Furthermore, worst-case modeling of typical injection sites by EPA formed the basis for the decision not to require routine modeling of dispersion in no-migration petitions.

Ground-water flow and, possibly, solute transport modeling are required to obtain a permit to inject hazardous wastes into Class I wells. Permitting decisions involving activities that may pose a threat to ground-water quality, such as landfills and surface storage of industrial wastes, commonly require ground-water simulations to demonstrate that no hazard exists. U.S. EPA (1987) provides a good overview of the use of models in managing ground-water protection programs.

Site Assessment and Remediation

Use of modeling and computer codes can be valuable in three phases of site-specific ground-water investigations: (1) site characterizaton, (2) exposure assessment, and (3) remediation assessment.

Site Characterization. Relatively simple models (such as analytic solutions) may be useful at the early stage for roughly defining the possible magnitude of a contaminant problem. Solute transport models that account for dispersion but not retardation may be useful in providing a worst-case analysis of the situation. They may help in defining the size of the area to be studied and in siting of monitoring wells. If more sophisticated computer modeling is planned, the specific code to be used will, to a certain extent, guide site characterization efforts by the aquifer parameters required as inputs to the model. Site characterization, particularly where water-quality samples are tested for possible organic contaminants, can generate large amounts of data. Computers are invaluable in compiling and processing these data.

Exposure Assessment. There is growing use of exposure assessments across EPA's regulatory programs (U.S. EPA, 1987). In the case of ground-water contamination, the results of an exposure assessment will often determine whether remediation will be required.

Remediation. Predictive models can be particularly valuable in estimating the possible effectiveness of alternative approaches to remediating ground-water contamination (Boutwell and others, 1985). Table 6-1 summarizes the types of modeling required for various remediation design features.

Hydrogeologic Model Parameters

All modeling involves simplifying assumptions concerning parameters of the physical system that is being simulated. Furthermore, these parameters will influence the type and complexity of the equations that are used to represent the model mathematically. There are six major parameters of ground-water systems that must be considered when developing or selecting a computer code for simulating ground-water flow and six additional parameters for contaminant transport.

Ground-Water Flow Parameters

Type of Aquifer. Confined aquifers of uniform thickness are easier to model than unconfined aquifers because the transmissivity remains constant. The thickness of unconfined aquifers varies with fluctuations in the water table, thus complicating calculations. Similarly, simulation of variable-thickness confined aquifers is complicated by the fact that velocities will generally increase in response to reductions and decrease in response to increases in aquifer thickness.

Matrix Characteristics. Flow in porous media is much easier to model than in rocks with fractures or solution porosity. This is because (1) equations governing laminar flow are simpler than those for turbulent flow,

Table 6-1. Modeling Designed-System Alterations and Corrective Action

Design Feature	Effects on Ground Water	Type of Model Required	Typical Modeling Problems
Capping, grading and revegetation	Reduction of infiltration Reduction of successive leachate generation	Unsaturated zone model, vertical layered	Parameters related to leaching characteristics of reworked soil
Ground-water pumping (and optional reinjection of treated water)	Changes in heads, direction of flow, and contaminant migration Controlled plume removal	Saturated zone model, two-dimensional areal, axisymmetric or three-dimensional; well or series of wells assigned to individual node	Representing partial penetration
Wastewater injection	Changes in heads and direction of flow Plume generation	Saturated zone model, two-dimensional areal, axisymmetric or three-dimensional; density-dependent flow; temperature difference effects	Representing density-dependent effects
Interceptor trenches	Changes in heads, direction of flow, and contaminant migration Plume removal	Saturated zone model, two-dimensional areal or cross-sectional, or three dimensional; trenches are represented by line of notes with assigned heads	Representing partial penetration, resolution near trenches

Table 6-1. Continued

Design Feature	Effects on Ground Water	Type of Model Required	Typical Modeling Problems
Impermeable barrier (optional drainage system to prevent mounding)	Containment of polluted water Routing unpolluted ground water around site Changes in heads and direction of flow	Saturated zone model, two-dimensional areal or cross-sectional, or three-dimensional; possibly two-dimensional cross-sectional unsaturated zone for liners	Representing partial penetration, flow and transport around end of barrier(s) Conductivity liner or barrier material Large changes in conductivity between neighboring elements Differences in required grid resolution
Subsurface drains	Removal of leachate Changes in heads, direction of flow, and contaminant migration	Saturated or combined unsaturated-saturated zone model, two-dimensional cross-sectional or three-dimensional	Resolution near drain
Solution mining	Removal of contaminants after induced mobilization	Saturated or combined unsaturated-saturated zone model, two-dimensional areal, cross-sectional or three-dimensional Lines of Sources (injection) and sinks (removal)	Parameters related to mobilization (sorption coefficient, retardation coefficient)
Excavation	Removal of waste material and pollutes soil changes in hydraulic characteristics and boundary conditions changes in heads and direction of flow	Unsaturated, saturated, or combined unsaturated-saturated zone model; for unsaturated some models minimal one-dimensional vertical, for other types minimal two-dimensional, cross-sectional.	Parameters of backfill material

Source: Adapted by van der Heijde et al. (1988) from Boutwell et al. (1985).

which may occur in fracture; and (2) effective porosity and hydraulic conductivity can be more easily estimated for porous media.

Homogeneity and Isotropy. Homogeneous and isotropic aquifers are easiest to model because their properties do not vary in any direction. If hydraulic properties and concentrations are uniform vertically, and in one of two horizontal dimensions, a one-dimensional simulation is possible. Horizontal variations in properties combined with uniform vertical characteristics can be modeled two-dimensionally. Most aquifers, however, show variation in all directions and, consequently, require three-dimensional simulation, which also necessitates more extensive site characterization data. The spatial uniformity or variability of aquifer parameters such as recharge, hydraulic conductivity, effective porosity, transmissivity, and storativity will determine the number of dimensions to be modeled.

Phases. Flow of ground water and contaminated ground water in which the dissolved constituents do not create a plume that differs greatly from the unpolluted aquifer in density or viscosity are fairly easy to simulate. Multiple phases, such as water and air in the vadose zone and NAPLs in ground water, are more difficult to simulate.

Number of Aquifers. A single aquifer is easier to simulate than multiple aquifers.

Flow Conditions. Steady-state flow, where the magnitude and direction of flow velocity are constant with time at any point in the flow field, is much easier to simulate than transient flow. Transient, or unsteady flow, occurs when the flow varies in the unsaturated zone in response to variations in precipitation, and in the saturated zone when the water table fluctuates.

Contaminant Transport Parameters

Type of Source. For simulation purposes, sources can be characterized as point, line, area, or volume. A point source enters the ground water at a single point, such as a pipe outflow or injection well, and can be simulated with either a one-, two-, or three-dimensional model. An example of a line source would be contaminants leaching from the bottom of a trench. An area source enters the ground water through a horizontal or vertical plane. The actual contaminant source may occupy three dimensions outside of the aquifer, but contaminant entry into the aquifer can be represented as a plane for modeling purposes. Leachate from a waste lagoon or an agricultural field are examples of area sources. A volume source occupies three dimensions within an aquifer. A DNAPL that has sunk to the bottom of an aquifer would be a volume source. Line and area sources may be simulated by either two- or three-dimensional models, whereas a volume source would require a three-dimensional model. Figure 6-1 illustrates the type of contaminant plume that results from a landfill in the following cases: (1) area source on top of the aquifer, (2) area source within the aquifer and perpendicular to the direction of flow, (3) vertical line source in the aquifer, and (4) point source on top of the aquifer.

Type of Source Release. Release of an instantaneous pulse, or slug, of contaminant is easier to model than a continuous release. A continuous release may be either constant or variable.

Dispersion. Accurate contaminant modeling requires incorporation of transport by dispersion. Unfortunately, the conventional convective-dispersion equation often does not accurately predict field-scale dispersion (U.S. EPA, 1988).

Adsorption. It is easiest to simulate adsorption with a single distribution or partition coefficient. Nonlinear adsorption and temporal and spatial variation in adsorption are more difficult to model.

Degradation. As with adsorption, simulation of degradation is easiest when using a simple first-order degradation coefficient. Second-order degradation coefficients, which result from variations in various parameters, such as pH, substrate concentration, and microbial population, are much more difficult to model. Simulation of radioactive decay is complicated but easier to simulate with precision because decay chains are well known.

Density/Viscosity Effects. If temperature or salinity of the contaminant plume is much different than that of the pristine aquifer, simulations must include the effects of density and viscosity variations.

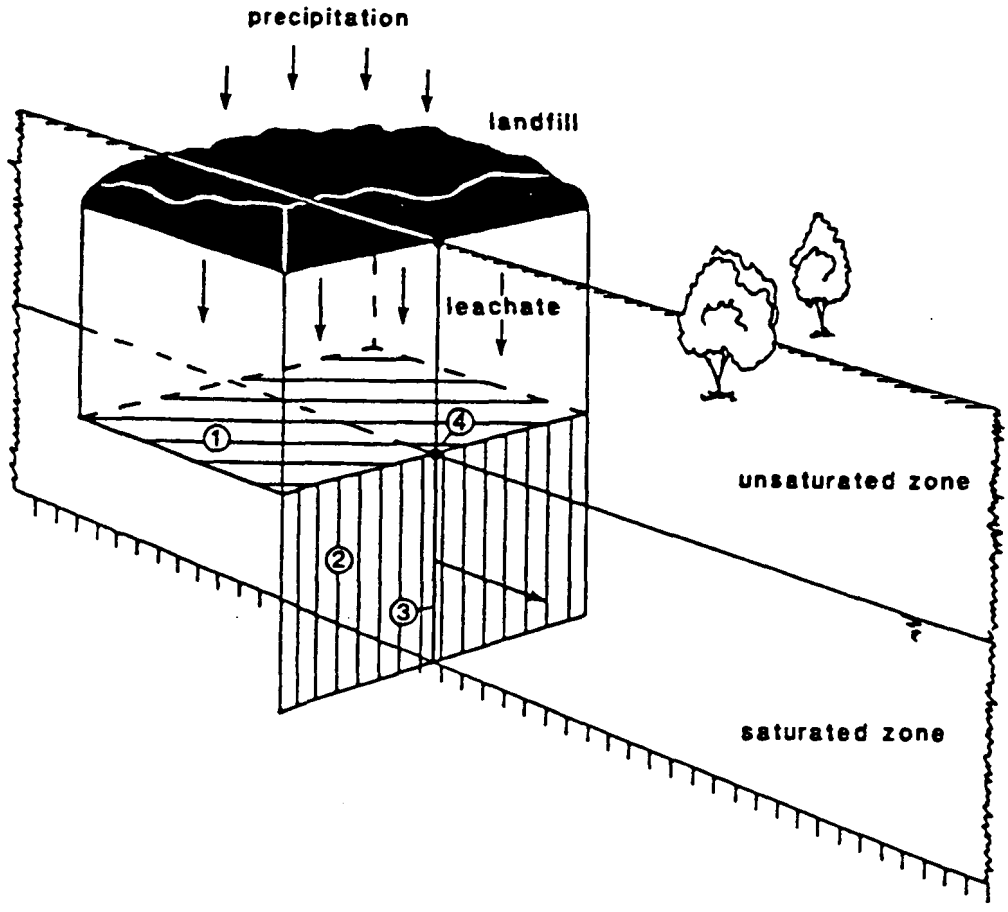
Types of Models and Codes

Ground-water models and codes can be classified in many different ways, including the mathematical approaches used to develop computer codes, as computer prediction codes, and as manipulative codes.

Mathematical Approaches

Models and codes are usually described by the number of dimensions simulated and the mathematical approaches used. At the core of any model or computer code are governing equations that represent the system being modeled. Many different approaches to formulating and solving the governing equations are possible. The specific numerical technique embodied in a computer code is called an algorithm. The following

a. various ways to represent source.



b. horizontal spreading resulting from various source assumptions.

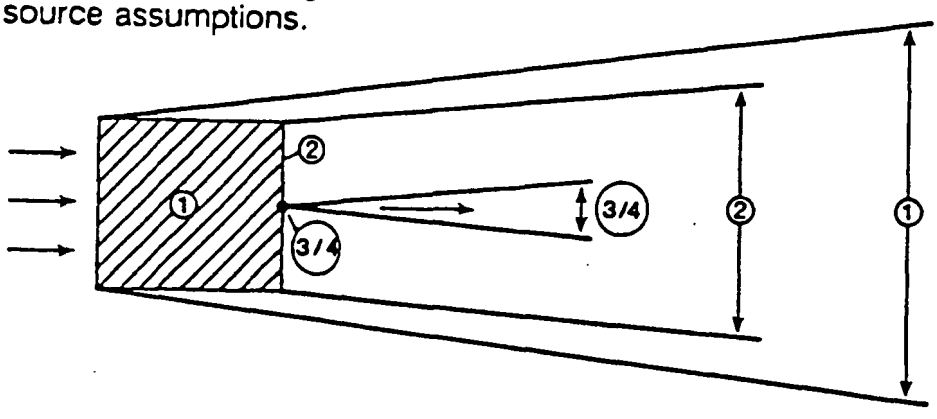
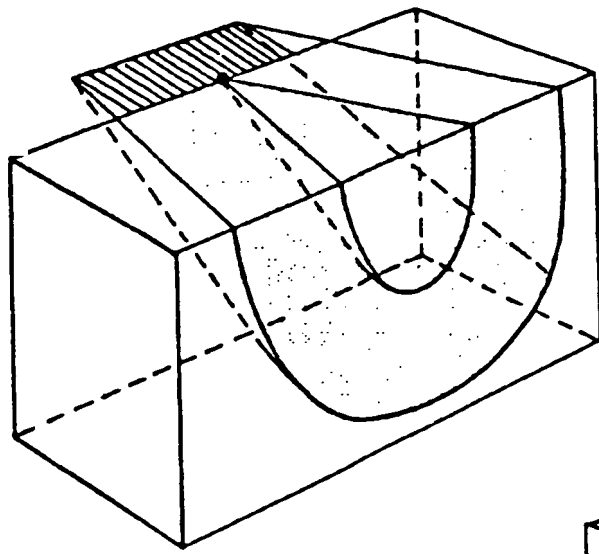
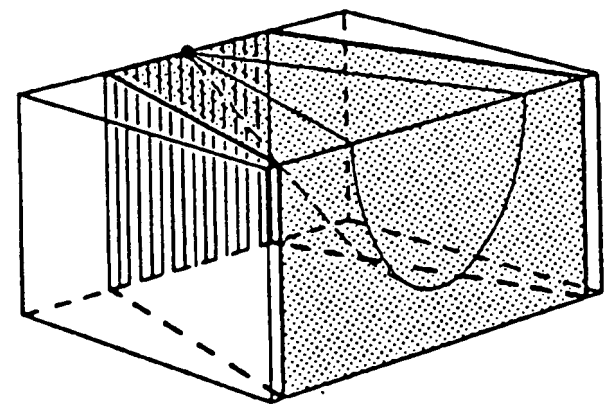


Figure 6-1. Definition of the Source Boundary Condition Under a Leaking Landfill (numbers 1 to 4 refer to cases 1 to 4) (from van der Heijde and others, 1988)

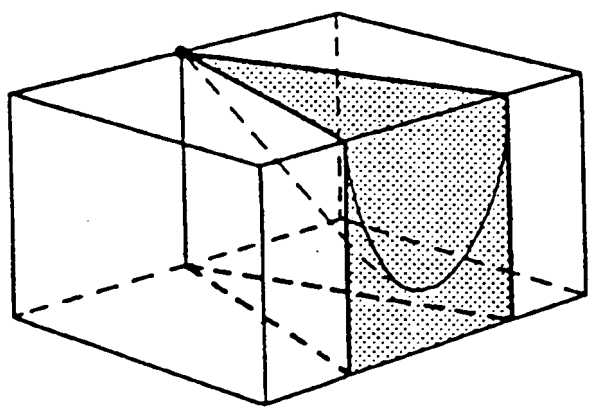
c. detailed view of 3D spreading for various ways to represent source boundary.



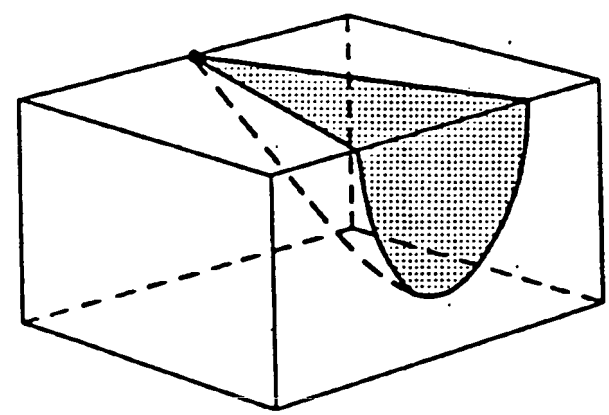
Case 1:
horizontal 2D-areal source at top of aquifer (for 3D modeling)



Case 2: vertical 2D-source in aquifer (for 2D horizontal, vertically averaged, or 3D modeling)



Case 3:
1D vertical line source in aquifer (for 2D horizontal, vertically averaged, 2D cross-sectional, or 3D modeling)



Case 4: point source at top of aquifer (for 2D or 3D modeling)

Figure 6-1. Continued

discussion compares and contrasts some of the most important choices that must be made in mathematical modeling.

Deterministic vs. Stochastic Models. A deterministic model presumes that a system or process operates so that a given set of events leads to a uniquely definable outcome. The governing equations define precise cause-and-effect or input-response relationships. In contrast, a stochastic model presumes that a system or process operates so that a given set of events leads to an uncertain outcome. Such models calculate the probability, within a desired level of confidence, of a specific value occurring at any point.

Most available models are deterministic. However, the heterogeneity of hydrogeologic environments, particularly the variability of parameters, such as effective porosity and hydraulic conductivity, plays a key role in influencing the reliability of predictive ground-water modeling (Smith, 1987; Freeze and others, 1989). Stochastic approaches to characterizing variability with the use of geostatistical methods, such as kriging, are being used with increasing frequency to characterize soil and hydrogeologic data (Hoeksma and Kitandis, 1985; Warrick and others, 1986). The governing equations for both deterministic and stochastic models can be solved either analytically or numerically.

Analytical vs. Numerical Models. A model's governing equation can be solved either analytically or numerically. Analytical models use exact closed-form solutions of the appropriate differential equations. The solution is continuous in space and time. In contrast, numerical models apply approximate solutions to the same equations.

Analytical models provide exact solutions, but employ many simplifying assumptions in order to produce tractable solutions; thus placing a burden on the user to test and justify the underlying assumptions and simplifications (Javendel and others, 1984).

Numerical models are much less burdened by these assumptions and, therefore, are inherently capable of addressing more complicated problems, but they require significantly more data, and their solutions are inexact (numerical approximations). For example, the assumptions of homogeneity and isotropicity are unnecessary because the model can assign point (nodal) values of transmissivity and storativity. Likewise, the capacity to incorporate complex boundary conditions provides greater flexibility. The user, however, faces difficult choices regarding time steps, spatial grid designs, and ways to avoid truncation errors and numerical oscillations (Remson and others, 1971; Javendel and others, 1984). Improper choices may result in errors unlikely to occur with analytical approaches (e.g., mass imbalances, incorrect velocity distributions, and grid-orientation effects).

Grid Design. A fundamental requirement of the numerical approach is the creation of a grid that represents the aquifer being simulated (see Figures 6-2 and 6-3). This grid of interconnected nodes, at which process input parameters must be specified, forms the basis for a matrix of equations to be solved. A new grid must be designed for each site-specific simulation based on data collected during site characterization. Good grid design is one of the most critical elements for ensuring accurate computational results.

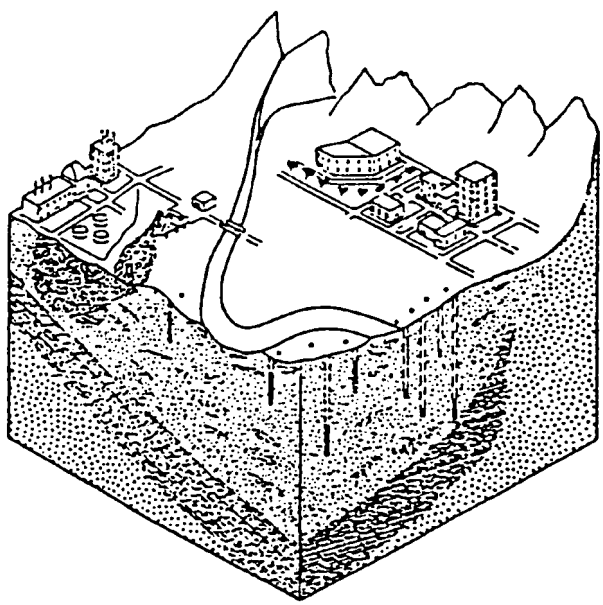
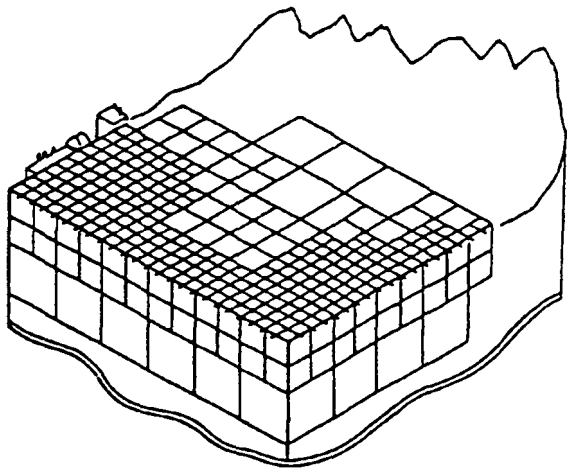


Figure 6-2. Typical Ground-Water Contamination Scenario. Several Water-Supply Production Wells are Located Downgradient of a Contaminant Source and the Geology Is Complex.

The grid design is influenced by the choice of numerical solution technique. Numerical solution techniques include (1) finite-difference methods (FD); (2) integral finite-difference methods (IFDM); (3) Galerkin and variational finite element methods (FE); (4) collocation methods; (5) boundary (integral) element methods (BIEM or BEM); (6) particle mass tracking methods, such as the RANDOM WALK (RW) model; and (7) the method of characteristics (MOC) (Huyakorn and Pinder, 1983; Kinzelbach, 1986). Figure 6-4 illustrates grid designs involving FD, FE, collocation, and boundary methods. Finite-difference and finite-element methods are the most frequently used and are discussed further below.



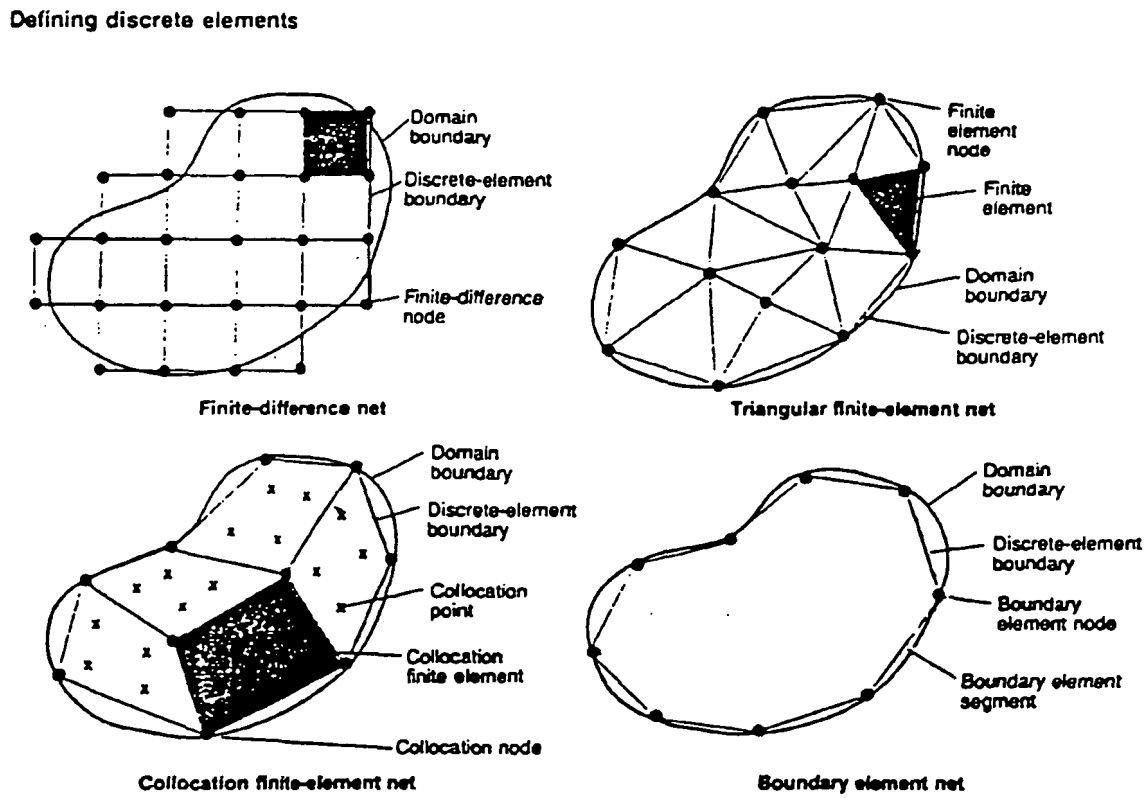
Values for natural process parameters would be specified at each node of the grid in performing simulations. The grid density is greatest at the source and at potential impact locations.

Figure 6-3. Possible Contaminant Transport Model Grid Design for the Situations Shown in Figure 6-2

Finite Difference vs. Finite Element. The finite-element method approximates the solution of partial differential equations by using finite-difference equivalents, whereas the finite-difference method approximates differential equations by an integral approach. Figure 6-5 illustrates the mathematical and computational differences in the two approaches. Table 6-2 compares the relative advantages and disadvantages of the two methods. In general, finite-difference methods are best suited for relatively simple hydrogeologic settings, whereas finite-element methods are required where hydrogeology is complex.

Ground-Water Computer Prediction Codes

Terminology for classifying computer codes according to the kind of ground-water system they simulate is not uniformly established. There are so many different ways that such models can be classified (i.e., porous vs. fractured-rock flow, saturated vs. unsaturated flow, mass flow vs. chemical transport, single phase vs. multiphase, isothermal vs. variable temperature) that a systematic classification cannot be developed that would not require placement of single codes in multiple categories.



Note: Each node represents one equation per independent variable, except in the case of collocation, in which each collocation point represents one equation. The boundary element, collocation, and finite-element methods offer flexibility in geometric representation.

Figure 6-4. Influence of Numerical Solution Technique on Grid Design (from Pinder, 1984)

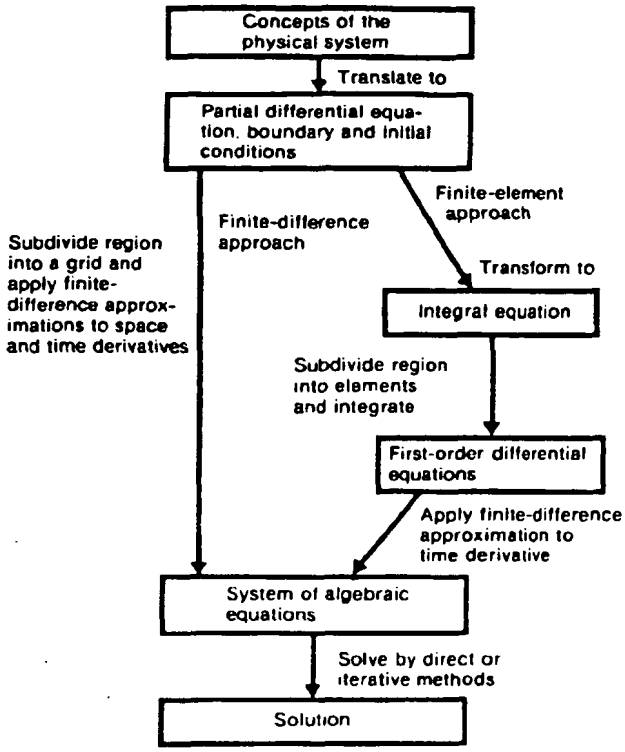


Figure 6-5. Generalized Model Development by Finite-Difference and Finite-Element Methods (from Mercer and Faust, 1981)

Table 6-3 identifies four major categories of codes and 11 major subdivisions, which are discussed below. This classification scheme differs from others (see, for example, Mangold and Tsang, 1987; van der Heijde and others, 1988), by distinguishing among solute transport models that simulate (1) only dispersion, (2) chemical reactions with a simple retardation or degradation factor, and (3) complex chemical reactions.

The literature on ground-water codes often is further confused by conflicting terminology. For example, the term "hydrochemical" has been applied to completely different types of codes. Van der Heidje and others (1988) used the term hydrochemical for codes listed in the geochemical category in Table 6-3, whereas Mangold and Tsang (1987) used the same term to describe coupled geochemical and flow models (chemical-reaction transport codes in Table 6-3).

Porous Media Flow Codes. Modeling of saturated flow in porous media is relatively straightforward; consequently, by far the largest number of codes are available in this category. Van der Heijde and others (1988) summarize 97 such models. These models are not suitable for modeling contaminant transport if dispersion is a significant factor, but they may be required for evaluating hydrodynamic containment of contaminants and pump-and-treat remediation efforts. Modeling variably saturated flow in porous media (most

Advantages	Disadvantages
Finite-Difference Method	
Intuitive basis	Low accuracy for some problems
Easy data entry	Regular grids required
Efficient matrix techniques	
Programming changes easy	
Finite-Element Method	
Flexible grid geometry	Complex mathematical basis
High accuracy possible	Difficult data input
Evaluates cross-product terms better	Difficult programming

Source: Adapted from Mercer and Faust (1981).

Table 6-2. Advantages and Disadvantages of FDM and FEM Numerical Methods

typically soils and unconsolidated geologic material) is more difficult because hydraulic conductivity varies with changes in water content in unsaturated materials. Such codes typically must model processes, such as capillarity, evapotranspiration, diffusion, and plant water uptake. Van der Heijde and others (1988) summarized 29 models in this category.

Solute Transport Codes. The most important types of codes in the study of ground-water contamination simulate the transport of contaminants in porous media. This is the second largest category (73 codes) identified by van der Heidje and others (1988) as being readily available. Solute transport codes fall into three major categories (see Table 6-3 for descriptions): (1) dispersion codes, (2) retardation/degradation codes, and (3) chemical-reaction transport codes. Dispersion codes differ from saturated flow codes only in having a dispersion factor, and they have limited utility except perhaps for worst-case analyses, since few contaminants act as conservative tracers. Retardation/degradation codes are slightly more sophisticated because they add a retardation or degradation factor to the mass transport and diffusion equations. Chemical reaction-transport codes are the most complex (but not necessarily the most accurate) because they couple geochemical codes with flow codes. Chemical reaction-transport codes may be classified as integrated or two-step codes.

Geochemical Codes. Geochemical codes simulate chemical reactions in ground-water systems without considering transport processes. These fall into three major categories (see Table 6-3): (1) thermodynamic

Type of Code	Description/Uses
Flow (Porous Media)	
Saturated	Simulates movement of water in saturated porous media. Used primarily for analyzing ground-water availability.
Variable saturated	Simulates unsaturated flow of water in the vadose (unsaturated) zone. Used in study of soil-plant relationships, hydrologic cycle budget analysis.
Solute Transport (Porous Media)	
Dispersion	Simulates transport of conservative contaminants (not subject to retardation) by adding a dispersion factor into flow calculations. Used for nonreactive contaminants such as chloride and for worst-case analysis of contaminant flow.
Retardation/ Degradation	Simulates transport contaminants that are subject to partitioning of transformation by the addition of relatively simple retardation or degradation factors to algorithms for advection-dispersion flow. Used where retardation and degradation are linear with respect to time and do not vary with respect to concentration.
Chemical-reaction	Combines an advection-dispersion code with a transport geochemical code (see below) to simulate chemical speciation and transport. Integrated codes solve all mass momentum, energy-transfer, and chemical reaction equations simultaneously for each time interval. Two-step codes first solve mass momentum and energy balances for each time step and then requilibrate the chemistry using a distribution-of-species code. Used primarily for modeling behavior of inorganic contaminants.

Table 6-3. Classification of Types of Computer Codes

Type of Code	Description/Uses
Geochemical Codes	
Thermodynamic	Processes empirical data so that thermodynamic data at a standard reference state can be obtained for individual species. Used to calculate reference state values for input into geochemical speciation calculations.
Distribution-of-species (equilibrium)	Solves a simultaneous set of equations that describe equilibrium reactions and mass balances of the dissolved elements.
Reaction progress (mass-transfer)	Calculates both the equilibrium distribution of species (as with equilibrium codes) and the new composition of the water, as selected minerals are precipitated of dissolved.
Specialized Codes	
Fracture rock	Simulates flow of water in fractured rock. Available codes cover the spectrum of advective flow, advection-dispersion, heat, and chemical transport.
Heat transport	Simulates flow where density-induced and other flow variations resulting from fluid temperature differences invalidate conventional flow and chemical transport modeling. Used primarily in modeling of radioactive waste and deep-well injection.
Multiphase flow	Simulates movement of immiscible fluids (water and nonaqueous phase liquids) in either the vadose or saturated zones. Used primarily where contamination involves liquid hydrocarbons or solvents.

Source: Adapted from van der Heijde and others (1988) and U.S. EPA (1989).

Table 6-3. Continued

codes, (2) distribution-of-species codes, and (3) reaction progress codes. Thermodynamic codes perhaps would be classified more properly as manipulative codes, but are included here because of their special association with geochemical codes. Such codes are especially important for geochemical modeling of deep-well injection where temperatures and pressures are higher than near-surface conditions for which most geochemical codes were developed. Apps (1989) reviews the availability and use of thermodynamic codes

.

By themselves, geochemical codes can provide qualitative insights into the behavior of contaminants in the subsurface. They also may assist in identifying possible precipitation reactions that might adversely affect the performance of injection wells in pump-and-treat remediation efforts. Chemical transport modeling of any sophistication requires coupling geochemical codes with flow codes. Over 50 geochemical codes have been described in the literature (Nordstrom and Ball, 1984), but only 15 are cited by van der Heijde and others (1988) as passing their screening criteria for reliability and usability.

Specialized Codes. This category contains special cases of flow codes and solute transport codes (see Table 6-3), including (1) fractured rock, (2) heat transport, and (3) multiphase flow. Fractured rock creates special problems in the modeling of contaminant transport for several reasons. First, mathematical representation is more complex due to the possibility of turbulent flow and the need to consider roughness effects. Furthermore,

precise field characterization of fracture properties that influence flow, such as orientation, length, and degree of connection between individual fractures, is extremely difficult. In spite of these difficulties, much work is being done in this area (Schmelling and Ross, 1989). Van der Heijde and others (1988) identified 27 fractured rock models.

Heat transport models have been developed primarily in connection with enhanced oil-recovery operations (Kayser and Collins, 1986) and programs assessing disposal of radioactive wastes. Van der Heijde and others (1988) summarized 36 codes of this type. Early work in multiphase flow centered in the petroleum industry focusing on oil-water-gas phases. In the last decade, multiphase behavior of nonaqueous phase liquids in near-surface ground-water systems has received increasing attention. However, the number of codes capable of simulating multiphase flow is still limited.

Manipulative Codes

Manipulative codes that may be of value in ground-water investigations include (1) parameter identification codes, (2) data processing codes, and (3) geographic information systems.

Parameter Identification Codes. Parameter identification codes most often are used to estimate the aquifer parameters that determine fluid flow and contaminant transport characteristics. Examples of such codes include annual recharge (Pettyjohn and Henning, 1979; Puri, 1984), coefficients of permeability and storage (Shelton, 1982; Khan, 1986a and 1986b), and dispersivity (Guven and others, 1984; Strecker and Chu, 1986).

Data Processing Codes. Data manipulation codes specifically designed to facilitate ground-water modeling efforts have received little attention until recently. They are becoming increasingly popular, because they simplify data entry (preprocessors) to other kinds of models and facilitate the production of graphic displays (postprocessors) of the data outputs of other models (van der Heijde and Srinivasan, 1983; Srinivasan, 1984; Moses and Herman, 1986). Other software packages are available for routine and advanced statistics, specialized graphics, and database management needs (Brown, 1986).

Geo-EAS (Geostatistical Environmental Assessment Software) is a collection of interactive software tools for performing two-dimensional geostatistical analyses of spatially distributed data. It includes programs for data file management, data transformations, univariate statistics, variogram analysis, cross validation, kriging, contour mapping, post plots, and line/scatter graphs in a user-friendly format. This package can be obtained from the Arizona Computer Oriented Geological Society (ACOGS), P.O. Box 44247, Tucson, AZ, 85733-4247.

Geographic Information Systems. Geographic information systems (GIS) provide data entry, storage, manipulation, analysis, and display capabilities for geographic, environmental, cultural, statistical, and political data in a common spatial framework. EPA's Environmental Monitoring System Laboratory in Las Vegas (EMSL-LV) has been piloting use of GIS technology at hazardous waste sites that fall under RCRA and CERCLA guidance. The American Society for Photogrammetry and Remote Sensing is a primary source of information on GIS.

Management Considerations for Code Use

The effective use of ground-water models is often inhibited by a communication gap between managers who make policy and regulatory decisions and technical personnel who develop and apply the models (van der Heijde and others, 1988). This section focuses on the following management considerations for using models and codes: personnel and communication requirements, cost of hardware and software options, selection criteria, and quality assurance.

Personnel/Communication

The successful use of mathematical models depends on the training and experience of the technical support staff applying the model to a problem, and on the degree of communication between these technical persons and management. Managers should be aware that a fair degree of specialized training and experience are necessary to develop and apply mathematical models, and relatively few technical support staff can be expected currently to have such skills (van der Heijde and others, 1985). Technical personnel need to be familiar with a number of scientific disciplines, so that they can structure models to faithfully simulate real-world problems.

A broad, multidisciplinary team is mandatory for adequate modeling of complex problems, such as contaminant transport in ground water. No individual can master the numerous disciplines involved in such an effort; however, staff should have a working knowledge of many sciences so that they can address appropriate questions to specialists, and achieve some integration of the various disciplines involved in the project. In practice, ground-water modelers should become involved in continuing education efforts, which managers should expect and encourage. The benefits

of such efforts are likely to be large, and the costs of not engaging in them may be equally large.

Technical staff also must be able to communicate effectively with management. As with statistical analyses, an ill-posed problem yields answers to the wrong questions. Tables 6-3 through 6-5 list some useful questions managers and technical support staff should ask each other to ensure that the solution being developed is appropriate to the problems. Table 6-3 consists of "screening level" questions, Table 6-4 addresses correct conceptualizations, and Table 6-5 contains questions of sociopolitical concern.

Cost of Hardware and Software Options

The nominal costs of the support staff, computing facilities, and specialized graphics' production equipment associated with numerical modeling efforts can be high. In addition, quality control activities can result in substantial costs, depending on the degree to which a manager must be certain of the model's characteristics and accuracy of output.

As a general rule, costs are greatest for personnel, moderate for hardware, and minimal for software. An optimally outfitted business computer (e.g., VAX 11/785 or IBM 3031) costs about $100,000, but it can rapidly pay for itself in terms of dramatically increased speed and computational power. In contrast, a well-complemented personal computer (e.g., IBM-PC/AT or DEC Rainbow) may cost $10,000, but the significantly slower speed and limited computational power may incur hidden costs in terms of its inability to perform specific tasks. For example, highly desirable statistical packages like SAS and SPSS are unavailable or available only with reduced capabilities for personal computers. Many of the most sophisticated mathematical models are available in their fully capable form only on business computers.

Figure 6-6 compares typical software costs for different levels of computing power. Obviously, the software for less capable computers is less expensive, but the programs are not equivalent; managers need to seriously consider which level is appropriate. If the modeling

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Assumptions and Limitations

What are the assumptions made, and do they cast doubt on the model's projections for this problem?
What are the model's limitations regarding the natural processes controlling the problem? Can the full spectrum of probable conditions be addressed?
How far in space and time can the results of the model simulations be extrapolated?
Where are the weak spots in the application, and can these be further minimized or eliminated?

Input Parameters and Boundary Conditions

How reliable are the estimates of the input parameters? Are they quantified within accepted statistical bounds?
What are the boundary conditions, and why are they appropriate to this problem?
Have the initial conditions with which the model is calibrated been checked for accuracy and internal consistency?
Are the spatial grid design(s) and time-steps of the model optimized for this problem?

Quality Control and Error Estimation

Have these models been mathematically validated against other solutions to this kind of problem?
Has anyone field verified these models before, by direct applications or simulation of controlled experiments?
How do these models compare with others in terms of computational efficiency, and ease of use or modification?
What special measures are being taken to estimate the overall errors of the simulations?

Source: Keely (1987).

Table 6-4. Conceptualization Questions for Mathematical Modeling Efforts

Demographic Considerations

Is there a larger population endangered by the problem than we are able to provide sufficient responses to?
Is it possible to present the model's results in both nontechnical and technical formats, to reach all audiences?
What role can modeling play in public information efforts?
How prepared are we to respond to criticism of the model(s)?

Political Constraints

Are there nontechnical barriers to using this model, such as "tainted by association" with a controversy elsewhere?
Do we have the cooperation of all involved parties in obtaining the necessary data and implementing the solution?
Are similar technical efforts for this problem being undertaken by friend or foe?
Can the results of the model simulations be turned against us? Are the results ambiguous or equivocal?

Legal Concerns

Will the present schedule allow all regulatory requirements to be met in a timely manner?
If we are dependent on others for key inputs to the model(s), how do we recoup losses stemming from their nonperformance?
What liabilities are incurred for projections that later turn out to be misinterpretations originating in the model?
Do any of the issues relying on the applications of the model(s) require the advice of attorneys?

Source: Keely (1987).

Table 6-5. Sociopolitical Questions for Mathematical Modeling Efforts

decisions will be based on very little data, it may not make sense to insist on the most elegant software and hardware. If the intended use involves substantial amounts of data, however, and sophisticated analyses are desired, it would be unwise to opt for the least expensive combination.

There is an increasing trend away from both ends of the hardware and software spectrum and toward the middle; that is, the use of powerful personal computers is increasing rapidly, whereas the use of small programmable calculators and large business computers alike is declining. In part, this trend stems from significant improvements in the computing power and quality of printed outputs obtainable from personal computers. It also is due to the improved telecommunications capabilities of personal computers, which are now able to emulate the interactive terminals of large business computers so that vast computational power can be accessed and the results retrieved with no more than a phone call. Most importantly for ground-water managers, many of the mathematical models and data packages have been "down-sized" from mainframe computers to personal computers; many more are now being written directly for this market. Figure 6-7 provides some idea of the costs of available software and hardware for personal computers.

Code Selection Criteria

Technical criteria for selecting ground-water modeling codes have been formulated by U.S. EPA (1988) in the form of a decision tree (Figure 6-8). These technical criteria correspond roughly to the hydrogeologic model parameters discussed earlier. Table 6-6 summarizes information with respect to these technical criteria for 49 analytical and numerical ground-water codes. More detailed information about these codes can be found in U.S. EPA (1988).

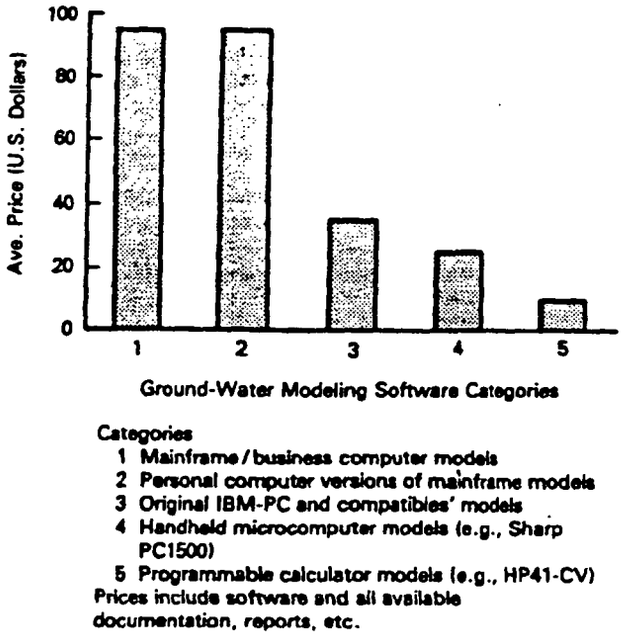

Figure 6-6. Average Price per Category for Ground-Water Models from the International Ground Water Modeling Center

A code might meet all of the above technical criteria and still not be suitable for use due to deficiencies in the code itself. An ongoing program at the International Ground Water Modeling Center evaluates codes using performance standards and acceptance criteria (van der Heijde, 1987). The Center has rated 296 codes in seven major categories using a variety of usability and reliability criteria (van der Heijde and others, 1988). Favorable ratings for the usability criteria include:

Pre- and Postprocessors. Code incorporates one or more of this type of code.

Documentation. Code has an adequate description of user's instructions and example data sets.

Support. Code is supported and maintained by the developers or marketers.

Hardware Dependency. Code is designed to function on a variety of hardware configurations.

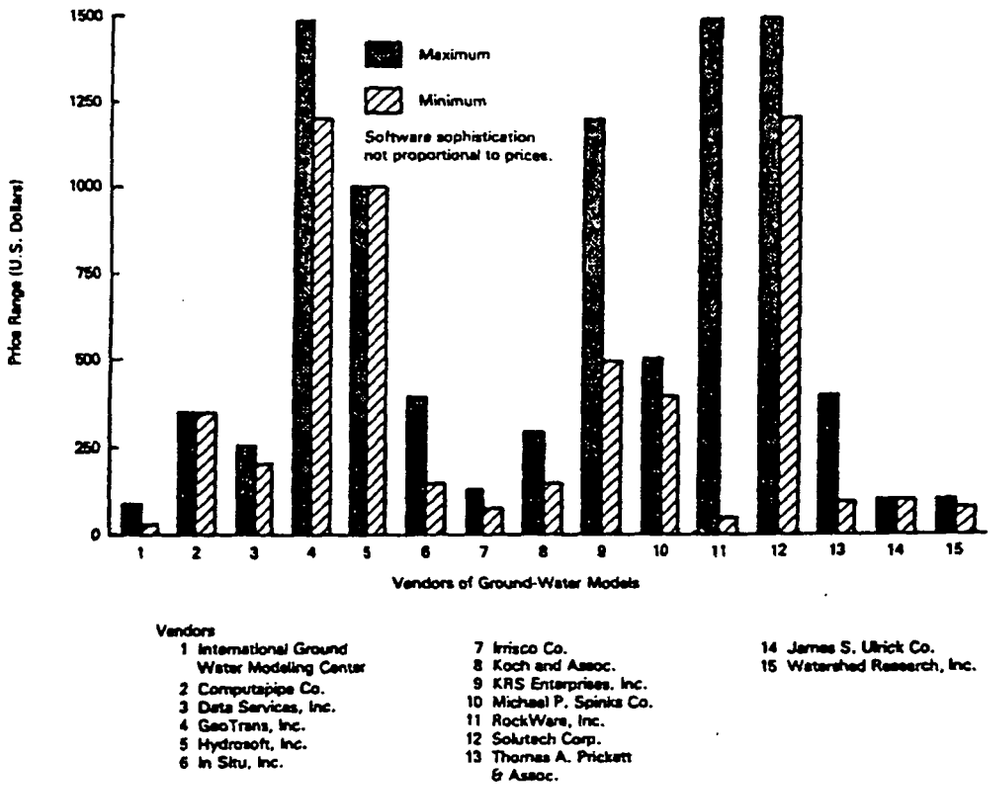

Figure 6-7. Price Ranges for IBM-PC Ground-Water Models Available from Various Sources (from Graves, 1986)

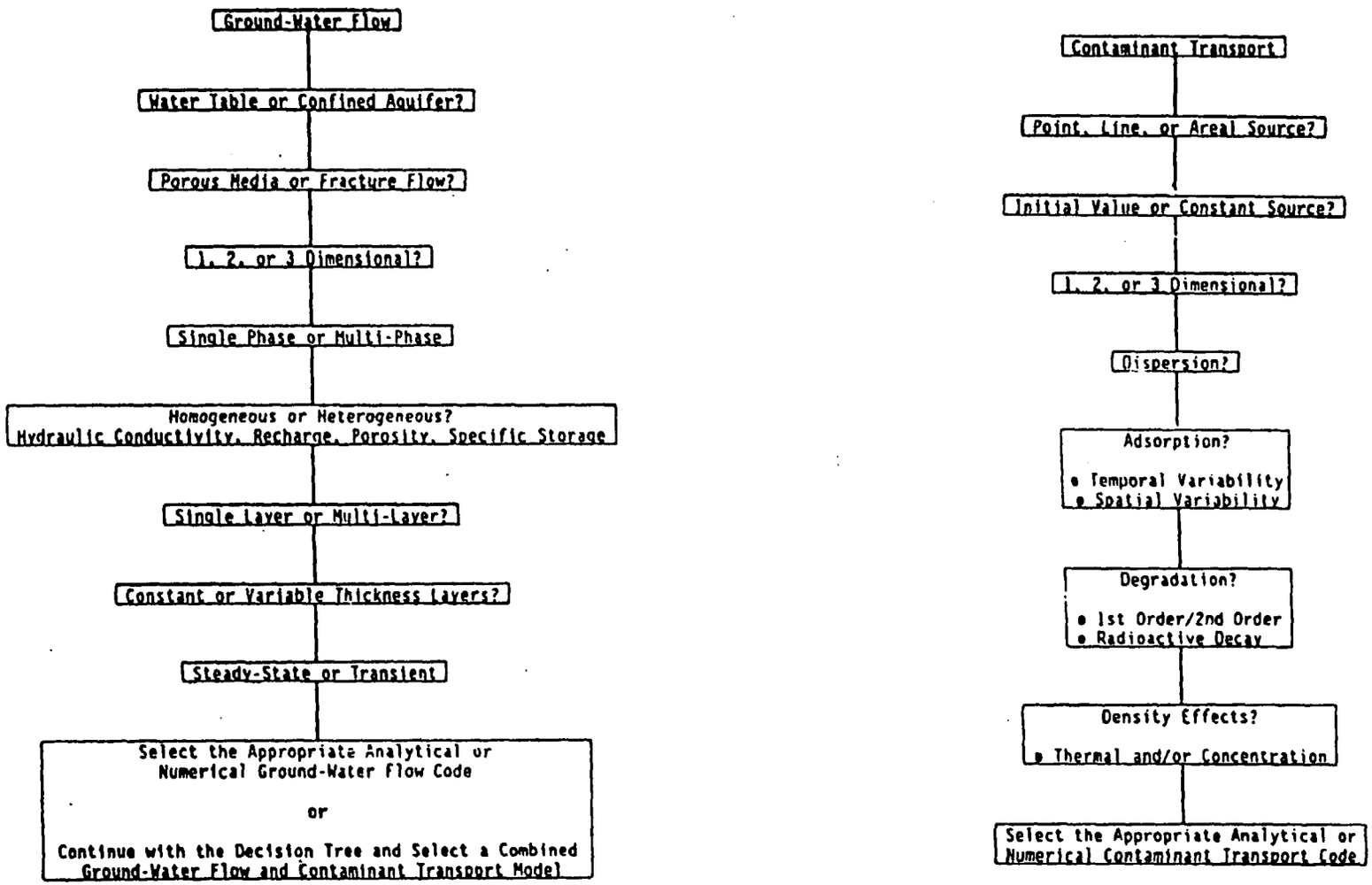

Figure 6-8. Ground-Water Computer Code Selection Decision Tree (from U.S. EPA, 1988)

Model Names	Water Table Aquifer	Confined Aquifer	Porous Media	Fracture Flow	Dimension Flow	Single-Phase	Multi-Phase	Homogeneous Hydraulic Parameters	Heterogeneous Hydraulic Parameters	Single-Layer	Multi-Layer	Constant Thickness	Variable Thickness	Steady State	Transient	Point, Line, or Areal Source	Initial Value Source	Constant Source	Dimension Transport	Dispersion	Absorption	Degradation 0,1,2, - order radio active	Density Effects	Solution Technique
Analytical Flow																								
PATHS		X	X		2	X		X		X		X		X	X	A	X	X	2		X			AN
Analytical Transport																								
AT123D			X		1,2,3	X		X		X		X		X		A	X	X	1,2,3	X		1		AN
CHAIN			X		1	X		X		X		X		X		P	X	X	1	X	X	1,R		AN
GETOUT			X		1	X		X		X		X		X		P		X	1	X		1,R		AN
GWMTM1&2			X		1,2	X		X		X		X		X		P,L		X	1,2	X		1		AN
MUTRAN			X		1,2,3	X			X	X		X		X		P,LA		X	1,2,3	X	X	R		AN
NWFT/DVM			X		1	X			X	X		X		X				X	1	X	X	R	SALT	AN
Numerical Flow (Saturated/Unsaturated)																								
UNSAT1			X		1	X			X		X		X		X									FE
FEMWATER1			X		2	X		X	X	X	X	X	X	X	X									FE
UNSAT 2			X		2	X			X		X		X		X									FE
FREEZE	X	X	X		3	X			X		X		X		X									FD
Numerical Flow (Saturated Only)																								
BEWTA	X		X		2	X		X		X		X			X									FD
COOLEY	X	X	X		2	X			X	X		X		X	X									FE
FEBDQW	X	X	X		3	X			X		X		X	X	X									FE
FLUMP	X		X		2	X			X	X		X		X	X									FE
FRESURF 1 &2	X		X		2	X			X	X	R-2	X	R-2		X									FE
											X		X											
TEM/AGI	X		X		1	X			X		X		X		X									FD
USGS2D	X	X	X		2	X			X	X		X			X									FD
VTT	X	X	X		2	X			X		X		X	X	X									FD
V3	X	X	X		2	X			X		X		X		X									FD
USGS3D - MODULAR	X	X	X		3	X			X		X		X	X	X									FD
USGS3D - TRESCOTT	X		X		3	X			X		X		X	X	X									FD
Numerical Transport (Saturated/Unsaturated)																								
FEMWASTE 1	X		X		2	X			X	X	X	X	X	X	X	A	X	X	2	X	X	1		FE
PERCOL	X		X		1	X		X			X		X	X	X	P	X		1	X	X	R		NR
SATURN	X		X		2	X			X		X		X	X	X	A	X	X	2	X	X	1,R		FE
SEGOL	X		X		3	X			X		X		X	X	X	A	X	X	3	X	X			FE
SUMATRA - 1	X		X		1	X			X		X		X	X	X	P	X	X	1	X	X	0,1	X	FE
SUTRA	X		X		2	X			X		X		X	X	X	A	X	X	2	X	X	1	X	FE, IFD
TRANUSAT	X		X		1,2	X			X		X		X	X	X				1,2	X	X	1		FE
TRUST	X		X		1,2,3	X			X		X		X	X	X	P,LA	X	X	1,2,3	X	X			FD
Numerical Transport (Saturated Only)																								
CHAINT			X	X		X		X		X		X			X	A	X	X	2	X	X	R,CH		FE
DUGUID-REEVES			X			X			X	X	X		X	X	X	A	X	X	2	X	X	R		FE
GROVE/GALERKIN			X		3	X			X		X		X	X	X	A	X	X	3	X	X	1		FE
ISOQUAD,ISOQUAD2			X			X			X	X		X		X	X	A		X	2	X				FE
COMBRED					1,2				X	X			X	X	X	A		X	1,2	X	X	1		FD, NR
(USGS2C - MOC)																								
DPCT	X		X		2	X			X		X		X	X	X	A	X	X	2	X	X	R		FE, RW
MMT			X			X		X		X		X		X	X	P	X	X	1	X		I,R, CH		RW
PINDER		X	X			X			X		X		X	X	X	A		X	3	X	X			FE
ROBERTSON1	X		X			X			X		X		X	X	X	P			2	X		R		FD
ROBERTSON2		X	X		2	X			X	X		X		X	X	A		X	2	X	X	R		FD, MOC
SWENT		X	X		1,2,3	X			X		X		X	X	X	A			1,2,3	X	X	1,R, CH	X	FD
TRANS (Prickett-Lonnquist)	X	X	X		2	X		X		X		X		X	X	A			2	X		1		RW
TRANSAT 2		X	X		3	X			X		X		X	X	X				3	X	X	1		FE
Numerical Coupled Solute and Heat Transport																								
CFEST		X	X		1,2,3	X			X		X		X		X	A	X	X	1,2,3	X	X	1	X	FE
GWTHERM			X		2	X			X		X		X		X	A	X	X	2	X	X		X	FD
OCRE			X		1,2	X		X			X	X			X	A		X	1,2	X	X	R		FE
SHALT			X		2	X			X		X		X		X	A		X	2	X	X	1	X	FE
SWFT	X	X	X		3	X			X		X		X	X	X	A		X	3	X	X	R,CH	X	FD
SWIP2	X	X	X		3	X			X		X	X			X	A	X		3	X	X	1	X	FD

Source: U.S. EPA, 1988

LEGEND

AN - Analytical

Degradation
- 0 - Zero Order Decay
- 1 - First Order Decay
- 2 - Second Order Decay
- R - Radioactive Decay
- CH - Radioactive chain Decay

Dimensionality
- 1 - One Dimensional
- 2 - Two Dimensional
- 3 - Three Dimensional

FD - Finite Difference
FE - Finite Element
IFD - Integrated Finite Difference
MOC - Method of Characteristics
NR - Newton Raphson

Source Type
- A - Areal Source
- L - Line Source
- P - Point Source

Table 6-6. Analytical and Numerical Models Worksheet

Favorable ratings for the reliability criteria include:

Review. Both theory behind the coding and the coding itself are peer reviewed.

Verification. Code has been verified.

Field Testing. Code has been extensively field tested for site-specific conditions for which extensive datasets are available.

Extent of Use. Code has been used extensively by other modelers.

Quality Assurance/Quality Control

The increasing use of modeling and computer codes in regulatory settings where decisions may be contested in court requires careful attention to quality assurance and quality control in both model development and application. The American Society for Testing and Materials (ATSM) defines several important terms that relate to QA/QC procedures for computer code modeling (ASTM, 1984):

Verification involves examination of the numerical technique in the computer code to ascertain that it truly represents the conceptual model and that there are no inherent numerical problems associated with obtaining a solution.

Validation involves comparison of model results with numerical data independently derived from experiments or observations of the environment.

Calibration is a test of a model with known input and output information that is used to adjust or estimate factors for which data are not available.
Sensitivity is the degree to which the model result is affected by changes in a selected input parameter.

Huyakorn and others (1984) identified three major levels of quality control in the development of groundwater models:

1. Verification of the model's mathematics by comparison of its output with known analytical solutions to specific problems.

2. Validation of the general framework of the model by successful simulation of observed field data.

3. Benchmarking of the model's efficiency in solving problems by comparison with other models.

These levels of quality control address the soundness and utility of the model alone, but do not treat questions of its application to a specific problem. Hence, at least two additional levels of quality control appear justified:

1. Critical review of the problem's conceptualization to ensure that the modeling effort considers all physical and chemical aspects that may affect the problem.

2. Evaluation of the specifics of the application, e.g., appropriateness of the boundary conditions, grid design, time steps, etc. Calibration and sensitivity analysis to determine if the model outputs vary greatly with changes in input parameters are important aspects of this process.

Verification of the mathematical framework of a numerical model and of a code for internal consistency is relatively straightforward. Field validation of a numerical model consists of first calibrating the model using one set of historical records (e.g., pumping rates and water levels from a certain year), and then attempting to predict the next set of historical records. In the calibration phase, the aquifer coefficients and other model parameters are adjusted to achieve the best match between model outputs and known data; in the predictive phase, no adjustments are made (excepting actual changes in pumping rates, etc.). Presuming that the aquifer coefficients and other parameters were known with sufficient accuracy, a mismatch means that either the model is not correctly formulated or that it does not treat all of the important phenomena affecting the situation being simulated (e.g., does not allow for leakage between two aquifers when this is actually occurring).

Field validation exercises usually lead to additional data gathering efforts, because existing data for the calibration procedure commonly are insufficient to provide unique estimates of key parameters. Such efforts may produce a "black box" solution that is so site-specific that the model cannot be readily applied to another site. For this reason, the blind prediction phase is an essential check on the uniqueness of the parameter values used. Field verification is easiest if the model can be calibrated to data sets from controlled research experiments.

Benchmarking routines to compare the efficiency of different models in solving the same problem have only recently become available (Ross and others, 1982; Huyakorn and others, 1984). Van der Heijde and others (1988) discuss, in some detail, procedures for developing QA plans for code development/maintenance and code application.

Limitations of Computer Codes

Mathematical models are useful only within the context of the assumptions and simplifications on which they

are based and according to their ability to approximate the field conditions being simulated. Faust and others (1981) rated the predictive capabilities of available models with respect to 10 issues involving quantity and quality of ground water (Table 6-7). A four-tiered classification scheme for models is shown in Table 6-7: (1) geographic scope (site, local, regional); (2) pollutant movement (flow only, transport without reactions, and transport with reactions); (3) type of flow (saturated or unsaturated); and (4) type of media (porous or fractured). The rating scale by Faust and others (1981) in Table 6-7 also can be viewed as stages of model development:

0 = No model exists.

1 = Models are still in the research stage.

2 = Models can serve as useful conceptual tools for synthesizing complicated hydrologic and quality data.

3 = Models can make short-term predictions (a few years) with a moderate level of credibility, given sufficient data.

4 = Models can make predictions with a high degree of reliability and credibility, given sufficient data.

The most advanced model is only able to simulate available supplies and conjunctive use at the local level. Contaminant transport modeling is generally at stage 3 for transport without reactions in saturated porous flow at the site and local level. Models at the stage 2 level of development generally include transport without reactions (saturated fractured, unsaturated porous), and transport with reactions (saturated porous) at the site and local level. Models at the earliest stage of development involve transport with reactions in saturated, fractured media.

Advances have been made in all areas of modeling since the ratings in Table 6-7 were made, but the basic relationships are essentially unchanged. This is illustrated in Table 6-8, which shows the percentage of computer codes in seven categories that received favorable usability and reliability ratings by van der Heijde and others (1988). The heat transport and geochemical model categories do not have direct

	Model Types																			
Spatial considerations:	*Site*										*Local*						*Regional*			
Pollutant movement, if any:	*Flow only*				*Transport w/o reactions*			*Transport w/ reactions*			*Flow only*		*Transport w/o reactions*		*Transport w/ reactions*		*Flow only*		*Transport w/o reactions*	
Flow conditions:	*sat P*	*sat F*	*unsat P*	*multi fluid*	*sat P*	*sat F*	*unsat P*	*sat P*	*sat F*	*unsat P*	*sat P*	*sat F*	*sat P*	*sat F*	*sat P*	*sat F*	*sat P*	*sat F*	*sat P*	*sat F*
Issues																				
Quantity Available supplies	3	2									4	3					3	3		
Quantity Conjunctive use	3	1									4	3					3	3		
Quality Accidental Petroleum products				1	3	2	1						2	1						
Quality Accidental Road salt					3	2	2													
Quality Accidental Industrial chemicals					3	2	2	2	1	0			3	2	2	0				
Quality Agriculture Pesticides & herbicides					3	2	2	2	1	0			3	2	2	0				
Quality Agriculture Salt buildup					3	2	2						3	2						
Quality Waste disposal Landfills					3	2	2	2	1	0			3	2	2	0				
Quality Waste disposal Injection					3	2	2	2	1	0			3	2	2	0				
Quality Sea-water intrusion				3	3	2	2						3	2					2	2

Table 6-7. Matrix Summarizing Reliability and Credibility of Models Used in Ground-Water Resource Evaluation

Key to Matrix	
Rows	issue and subissue areas.
Columns	model types and scale of applications; for example, sixth column applies to a site-scale problem in which pollutant movement is described by a transport model without reactions and with saturated flow in fractured media.
Application scale	
Site	area modeled less than a few square miles.
Local	area modeled greater than a few square miles but less than a few thousand square miles.
Regional	area modeled greater than a few thousand square miles.
Abbreviations	
w/	with.
w/o	without.
sat	saturated ground-water flow conditions.
unsat	unsaturated flow conditions.
P	porous media.
F	fractured, fissured, or solution cavity media.
Entries	
4	a useable predictive tool having a high degree of reliability and credibility given sufficient data.
3	a reliable conceptual tool capable of short-term (a few years) prediction with a moderate level of credibility given sufficient data.
2	a useful conceptual tool for helping the hydrologist synthesize complicated hydrologic and quality data.
1	a model that is still in the research stage.
0	no model exists.
blank	model type not applicable to issue area.

Table 6-7. Continued

counterparts in Table 6-7. The multiphase flow category is closest to the accidental petroleum products quality category in Table 6-8.

Not surprisingly, the largest number of codes are in the saturated flow category (97), followed by the saturated solute-transport category (73). The more limited availability of models for unsaturated flow, fractured rock, multiphase flow, and geochemistry primarily reflects the difficulties in mathematical formulation due to complexity of processes, process interactions, and field heterogeneities.

Table 6-8 also provides an overview of the status of ground-water modeling from a quality assurance perspective. In general, a high percentage of codes have been peer reviewed in terms of the basic theory. The exceptions are fractured-rock (44%) and multiphase flow models (21%). In contrast, relatively few models have been reviewed in terms of actual coding. Only the geochemical model category has more than half its models (60%) meeting this criterion. As was noted earlier, model verification is a relatively straightforward procedure, which is demonstrated in Table 6-8 where high percentages of all categories have been verified. In contrast, very few codes have had any significant amount of field testing. Less than a third of the codes in the saturated flow category have been extensively field tested, and field testing of codes in the other categories ranges from none for fractured rock and geochemical to 21% for variable saturated flow. The percentages in Table 6-8 should be viewed with the following caveats: (1) many codes received an "unknown" rating, which means that the percentages may underestimate the number of codes with actual favorable ratings; and (2) many of the codes have been subjected to limited field testing.

A number of possible pitfalls will doom a ground-water modeling effort to failure (OTA, 1982; van der Heijde and others, 1985):

1. Inadequate conceptualization of the physical system, such as flow in fractured bedrock
2. Use of insufficient or incorrect data
3. Incorrect use of available data
4. Use of invalid boundary conditions
5. Selection of an inadequate computer code
6. Incorrect interpretation of the computational results
7. Imprecise or wrongly posed management problems

Type of code	Total	Support	Theory Rev.	Code Rev.	Verifi-cation	Field Tested
Saturated flow	97	65%	74%	12%	90%	32%
Solute transport	73	67%	68%	29%	96%	14%
Heat transport	36	78%	78%	42%	97%	6%
Variable saturated flow	29	48%	72%	21%	83%	21%
Fractured rock models	27	7%	44%	33%	100%	0%
Multiphase flow	19	5%	21%	11%	89%	11%
Geochemical	15	33%	60%	60%	100%	0%

Source: Adapted from van der Heijde and others (1988).

Table 6-8. Percentage of Computer Codes with Favorable Usability and Reliability Ratings

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*U.S. GOVERNMENT PRINTING OFFICE: 1994-550-001/80339