

Part I:

Introduction



*Chattooga River, Nantahala National Forest, North Carolina.
Photo by Bill Lea*

Chapter 1

Goals of this Report

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The Importance of Safe Public Drinking Water

The U.S. Congress justified passing the Safe Drinking Water Act Amendments of 1996 (SDWA) (Public Law 104–182) codified at 42 U.S.C. sec. 300j–14, by stating “safe drinking water is essential to the protection of public health.” For over 50 years, a basic axiom of public health protection has been that safe drinking water reduces infectious disease and extends life expectancy (American Water Works Association 1953). Although most U.S. residents take safe public drinking water for granted, assuring its safety remains a high national priority. Large investments are made by all levels of government to maintain and upgrade our public water systems.

To strengthen that process, the SDWA mandates that greater protection and information be provided for the 240 million Americans who are served by public water supplies. Section 1453 of the SDWA requires all States to complete source water assessments (SWA’s) of their public drinking water supplies by 2003. To meet this requirement, each State and participating tribe will delineate the boundaries of areas that serve as sources for individual public drinking water systems, identify significant potential sources of contamination, and determine how susceptible each system is to contamination. Source water assessments are required for all public drinking water supplies regardless of the ownership of the drinking water system or the land that comprises its source area. Results of SWA’s will be made public and will assist local planners, tribes, and Federal and State Governments to make more informed decisions to protect drinking water sources.

To get information about a source water assessment program (SWAP) from a particular State, go to the U.S. Environmental Protection Agency (EPA) homepage to view the SWAP contact list. This site includes names and telephone numbers of State source water contacts and hotlinks to existing State homepages for more information. The EPA homepage can be found at <http://epa.gov/OGWDW/protect.html>.

U.S. Congress chose source water protection as a strategy for ensuring safe drinking water because of its high potential to be cost-effective. A poor source of water can substantially increase the cost of treatment to make the water drinkable. When source water is so contaminated that treatment is not feasible, developing alternative water supplies can be expensive and cause delays in providing safe, affordable water. Delineating areas that supply water and inventorying potential sources of contamination will help communities know the threats to their drinking water. Communities can then more effectively and efficiently address these threats.

Drinking Water from Forests and Grasslands

Forests and grasslands have long been relied upon as sources of clean drinking water for two reasons: (1) forests mainly grow under conditions that produce relatively reliable water runoff, and (2) properly managed forests and grasslands can yield water relatively low in contaminants when compared with many urban and agricultural land uses. We estimate that at least 3,400 towns and cities currently depend on National Forest System watersheds for their public water supplies. In addition, the national forests and grasslands have over 3,000 public water supplies for campgrounds, administrative centers, and similar facilities. Communities that draw source water from national forests and grasslands provide a public water supply to 60 million people, or one-fourth of the people served by public water supplies nationwide. Since 70 percent of the forest area in the United States is outside of the National Forest System, the number of people served by all forests and grasslands is far greater.

With the large number of public water supplies on forests and grasslands, there is a high likelihood that many forest and grassland managers will be involved in the process of planning, implementing, or reacting to public concerns related to SWA’s. The level of involvement in this process will probably vary from place to place depending on the requirements of each State, the degree of public attention that particular management activities receives, and the potential of specific land uses to affect source waters. At the time of writing this document, it is difficult to predict to

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what degree particular managers may become involved with this process. We have assembled current scientific knowledge in a useful form that will help managers protect the safety of drinking water sources and be better-informed participants in SWA's.

The Purpose and Scope of this Document

This document was written to assist forest and grassland managers in their efforts to comply with the SDWA by providing them with a review and synthesis of the current scientific literature about the effects of managing these lands on public drinking water sources. This is not a decision document. Its audience includes managers of national forests and grasslands as well as managers of public and private forests and grasslands. Managers of public water supplies and community groups concerned with drinking water may also find this document useful.

This report's focus is restricted to potential contamination of source water associated with ordinary land uses in national forests and grasslands. It does not treat the delineation of source areas because the EPA and the States will decide those criteria. We chose conventional land uses on national forests and grasslands because they clearly come under the mandate of the U.S. Department of Agriculture, Forest Service (Forest Service), the principal sponsor of this document, and because a significant portion of the public depends on national forests and grasslands for water. We did include grazing and land uses that occur where urban areas border on or intermix with forests and grasslands. The report does not address large urban developments, large industrial complexes, row crop agriculture, or concentrated animal feeding operations because they come more appropriately under the oversight of other agencies. We focus on issues for public water supplies, rather than those of small, private water sources for individual families, because only public supplies are examined in SWA's.

The processes reviewed in this report occur at spatial scales ranging from a few square yards (meters) to many millions of acres (hectares). Most scientific studies, however, have been done at relatively small scales. Inferences about larger areas are drawn mostly from models or extrapolations based on those small-scale studies. Where regional differences in effects of land management were reported in the literature, the authors indicated them in this document. If not, we did not make regional distinctions. Several conventions are used by the scientific and land management communities for classifying geographic, climatic, and ecological zones with similar characteristics into ecoregions, but no standard system of classification has been endorsed across relevant

scientific disciplines or Federal Agencies. For this reason, we cited whatever ecoregions were used in the literature.

How to Use this Document

This document is intended to be used by managers as a reference for assessing watersheds and planning programs to minimize the effects of land management practices on the quality of drinking water sources. When managers are concerned with the potential of a particular land management practice, they can consult the chapter summarizing what is known about the effects of that practice. Managers should note both what is known and what is not known from scientific studies. Known information may provide a means to estimate the effects of a particular practice. What is unknown is equally important because it may indicate which management actions entail risk because their effects are not well understood.

We wish to emphasize the importance of using scientific information as a basis for management. Managers often are forced by circumstances to make decisions based on incomplete knowledge. They compensate by filling information gaps with reasonable assumptions. Each such assumption carries the risk of unintended consequences. Use of scientific data in decision-making has the advantage that many of the important conditions that affect outcomes have been controlled or measured, and critical assumptions are often carefully spelled out. When decisions are based on anecdotal experience, less may be known about conditions that affect outcomes, and key assumptions about these conditions may not be explicit. Decisions that draw on scientific information, therefore, reduce the risk of unexpected outcomes.

The subjects covered are broadly and briefly summarized. When managers need to go more deeply into a topic, they should use the scientific literature that is cited in each chapter as an entry point into the larger body of knowledge that underlies each of the chapters. Wherever possible, the scientific information that is cited has been peer reviewed and published. Case studies presented are meant to illustrate the complexity of actual management situations and are not necessarily based on peer-reviewed literature.

To synthesize the scientific information into a form that answers questions relevant to managers required that the authors use their best professional judgement both to draw together diverse sources and to evaluate their validity. Exercising this judgement is necessary to make this document more useful than a mere compilation of data or annotated bibliography. We have made every effort to make

apparent the distinction between published scientific observations and logical synthesis on the part of the authors.

This document has undergone a rigorous peer review by professional scientists and managers from inside and outside government to critique the validity and currency of its sources, syntheses, and conclusions. The finished document has been revised to consider and respond to the comments of these reviewers.

Although this document is separated into chapters by types of land use, we recognize that in most practical situations effects on source waters result from the cumulative effects of multiple land uses that often overlap in space and change over time. To address this issue we direct readers to chapter 2, which covers the natural processes of watersheds that overlay all land uses, and to chapter 3, which summarizes the cumulative effects of multiple land uses distributed over space and time.

In this document we concentrate on issues that arise from the need of managers to comply with the SDWA. This is only one of the many policies and laws that currently govern the actions of national forest and grassland managers. A provision of the Organic Act of 1897 (30 Stat. 11), codified at 16 U.S.C. Subsec. 473–475, 477–482, 551, that established the national forests “for the purpose of securing favorable conditions of water flows,” has been interpreted to authorize managing this land for water resources. Administration of national forests is currently guided primarily by four laws: (1) the Multiple Use-Sustained Yield Act (Public Law 86–517), codified at 16 U.S.C. sec.525 *et seq.*; (2) the National Environmental Policy Act (Public Law 91–190), codified at 16 U.S.C. sec.4321 *et seq.*; (3) the Forest and Rangeland Renewable Resources Planning Act (Public Law 93–378), codified at 16 U.S.C. sec.1600 *et seq.*; and (4) the National Forest Management Act (Public Law 94–588). Forest and grassland managers also must comply with many environmental statutes including the Endangered Species Act (Public Law 93–205), codified at 16 U.S.C. sec.1531 *et seq.*; the Clean Water Act (Public Law 80–845), codified at 33 U.S.C. Sec.1251; and the Clean Air Act (Public Law 84–159), codified at 42 U.S.C. sec.7401 *et seq.* Activities of the Forest Service with State and private landowners were authorized by the Cooperative Forestry Assistance Act (Public Law 95–313) and amended in the 1990 Farm Bill (Public Law 101–624), codified at 16 U.S.C. Subsec. 582a, 582a–8, 1648, 1642 (note), 1647a, 2101 (note), 2106a, 2112 (note), 6601 (note). The Forest and Rangeland Renewable Resources Act (Public Law 93–378), with amendments in the 1990 Farm Bill (Public Law 101–624), provided authority for research by the Forest Service. For a more complete listing of relevant laws and the text of these laws, see U.S.

Department of Agriculture, Forest Service (1993). Over time, the laws and policies that guide public land use have evolved in response to changes in perceived public needs and will probably continue to change in the future.

A number of laws that affect forest and grassland management require the use of best management practices (BMP’s). These practices vary widely in their application and effectiveness from State to State and continually evolve in response to new environmental concerns, technology, and scientific evidence (Dissmeyer 1994). This document does not cite or endorse specific BMP’s but rather presents scientific evidence that has the potential to serve as a basis for developing practices that more effectively protect source water.

Some laws and prudent practice require that environmental monitoring be used to assess the outcomes of land management. We considered the broad topic of monitoring to be beyond the scope of our effort, but implicit throughout this document is the assumption that monitoring of outcomes should be an integral part of land management. Scientific evidence does not eliminate all risks of unforeseen outcomes, and where scientific studies are lacking, risks are likely to be higher. Monitoring land-use practices will help to protect public health and other important values.

This document focuses narrowly on protecting human health by protecting drinking water. We acknowledge that managers must consider a much wider range of values in most land-use decisions. It is not our intent to tell managers how to weigh a spectrum of values or how to decide among them. Rather we wish to inform managers about specific effects on drinking water so that they can better take these effects into consideration when they make land-use decisions.

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Chapter 2

Drinking Water Quality

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Introduction

Watersheds are topographically defined areas drained by connecting stream channels that discharge water, sediment, and dissolved materials through a common outlet. The term is synonymous with drainage basin and catchment and can refer to a large river basin or the area drained by a single ephemeral stream. Watersheds are commonly classified by physiography (headwater, steeplands, lowland, etc.), environmental condition (pristine, degraded, etc.), or their principal use or land cover (forest, urban, agricultural, municipal water supply, etc.).

Municipal watersheds are managed to provide a sustainable supply of high-quality, safe drinking water at minimum environmental and economic costs. Many activities within a watershed can contaminate water (table 2.1), and most supplies are not suitable for human consumption without some form of treatment. This chapter provides an overview of the chemical and physical processes that affect the chemistry and quality of water as it travels across the landscape. The appendix presents information on treatment techniques (appendix tables E.1–E.4) that are used for controlling common contaminants (National Research Council 1997).

Water quality is a relative concept that reflects measurable physical, chemical, and biological characteristics in relation to a specific use. The suitability of water for domestic use is typically defined by taste, odor, color, and the abundance of organic and inorganic substances that pose risks to human health (table 2.2). In the United States, suitability is formally defined in legally enforceable primary standards (table 2.3) and in recommended or secondary guidelines (table 2.4). The States will focus on the contaminants listed in tables 2.3 and 2.4 in their source water assessments.

Standards for drinking water apply to water that is delivered to consumers after it has been treated to remove contaminants, but not to source water as it is withdrawn from surface or ground water. Ambient standards set under the Clean Water Act (Public Law 80–845) for streams or lakes are not intended to ensure that water is drinkable without

treatment. Considerable treatment may be required to purify water meeting the ambient standard to comply with the drinking water standard. As effects on human health from exposure to contaminants in drinking water become better understood and as new substances are released to the environment, changes in drinking water standards can be expected in the future.

Chemical Properties

Water is formed by the covalent union of two hydrogen (H) atoms and one oxygen (O) atom. These atoms are joined in an unsymmetrical arrangement where the hydrogen end of the molecule has a slight positive charge and the oxygen end a slight negative charge. This arrangement of unbalanced electrical charges creates the dipolar characteristic that gives the molecule the remarkable ability to act as both an acid and a base and be a solvent for cations, anions, and some types of organic matter. This arrangement also allows water molecules to form hydrogen bonds with adjacent water molecules. These bonds are responsible for water's high viscosity, high cohesion and adhesion, high surface tension, high melting and boiling points, and the large temperature range through which it is a liquid.

As water travels across the landscape, it interacts with its environment through a variety of chemical processes (table 2.5). In the process, it picks up and transports dissolved gases, cations and anions, amorphous organics, trace metals, and particulates. The most common positively charged ions, or cations, include calcium (Ca^{+2}), magnesium (Mg^{+2}), sodium (Na^{+1}), potassium (K^{+1}), and ammonium (NH_4^{+1}). The most common anions, or negatively charged ions, include nitrate (NO_3^{-1}), sulfate (SO_4^{-2}), chloride (Cl^{-1}), and several different forms of phosphorus (P). Most amorphous substances are organic carbon-based compounds that readily adsorb and exchange cations. Common particulates include mineral particles, i.e., inorganic sediment, organic debris, and microscopic organisms (plankton, diatoms, etc.). Both the chemical behavior (table 2.6) and the origin of contamination (table 2.1) vary with the type of chemical contaminants.

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Table 2.1—Summary of common water pollutants by land-use activities

Land use and type of activity	Spatial distribution	Major types of pollution	Pollution indicators
Forests			
Harvesting	Diffuse	N, O	Sediment
Camping, hunting	Diffuse	FC, O, S	FC, garbage
Skiing	Diffuse, line	N, I, S	Salts, sediment
Rangeland			
Grazing	Diffuse	FC, N, O	NO ₃ ⁻¹ , sediment
Urbanization			
Unsewered sanitation	Point, diffuse	N, FC, O, S	NO ₃ ⁻¹ , NH ₄ ⁺¹ , FC, DOC, Cl ⁻
Leaking sewers	Point, line	N, FC, O, S	FC, NH ₄ ⁺¹ , NO ₃ ⁻¹
Leaking fuel tanks	Point	O	HC, DOC
Storm drainage	Line, diffuse	I, H, O, S	Cl ⁻ , sediment
Industrial			
Leaking tanks	Point	O, S, H	Variable, HC
Spills	Point, diffuse	O, S, H	Variable
Aerial fallout	Diffuse	S, I, N, O	SO ₄ ⁻² , NO ₃ ⁻¹ , HC
Agriculture			
Cropland	Diffuse	N, O, S, P	NO ₃ ⁻¹ , sediment
Livestock	Point, diffuse	FC, N, O	NO ₃ ⁻¹ , sediment
Mineral extraction	Point, diffuse	H, I	Variable, sediment

DOC = dissolved organic carbon; FC = fecal coliform; H = heavy metals; HC = hydrocarbons; I = inorganic salts; N = nutrient; NH₄⁺¹ = ammonium; NO₃⁻¹ = nitrate; O = organic load; P = phosphorous; S = synthetic organic compounds; SO₄⁻² = sulfate.
 Source: Updated from Foster and Gomes 1989.

Table 2.2—Common types of water contaminant guidelines for different water uses^a

Contaminant	Human				
	consumption	Irrigation	Livestock	Fisheries	Recreation
Coliform bacteria	*				*
Nematode eggs		*			
Particulate matter	*			*	
Dissolved oxygen (BOD, COD)				*	*
Nitrates	*	*	*		
Nitrites	*		*	*	
Salinity	*	*	*	*	
Inorganic pollutants (trace metals)	*	*	*	*	*
Organic pollutants	*			*	*
Pesticides	*			*	

BOD = biological oxygen demand; COD = chemical oxygen demand.

^aAn * indicates that guidelines typically exist for a particular use. The absence of an * indicates that no guidelines exist for a particular use.

Source: Adapted from GEMS 1991.

Table 2.3—National primary drinking water regulations^a (States are expected to focus attention on risks related to the contaminants listed in their source water assessments.)

Contaminants	MCLG	MCL or TT	Potential health effects from ingestion of water	Sources of contaminant in drinking water
--- Milligrams per liter ---				
Inorganic chemicals				
Antimony	0.006	0.006	Increase in blood cholesterol, decrease in blood glucose	Discharge from petroleum refineries, fire retardants, ceramics, electronics, solder
Arsenic	None ^b	.05	Skin damage, circulatory system problems, increased risk of cancer	Discharge from semi-conductor manufacturing, petroleum refining, wood preservatives, animal feed additives, herbicides, erosion of natural deposits
Asbestos (fiber > 10 µm)	7 million fibers/L	7	Increased risk of developing benign intestinal polyps	Decay of asbestos cement in water mains, erosion of natural deposits
Barium	2	2	Increase in blood pressure	Discharge of drilling wastes, discharge from metal refineries, erosion of natural deposits
Beryllium	.004	.004	Intestinal lesions	Discharge from metal refineries and coal-burning factories; discharge from electrical, aerospace, and defense industries
Cadmium	.005	.005	Kidney damage	Corrosion of galvanized pipes, erosion of natural deposits, discharge from metal refineries, runoff from waste batteries and paints
Chromium (total)	.1	.1	Some people who use water containing chromium well in excess of the MCL over many years could experience allergic dermatitis.	Discharge from steel and pulp mills, erosion of natural deposits
Copper	1.3	Action level ^c = 1.3, TT	Short-term exposure— gastrointestinal distress, long-term exposure— liver or kidney damage	Corrosion of household plumbing systems, erosion of natural deposits, leaching from wood preservatives
Cyanide (as free cyanide)	.2	.2	Nerve damage or thyroid problems	Discharge from steel and metal factories, discharge from plastic and fertilizer factories
Fluoride	4.0	4.0	Bone disease (pain and tenderness of the bones); children may get mottled teeth.	Water additive which promotes strong teeth, erosion of natural deposits, discharge from fertilizer and aluminum factories
Lead	Zero ^d	Action level ^c = 0.015, TT	Infants and children— delays in physical or mental development; adults—kidney problems, high blood pressure	Corrosion of household plumbing systems, erosion of natural deposits
Inorganic mercury	.002	.002	Kidney damage	Erosion of natural deposits, discharge from refineries and factories, runoff from landfills and cropland
Nitrate (measured as nitrogen)	10	10	Blue-baby syndrome in infants under 6 mo—life threatening without immediate medical attention	Runoff from fertilizer use; leaching from septic tanks, sewage; erosion of natural deposits

continued

Table 2.3—National primary drinking water regulations^a (States are expected to focus attention on risks related to the contaminants listed in their source water assessments.) (continued)

Contaminants	MCLG	MCL or TT	Potential health effects from ingestion of water	Sources of contaminant in drinking water
<i>--- Milligrams per liter ---</i>				
Inorganic chemicals				
(cont.)				
Nitrite (measured as nitrogen)	1	1	Blue-baby syndrome in infants under 6 mo—life threatening without immediate medical attention	Runoff from fertilizer use; leaching from septic tanks, sewage; erosion of natural deposits
Selenium	0.05	0.05	Hair or fingernail loss, numbness in fingers or toes, circulatory problems	Discharge from petroleum refineries, erosion of natural deposits, discharge from mines
Thallium	.0005	.002	Hair loss; changes in blood; kidney, intestine, or liver problems	Leaching from ore-processing sites; discharge from electronics, glass, and pharmaceutical companies
Organic chemicals				
Acrylamide	Zero ^d	TT	Nervous system or blood problems, increased risk of cancer	Added to water during sewage and wastewater treatment
Alachlor	Zero ^d	.002	Eye, liver, kidney, or spleen problems; anemia; increased risk of cancer	Runoff from herbicide used on row crops
Atrazine	.003	.003	Cardiovascular system problems, reproductive difficulties	Runoff from herbicide used on row crops
Benzene	Zero ^d	.005	Anemia, decrease in blood platelets, increased risk of cancer	Discharge from factories, leaching from gas storage tanks and landfills
Benzo(a)pyrene	Zero ^d	.0002	Reproductive difficulties, increased risk of cancer	Leaching from linings of water storage tanks and distribution lines
Carbofuran	.04	.04	Problems with blood or nervous system, reproductive difficulties	Leaching of soil fumigant used on rice and alfalfa
Carbon tetrachloride	Zero ^d	.005	Liver problems, increased risk of cancer	Discharge from chemical plants and other industrial activities
Chlordane	Zero ^d	.002	Liver or nervous system problems, increased risk of cancer	Residue of banned termiticide
Chlorobenzene	.1	.1	Liver or kidney problems	Discharge from chemical and agricultural chemical factories
2, 4-D	.07	.07	Kidney, liver, or adrenal gland problems	Runoff from herbicide used on row crops
Dalapon	.2	.2	Minor kidney changes	Runoff from herbicide used on rights-of-way
1, 2-Dibromo-3-chloropropane (DBCP)	Zero ^d	.0002	Reproductive difficulties, increased risk of cancer	Runoff and leaching from soil fumigant used on soybeans, cotton, pineapples, and orchards
o-Dichlorobenzene	.6	.6	Liver, kidney, or circulatory system problems	Discharge from industrial chemical factories
p-Dichlorobenzene	.075	.075	Anemia; liver, kidney, or spleen damage; changes in blood	Discharge from industrial chemical factories
1, 2-Dichloroethane	Zero ^d	.005	Increased risk of cancer	Discharge from industrial chemical factories

continued

Table 2.3—National primary drinking water regulations^a (States are expected to focus attention on risks related to the contaminants listed in their source water assessments.) (continued)

Contaminants	MCLG	MCL or TT	Potential health effects from ingestion of water	Sources of contaminant in drinking water
<i>--- Milligrams per liter ---</i>				
Organic chemicals (cont.)				
1-1- Dichloroethylene	0.007	0.007	Liver problems	Discharge from industrial chemical factories
cis-1, 2- Dichloroethylene	.07	.07	Liver problems	Discharge from industrial chemical factories
trans-1, 2- Dichloroethylene	.1	.1	Liver problems	Discharge from industrial chemical factories
Dichloromethane	Zero ^d	.005	Liver problems, increased risk of cancer	Discharge from pharmaceutical and chemical factories
1-2- Dichloropropane	Zero ^d	.005	Increased risk of cancer	Discharge from industrial chemical factories
Di (2-ethylhexyl) adipate	.4	.4	General toxic effects or reproductive difficulties	Leaching from PVC plumbing systems, discharge from chemical factories
Di (2-ethylhexyl) phthalate	Zero ^d	.006	Reproductive difficulties, liver problems, increased risk of cancer	Discharge from rubber and chemical factories
Dinoseb	.007	.007	Reproductive difficulties	Runoff from herbicide used on soybeans and vegetables
Dioxin (2,3,7,8-TCDD)	Zero ^d	.00000003	Reproductive difficulties, increased risk of cancer	Emissions from waste incineration and other combustion, discharge from chemical factories
Diquat	.02	.02	Cataracts	Runoff from herbicide use
Endothall	.1	.1	Stomach and intestinal problems	Runoff from herbicide use
Endrin	.002	.002	Nervous system effects	Residue of banned insecticide
Epichlorohydrin	Zero ^d	TT	Stomach problems, reproductive difficulties, increased risk of cancer	Discharge from industrial chemical factories, added to water during treatment process
Ethylbenzene	.7	.7	Liver or kidney problems	Discharge from petroleum refineries
Ethylene dibromide	Zero ^d	.00005	Stomach problems, reproductive difficulties, increased risk of cancer	Discharge from petroleum refineries
Glyphosate	.7	.7	Kidney problems, reproductive difficulties	Runoff from herbicide use
Heptachlor	Zero ^d	.0004	Liver damage, increased risk of cancer	Residue of banned termiticide
Heptachlorepoxyde	Zero ^d	.0002	Liver damage, increased risk of cancer	Breakdown of heptachlor
Hexachlorobenzene	Zero ^d	.001	Liver or kidney problems, reproductive difficulties, increased risk of cancer	Discharge from metal refineries and agricultural chemical factories
Hexachlorocyclo- pentadiene	.05	.05	Kidney or stomach problems	Discharge from chemical factories
Lindane	.0002	.0002	Liver or kidney problems	Runoff and leaching from insecticide used on cattle, lumber, gardens

continued

Table 2.3—National primary drinking water regulations^d (States are expected to focus attention on risks related to the contaminants listed in their source water assessments.) (continued)

Contaminants	MCLG	MCL or TT	Potential health effects from ingestion of water	Sources of contaminant in drinking water
- - - Milligrams per liter - - -				
Organic chemicals (cont.)				
Methoxychlor	0.04	0.04	Reproductive difficulties	Runoff and leaching from insecticide used on fruits, vegetables, alfalfa, livestock
Oxamyl (Vydate)	.2	.2	Slight nervous system effects	Runoff and leaching from insecticide used on apples, potatoes, and tomatoes
Polychlorinated biphenyls (PCB's)	Zero ^d	.0005	Skin changes, thymus gland problems, immune deficiencies, reproductive or nervous system difficulties, increased risk of cancer	Runoff from landfills, discharge of waste chemicals
Pentachlorophenol	Zero ^d	.001	Liver or kidney problems, increased risk of cancer	Discharge from wood-preserving factories
Picloram	.5	.5	Liver problems	Herbicide runoff
Simazine	.004	.004	Problems with blood	Herbicide runoff
Styrene	.1	.1	Liver, kidney, and circulatory problems	Discharge from rubber and plastic factories, leaching from landfills
Tetrachloroethylene	Zero ^d	.005	Liver problems, increased risk of cancer	Leaching from PVC pipes, discharge from factories and dry cleaners
Toluene	1	1	Nervous system, kidney, or liver problems	Discharge from petroleum factories
Total trihalomethanes (TTHM's)	None ^b	.10	Liver, kidney, or central nervous system problems; increased risk of cancer	By-product of drinking water disinfection
Toxaphene	Zero ^d	.003	Kidney, liver, or thyroid problems; increased risk of cancer	Runoff and leaching from insecticide used on cotton and cattle
2,4,5-TP (Silvex)	.05	.05	Liver problems	Residue of banned herbicide
1,2,4-Trichlorobenzene	.07	.07	Changes in adrenal glands	Discharge from textile finishing factories
1,1,1-Trichloroethane	.20	.2	Liver, nervous system, or circulatory problems	Discharge from metal degreasing sites and other factories
1,1,2-Trichloroethane	.003	.005	Liver, kidney, or immune system problems	Discharge from industrial chemical factories
Trichloroethylene	Zero ^d	.005	Liver problems, increased risk of cancer	Discharge from petroleum refineries
Vinyl chloride	Zero ^d	.002	Increased risk of cancer	Leaching from PVC pipes, discharge from plastic factories
Xylenes (total)	10	10	Nervous system damage	Discharge from petroleum factories, discharge from chemical factories

continued

Table 2.3—National primary drinking water regulations^a (States are expected to focus attention on risks related to the contaminants listed in their source water assessments.) (continued)

Contaminants	MCLG	MCL or TT	Potential health effects from ingestion of water	Sources of contaminant in drinking water
- - - Milligrams per liter - - -				
Radionuclides				
Beta particles and photon emitters	None ^b	4 millirems per yr	Increased risk of cancer	Decay of natural and man-made deposits
Gross alpha particle activity	None ^b	15 pCi/L	Increased risk of cancer	Erosion of natural deposits
Radium 226 and radium 228 (combined)	None ^b	5 pCi/L	Increased risk of cancer	Erosion of natural deposits
Microorganisms				
<i>Giardia lamblia</i>	Zero ^d	TT	Giardiasis—a gastroenteric disease	Human and animal fecal waste
Heterotrophic plate count	NA	TT	No health effects but can indicate how effective treatment is at controlling microorganisms.	NA
Legionella	Zero ^d	TT	Legionnaire's Disease—a form of pneumonia	Found naturally in water, multiplies in heating systems
Total coliforms (including fecal coliform and <i>E. coli</i>)	Zero ^d	5.0%	Used as an indicator that other potentially harmful bacteria may be present	Human and animal fecal waste
Turbidity	NA	TT	Turbidity has no health effects but can interfere with disinfection and provide a medium for microbial growth. It may indicate the presence of microbes.	Soil runoff, growth of algae
Viruses (enteric)	Zero ^d	TT	Gastroenteric disease	Human and animal fecal waste

MCL = maximum contaminant level or the maximum permissible level of a contaminant in drinking water delivered to any user;
MCLG = maximum contaminant level goal; NA = not available; pCi = picocuries; PVC = polyvinyl chloride; TT = treatment technique.

^a Water-quality regulations are subject to change. For the latest regulations, visit the Web site: <http://www.epa.gov/OGWDW/wot/appa.html>.

^b MCLG has not been defined.

^c The units vary with the contaminant and are defined by the U.S. Environmental Protection Agency.

^d MCLG is 0.0.

Source: U.S. EPA 1999b.

Table 2.4—National secondary drinking water regulations, which are nonenforceable guidelines for contaminants that may cause cosmetic effects (e.g., skin or tooth discoloration) or aesthetic effects (e.g., taste, odor, or color) in drinking water

Contaminant	Unit	Secondary standard
Aluminum	mg/L	0.05–0.2
Chloride	mg/L	250
Color	Color units	15
Copper	mg/L	1.0
Fluoride	mg/L	2.0
Foaming agents	mg/L	.5
Iron	mg/L	.3
Manganese	mg/L	.05
Odor	Threshold odor number	3
pH		6.5 –8.5
Silver	mg/L	.10
Sulfate	mg/L	250
Total dissolved solids	mg/L	500
Zinc	mg/L	5

Source: U.S. EPA 1999c.

Dissolved Gases

The most abundant dissolved gases in water are nitrogen (N_2), oxygen (O_2), carbon dioxide (CO_2), methane (CH_4), hydrogen sulfide (H_2S), and nitrous oxide (N_2O). The first three are abundant in the Earth's atmosphere. The second three are typically products of biogeochemical processes that occur in nonaerated, low oxygen environments. The solubility of most gases increases with decreasing water temperature and decreases with increasing concentrations of chlorides or other salts.

The concentration of dissolved oxygen (DO) is essential to aquatic life and can effect the water's color, taste, odor, and chemistry. Unpolluted surface waters are generally saturated with DO because of reaeration and the production of oxygen during photosynthesis by submerged aquatic plants. Ground water systems tend toward oxygen depletion and reducing conditions because the oxygen consumed during hydrochemical and biochemical reactions is not replenished by the atmosphere. Polluted surface waters tend to have lower DO concentrations because of oxygen consumption during the decomposition of organic matter.

The concentrations of DO strongly influence the solubility and stability of elements that readily gain or lose electrons including iron (Fe^{+3}), manganese (Mn^{+3}), nitrogen, sulfur (S), and arsenic (As^{+3}). When dissolved iron and manganese are exposed to air, they form insoluble precipitates that make water turbid, cause stains in laundry, and impart a bitter taste (Cox 1964). In water with little or no oxygen, iron minerals are reduced, and adsorbed phosphorus and other elements can be released into the water. The solubility of most arsenic and arsenic-sulfur compounds depends on the presence of DO and can have concentrations in water above the primary standard of 0.05 milligrams (mg) per liter (L) (Freeze and Cherry 1979).

Organic Compounds

Organic compounds have carbon and usually hydrogen and oxygen as the main components in their structural framework. They are typically nonpolar, have relatively low solubility, and are degraded by microorganisms, hydrolysis, oxidation-reduction, and volatilization. In natural waters, they are transported as dissolved phases and attached to particulates.

Commonly occurring natural organic compounds include plant and animal tissue and the products of their decomposition. Synthetic organics found in water include petroleum products, pesticides, and herbicides (table 2.3). Most synthetic toxic organic compounds originate from coal mining, petroleum refining, and manufacture of textile, wood pulp, and pesticides (table 2.1). In the environment, they are usually associated with roadways and industrial, urban, and agricultural land uses. Disinfecting some organic-rich waters with chlorine may also result in the formation of carcinogenic organic compounds such as trihalomethanes (Martin and others 1993, see chapter 5). Highly soluble, potentially carcinogenic organic compounds from gasoline spills and emissions are also found in water supplies and can make water distasteful and undrinkable (see chapter 6).

Trace Metals and Nonmetals

Primary and secondary water-quality standards have been developed for common trace metals and nontrace elements (tables 2.3, 2.4). Most of these elements occur in natural, uncontaminated waters in concentrations below 1 mg per liter. Metals have relatively low solubilities. Solubilities are usually lowest at neutral acidity and increase with increasing acidity and increasing alkalinity. A characteristic feature of metals is their tendency to form hydrolyzed species and

Table 2.5—Common chemical processes involved as water interacts with its environment

Process	Description
Acid-base reactions	Acid-base reactions are a common type of chemical reaction in aqueous environments that are important in the leaching and transport of cations. They are also important in certain water treatment processes and in the corrosion of water distribution systems. Acids are hydrogen-containing substances that supply protons in water, typically by liberating hydrogen ions. Bases are proton acceptors and are typically substances that contain hydroxide ions (OH ⁻) or hydroxyl groups, which dissociate in water. Acidity is usually measured using the logarithmic pH scale, which is defined as the concentration of hydrogen ions in water in moles per liter (see glossary of terms). Acidic soil or waters can have increased concentrations of metals and decreased phosphate availability and nitrification rates. The dissolution of carbon dioxide in water to form carbonic acid (H ₂ CO ₃) is the most common acid-producing reaction in natural waters.
Adsorption-desorption	Adsorption-desorption is the exchange of chemicals from solution and the surfaces of charged particles by chemical or physical bonding. When the adsorption bonds are chemical, they are relatively irreversible. If they are physical van der Waals type forces, they are easily broken and reversible. Particle type (organic or inorganic), particle size (clay, sand, etc.), and the presence of organic and inorganic coatings can have large effects on the amount of adsorption and desorption of organic waste, pesticides, ammonia, and phosphorus as they are transported by water through soils. In general, adsorption tends to increase with increases in the content of both clay and organic matter. The removal of contaminants in water by adsorption and subsequent settling of sediments is an important process in lakes, rivers, and water treatment plants.
Volatilization	Volatilization is the loss of a chemical from the soil-water system by vaporization into the atmosphere. The rate of volatilization depends on the concentration gradient above the volatilization surface and typically increases with temperature and the removal of vaporized chemicals away from the surface by wind or heat. This is a particularly important process in fires and after the application of pesticides or nutrients.
Reaeration	Reaeration is the transfer of gases, typically oxygen, from the atmosphere into water. The rate of reaeration increases with turbulence, exposed surface area, and the solubility and diffusivity of gas, both of which are temperature dependent. Oxygen is the most common dissolved gas in water and is essential for aquatic life and the decomposition of natural and synthetic organic matter.
Oxidation-reduction	Oxidation is the loss of electrons and reduction is the gain of electrons. The redox potential is used to express the tendency to exchange electrons and is measured as the voltage required to prevent the acceptance of electrons on a standard electrode. Oxidic environments are considered to have high redox potentials because O ₂ is available as an electron acceptor. In order to reduce inorganic constituents, some other constituents must be oxidized, typically organic matter in reactions that are catalyzed by bacteria or isolated enzymes.
Decomposition-mineralization-immobilization	Decomposition is a general term that refers to the breakdown of organic matter. Mineralization specifically refers to decomposition processes that release carbon as CO ₂ and nutrients in inorganic forms. This breakdown usually involves soil microbes and is caused by some combination of photolysis, hydrolysis, oxidation-reduction, and enzyme actions. Immobilization is the accumulation of N, P, and other nutrients in soil microbes.

Table 2.6—Summary of the chemical behavior of important water contaminants

Contaminant	Biochemical transformations ^a		Chemical reactions		Physiochemical retardation ^b	
	Aerobic	Anaerobic	Acid	Alkaline	Acid	Alkaline
Metals						
Aluminium (Al ⁺³)	M	P	P	D	P	D
Cadmium (Cd ⁺²)	P	P	P	D	P	M
Chromium (Cr ⁺³)	P	P	M	P	D	P
Copper (Cu ⁺¹)	P	P	M	D	D	M
Iron (Fe ⁺³)	D	P	P	D	M	D
Lead (Pb ⁺⁴)	P	P	M	D	M	D
Manganese (Mn ⁺³)	M	M	P	D	M	D
Mercury (Hg ⁺¹)	M	P	M	D	M	D
Silver (Ag ⁺¹)	P	P	M	D	M	D
Zinc (Zn ⁺²)	P	P	P	D	P	D
Inorganic nonmetals						
Ammonium (NH ₄ ⁺¹)	D	P	P	P	P	D
Nitrate (NO ₃ ⁻¹)	P	D	P	P	P	P
Sodium (Na)	P	P	P	P	P	M
Sulphate (SO ₄ ⁻²)	P	D	P	M	P	P
Fluoride (F ⁻¹)	P	P	M	M	D	P
Chloride (Cl ⁻¹)	P	P	P	P	P	P
Arsenic (As)	P	P	M	P	M	D
Selenium (Se)	P	P	D	M	D	P
Cyanide	P	P	P	P	D	P
Organic compounds						
Aliphatic	D	P	P	P	D	D
Hydrocarbons						
Phenols	D	M	P	P	M	M
Benzene	D	P	P	P	D	D
Toluene	D	P	P	P	D	D
Polynuclear aromatics	M	P	P	P	M	M
Halogenated organics						
Tri and tetra						
chloroethylene	P	M	P	P	M	M
Carbon tetrachloride	P	M	P	P	M	M
Chloroform	P	M	P	P	P	P
Methylene chloride	M	M	P	P	P	P
Chlorobenzene	D	P	P	P	D	D
Chlorphenols	D	M	P	P	P	M
Fecal organisms						
Fecal coliform	P	P	P	P	M	P
Pathogenic bacteria	P	P	P	P	M	P
Pathogenic virus	P	P	D	M	M	D

D = reactions do occur; M = reactions may occur; P = reactions probably occur.

^a Biochemical transformations involve biological organisms, usually microbes.

^b Physiochemical retardation involves physical and chemical bonds that are usually to mineral surfaces.

Source: Adapted and updated from Foster and Gomes 1989.

inorganic and organic complexes. These complexes typically absorb to suspended particulates or form insoluble precipitates. Therefore, the transport of metals across the landscape is often related to acidity, the presence of organic compounds, and the transport of sediment (table 2.6).

While trace metals and nonmetals occur naturally, their concentrations can be greatly increased over background levels by mining activities, waste dumps, acidic runoff, tanneries, and other industries. Some metals, such as copper and cadmium, are associated with automobiles and are concentrated on streets, parking lots, and industrial areas (Bannerman and others 1993). Major sources of lead include urban soil, lead-based paint, and some hair-coloring cosmetics (Mielke 1999).

Fluorine (F) is a trace nonmetal that occurs as fluoride and is undersaturated in nearly all natural water. Because it can have beneficial effects on dental health, fluorine is added to some municipal water supplies. Arsenic is a soluble trace nonmetal that can be naturally present in water from areas of recent volcanism. It is widely used in pigments, insecticides, herbicides, and metal alloys (Freeze and Cherry 1979). Selenium (Se) is a toxic nonmetallic element that has geochemical properties similar to sulfur. It can occur in appreciable concentrations in coal, uranium ore, certain shales, and discharges from petroleum refineries and mines. Like sulfur, it forms strong chemical bonds on the surface of minerals and can be reduced by anaerobic bacteria (Schlesinger 1997).

Nitrogen

Nitrogen, a major nutrient for vegetation, plays a dominant role in many biochemical reactions. However, in certain chemical forms, it can adversely affect humans, ecosystems, and water supplies. Since preindustrial times, fertilizer production and other human activities have more than doubled the global input of nitrogen to terrestrial ecosystems (Kinzig and Socolow 1994, Vitousek and others 1997). This increase has made nitrogen the most common water pollutant in the United States. In the Northeastern United States alone, anthropogenic activities have apparently increased the nitrate concentrations in major rivers threefold to tenfold since the early 1900's (Matson and others 1997). Anthropogenic alteration of nitrogen cycles has also affected forest and aquatic productivity and increased acid rain, photochemical smog, and greenhouse gases (Fenn and others 1998, Vitousek and others 1997).

Certain nitrogen compounds can have toxic effects at relatively low concentrations. Methaemoglobinemia (blue-baby syndrome) in bottle-fed babies and the elderly is a

human health hazard associated with nitrite (NO_2^{-1}) in drinking water (table 2.2). Nitrate in water can also present similar health hazards as can nitrate in many foods (GEMS 1991). Bacteria residing in vertebrate digestive tracts can convert the relatively benign nitrate into the toxic nitrite (Kinzig and Socolow 1994). Ammonia dissolved in drinking water is not toxic to humans but can be toxic to some aquatic invertebrates and fish depending on the concentration of DO temperature, acidity, and salinity, and the carbon dioxide-carbonic acid equilibrium of water. Because all forms of inorganic nitrogen are nutrients to green plants, excessive concentrations in water can lead to algal blooms, excessive growth of submerged aquatic plants, and eutrophication, particularly in coastal and marine ecosystems.

The global nitrogen cycle consists of three major reservoirs: (1) the atmosphere, (2) the hydrosphere, and (3) the biosphere (fig. 2.1). The flow between these reservoirs occurs in many forms and pathways (fig. 2.2). Inorganic nitrogen can be transported in water as dissolved nitrous oxide or nitrogen gas, ammonia, and cations or as anions of nitrite or nitrate. The concentrations of these compounds are low in most unpolluted freshwater and high in waters contaminated by organic wastes, sewage, or fertilizers. Worldwide, pristine rivers have average concentrations of ammonia and nitrate of 0.015 mg per liter and 0.1 mg per liter, respectively (GEMS 1991). Nitrate concentrations > 1 mg per liter generally indicate anthropogenic inputs. The lowest concentrations are generally found in deep ground water and surface waters draining pristine wildlands (GEMS 1991, Spahr and Wynn 1997). The highest levels are associated with surface runoff and ground water from fertilized agricultural and urban areas. In undisturbed watersheds, annual yields increase with annual runoff, and yields from savanna and rangeland are less than from forest (Lewis and others 1999).

Organic nitrogen is converted to inorganic nitrogen in a process called mineralization in the following oxidation sequence: organic nitrogen and ammonium to nitrite to nitrate. In water that is strongly oxidized, nitrate is the stable phase and is very mobile. As redox potential declines, nitrate is reduced or denitrified to nitrous oxide or nitrogen gas. Because of the potential adverse ecosystem and health effects associated with nitrites and nitrates, denitrification is desirable for water quality. Generally, the amount of net mineralization is directly related to the total content of organic nitrogen and carbon (Schlesinger 1997, Vitousek and Melillo 1979). Nitrification tends to be lower in soil with low acidity, low soil oxygen, low soil moisture, and low temperature, and high litter carbon to nitrogen ratios. At the watershed scale, rates of denitrification vary with landscape positions (Jordan and others 1993, Peterjohn and

The Global Nitrogen Cycle

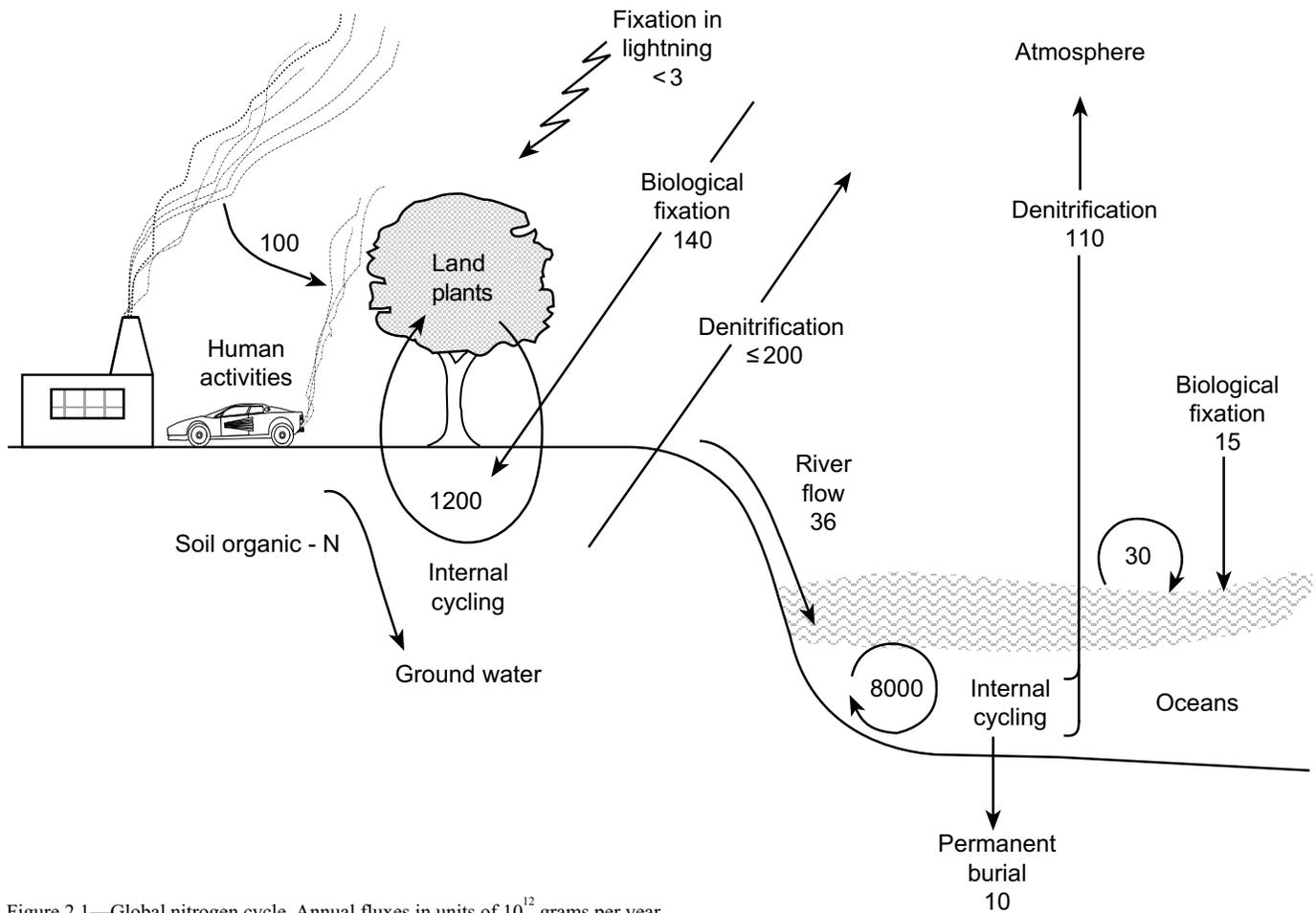


Figure 2.1—Global nitrogen cycle. Annual fluxes in units of 10^{12} grams per year.

Correll 1984). In general, relatively high denitrification rates are found in riparian forests and at the base of slopes where water, carbon, nitrogen, and phosphorus are readily available.

Because nitrogen is essential to the growth of plants, seasonal differences in plant uptake can cause measurable variations in the concentration of nitrogen in soil and surface water. In general, the lowest nitrogen levels in surface or ground water occur during the early growing season when plant uptake is greatest (Boyd 1996). Maximum nitrogen concentrations typically occur in the winter when plant uptake is reduced, and the dissolved fraction is concentrated in unfrozen water. However, seasonal trends

can be reversed or diminished in areas with large anthropogenic inputs.

Phosphorus

The presence of phosphorus in drinking water is not considered a human health hazard, and no drinking water-quality standards are established for phosphorus. Nevertheless, phosphorus can affect the water's color and odor and indicate the presence of other organic pollution. Furthermore, because phosphorus can accelerate the growth of algae and aquatic vegetation, it contributes to the eutrophication and associated deterioration of municipal water supplies. Whereas excess nitrogen is responsible for most of

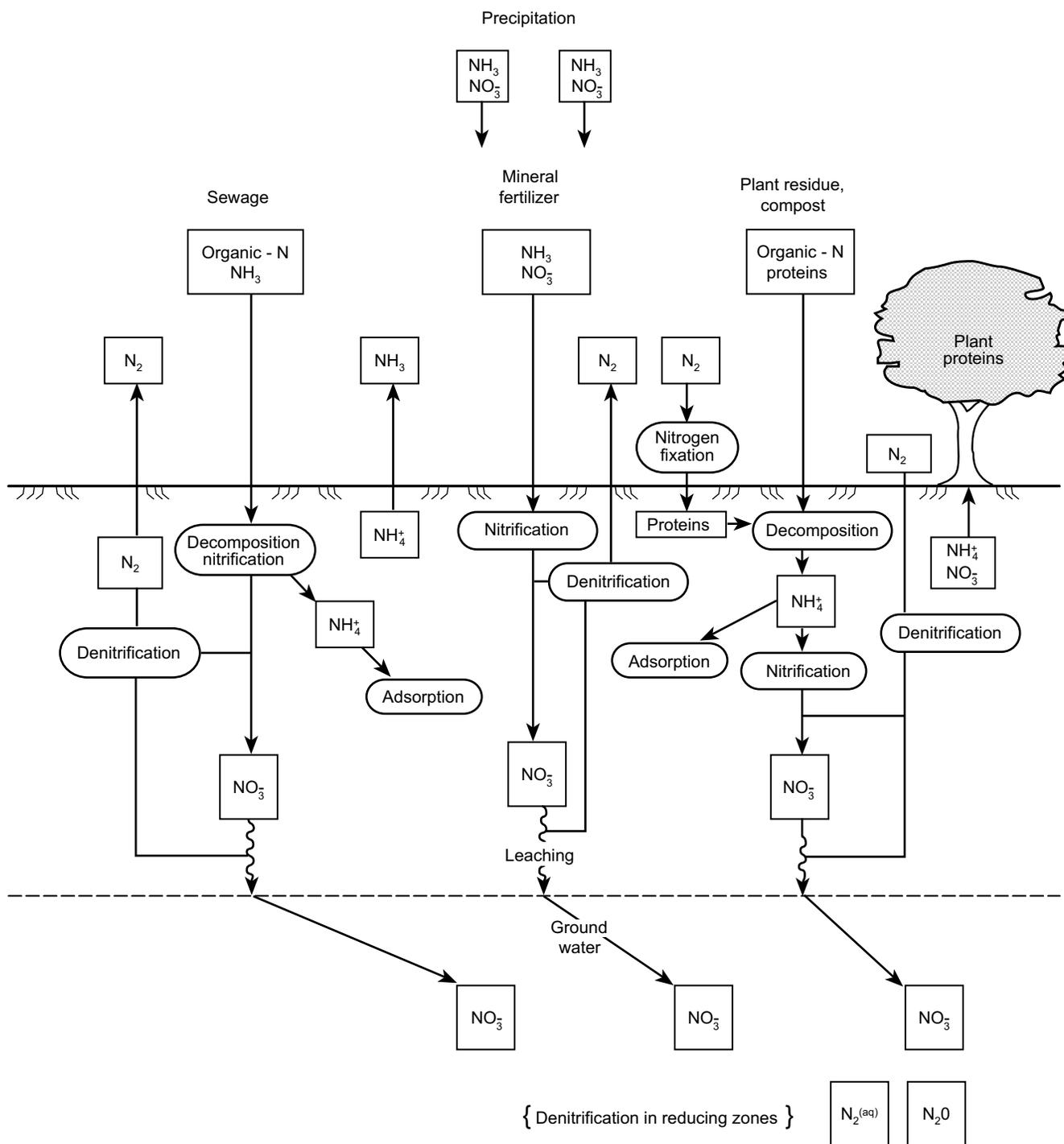


Figure 2.2—Sources and pathways of nitrogen in the subsurface environment.

the coastal and marine eutrophication, agricultural sources of phosphorus dominate the eutrophication processes in many freshwater aquatic systems (Matson and others 1997).

Nearly all the phosphorus in terrestrial ecosystems is originally derived from the weathering of minerals (fig. 2.3). The most common phosphorus-rich mineral is apatite, a calcium orthophosphate that is present in some igneous rocks and marine sediments. In natural freshwater, phosphorus exists in both dissolved and particulate fractions. Dissolved phases typically originate from excretions by organisms, whereas particulate fractions can have organic or inorganic origins. In streams, a large fraction of phosphorus is adsorbed on and transported with organic and inorganic particulates. In lakes, a large proportion of the phosphorus in oxygen-rich surface waters is held in plankton biomass (Schlesinger 1997). In deeper, anoxic lakes, phosphorus is adsorbed to sediments and particulates but can be released during the reduction of iron compounds. Unlike nitrogen, carbon, and hydrogen, phosphorus does not have a significant gaseous component.

Chemical Evolution of Water

As water moves across the landscape, it interacts with the surfaces it contacts and chemically evolves toward the composition of seawater [for detailed explanations see Stumm and Morgan (1970) and Freeze and Cherry (1979)]. In general, the evolution of deep ground water typically involves increases in dissolved solids and decreases in DO, organic waste, pesticides, phosphorus, and nitrogen. In contrast, the concentrations of organic waste, pesticides, phosphorus, and nitrogen increase as surface water travels across the landscape and interacts with both natural and anthropogenic systems.

Fresh, young water that has had little contact with its surroundings is generally low in total dissolved solids and rich in bicarbonate anions derived from soil carbon dioxide and the dissolution of carbonate minerals. Sulfate anions tend to dominate in intermediate age ground water while chloride anions dominate in older, deep ground water that has traveled long distances. These sulfate and chloride anions are derived from the dissolution of soluble sedimentary minerals. Because these minerals are present only in small amounts in most rocks, water usually has to travel considerable distances before it is dominated by either sulfate or chloride anions.

The DO content and redox potential tend to decrease as water travels across the landscape. Rain and snow are exposed to atmospheric oxygen and have relatively high DO and redox potentials. As water passes through organic-rich

forest litter and soil, the DO is removed, redox potential declines, and large amounts of organic acids are generated. Nutrient immobilization predominates in the upper layers of fresh litter, while mineralization of nitrogen, phosphorus, and sulfur is usually greatest in the upper mineral soil. As water travels through the subsurface, all the DO is consumed by bacterially catalyzed reactions that oxidize organic matter. Eventually the aerobic bacteria involved in these reactions can no longer thrive, and anaerobic conditions prevail. Then ammonia, manganese, ferrous iron, and sulfate become oxidizing agents.

Cation concentrations in water vary considerably in space and time and do not follow well-defined, theoretically based sequences like anions or redox potentials. Nevertheless, cations enter the aquatic system from the weathering of minerals and the breakdown of organic materials. Their concentrations typically increase with travel distance in both surface and ground water. The most abundant cations in water supplies are calcium and magnesium, which can be removed by chemical treatments to prevent scaling of pipes and to reduce the amount of soap needed for washing.

Physical Properties

The physical characteristics of concern in drinking water are temperature, color, turbidity, sediments, taste, and odor.

Temperature

Because of its hydrogen bonds and molecular structure, water has an unusual trait—the density of its solid phase (ice) is lower than that of its liquid phase (water). Because of this trait, ice floats, and pipes and plant tissues rupture when the water within them freezes and expands.

The rates of chemical and metabolic reactions, viscosity and solubility, gas-diffusion rates, and the settling velocity of particles depend on temperature. Metabolism, reproduction, and other physiological processes of aquatic organisms are controlled by heat-sensitive proteins and enzymes (Ward 1985). A 10 °C increase in temperature will roughly double the metabolic rate of cold-blooded organisms and many chemical reactions. A permanent 5 °C change in temperature can significantly alter the structure and composition of an aquatic population (MacDonald and others 1991, Nathanson 1986). Temperature increases also decrease DO concentrations but can increase the oxidation rate and efficiency of certain biological, wastewater treatment systems.

The Global Phosphorus Cycle

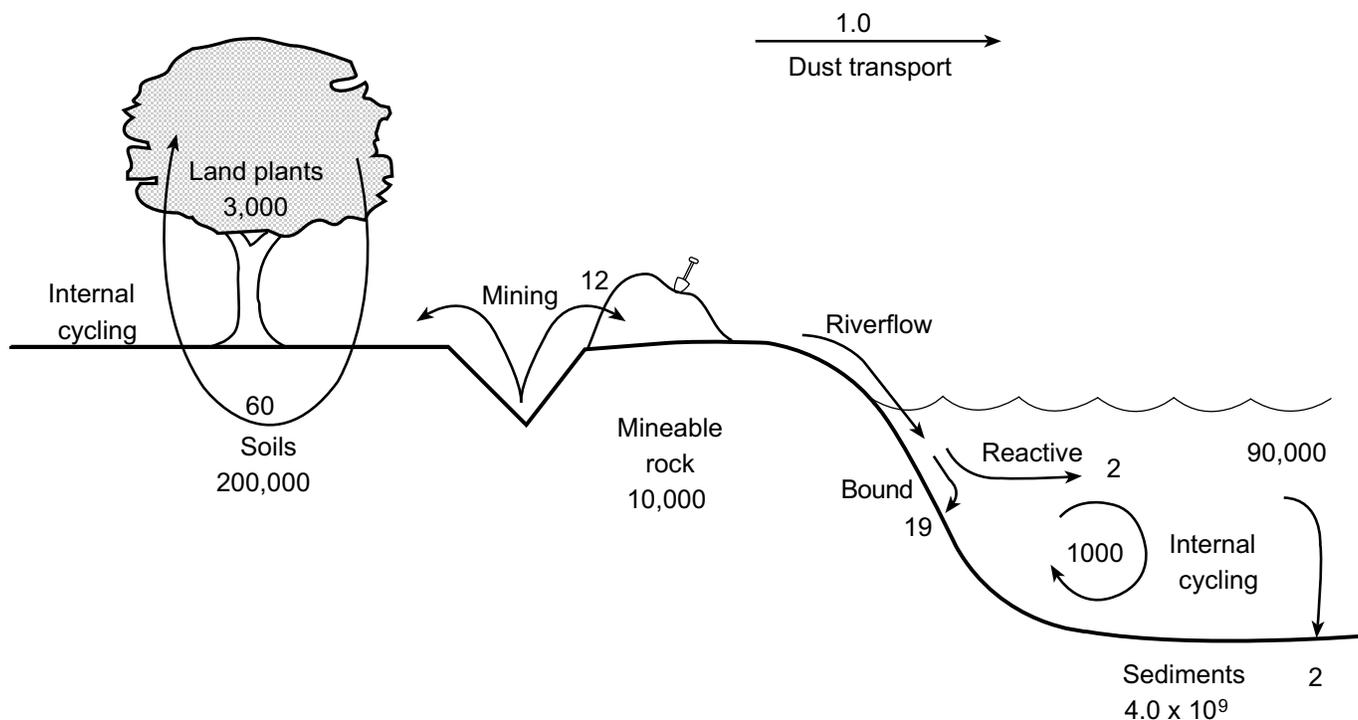


Figure 2.3—Global phosphorus cycle. Annual fluxes in units of 10^{12} grams per year.

The temperature of water naturally varies with time of day, season, and the type of water body. Changes in surface water temperatures reflect seasonal changes in net radiation, daily changes in air temperature, and local variations in incoming radiation. Temperature variations in ground water are less than in surface water. Except in the winter, surface water is usually warmer than ground water, and most anthropogenic activities increase water temperatures. Removal of vegetative canopies over streams influences water temperatures by affecting energy inputs, evaporative cooling, and the way water flows across the landscape. The cooling rate for surface water depends on heat transfer to the atmosphere.

Seasonal and spatial variations in the temperature in water supply reservoirs can have large effects on the quality of raw municipal water (Cox 1964). Water in deep reservoirs is commonly divided into three zones: the upper circulating zone, the middle transition zone, and the deepest zone of stagnation. Water in the upper surface zone is aerated and mixed by wind action and typically has abundant DO. In contrast, the deepest, stagnant water contains little or no DO because it has been removed during the oxidation of organic

matter. The breakdown of organic matter also makes deep water acidic and rich in carbonic acid. Consequently, stagnate, deep water has the chemical conditions necessary to dissolve iron, manganese, sulfur, and other taste- and odor-producing substances. To avoid the objectionable taste and odor of the deep water, municipal water is usually drawn from the surface of the reservoir. However, when the temperature of the surface water falls rapidly, it can become denser than the bottom water, causing the entire water column of the reservoir to mix or “turn over.” During these mixing events, the DO content of the entire lake can decrease, causing massive fish kills and foul smelling and poor tasting water. Similar mixing can occur in stratified lakes or estuaries during periods of intense runoff.

Color and Turbidity

Pure water is colorless in thin layers and bluish green in thick layers. Particulates and insoluble compounds typically add color and reduce transparency. Consequently, the presence of light-dependent aquatic organisms can affect esthetic appeal and taste of water as well as the effectiveness of certain wastewater treatment processes.

Turbidity is an optical property related to the scattering of light and clarity. It is typically controlled by the presence of suspended particles or organic compounds. Turbidity itself is not injurious to human health. Approximately 50 percent of the total incident light is scattered or transformed into heat within the first meter of water. As turbidity increases, it reduces the depth of sunlight penetration, thereby altering water temperature and stratification, the photosynthesis of aquatic organisms, the DO content of the water body, and the cost of water treatment. In addition, turbid water can contain particulate of soil or fecal matter that harbors microorganisms and/or carries absorbed contaminants. The removal of particulates by gravity or by addition of chemicals is typically the first step in treating water for human consumption. The sedimentation of particles and the bleaching action of sunlight during reservoir storage can reduce both the color and turbidity of water (Cox 1964).

Sediment

Sediment is a major water-quality concern because of its ability to transport harmful substances and its impacts on the cost of water treatment and the maintenance of water distribution systems. While sediment is derived during the natural weathering and sculpturing of the landscape, accelerated levels of erosion and sedimentation are associated with many anthropogenic activities (table 2.1).

The general term sediment includes both organic and inorganic particles that are derived from the physical and chemical weathering of the landscape. Individual particles are eroded, transported, and deposited. Erosion can be either physical or chemical. Transport can be by wind, gravity, or water. In water and air, particles can be transported in suspension (suspended load) or along the substrate (bed load). Sediment load is the total quantity of sediment that is transported through a cross-section of a stream during a specific time period. The actual amount of sediment transported at any place or time depends on the supply of sediment and the transport capacity of the stream. Sediment is usually measured as mass per unit area (tons per acre per year or metric tonnes per hectare per year), concentration (parts per million or milligrams per liter), or lowering of the landscape (inches per 1,000 years or millimeters per 1,000 years). In general, high sediment loads increase water treatment costs and reduce the storage volume and life span of water storage facilities.

Biological Properties

Aquatic organisms are usually grouped into those that (1) obtain the carbon they need for biosynthesis from carbon

dioxide (autotrophs) and (2) use existing organic compounds as their carbon source (heterotrophs). Generally, autotrophs increase DO concentrations in water through photosynthesis, while heterotrophs are responsible for breakdown and recycling of dead organic materials and decreased DO concentrations.

Most microbial contaminants in water are caused by heterotrophs that are transmitted to a water system via human and animal fecal matter (U.S. EPA 1999a). Most waterborne pathogenic microorganisms are bacteria or viruses that survive in sewage and septic leachate (table 2.7). Bacterial pathogens are generated by both animal and human sources, while viral pathogens are usually only generated by human sources. Viruses that infect animals normally do not cause illness in humans. However, animal sources for some viruses that effect humans are suspected, particularly viruses that infect the respiratory system like the sin nombre virus, hantavirus, influenza virus, and Ebola virus.

Common bacterial diseases spread by aquatic microorganisms include Legionnaire's disease, cholera, typhoid, and gastroenteritis. Waterborne viral diseases include polio, hepatitis, and forms of gastroenteritis. Waterborne parasitic diseases include amoebic dysentery, flukes, and giardiasis. *Giardia* spp. and *Cryptosporidium* spp. are parasitic protozoans that are transferred between animals and humans via the fecal-oral route and are significant sources of gastrointestinal illness. They are common in surface water in back-country areas, including in many national forests and parks. These back-country areas, which provide animal habitat, experience low human use (Monzingo and Stevens 1986) (see chapter 15). Unfortunately, some parasitic protozoans are not removed in most water treatment plants because they are small enough to pass filtration systems and are very resistant to disinfectants.

The analytical procedures for detecting waterborne viral diseases are costly and time consuming. Therefore most drinking and recreational waters are routinely tested for microbes that are easier to detect but whose presence is highly correlated with human health hazards. Coliforms are the most common type of microbes used in this type of testing. All coliforms are aerobic and facultative anaerobic, gram-negative, nonspore-forming, rod-shaped bacteria that ferment lactose. Their presence and abundance in raw water is used to screen for fresh fecal contamination (Cox 1964). Their presence in treated water is used to determine treatment plant efficiency and the integrity of the distribution system.

Many environmental factors can affect the transport of microbes across the landscape (table 2.8). Relatively

Table 2.7—Common waterborne pathogenic and indicator bacteria and viruses

Waterborne pathogenic bacteria	Waterborne pathogenic viruses
Legionella <i>Mycobacterium avium</i> intracellular (MAC) <i>Shigella</i> (several strains) <i>Helicobacter pylori</i> <i>Vibrio cholerae</i> <i>Salmonella typhi</i> <i>S. typhimurum</i> Yersinia <i>Campylobacter</i> (several strains) <i>Escherichia coli</i> (several pathogenic strains)	Enteroviruses Coxsackieviruses Echoviruses Poliovirus Enterovirus 70 and 71 Hepatitis A virus Hepatitis E virus Enteric adenoviruses Rotavirus Norwalk virus Small round structured viruses (SRSV) Astrovirus Caliciviruses
Waterborne indicator bacteria	Waterborne indicator viruses
Total coliform Fecal coliform <i>E. coli</i> (both nonpathogenic and pathogenic strains) Enterococci Fecal streptococci <i>Clostridium perfringens</i> (anaerobic spores) <i>Klebsiella pneumoniae</i> <i>Aeromonas hydrophila</i>	Bacteriophage Bacteroides phage Coliphage Male-specific coliphage FRNA phage FDNA phage Host <i>Salmonella</i> WG-49 Host <i>E. coli</i> C-3000 Host <i>E. coli</i> FAMP Host <i>E. coli</i> 15597 Somatic coliphage Host <i>E. coli</i> C 13706, C-3000 Host <i>Salmonella</i> WG-49

Source: U.S. EPA 1999a.

coarse-grained or sandy soils are poor adsorbers of microbes (Keswick and Gerba 1980, U.S. EPA 1999a). Fine-textured clay soils or soils with abundant colloidal organic material are very adsorbent because their negatively charged surfaces and large surface area per-unit volume increase the number of potential adsorption sites for microbial contaminants. As a result, clay soils slow the migration but can enhance the survival of certain microbes (Bitton and others 1986, Keswick and Gerba 1980). In contrast, the absorption of viruses to organic soils or in environments with high concentrations of dissolved organic matter or organic acids is relatively poor, probably because of competition for adsorption sites. The presence of humic and fulvic acids may reduce virus infectivity.

The acidity and ionic strength of liquids percolating past adsorbed microbes can influence their sorption and desorption. Moreover, a reduction in the ionic strength of pore water weakens the virus-soil adsorption forces and increases their entrainment and concentrations in percolating water (Bitton and others 1986). Therefore, natural rainwater with its extremely low ionic strength can mobilize and transport viruses that have sorbed to the upper layers of the soil.

Fecal contamination of surface and ground water can occur by several pathways (table 2.1). The concentration of microbes in surface runoff is generally higher in warmer months and higher in runoff from grazed rather than ungrazed land (Edwards and others 1997). Lawns and residential streets are important sources of fecal coliforms from domestic animals (Bannerman and others 1993). Leaking sewer lines and failed septic systems are also common sources (U.S. EPA 1999a), and water distribution systems can harbor bacterial or fecal contamination. This contamination enters distribution systems when controls fail or when negative pressure in a leaking pipe allows contaminants to infiltrate.

Storage in reservoirs can increase or decrease the microbial content of surface water. Sedimentation of particles with adsorbed microbes and the germicidal action of sunlight can lower microbial content (Cox 1964). However, these effects are spatially and seasonally variable and are influenced by microclimate and the morphology and chemistry of a water body (James and Havens 1996). Eutrophic conditions that reduce DO concentrations or produce toxic blue-green algae blooms may decrease water quality (see Hebgen Lake case in chapter 5).

Table 2.8—Factors influencing virus transport and fate in the subsurface

Factor	Influence on fate of virus	Influence on transport
Light	Minor factor in virus inactivation, effective only at the soil's surface	Unknown
Temperature	Viruses survive in soil and water longer at lower temperatures.	Unknown
Hydrogeologic conditions and well pumping rate	A short ground water time of travel indicates that viruses may be transported to water supply wells before dying off or becoming inactivated. High pumping rates decrease ground water travel times.	Relatively slow flow reduces the rate of virus migration while conduit, fracture flow, or rapid flow in coarse-grained, porous media enhances transport.
Soil properties; iron-oxide coatings on soil or aquifer grains	Effects on survival are probably related to the degree of virus adsorption. Iron oxides probably increase inactivation.	High degree of virus retention by the clay fraction of soil; iron coatings may be especially efficient in providing an attractive surface for virus attachment.
pH	Most enteric viruses are stable between a pH range of 3 to 9. Survival may be prolonged at near-neutral pH values.	Generally, low pH favors virus adsorption and high pH results in virus desorption from particles.
Inorganic ions/salt species and concentration	Some viruses are protected from inactivation by certain cations; the reverse is also true.	Generally, increasing the concentration of ionic salts and increasing cation valencies enhance virus adsorption.
Organic matter	Presence of organic matter may protect viruses from inactivation; others have found that it may reversibly retard virus infectivity.	Soluble organic matter competes with viruses for adsorption sites on soil particles.
Virus type	Different virus types vary in their susceptibility to inactivation by physical, chemical, and biological factors.	Virus adsorption to soils is probably related to physicochemical differences in virus capsid surfaces.
Microbial activity	Some viruses are inactivated more readily in the presence of certain microorganisms; however, adsorption to the surface of bacteria can be protective.	Unknown
Iron content in shallow soil or aerobic aquifers	May increase virus attachment and inactivation	Iron-oxidizing bacteria may form a biomass layer that filters out viruses. Heavy precipitation events may cause the ionic strength of the water to decline and the biofilms to release the filtered organisms.
Soil moisture content	Influences inactivation and adsorption to particle surfaces; survival may increase in unsaturated conditions.	Increased saturation promotes desorption of viruses from particle surfaces and migration in ground water.

Source: U.S. EPA 1999a.

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Chapter 3

Watershed Processes—Fluxes of Water, Dissolved Constituents, and Sediment

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Introduction

The quantity and quality of drinking water coming from a watershed depend on processes involving surface water, ground water, biogeochemistry, biota, atmospheric deposition, and sedimentation. Quality of drinking water supplies, therefore, hinges on understanding of the routing of water, its dissolved constituents, and its entrained sediments through watersheds and ground water systems (Dunne and Leopold 1978). Fluxes and storages of water, chemical constituents, and sediment can be described in terms of the average properties and variability of the cycling systems involved. Natural forces, such as floods, can cause changes in water flows and water quality. At times these changes can overwhelm effects of land-use practices. It is important, therefore, to understand natural variability of water systems in order to have realistic expectations about the quantity and quality of water yields from specific watersheds. Furthermore, natural variation in streamflow and water quality is integral to the health of aquatic ecosystems (Poff and others 1997) and, thus, must be considered when attempting to balance consumptive uses with environmental protection.

In this chapter, we provide general background information on hydrology, dissolved constituents, atmospheric deposition, sedimentation, nitrogen impacts of surface and ground water, cumulative watershed effects of land uses, management and policy implications, research needs, and, finally, key points. This information is useful in assessing drinking water issues.

The Integrated Hydrologic System

Ecosystems are energy-processing units that are continually cycling and being regulated by essential nutrients and water. Some cycles, like the hydrologic cycle, are global and, thus, involve transport over great distances. Other cycles occur locally among biotic elements, forest litter, and soil. In most forests, large pools of tightly bound, relatively unavailable nutrients are linked with small pools of available nutrients that are rapidly cycled through the ecosystem.

The circulation of water through the hydrologic cycle is the largest movement of a chemical substance at the surface of the earth (Schlesinger 1997). The hydrologic cycle describes the constant exchange of water among the land, sea, and atmosphere. A water budget is the balance of inflows, outflows, and changes in storage over a defined time period at a specific location. Both water cycles and budgets consider water in solid, liquid, or gaseous form and are typically viewed in a sequence from precipitation to streamflow (fig. 3.1). Since most chemicals are somewhat water soluble, the hydrologic cycle strongly influences nutrient cycling, weathering, chemical and sediment transport, and water quality. Furthermore, water plays vital roles in mobilization and transport of sediment downslopes and through stream networks.

The basic equation that describes a hydrologic budget is

$$Q = P - I - T - E - G - W + R + /- S,$$

where

Q = streamflow,
P = precipitation,
I = interception,
T = transpiration,
E = evaporation,
G = ground water recharge,
W = water withdrawals for consumptive use,
R = return flow from outside sources, and
S = change in storage over measurement period.

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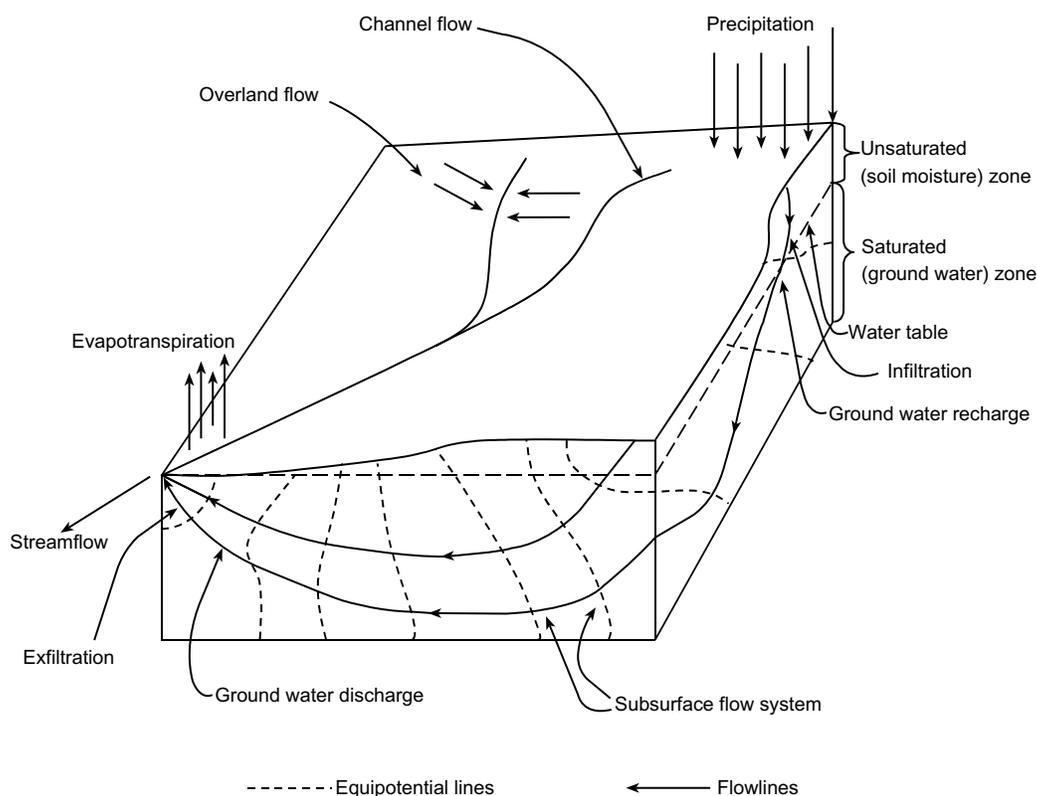


Figure 3.1—Hydrologic cycle within a watershed.

Units in the equation are expressed in terms of volume or depth per-unit time (million gallons or liters per day or inches or centimeters per year). Like all budgets, the magnitudes of the components depend on the spatial and temporal scale considered, and their evaluation involves errors due to measurement and interpretation. Water budgets for small watersheds typically have combined measurement errors of 20 percent or greater (Winter 1981). Likewise, municipal water that is not accounted for, that is, the difference between the amount of water produced by treatment plants and the amount legitimately consumed, is commonly 20 to 40 percent of total water treatment plant production (World Bank 1993). This difference includes leakage from the storage and distribution system and illegitimate uses.

For ease of communication, each component is discussed below. However, it is important to recognize that components are tightly coupled; that is, ground water and surface water systems, as well as atmospheric and biotic influences on them, must be viewed and managed as a single system and a single resource (Winter and others 1998). Many of our past and present water-use practices and policies have

ignored these linkages. Modification in one part of the system is likely to significantly affect other parts.

Precipitation and Atmospheric Deposition

Precipitation is often classified by physical form (liquid, solid, or gas), size (rain, drizzle, or mist), the responsible weather system (cyclonic, warm front, cold front, convective, or orographic, etc.), and chemistry (acidic). In general, the amount of annual precipitation varies with elevation and aspect relative to the prevailing winds.

Because the water molecule is dipolar and attracts other molecules, natural precipitation contains dissolved gases in amounts proportional to their concentrations in the atmosphere, their solubility, and ambient temperature. Uncontaminated precipitation also has low concentrations of solutes, is slightly to moderately acidic, and has a high redox potential. The equilibrium pH for nonsaline water in contact with atmospheric carbon dioxide is 5.7; and rainwater and melted snow in nonurban, nonindustrial areas typically have pH levels between 5 and 6 (Freeze and

Cherry 1979, Park 1987). In contrast, rainfall contaminated by urban or industrial inputs can frequently have a pH as low as 3 to 4. This acid rain is typically a result of nitrate (NO_3^-) and sulfate (SO_4^{2-}) that are derived from the incorporation of gaseous pollutants in raindrops (Schlesinger 1997). This increased acidity can increase the rate of weathering and release of cations from exchange sites. Consequently, the concentrations of metals in source water and the corrosion of water storage and distribution systems also increase, causing higher metal concentrations in drinking water (McDonald 1985, Park 1987). The constituents of concern are high acidity levels ($\text{pH} \leq 4.5$) in precipitation, high nitrate in soil or ground water, and the interaction of these in soil water to yield high concentrations of aluminum (Al) and lead (Pb). In addition, highly acidic water can dissolve lead in solder joints where copper pipes are used for plumbing.

Atmospheric deposition is a primary source of mercury (Hg) that can cause adverse health effects. The dangerous form, methyl mercury, is bioconcentrated in fish that must be eaten to endanger health. Methyl mercury in precipitation or surface waters usually does not occur at toxic concentrations (Nriagu and Pacyna 1988) (see chapter 2; tables 2.3, 2.4).

Some people in the Eastern United States use cisterns, shallow wells, or ponds for their water supply. Some sites in coal regions can use only cistern water sources because local ground water is extremely acidic ($\text{pH} < 4.0$) from acid mine drainage. The sites at risk are those where precipitation pH is ≤ 4.5 , and where surface soils do not contain enough bases (calcium and magnesium bicarbonates) to neutralize the precipitation acidity. Granitic bedrock, base-poor quartz sandstones, and sandy soils derived from them have low amounts of neutralizing bases. Basaltic rocks, sandstones with high amounts of calcite cement, and marine, sedimentary rocks have high amounts of neutralizing bases. In areas with acidic source water, public water supplies often adjust pH as part of the water purification treatment, but this may not occur in some small, private drinking water systems.

Areas in the United States where precipitation pH averages ≤ 4.5 are restricted to the Upper Midwest and Eastern United States (fig. 3.2). Areas with acid surface soils are in the East, Southeast, Upper Midwest, and Northwest (fig. 3.3). These soils correspond to areas where lakes and streams are acid, and, thus, shallow ground water is assumed to be acid (Church 1983).

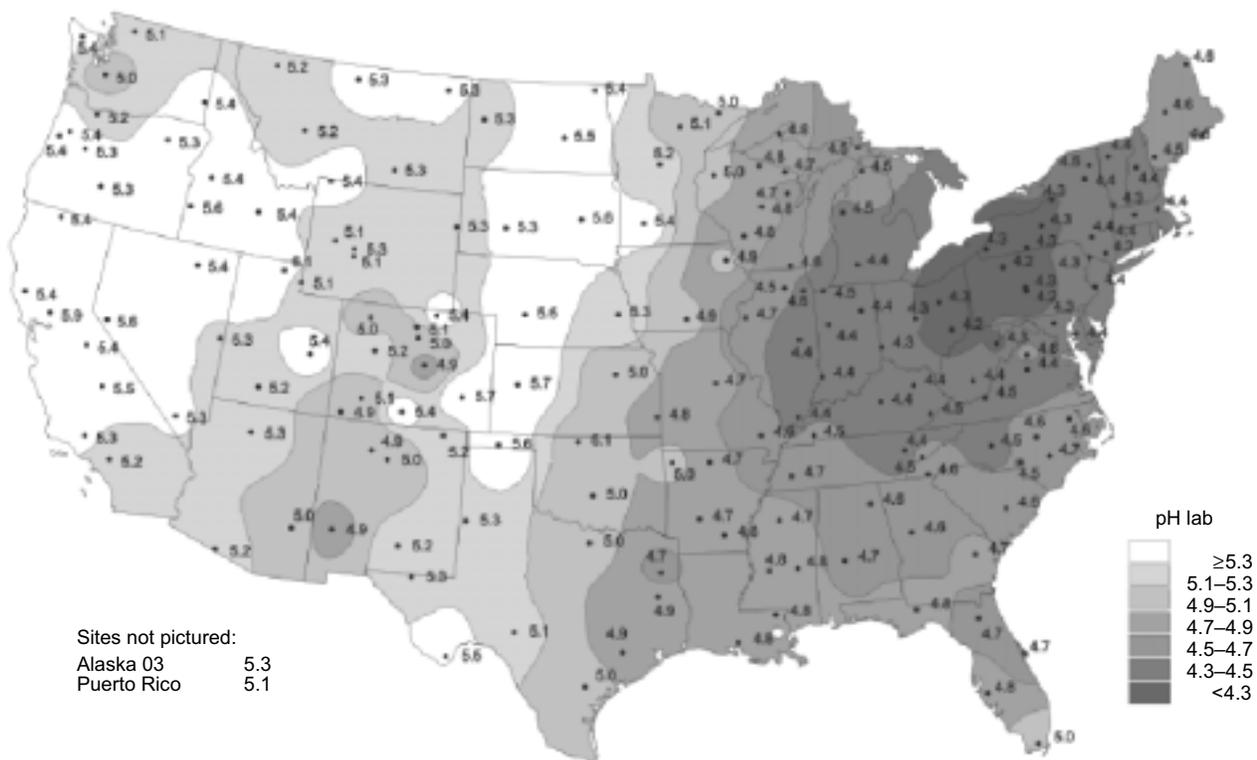


Figure 3.2—Average acidity (pH) of precipitation in the United States from 1988 through 1997 (National Atmospheric Deposition Program 1999).

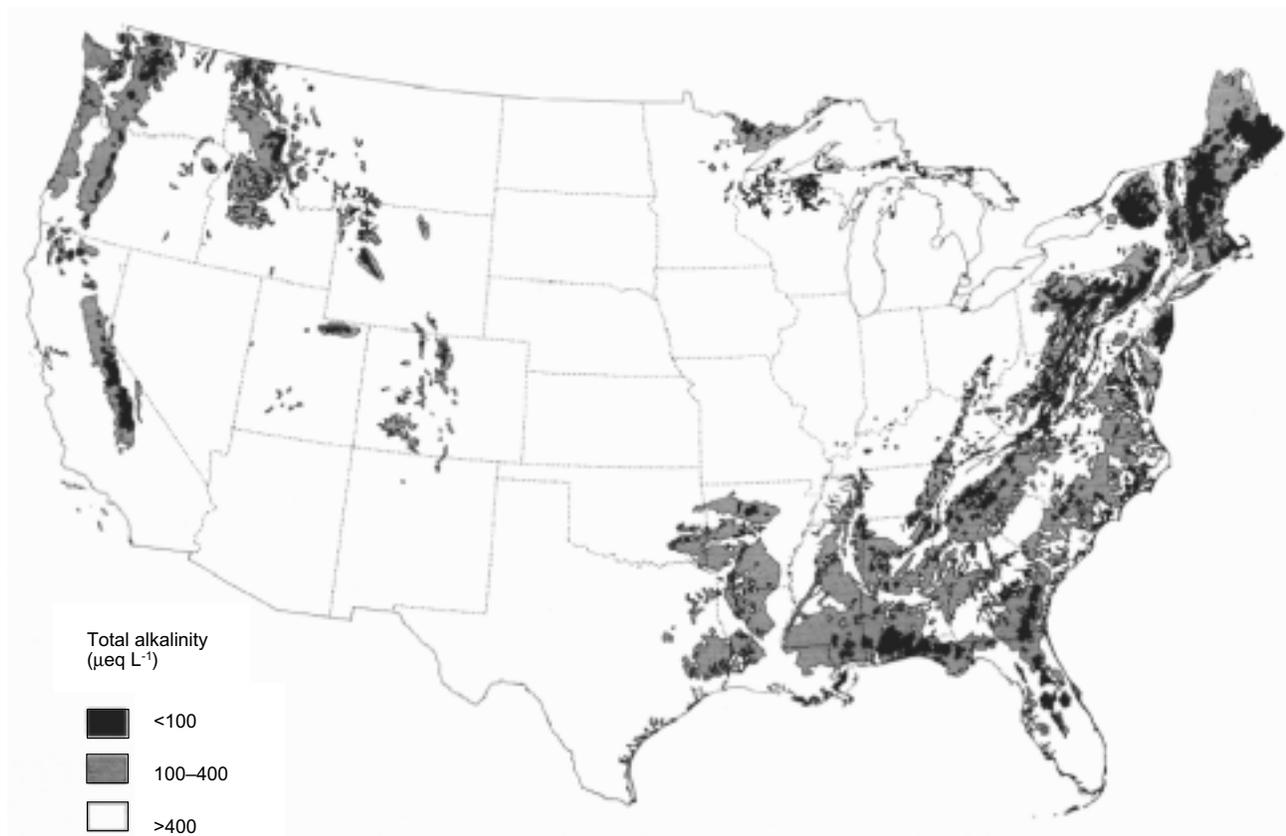


Figure 3.3—The acid buffering capacity (total alkalinity) in lake water and, by inference, the surrounding water in soils and groundwater for the United States. Where lakes have low or negative alkalinities in the black areas, surface water pH values may be <4.5. (Map prepared by J.M. Omernick, G.E. Griffith, J.T. Irish, and C.B. Johnson with the U.S. Environmental Protection Agency.)

Evaporation, Transpiration, and Evapotranspiration

Evaporation is the process of converting water from a liquid or solid state to a gaseous state. Evaporation occurs from lakes and wetlands, large rivers, soil surfaces, and accumulations of water on vegetative matter or other surfaces. Sublimation is evaporation from snow and ice surfaces. The rate of evaporation depends primarily on solar radiation, temperature, wind, and the humidity gradient above the evaporating surface. Because evaporation losses from open water can be large, efforts have been made to reduce losses from municipal water supplies by covering reservoirs or storage tanks, using underground storage, controlling aquatic growth, reducing surface area, and applying chemicals (Viessman and others 1977).

Transpiration is the process by which water is released as vapor from plants through leaves to the atmosphere and is influenced by soil moisture, the type of vegetation, vapor pressure gradients across leaf surfaces, and the same factors that affect evaporation: solar radiation, temperature,

humidity, and wind. In many cases, evaporation and transpiration are summed and reported as one process, termed evapotranspiration (ET). Unlike other pathways, ET returns water to the atmosphere without solutes and, thus, increases the concentrations of solutes in the water remaining in terrestrial or aquatic systems.

Interception, Throughfall, and Stemflow

Interception is the process whereby precipitation collects on vegetation and evaporates instead of falling directly or indirectly to the ground. Throughfall is water that may or may not contact vegetation as it passes through the vegetative canopy and eventually falls to the ground. Stemflow is water that reaches the ground flowing along the stems of vegetation. The amount of interception varies with the magnitude and intensity of rainfall, the structure and composition of the canopy, the season, and the form of precipitation (Anderson and others 1976). In general, forest vegetation intercepts more precipitation than grasslands, and conifers intercept more water than hardwoods.

Solutes in throughfall and stemflow consist of both new and recycled constituents. New inputs are chemicals and particulates that originated outside the area of interest and collect on vegetative surfaces through a process called dry deposition. Recycled constituents come from decomposition and leaching of plant tissue. In general, the cation and anion concentrations in throughfall are 2 to 100 times those of rainfall (Wenger 1994). In deciduous forests, throughfall and stemflow have the highest concentrations during the summer when the forest has the largest leaf area.

Soil Water

Once water passes through the vegetative canopy, it comes in contact with the forest litter layer and soil surface, where it either infiltrates, evaporates, temporarily ponds, or leaves the area as surface runoff. Water that infiltrates can reside in many subsurface areas (fig. 3.4) and remain below the surface for a period ranging from seconds to millennia. The rate at which water enters the soil—called the infiltration or percolation rate—is influenced by the magnitude and

intensity of rainfall, the type and extent of vegetation cover, and the temperature and condition of the surface. In general, the amount of infiltration in a watershed decreases with the amount of pavement and increases with forest cover and soil organic matter content.

Ground Water

As water flows through the upper soils and ground water system, it interacts with its surroundings and undergoes chemical changes. Typically, organic acids are produced and nutrients are immobilized in the upper layers of fresh forest litter (Schlesinger 1997). Mineralization of nitrogen (N), phosphorus (P), and sulfur (S) is usually greatest in the lower forest floor and upper mineral soil. As water passes through these layers, organic acids and other decomposition products can produce undesirable odors or taste and can increase water hardness (Freeze and Cherry 1979).

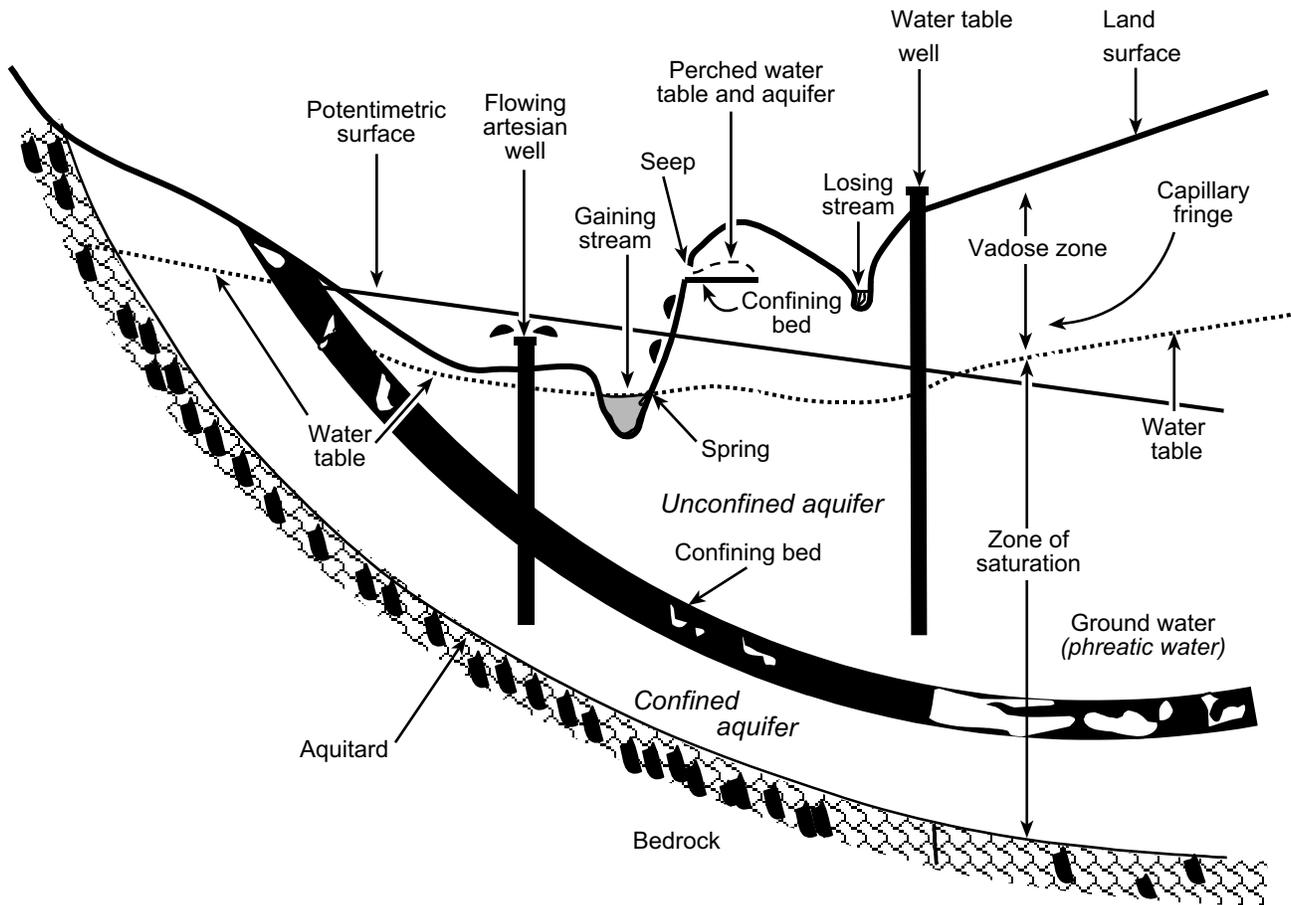


Figure 3.4—Schematic illustrating ground water terms and concepts.

Exchanges between ground water and mineral particles generally increase the concentrations of total dissolved solids (TDS), cations, calcium bicarbonate ($\text{Ca}(\text{HCO}_3)_2$), magnesium bicarbonate ($\text{Mg}(\text{HCO}_3)_2$), calcium sulfate (CaSO_4), and magnesium sulfate (MgSO_4). Microbial processes are usually responsible for methanogenesis, denitrification, sulfate (SO_4^{2-}) and ferrous iron (Fe^{+3}) reduction and the breakdown of natural and synthetic organic compounds. Denitrifying bacteria have been collected from depths of 1,000 feet [300 meters (m)] and sulfate-reducing bacteria can remove most sulfate within 50 feet (15 m) of the soil surface (Schlesinger 1997). The travel time and distance needed to remove viruses or synthetic organics vary considerably, but typically are on the order of days to years and feet to miles (meters to kilometers). Because of these exchanges and microbial processes, older and deeper ground water generally has greater hardness but fewer organic pollutants and often needs less treatment than surface water or shallow, young ground water.

Streamflow

Streamflow is broadly divided into two types, stormflow and baseflow. Water flows to streams by three processes: (1) overland flow (or surface runoff), (2) interflow (or subsurface stormflow), and (3) ground water flow (Linsley and others 1982). Overland flow involves water that travels over the ground surface to a stream channel. Interflow involves water that infiltrates into the upper soil layers and moves laterally until it enters a stream channel. In most forested watersheds, the rate at which water can infiltrate into the soil is greater than the rate of rainfall. Therefore, overland flow is relatively rare or is limited to areas with shallow, degraded soils or saturated areas in a watershed. In contrast, interflow is common, especially in areas with thin, porous soils that become saturated during storms or in areas where subsurface soil pipes or macropores have developed. These subsurface conduits can have diameters that range from fractions of an inch (centimeter) to several feet (meters). In some instances, the flow velocities within pipes are sufficient to cause *in-situ* erosion. Subsurface conduits can eventually become so large that they collapse and form incipient stream channels. Because water in these pipes is rapidly transferred to streams, the purification that commonly occurs as water slowly travels through microscopic soil pores does not take place.

The chemical characteristics of streamwater depend on its source and the flow path and transit time to the stream. In general, the concentrations of dissolved solids decrease with increasing discharge and increase with the length of the flow path and the amount of time the water has traveled across the landscape. These generalizations are especially true of

highly soluble and typically nonbiologically limiting ions—like calcium (Ca^{+2}), magnesium (Mg^{+2}), sodium (Na^{+1}), silica (SiO_2), chloride (Cl^{-1}), bicarbonate (HCO_3^{-1}), and sulfate SO_4^{2-} —associated with chemical weathering (Schlesinger 1997). They are also generally true for chemicals derived from point sources that enter streams at relatively constant rates. In contrast, the concentrations of sediment and particulate matter derived from physical detachment and chemicals derived from the flushing of the land surface or shallow subsurface tend to increase with stream discharge and the proportion of surface or storm runoff in the stream.

Once in the stream, constituents may be transported in solution, in suspension, or attached to particles. Metabolic activity in a stream depends on upstream inputs, internal (algae, aquatic plants), and external (leaves, dissolved organic carbon) sources of food and nutrients. The major processes affecting dissolved oxygen (DO) in a stream are reaeration, carbonaceous and nitrogenous deoxygenation, sediment oxygen demand, and plant photosynthesis and respiration (Marzolf and others 1994, Newbold and others 1982, Vannote and others 1980). In streams with large, standing crops of submerged aquatic plants, the uptake of carbon dioxide during photosynthesis can remove enough carbonic acid from water to increase daytime pH by several units.

From headwaters to lowlands, streams change in their morphology, water chemistry, and biotic communities (Vannote and others 1980). In general, headwater streams are shaded by terrestrial vegetation and have biotic communities that depend on leaf litter and other natural terrestrial sources of organic matter. The water in these streams also tends to have low concentrations of TDS. In downstream areas, the amount of light entering the channel, the contribution of ground water, and anthropogenic contaminants generally increase. Consequently, TDS and contaminants tend to increase and aquatic plants and algae rather than terrestrial plants become the major source of organic matter inputs to streams. As dissolved constituents are transported downstream, they are converted to organic forms and accumulated in organisms until they die and are recycled. This change between organic and inorganic forms may occur several times as nutrients “spiral” down the channel from the headwaters to the lowlands (Newbold and others 1982).

The physical and biotic changes in water quality that occur along a river can also affect the operation and cost of municipal water treatment. Moreover, because concentrations of TDS and pollutants increase downstream, water withdrawn from lower reaches of streams typically needs

more treatment than water from undisturbed, forested headwater areas. Nevertheless, because they usually have greater volumes of water and less seasonal variability in supply, lowland water intakes are often more reliable sources of water.

Water Withdrawals and Return Flow

Water withdrawal is the process of removing water from a hydrologic system and conveying it to a place for offstream use. Nonwithdrawal or instream uses include navigation, hydropower generation, recreation, and the maintenance of aquatic habitat. Return flow includes water that is added to a hydrologic system after it has been withdrawn and used or leakage from storage and distribution systems. In municipal watersheds, return flows are typically from point discharges from sewage treatment plants, irrigation systems, or industrial sources. However, in some urban areas, nonpoint discharges and conveyance losses from leaky pipes or irrigation ditches may contribute significant volumes of return flow.

The influence of return flow on water quality is a function of the quality and quantity of the return flow, the quality of the receiving waters, and the distance below the discharge point and turbulence of flow in the receiving waters. In general, a streamflow to wasteflow ratio of about 40 to 1 is needed to safely dilute most raw, untreated waste (Gupta 1995). A ratio of about 2 to 1 is needed to dilute waste from most secondary water treatment plants. Unless the effluent is disinfected, most wastewater treatment does not markedly reduce pathogens. In the United States, most effluent is disinfected with chlorine. Excessive chlorination, however, may lead to toxicity problems for aquatic organisms in receiving waters. See chapter 5 for more discussion of withdrawals and return flows on drinking water quality.

Effects of Nitrogen Deposition on Stream and Ground Water Quality

An increasing number of studies from wildland watersheds in many parts of the World demonstrate a link between chronic nitrogen (N) inputs from air pollution and nitrate levels in streamwater and ground water emanating from these watersheds (Fenn and Poth 1999, Stoddard 1994). Nitrogen saturation is the term now commonly applied to the phenomenon of ecosystems, which export high nitrogen levels as a result of available nitrogen in excess of biotic demand and of watershed nitrogen retention capacity (Aber and others 1989). The excess nitrogen is predominantly exported as nitrate in drainage waters, but gaseous losses of nitrogen from soil and in the riparian zone can also be

important. The source of the excess nitrogen is usually elevated nitrogen inputs from the atmosphere (nitrogen deposition), but nitrogen fertilizer application and nitrogen-fixing plant species, which convert free dinitrogen gas (N_2) in the atmosphere into organic forms of nitrogen in plant tissue, are other sources of excess nitrogen.

Surface water and ground water are commonly contaminated with elevated nitrate in nitrogen saturated watersheds (Berg and Verhoef 1998, Fenn and Poth 1999). However, watershed-level studies of nitrogen saturation tend to focus on nitrate concentrations in streams, which is generally easier to access than ground water. Ground water can be sampled from wells or from ground water-fed springs where they occur. In many instances, streamflow originates from springs. If the watershed is nitrogen saturated, stream nitrate may come from contaminated ground water. For example, hydrologic studies in the nitrogen-exporting Neversink River watershed in the Catskill Mountains, NY, found that during the summer low-flow period, streamflow originated from perennial springs. The springs discharged deep ground water that was recharged during the dormant season 6 to 22 months earlier when soil nitrate levels are highest (Burns and others 1998). During the summer, nitrate concentrations in these streams were higher than in shallow ground water, which was recharged during that growing season when plants take up nitrogen and because the streamwater originated from deep ground water. In the summer-dry climate of southern California, nitrate concentration in springs was an excellent indicator of watershed nitrogen status (Fenn and Poth 1999). Nitrate concentrations in springwater did not vary seasonally, suggesting that springwater or ground water may be a more useful indicator of nitrogen saturation due to its greater temporal stability compared to surface runoff.

Water from forested watersheds is commonly used to improve drinking water quality by blending it with lower quality water from other sources. Water from nitrogen-saturated watersheds has high nitrate concentrations that only exceed the Federal drinking water standard following a major disturbance, such as fire, harvesting, etc. (Riggan and others 1994). Water from nitrogen-saturated watersheds has reduced dilution power, leaving water resource managers with the need to implement much more expensive water treatment options. A high nitrate level in drinking water is an important human health concern (see chapter 2). In addition, excess nitrate exported to lakes and estuaries contributes to eutrophication of these bodies.

The geographic extent of nitrogen-saturated watersheds in North America (Fenn and others 1998, Stoddard 1994) is only partially known, largely because research on this topic

in North America is still in the relatively early stages. Nitrogen saturation cannot be predicted based solely on the amount of nitrogen deposition. The rate of total nitrogen deposition is a factor contributing to nitrogen saturation. However, for most areas, only nitrogen deposition in rain and snow have been measured. Dry deposition of nitrogen in gaseous or particulate forms has been measured in relatively few places because measuring dry deposition is still in the experimental stage of development. In areas of high air pollution, especially in dry climates, dry deposition may be a large contributor of nitrogen to forests.

The percentage of forest land cover in North America exhibiting severe symptoms of nitrogen saturation, such as large nitrate export losses, is relatively low. Much larger areas of forested lands exhibit moderate nitrate export, a sign that they may be vulnerable to nitrogen saturation in the future. Forest production may be enhanced in some of these areas as a result of the inadvertent atmospheric nitrogen fertilization (Fenn and others 1998). However, the problem of excess nitrogen is not trivial. Signs of nitrogen saturation have been reported in the headwater streams in forests in the Catskill Mountains (Murdoch and Stoddard 1992) and the Transverse Ranges in southern California, both of which supply drinking water to millions of inhabitants (Fenn and Poth 1999, Riggan and others 1994).

In many nitrogen-saturated watersheds, most of the nitrate leached from the ecosystem is cycled through plant litter, organic matter, and microbes prior to being exported. Although some studies suggest that nitrogen deposition above a threshold level can eventually lead to elevated nitrate loss in temperate forests (Dise and Wright 1995), there are also clear exceptions to the pattern. The relationship between nitrogen deposition and nitrate leaching is confounded by complex, nitrogen-cycling processes and the biological and physical characteristics of forested watersheds. Ecosystem controls on nitrogen processing and nitrogen loss are poorly understood at the mechanistic level. Thus, our ability to predict nitrogen losses from watersheds exposed to chronic atmospheric nitrogen deposition is limited. Plant and soil indicators of ecosystem nitrogen status are available, and they can be used to monitor and identify ecosystems for symptoms of nitrogen saturation (Fenn and Poth 1998). Suggested indicators include litter carbon-to-nitrogen ratio, foliar nitrogen-to-phosphorus or nitrogen-to-magnesium ratio, and ratios of rates of soil nitrification-to-mineralization.

Although our knowledge is incomplete of how different ecosystem types process nitrogen, certain characteristics that are known to predispose ecosystems to nitrate loss can be used to identify watersheds at risk of elevated streamwater

nitrate concentrations (Fenn and others 1998). Such factors include steep slopes and coarse-textured, shallow soils, or both that encourage rapid runoff with little opportunity for biological uptake and retention of dissolved nitrate. Mature forest stands may have large stores of organic nitrogen in the soil, the forest floor litter layers, and the old trees. Plant nitrogen demand from the soil is lower in old stands than vigorously growing younger stands with lower ecosystem nitrogen stores. Older stands, therefore, are particularly prone to nitrogen saturation, even where rates of atmospheric nitrogen deposition are low to moderate (Foster and others 1989).

Forest type may influence ecosystem susceptibility to nitrogen saturation. Recent studies suggest conifer stands are more prone to nitrogen saturation and nitrogen loss than hardwood stands (Aber and others 1995). Preliminary results indicate elevated nitrogen inputs may convert some conifer stands to deciduous forests with high nitrogen cycling rates (McNulty and others 1996).

Low soil cation capacity may predispose forests to symptoms of nitrogen saturation if other macronutrients, such as calcium or magnesium, become limiting. This can result in nutrient imbalance in some plants, disruption of plant function, forest decline, decreased nitrogen demand, and increased nitrate leaching (Durka and others 1994).

High elevation ecosystems, which include some class I wilderness areas in national forests and national parks, are especially prone to high nitrate losses, even where atmospheric nitrogen deposition is moderate. High elevation systems are often characterized by steep slopes, coarse-textured soils, exposed bedrock, and sparse vegetation with low plant nitrogen demand. Low temperatures also result in reduced plant and microbial nitrogen retention. Nitrate runoff in these systems is particularly high during high runoff periods, such as during spring snowmelt and after large storms. High elevation tundra ecosystems in the Front Range of the Colorado Rockies are nitrogen saturated with low-to-moderate nitrogen deposition rates (Williams and others 1996).

Since the norm for most watersheds of the Northern Hemisphere is nitrogen limitation rather than nitrogen excess, land managers have little experience dealing with the problem of nitrogen saturation. The causes and effects of nitrogen saturation are areas of active research. The generalizations discussed in this section are supported by many recent studies, but little if any research in North America has focused on the effectiveness of silvicultural treatments for reducing high nitrate concentrations in runoff water. Previous studies on the effects of fire, harvesting activities,

and other silvicultural treatments on nitrate runoff provide clues as to possible management options in nitrogen-saturated watersheds. Further research is needed, but it seems likely that increasing plant nitrogen and water demand by encouraging the growth of young, fast-growing deciduous forests, which are increasing in biomass, is likely to reduce nitrate runoff. Other promising strategies for reducing nitrate in runoff focus on the riparian zone.

Reducing the amount of nitrogen stored in the ecosystem is another strategy for reducing high nitrogen losses in runoff. For example, forest harvest intensity affects the amount of nitrogen left in the system and, thus, the amount of nitrogen that can leach from the watershed. Whole-tree harvesting was found to reduce the amount of nitrate in runoff compared to less intensive harvests in which slash was left in the forest after the harvest (Hendrickson and others 1989). Removing slash in harvest operations not only reduces the amount of nitrogen in the system, it also allows for more rapid regeneration of vegetation following the harvest, resulting in greater vegetative nitrogen demand and nitrogen retention. However, in Eastern North America, cation depletion in soil is thought to be a serious threat to forest sustainability and productivity in some areas (Federer and others 1989). The more intensive harvesting regimes would likely exacerbate this problem, and could only be used effectively if fertilizer is applied to replace the limiting nutrients. In fact, if a forest is growth-limited by a nutrient other than nitrogen, e.g., phosphorus, sulfur, or calcium, fertilizing with that nutrient will likely increase plant nitrogen demand and should result in lower levels of nitrate runoff (Stevens and others 1993). Nitrogen fertilization should be avoided in forests showing signs of nitrogen saturation because it is likely to exacerbate nitrate levels in runoff. There is evidence that some plant species are associated with more rapid rates of nitrate production (nitrification) and, thus, increase the risk of elevated nitrate runoff. Replanting with species with lower nitrification rates and greater nitrogen consumption or both and storage rates is another option for reducing nitrogen-saturation effects. Use of this approach, however, will require information on the nitrogen-cycling properties of the tree species under consideration.

Prescribed burning may serve a similar function to harvesting in removing organic nitrogen stores and stimulating more vigorous vegetation growth after burning. It has been proposed as a management alternative in nitrogen-saturated watersheds (Riggan and others 1994). However, nitrate concentrations may increase dramatically for a time after burning in nitrogen-saturated sites, and care must be used to avoid erosion and high sediment transport. Over the long term, however, nitrate concentrations are expected to

decrease following moderate burns. More research will be needed to determine if this approach is effective in different ecosystem types. Even if prescribed fire is shown to be effective in reducing nitrate runoff, other political, sociological, logistical, environmental, and economic restraints can sometimes make this approach difficult to implement.

In many cases, the best opportunity for reducing nitrate concentrations in runoff will likely be in the riparian zone, where nitrogen cycling is particularly dynamic. Nitrate levels can be reduced as nitrogen is taken up by riparian vegetation or by aquatic biota. The other major mechanism for reducing nitrate levels is denitrification, which is the conversion of nitrate to gaseous forms of nitrogen by a specialized group of anaerobic microorganisms. Buffer strips of riparian vegetation can be managed for maximum nitrogen retention and as a carbon source for denitrifying bacteria. Buffer strips 15 to 100 feet (5 to 30 m) wide have been shown to be highly effective in nutrient retention in surface runoff and in subsurface flow (Haycock and others 1993). Wetlands can also serve as effective nitrogen sinks, and restoration or creation of wetlands is another option for management of high nitrate runoff to coastal areas (Fleischer and others 1991). Although some active management practices have the potential to reduce impacts of nitrogen-saturation on drinking water, none have been tested in nitrogen-saturated watersheds. Active management options may be limited or inappropriate in areas such as alpine zones or wilderness.

Sediment Production and Transport

Sediment is moved from slopes to stream channels and through stream networks by a great variety of processes. Some of these processes are pervasive and persistent, such as the removal of fine-grained weathering products in suspension. Other processes operate infrequently and even catastrophically, as in the case of rapid landslides. Sediment transport through stream systems involves a variety of processes ranging from transfer of dissolved material, to movement of fine particulate material in the water column, to rolling of coarse particles along the streambed. Thus, the movement of these materials through a watershed involves a series of linked transfer processes and storage sites, such as gravel bars and floodplains. As with hydrological and biogeochemical cycling, the routing of sediment through watersheds has both long-term, average properties and very significant fluxes during extreme events.

Small sediment [<0.06 millimeters (mm), silt size] tends to move relatively rapidly through the channel system as wash load. Fine sediment is a major cause of turbidity. Larger

sediment moves as bed material load and can have long residence times. Bunté and MacDonald (1999) comprehensively reviewed the literature dealing with sediment transport distance as a function of particle size. Travel distance for suspended load (wash load plus some sand) ranges from 1.2 to 12 miles [2 to 20 kilometers (km)] per year, whereas bed load (pebbles and cobbles) travels only 0.012 to 0.3 miles (0.02 to 0.5 km) per year. In low-gradient channels, such as those found in portions of the Lake States and the Southeastern United States, residence times for sands can range from 50 to 100 years (Phillips 1993, Trimble 1999). Studies in the Western United States show sediment storage times in active stream channels ranging from 5 years to hundreds of years, depending on particle size and the type of sediment deposit (Madej and Ozaki 1996, Megahan and others 1980, Ziemer and others 1991).

Effects of floods, landslides, and chronic processes on sediment production have been widely studied and are highly relevant to evaluating the effects of forest and rangeland management on drinking water supplies. However, little work has linked results of sedimentation studies directly with the quality of drinking water. The relevant approaches to studies have included small watershed experiments (Binkley and Brown 1993, Fredriksen and others 1975, Likens and Bormann 1995, Swank and Crossley 1988), landslide inventories (Sidle and others 1985), sediment budget analyses (Reid and Dunne 1996, Swanson and others 1982), magnitude-frequency analysis (Wolman and Miller 1960), and studies directly targeting water-quality issues for particular storm events (Bates and others 1998). The latter type of study is most germane to our topic here but commonly resides in the gray literature and consulting reports. The other study approaches listed commonly present results in terms of annual or longer time scales because they typically address questions of soil loss, nutrient balances, and landscape denudation, rather than drinking water quality where problems typically develop on the time scale of individual storm events.

The capacity of watersheds with near-natural vegetation to produce sediment that reduces drinking water quality depends on soil properties, topography, climate, and vegetation conditions. Steeper slopes, of course, favor sediment production. Certain rock and soil types are prone to landslides (Sidle and others 1985) and to produce distinctive clay minerals that can cause persistent turbidity (Bates and others 1998, U.S. General Accounting Office 1998, Youngberg and others 1975). Effects of climate are complex. More precipitation favors water-driven erosion processes, but wetter conditions also favor vegetation development. Vegetation suppresses soil erosion by developing a litter layer that protects the soil from surface erosion and by developing root systems that contribute to soil strength.

Numerous studies in steep, unstable mountain land have documented that a substantial share of long-term sediment production occurs during extreme events, particularly when landslides are triggered (Swanson and others 1987). Inventories of small, rapid landslides reveal that these natural processes occur in forested terrain, as well as in areas disturbed by land management activities (Sidle and others 1985). Large, slow-moving landslides, commonly termed earthflows, are also natural, and, in some cases, they persist for millennia. Earthflow areas may be more prone to produce persistent turbidity because the montmorillonite clays that degrade water quality also cause the slow, creeping deformation characteristic of this type of landslide (Taskey and others 1978). Earthflows slowly encroach on stream channels, constricting them over periods of years. Then, floodwater undercuts the toe of the earthflow, causing streamside slides and delivering turbidity-producing sediment.

Interactions among geomorphic processes can increase the availability of sediment for many years. Major floods can deliver massive quantities of sediment to channels often by initiating landslides. For many years afterwards, suspended sediment loads may be unusually high during storms. In these cases, large amounts of sediment build up in transient storage sites along the stream, where they are mobilized by subsequent storms (Brown and Ritter 1971). Large sediment input to rivers causes channel aggradation, widening, and lateral cutting into floodplain deposits and toes of hillslopes, thus entraining stored sediment. In some cases, stored sediment and colluvium may have weathered sufficiently to contain clay minerals that cause high levels of turbidity. Thus, a major flood can affect erosion and sediment transport processes during interflood periods. These processes are more evident in areas of extreme sedimentation (Kelsey 1980), but these interactions probably operate less conspicuously in systems with lower overall rates of sediment input.

North Santiam River Case Study

Many of these complex interactions among natural processes, land use, water management, and drinking water are exemplified by the case of the city of Salem, the capital of Oregon. High levels of turbidity led Salem to temporarily suspend use of its drinking water treatment facilities that draw water from the 766-square mile (1960-km²) North Santiam River Basin during a major flood in February 1996 (Bates and others 1998). The x-ray diffraction analysis of suspended sediment in the turbid water revealed smectite clay, which forms exceedingly small particles with surface

electrical properties that permit them to remain in suspension for many weeks. Using the clay mineral analysis, it is possible to “fingerprint” large, slow-moving landslides locally termed earthflows as a major source of turbidity-producing smectite (Bates and others 1998). Thus, natural geomorphic features (earthflows) and processes (earthflow movement and flooding, eroding earthflows) play a strong role in the elevated turbidity. The degree to which current land-use practices affect earthflow movement and the floods eroding the toes of these ancient landslides are debatable. These relations between rock and soil types or both, processes of sediment delivery, and downstream water quality are common in other areas of the Cascade Range in Oregon (Taskey and others 1978, Youngberg and others 1975), and the general approach to fingerprinting causes of water-quality degradation can be applied more broadly.

In addition, a large flood-control reservoir in the middle of the North Santiam watershed, which, while reducing flood levels downstream, releases turbid water over a period of many days, thus exacerbating water-quality problems. As the city of Salem moved to increase chemical treatment of water from the North Santiam River, computer chip manufacturers expressed concern that the introduced chemicals would degrade water quality from the perspective of their uses.

Geographic and temporal variation in watershed response to floods and land use is great, as are the implications for drinking water supplies (U.S. General Accounting Office 1998). While Salem’s water treatment system was temporarily shut down due to high turbidity levels, more advanced treatment facilities, such as those of Eugene, OR, were treating water with higher turbidity (U.S. General Accounting Office 1998). However, in the generally stable watershed supplying Portland, OR, a wet winter triggered a single, natural landslide in an unmanaged area that interrupted water supplies because of high turbidity levels. Logging and roads in the watershed have been controversial, but it has been difficult to demonstrate they have degraded water quality. Watersheds with extensive areas of unstable rock and soil types are likely to have lower water quality, even if land-use activities were absent.

Natural Disturbance Processes

Natural processes that severely disturb vegetation, such as fire and extensive wind toppling of forests, can affect drinking water quality. Windstorms in the Eastern United States range in scale from localized storms (Hack and Goodlett 1960) to regional hurricanes (Foster and others 1997). The potential of wildfire to degrade drinking water

supplies is a prevalent problem in western mountain landscapes, where fire strongly affects both pulses and long-term patterns of sediment production (Swanson 1981) as well as nitrogen concentrations in runoff (Beschta 1990). Fire is also prevalent in grassland systems, but its effects on sediment production can be quite limited if the fire does not kill the vegetation or change the roughness of the ground surface (Gray and others 1998: 162) (see chapter 12). Effects of these vegetation disturbances on downstream water quality depend on the severity of disturbance to vegetation and soil, the timing of precipitation in relation to vegetation disturbance, and the propensity of the landscape and ecosystem to produce compounds that degrade water quality. However, we know of no studies directly addressing drinking water quality in response to these processes.

In many regions of the country, streams are currently transporting sediment from past land uses and management practices as well as sediment from past catastrophic events, such as wildfires, large storms, and landslides. The rate of transport depends on the size of sediment particles, gradient of streams, and streamflow. At many locations, sediment from past erosion is influencing present-day channel conditions and sediment transport. In several regions, forests were cleared for grazing, mining, and agriculture in the 1800’s and early 1900’s. For example, forests in the Piedmont region of the Southeast were cleared for agriculture and were abusively treated, causing large increases in erosion (Trimble 1969, 1974). The excessive sediment supply exceeded the transport ability of the streams. Huge volumes of sediment were deposited in the stream channels and floodplains. The severely eroded fields were eventually abandoned and reverted naturally to forest or were planted to trees and pasture under conservation programs in the mid-to-late 1900’s. The landscape stabilized and sediment yields to streams were greatly reduced. Because the runoff from the landscape carried little sediment, the streams had more energy available to transport sediment and began transporting sediment released from floodplain storage as the streams have cut downward and headward through the stored sediment (Trimble 1999). The process continues today in many river systems. For these streams, much of the sediment being transported today is from long-abandoned land uses.

Several issues and risks that may result from sediment transport from past and abandoned land uses include (1) sediment yields from a watershed may be higher than expected from present forest and grassland management.; (2) streams remobilizing stored sediment often have unstable channels and banks; and (3) stored sediment from past land uses may contain chemicals and metals that impair water quality.

Cumulative Watershed Effects

The National Environmental Policy Act (NEPA) of 1969 stipulates that cumulative effects must be considered in evaluating environmental impacts of proposed Federal projects. To implement this legislation, the Council on Environmental Quality (CEQ Guidelines, 40 CFR 1508.7, issued 23 April 1971) provided the relevant definition:

Cumulative impact is the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions . . . Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time.

A cumulative watershed impact influences or is influenced by the flow of water through a watershed (Reid 1998). Cumulative watershed effects, a phrase which has widely replaced reference to “impacts,” can be additive or synergistic and involve modification of water, sediment, nutrients, pollutants, and other watershed system components. An example of such effects would be where forest roads and timber cutting contributes to increased peak streamflows and sediment loads, leading to aggradation of downstream areas, which in turn results in lateral channel migration causing streambank and floodplain erosion, which entrains additional sediment.

Reid (1993) provides a broad and detailed summary of cumulative watershed effects of diverse land-use activities, such as grazing, roads, logging, recreation, and water extraction. She also addresses alternative approaches for assessing cumulative effects (Reid 1993, 1998). Cumulative effects can be addressed by examining the changes triggered by a particular land-use activity and how these changes interact with effects of other land uses and natural processes. Such an approach is best undertaken as a long-term study with substantial focus on mechanisms of transport, transformation, and storage within the watershed. An alternative approach is to work backward from a detected impact and attempt to interpret the chain of events and processes responsible. Each approach has strengths and weaknesses.

An important development in anticipating and hopefully minimizing cumulative watershed effects has been the watershed analysis developed for use by Federal (e.g., Regional Ecosystem Office 1995) and State (Washington Forest Practices Board 1995) agencies in the Pacific Northwest. The general objective of the Federal watershed analysis procedure is to gain an understanding of present and prospective future mechanisms affecting watershed conditions. Thus, watershed analysis provides a useful starting point for assessments of cumulative watershed effects. However, Reid (1998) asserts that neither of these

“widely used watershed analysis methods provides an adequate assessment of likely cumulative effects of planned projects.”

See appendix C for a case study on the cumulative impacts of land use on water quality in a Southern Appalachian watershed. Watershed analysis is in an early stage of development and application. It recognizes that water supply and watershed management issues must be addressed from an interdisciplinary, whole-system perspective. Although watershed analysis may provide a useful first step for assessing how multiple, simultaneous forms of management affect sources of drinking water, there is a need to develop better models to predict watershed cumulative effects.

Management and Policy Considerations

Existing information on the hydrologic cycle and sediment routing systems is generally good in terms of understanding natural controls on water flow and quality. This knowledge is based in part on a long history of water use, detection of problems, and studies to build a basis for problem solution (Anderson and others 1976, Binkley and Brown 1993). Long-term studies in experimental watersheds, including control watersheds, give a lengthening record of variability in water quality; but records seldom include the instantaneous sample concentrations that are most useful in addressing questions about drinking water quality. These and other long-term and short-term studies generally corroborate results of earlier work.

Despite our growing knowledge of natural patterns of water flows and quality, new land management practices are stretching the reliability of existing information. For example, long-term studies of water quality from small watersheds involve forest land-use treatments that are unlike those being used today. These new practices involve lower intensities of site treatments, e.g., partial cutting vs. clearcutting, lower intensities of slash fires, and longer rotations, so the treated and control watersheds in experimental watershed studies bracket the conditions created by newer treatments, providing a basis for estimating effects. Also, some new management treatments are aimed at ecosystem and watershed restoration, which may include use of fire in fire-prone systems where fire has been excluded for many years. Reintroduction of fire into forests where it has been suppressed for many decades will require evaluating short-term risks of degraded water quality against the expectation of reducing longer term risks of high-severity wildfire resulting from higher fuel loads (see chapter 12). In these cases, water-quality objectives will compete with other ecological and management objectives.

Furthermore, water-quality standards are changing for a host of reasons, not only for drinking water use, but also to meet refined ecological objectives such as protection of threatened and endangered species and to supply high-technology companies, which may not want water subjected to the standard chemical treatments for drinking water. These factors, in the evolving social and biophysical environment of drinking water issues, indicate the importance of explicitly revealing the limits of knowledge and possibly taking a risk assessment perspective in addressing emerging drinking water issues.

Because present knowledge pertains to the specific geophysical and biological conditions of study sites, we have limited ability to extrapolate findings more broadly. However, various efforts to develop regional and national spatial data bases on soil, vegetation, and topography in relation to watersheds supplying drinking water are building a basis for extrapolating findings across much larger areas (Hunsaker and others 1992). These data compilation efforts are a common factor in many bioregional assessments (Johnson and others 1999).

Important challenges are emerging in cases where competing objectives call for integrated understanding of ecological, geophysical, and human factors over large watersheds. Bases for carrying out this integration are being developed in watershed analyses conducted in a variety of contexts, including dam relicensing procedures under Federal Energy Regulatory Commission and in the Northwest Forest Plan in the Pacific Coast. These large-scale, integrative assessments, which form the basis for addressing management and policy issues around major water supplies, are substantially advancing knowledge.

Research Needs

1. Studies are needed of key hydrological, biogeochemical, and sediment transport processes that affect drinking water quality. Research needs include (a) development of reliable methods to analyze routing of these materials through watersheds; (b) determining the chemical processes associated with sediment in transport and storage; and (c) refining understanding of the roles of past and present land-use practices on water quality and sediment production, including land-use-related sediment released from long-term storage. The target processes may vary among ecological, geological, and climatic settings across the country.
2. Better understanding is needed of the overall cycling and routing of water, dissolved constituents, soil, and sediment in natural and managed watersheds. Studies to gain this understanding need to be framed so that questions such as the following can be addressed: How has management of ecosystems and water systems altered natural, historical water flow regimes, biogeochemistry, and sediment routing? How have the types and degrees of these past and prospective future alterations of these systems altered their ability to meet objectives for water supplies, ecosystem health, and other goods and services? How might climate change alter these systems?
3. Watershed-scale assessments are needed of water pollution and sediment sources operating during and after extreme events. It is important to better quantify the significance of these events by maintaining long-term studies, by monitoring the quality of source water at drinking water treatment facilities, and short-term, intensive studies targeting effects of particular storm events. While many of these assessments are conducted by management agency personnel, there is a continuing need for participation by researchers to foster development of science at this geographic scale and scope of interdisciplinary work.
4. There is need for integration of information from specific watershed studies to broad-scale management applications. This sort of regional analysis is occurring in a variety of management and research sectors on topics relevant to drinking water quality. Relevant tools, such as Geographic Information Systems, analytical approaches, and data bases are available.
5. Good records of raw and treated water at treatment facilities would provide researchers and others with much improved data bases for evaluating long-term trends in water quality from watersheds used as drinking water sources. Existing records should be examined for trends in water quality. Though not a research need itself, this is an important step in ultimately furthering research into causes and cures of water-quality problems.
6. Management options for controlling streamwater nitrate levels need to be tested for efficacy. Examples of options include tree harvesting; planting more rapidly growing and nitrogen demanding species; thinning, or other vegetation management approaches; prescribed burning; fertilizer application; and vegetation buffer strips in the riparian zone. More research is needed on vegetation type or species' effects on nitrification, since nitrate production rates are key in controlling nitrate losses. Information is also needed on tree species with the capacity to consume and store high levels of nitrogen in nitrogen-saturated watersheds. Such species can be favored, thus increasing site nitrogen retention and reducing export. Greater understanding is needed of the mechanisms and capacities for nitrogen retention in various soils and

ecosystems (Fenn and others 1998). Key indicators of ecosystem nitrogen status need to be tested and implemented in monitoring networks in order to more fully identify sites impacted by excess available nitrogen in the ecosystem.

Key Points

1. The hydrologic cycle is highly coupled, so modifications of one part of the system are likely to affect other parts that may be far removed in time and space. It is important to recognize the close coupling of surface water and ground water systems and resources—failure to do so in many past and present practices and policies has created difficult problems in water allocation and environmental protection.
2. Sediment production, transport, and storage should be viewed as a complex system in which modification of one part will affect other parts. On steep land, extreme events commonly have profound, long-lasting effects on sediment routing. Sediment impacts on drinking water may not be strictly associated with present land management. Impacts may be partly attributed to land uses and events that occurred many years previously.
3. For significant Federal projects, NEPA requires analysis of the cumulative watershed effects, the aggregate consequences of multiple land-use activities within a watershed. Watershed effects can be addressed through several complementary approaches. Watershed analysis can provide broad, historical context for evaluating potential cumulative effects of proposed land-use activities. Thoughtful reviews of the issue (Reid 1993, 1998) describe prospects and pitfalls in addressing cumulative watershed effects.
4. Watersheds in areas influenced by high atmospheric nitrogen pollution from high population urban zones, industrial areas, or in areas of mixed forest and intensive irrigated and nitrogen-fertilized agricultural areas are at risk of degraded water quality from nitrate concentrations in surface and subsurface runoff. Equally important risk factors include steep slopes and coarse-textured, shallow soils; mature forests or vegetation with low-nitrogen demand; high accumulation of nitrogen in organic matter; rapid nitrogen cycling rates in soil and vegetation; and the abundance of vegetation with high nitrogen fixation rates, e.g., alder (*Alnus* spp.).
5. Management strategies for nitrogen-saturated watersheds have not been adequately tested, but ecological principles and past studies of nitrate runoff responses to silvicultural treatments suggest reasonable strategies for reducing

nitrate runoff. The basic strategies include: (1) increasing plant nitrogen demand, (2) reducing the amount of nitrogen in the ecosystem, or (3) enhancing gaseous losses of nitrogen (denitrification)—usually from the riparian zone. These objectives may be accomplished by: (1) stimulating forest production through thinning, planting, harvest and regeneration, and fertilizing with limiting nutrients other than nitrogen; (2) removing nitrogen through prescribed burning and whole-tree harvesting; and (3) discouraging transport by maintaining effective vegetation buffer strips in the riparian zone. Field studies are needed to test the effectiveness of these approaches in a variety of ecosystem types and conditions. Most management options to mitigate nitrogen-saturation effects are probably not applicable in wilderness areas.

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Chapter 4

Economic Issues for Watersheds Supplying Drinking Water

Thomas C. Brown¹

Introduction

No other resource serves as many purposes as water. It is widely used in industry, in electric energy production, in farming and ranching, and, of course, by households for drinking, washing, and gardening. Water is essential to the health of ecological systems, supports numerous forms of recreation, and provides important amenity values. In addition, water is valuable in flushing and treating wastes, both from contained sites such as industrial plants, commercial establishments, and houses, and from land areas such as lawns, farms, and forests. Unfortunately, the processing of wastes often leaves water unsuitable for other uses without restoration of purity.

Water is essential to the viability of forests, farms, pastures, and other land areas, but, as it runs off, water carries soil from the land. Excess soil reaching streams impairs fish habitat, accumulates in reservoirs and other water management facilities, and increases costs of water treatment. In addition, pesticides, nutrients, and other contaminants attached to soil particles often leave the site.

Water supply and water quality are thus integrally linked. Most water users—whether they be boaters, farmers, industries, or households—are affected by the water's quality and in turn affect quality of the water that others use. These interdependencies make both water treatment and watershed management essential.

Sources of water pollution are usually grouped into point and nonpoint categories. Point sources, which emit from pipes or canals, include municipal wastewater treatment plants and industrial facilities. Nonpoint sources, which are diffuse and difficult to monitor, include runoff from farms, pastures, forests, cities, and highways, as well as rural septic systems and landfills. Watershed management is, in large part, the management of nonpoint sources of water pollution.

Nonpoint sources have long been recognized as the primary causes of some types of water pollution. For example, Gianessi and Peskin (1981) estimated that in the 1970's, 98 percent of the total suspended solids, over 85 percent of the phosphorus and nitrogen, and 57 percent of the 5-day biochemical oxygen demand in U.S. waters were attributable to nonpoint sources. For 1986, the U.S. Environmental Protection Agency (EPA) (1987) reported that nonpoint-source pollution was the cause of 65 percent of the water-quality-impaired stream miles and 76 percent of the impaired lake acres. The most recent EPA water-quality inventory, for 1996, reports a similar finding and shows that although agriculture is by far the largest nonpoint source of water pollution in the United States, forestry and other activities are important sources in some areas (U.S. EPA 1998).

Since the 1972 Clean Water Act was passed, some progress has been made in improving the Nation's water quality. For example, Lettenmaier and others (1991) examined trends from 1978 to 1987 at 403 stations in the U.S. Geological Survey's National Stream Quality Accounting Network and found significantly more stations with decreases than increases in pathogens, oxygen deficit, phosphorus, and some heavy metals. However, increases outnumbered decreases for total nitrogen, and suspended sediment had remained largely unchanged. In general, the successes are associated with point-source controls and the lack of success with nonpoint sources. Such findings suggest that the Nation's water-quality goals will not be met without increased emphasis on nonpoint-source pollution.

The provision of high-quality drinking water is affected by a host of natural events and human activities occurring on watersheds. The natural events include extreme precipitation events, forest fire, landslides, and transmission of pathogens by wild animals, e.g., *Giardia* spp. The human activities include mining, agricultural tillage, industrial production, timber harvest, livestock grazing, automobile use, road construction and maintenance (including deicing), and use of fertilizers and pesticides (whether in agriculture, forestry, range management, or by homeowners). The interactions among these factors, and the unpredictable nature of some factors, make water-quality protection a challenging task.

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The costs of water-quality control in the United States are substantial and rising. In 1985, households obtaining their water from municipal systems spent 0.6 percent of their income for water, plus an additional 0.4 percent for wastewater treatment (Singh and others 1988). These costs were expected to increase by about 30 percent in response to stricter standards implemented since the late 1980's. A recent EPA survey indicates that community water systems in the United States will need to invest \$138 billion over the next 20 years (Hertzler and Davies 1997). Additional expenditures will be necessary by industry, agriculture, and other sectors to protect water quality. These costs highlight the importance of considering the economics of water quality.

Perhaps the most fundamental economic question regarding drinking water quality is whether the benefits of drinking water standards exceed the costs. The benefits consist of averted losses of two general kinds. First, a water-quality standard can avert losses from drinking unclean water, including human health losses and associated health care costs. Second, where meeting the standard involves controlling upstream sources of pollution, the benefits also include averted losses between the pollution source and the drinking water diversion, including fish population losses, costs of removing sediment from canals and reservoirs, and decreases in recreation quality and use. The costs to be compared with such benefits include those at water treatment plants or by rural households that must treat their own water, and costs of controlling pollution emissions upstream of the drinking water diversion. Potential upstream pollution control costs include, for example, crop losses from decreased pesticide use; costs of controlling erosion from fields, forests, and roads; reduced beef production associated with fencing cattle out of riparian areas; and costs at upstream wastewater treatment plants.²

Despite serious efforts to estimate the benefits of drinking water standards and other water-quality controls (Freeman 1982, 1993), the estimates remain rough. Because of imprecise benefit estimates and reluctance to compromise on the safety of public drinking water, drinking water standards are often set without definitive economic analysis.

² For examples of such costs, see Easter (1993) on effects of reduced herbicide use; Chang and others (1994), Lyon and Farrow (1995), and Young and others (1991) on agricultural erosion costs; and Binkley and Brown (1993b) on erosion control costs in forestry.

³ Another economic issue, which under conditions of full employment is largely a matter of equity or distribution, is the economic impact of pollution in terms of jobs and income. These economic impacts are not discussed herein.

⁴ Several excellent books cover the topics summarized here, including Freeman and others (1973), Kneese and Bower (1968), and Tietenberg (1988).

Although benefit-cost comparison of drinking water standards remains an important issue, a more limited—though still challenging—role for economics is perhaps of more immediate relevance. That role and the focus of this chapter is helping to determine how the standards, once set, should be met.

To avoid waste of resources, standards should be met efficiently. A drinking water standard may be met solely by treating existing water prior to use, or by a combination of water treatment at points of use and pollution control upstream where the water-quality problems originate. Because pollution may occur at various points in the watershed, corrective action may involve many different costs. And because the costs of alternative actions can differ considerably, opportunities for inefficiencies (or conversely for cost savings) abound.

A related economic issue is the equity of options for implementing the efficient cost allocation.³ Expecting each actor to bear the cost of any change required to minimize the total cost of reaching the downstream water-quality standard may unfairly allocate the costs. If so, options for cost sharing, including the use of economic subsidies, should be explored. These two issues, efficiency and equity, are addressed below.⁴

Cost Minimization

Concerns about drinking water quality involve a relation between upstream emitters of a pollutant and downstream receptors who must treat the water before it can be safely used. An emitter is any pollution source, such as a forest area, a farm, or an urban wastewater treatment plant. A receptor is any drinking water provider or rural domestic user not served by a water provider.

The goal of a drinking water provider at a given use point j is to reduce the concentration of a pollutant in water delivered to users (X_d) to a level at or below the standard (X_s):

$$X_d j \leq X_s \quad (1)$$

For a water provider, achieving the desired water quality is a function of the concentration of a pollutant at the reception point (X_r) and of the reduction in that concentration by treatment (T) before the water is used:

$$X_d j = X_r j - T_j \quad (2)$$

The receptor must react to Xr , increasing the level of treatment to compensate for an increase in Xr .

Pollutant concentrations at the reception point are the result of many upstream management actions and natural events. For example, in figure 4.1, the city’s water treatment plant receives pollution from the forest, the recreation area, the upstream town’s wastewater treatment plant and storm runoff, septic systems of rural households, and farms. Pollutants from land areas such as forests and farms may result from both natural (sometimes called background) and management-caused emissions. In addition, upstream consumptive use, such as by farms, towns, and transbasin diversions, can increase the concentration of pollutants reaching the receptor, and natural processing of pollutants occurring in the stream or the adjacent alluvium decreases the concentration.

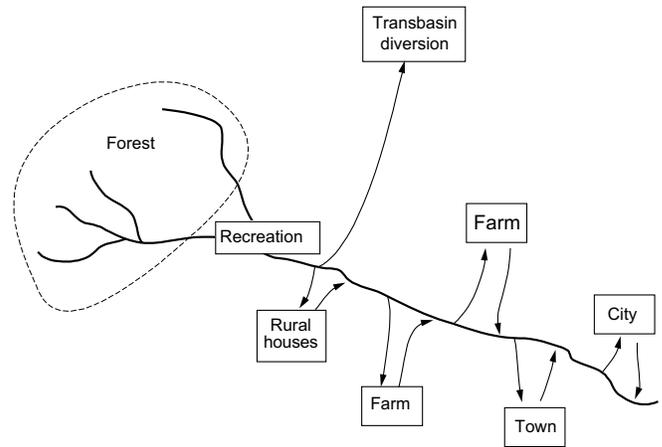


Figure 4.1—Hypothetical river basin.

Therefore, the concentration of a pollutant at reception point j (Xr_j) is a function of the emissions (e) of each upstream source (i), the transfer coefficients between each source and the receptor (α_{ij}), and the amount of streamflow at the reception point (Q_j):

$$Xr_j = \sum_i e_i \alpha_{ij} / Q_j \quad (3)$$

The streamflow amount is equal to the natural, i.e., virgin, flow minus upstream consumptive use resulting from each upstream diversion. The transfer coefficient α varies from 0 to 1 and reflects the water treatment that naturally occurs between the emission and the receptor, plus any removal of pollutants by diversions. For degradable pollutants, naturally occurring treatment increases (causing α to decrease) with distance, all else equal; for nondegradable pollutants α approaches 1. Removal of pollutants by diversions is most common with transbasin diversions; other diversions may temporarily remove some pollutants, but they often eventually return to the stream with return flows.

If the water body is a lake rather than a stream, equations (1) through (3) apply, but Q is storage rather than flow; all receptors on the lake are potentially affected by all emitters, and α for a given pollutant will not differ among emissions to the extent that mixing occurs.

The economic task is to determine the most cost-effective way to reach the goal characterized in equation (1). Pollution can be controlled at its source or removed at the point of reception and can be lessened by dilution. Hence, upstream emitters, upstream consumptive users, and downstream receptors are all candidates for actions to help meet the drinking water standard.⁵ Each actor has a cost of reducing the concentration of a pollutant to the required level. Ideally, from an efficiency point of view, control efforts would occur at the most cost-effective points.

The cost at upstream pollution source i (Ce_{ij}) depends on the reduction in profit or benefit caused by reducing the concentration of the emission that reaches receptor j . The cost at upstream consumptive use point k (Cu_{kj}) is the reduction in profit or benefit caused by reducing consumptive use so that more water reaches receptor j , thereby reducing the concentration of pollutants.⁶ The cost at the point of reception (Ct_j) is the cost of water treatment prior to use. The objective for use point j is to minimize the total cost of meeting the standard (C_j) where:

$$C_j = \sum_i Ce_{ij} + \sum_k Cu_{kj} + Ct_j \quad (4)$$

⁵ Emitters may also be diverters, and, therefore, potentially consumptive users, e.g., agricultural irrigators and cities, but not all emitters are diverters, e.g., forests, and not all diverters are emitters, e.g., transbasin water diversions.

⁶ The cost for an upstream consumptive user is more complicated than expressed in equation (4) when the water pollutants removed from the stream with the diversion do not all return to the stream in return flow. In this case, the reduction-in-treatment cost at the downstream drinking water treatment plant caused by the upstream removal of pollutants with the diversion must be subtracted from the increase in cost at the treatment plant caused by decreases in streamflow that occurs with the consumptive use of the diversion. Obviously, when the diverted pollutants do not return to the stream, the more polluted is the diverted water, the lesser is the cost imposed by the consumptive use on the downstream drinking water treatment plant.

The costs of each entity are a function of prices of inputs and outputs and of the entities' production functions and how their production actions affect Xd_i . Of course, if a watershed has more than one reception point, the overall cost efficiency goal is to minimize the sum of the various C_j .

Opportunities for Cost Savings

To find opportunities for cost savings, we must understand the costs of emitters, consumptive users, and receptors. In the short run, most of the cost of a water treatment physical plant is fixed, and only variable costs (for labor, materials, and supplies) change with changes in the concentration of pollutants entering the plant or with the volume of water treated. Similarly, in the short run, emitters' and consumptive users' facilities or equipment, such as timber harvest machinery, road designs, livestock fences, agricultural irrigation machinery, homeowners' septic systems, and canal sizes, are fixed. However, in the long run, fixed costs change to permit expansion of existing treatment facilities, introduction of new machinery, fencing, canal lining, etc. Thus, flexibility to adapt to changing levels of pollutant concentration, changing drinking water demands, or changing water-quality standards is much greater in the long run.

⁷ The appropriate marginal cost curve for an emitter takes account of the natural assimilative capacity of the environment for the pollutant at issue [α in equation (3)]; it depicts the marginal cost of reducing the pollutant load at the point of reception (Xr), not at the point of emission. If two emitters yield identical amounts of pollution but have different transfer coefficients, their effective marginal cost curves from the standpoint of meeting the drinking water-quality standard, are different.

⁸ Marginal cost curves are typically drawn with movement to the right along the horizontal axis indicating increasing producer effort, so that the marginal cost curve has a positive slope. Because the horizontal axis in figure 4.2 is concentration of a pollutant, producer effort increases to the left and, thus, the marginal cost curve has a negative slope.

⁹ The marginal cost curve of a consumptive user who is not an emitter, such as a transbasin diversion of pristine water, is likely to be similar in shape to that of the emitter shown in figure 4.2. That is, initial reductions in diversion are likely to be inexpensive, especially where water use is subsidized, as is much irrigation in the West. However, further reductions will only be possible at increasing costs.

¹⁰ This curve assumes a given volume of water treated to the concentration level indicated on the horizontal axis. The entire curve shifts up as water volume increases. A treatment plant's marginal cost curve could also be expressed as a function of volume of water treated assuming a constant level of treatment, i.e., a constant level of concentration reduction, per unit of water volume. The marginal cost curve in this case would have a positive slope, and would shift vertically with changes in the treatment level. A three-dimensional graph could, of course, show marginal cost as a function of both volume of water treated and treatment level.

¹¹ Moore and McCarl (1987) provide data for plotting a water treatment plant marginal cost curve. They estimated the marginal costs of removing sediment at a municipal water treatment plant in Corvallis, OR. The principal costs modeled were for alum, lime, and sediment disposal. Over a wide range in sediment concentration, marginal cost increased only slightly as sediment concentration decreased, but as the concentration approached zero the marginal cost abruptly increased.

Short-Run Costs

Marginal cost curves, showing the change in cost with a change in some measure of output, can be estimated for the short or long run. Consider first the short-run cost curve of an upstream pollution source such as a forest road, expressed as a function of pollutant concentration (fig. 4.2).⁷ If no effort is made to control emissions (in this case, sediments), the concentration of the pollutant reaching a water-use reception point is Xr' and, of course, the emitter's marginal cost of control is \$0. Initial reductions in the concentration of the pollutant reaching the reception point are likely to be relatively inexpensive, perhaps brought about by cleaning culverts and drainage ditches. However, further reductions in the pollutant concentration are likely to be progressively more expensive, as indicated by the increasing emitter marginal cost in figure 4.2.⁸ Reducing the concentration to zero may be quite expensive, and could require closing the road altogether.⁹

Now consider the short-run marginal cost curve of a downstream water treatment plant. This curve (the receptor treatment cost curve in figure 4.2)¹⁰ also is likely to rise as the pollutant concentration level is lowered because even lower concentrations are more and more costly to achieve.¹¹ However, the receptor's marginal cost is unlikely to drop to zero at a high level of concentration, as does the emitter's cost curve because of the need to maintain the labor and

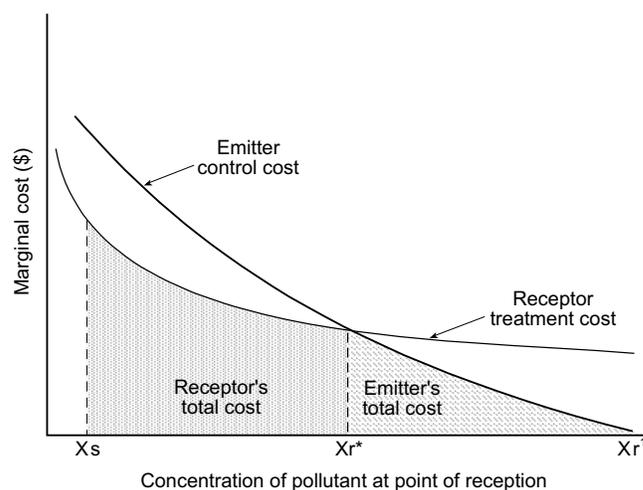


Figure 4.2—Efficient allocation of cost of meeting drinking water quality standard, with one emitter and one receptor.

materials necessary to meet a water-quality threat at all times. Even if the water-quality standard was set as low as X_r' , the provider would still need to maintain the daily capability of handling water withdrawals with pollutant concentrations that exceeded X_r' . Thus, as shown in figure 4.2, the receptor's marginal cost curve flattens out to the right but remains above the horizontal axis.

Assuming a single emitter and single receptor represented by the two short-run marginal cost curves of figure 4.2, and a water-quality goal no greater than X_s , the efficient allocation of treatment and control costs is indicated by the intersection of the two cost curves at a concentration of X_r^* . To the right of X_r^* , the emitter's marginal cost is lower than the receptor's, whereas to the left the reverse is true. Requiring the emitter to reduce the concentration at the point of reception below X_r^* costs the emitter more than it saves the receptor, and not requiring the emitter to reduce the concentration to at least X_r^* costs the receptor more than it saves the emitter.

The sum of the total costs, equivalent to C_j in equation (4), is minimized by finding the equimarginal point along the relevant marginal cost curves. Total cost is equal to the area below the relevant marginal cost curve. Assuming an efficient distribution of costs as in figure 4.2, the emitter's total cost is represented by the area below the emitter's marginal cost curve to the right of X_r^* , and the receptor's total cost is the area below the receptor's marginal cost curve to the left of X_r^* and right of X_s .

Although the receptor's marginal cost curve will always be above the emitter's at concentration level X_r' (fig. 4.2), the emitter's marginal cost curve will not necessarily rise above the receptor's as the concentration level is reduced. If the emitter's marginal cost curve remains below the receptor's curve at all concentration levels, costs are minimized by focusing all pollution control efforts on the emission source.

The precise placement of the emitter's and receptor's cost curves may be difficult to determine. And the marginal costs of the two entities could be quite similar over some range in concentration, further complicating determination of the equimarginal point. However, in some cases the opportunities for cost savings will be obvious; it is these cases where cost control efforts should initially focus. For example, consider costs of phosphorus reduction for agriculture versus municipal treatment plants. Schleich and others

(1996) report average costs to reduce a kilogram of phosphorus of \$26 using onsite pollution control practices in agriculture and \$169 at municipal treatment plants (1990 dollars).

Long-Run Costs

Often, cost minimization involves long-run decisions. Long-run cost curves of water treatment plants depict how costs change as plant capacity increases to handle a given pollutant. Most such curves have focused on changing water volumes; they typically show economies of scale, with considerable decreases in average costs as plant size increases, along with decreasing or relatively constant long-run marginal cost curves. For example, figure 4.3 shows construction cost for a pressure filtration plant as estimated by Gummerman and others (1978), expressed in 1978 dollars.¹²

When comparing treatment plant costs with costs of controlling pollution at its source, the most relevant issue is pollutant concentration rather than water volume. The relevant long-run marginal cost at the treatment plant may be the cost of adding or altering, not simply expanding, a treatment capability to deal with increased pollution concentrations (rising X_r) or tightening of water-quality standards (lowering of X_s). For example, the oocysts of the protozoan *Cryptosporidium* spp. are not inactivated by chlorine at dosages that are feasible in drinking water treatment. If these oocysts must be removed at a treatment plant that has relied on chlorine to control pathogens, new processes, such as filtration or use of ozone, will be necessary. In such a case, the long-run marginal cost curve as a function of concentration in the received water rises abruptly at a concentration equal to the water-quality goal (X_s) of the drinking water provider, as in figure 4.4. As

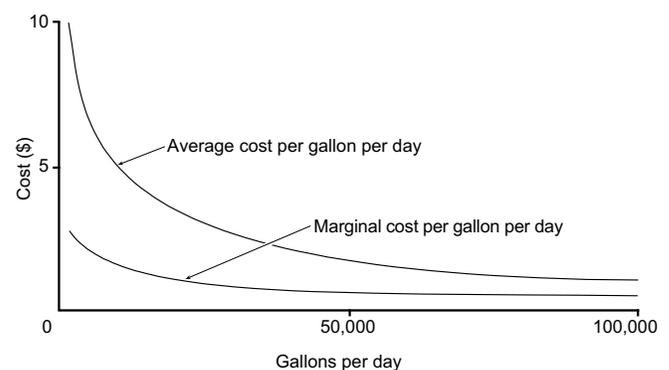


Figure 4.3—Construction cost of pressure filtration plant with an infiltration rate of 2 gallons per minute per square foot (Gummerman and others 1978).

¹² Computer models now exist for estimating treatment costs for a variety of water treatment processes; see Raucher and others (1995) for summaries of several such models.

discussed later regarding New York City, upstream pollution control may help avoid such upward jumps in marginal cost.

Complexity

Although straightforward in concept, minimizing cost from the nonpoint-source-pollution context is extremely difficult in practice, principally because of the complexity of the physical processes involved. Numerous factors complicate the cost minimization. First and foremost, nonpoint-source pollution, by its very nature, is difficult to monitor at its source, especially on a continuous and widespread basis.

Downstream water quality may be assessed, but linking that water quality to upstream events and locations is inexact at best. Even in the case of sediment and other natural pollutants, it is often difficult to separate user emissions from background levels.

Six additional factors further complicate assessment and minimization of the costs of meeting drinking water goals:

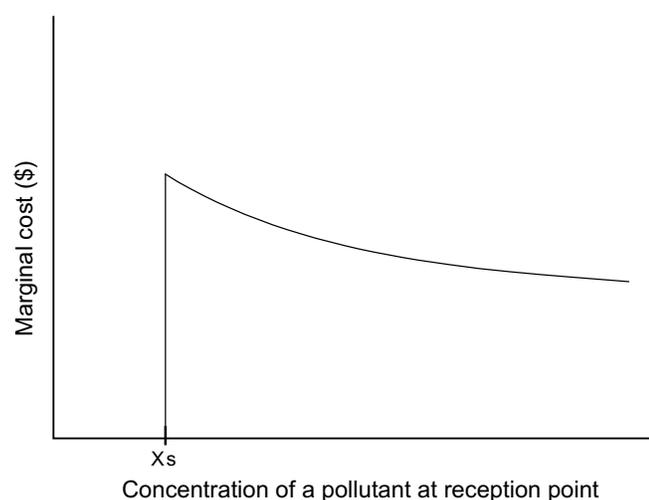


Figure 4.4—Long-term marginal cost as a function of pollutant concentration.

¹³ Forest lands demonstrate this point. Although not generally a significant source of nonpoint-source pollution (Binkley and Brown 1993a), soil loss from such lands can be substantial in the wake of severe weather events. Erosion can be particularly serious if severe weather happens to coincide with activities that temporarily expose soil, such as forest fire, timber harvest, and road construction. Also, protecting the forest from harvest and associated roads is not necessarily the best policy for protecting water quality, because natural fuel buildup may lead to more devastating fires and, thus, to greater eventual soil loss. See Brown and others (1993) for more on the policy and economics of nonpoint-source pollution control in forest areas.

- There may be numerous pollution sources and numerous points of consumptive use, so computing the minimum cost for a given receptor may require estimating many different costs.
- A basin is likely to have numerous drinking water reception points.
- Each emitter and receptor must be concerned with numerous, different pollutants and treatment or control of one pollutant may affect other pollutants. For example, treatment for *Giardia* may remove other pathogens, and erosion control will reduce transmission of pesticide residues.
- Each emitter, consumptive user, and diverter may have several options for lowering X_r . For example, a forest manager may lower stream sediment loads by more careful placement of skid trails, improved engineering of roads, and avoiding harvest near streams.
- X_r is stochastic, depending on unpredictable (and perhaps highly intermittent) weather events and uncertain actions of upstream landowners.¹³
- Uses beyond drinking water, such as fish habitat, recreational swimming, and industrial water use, are affected by the quality of the water in the stream or lake. If society's cost efficiency objective is to minimize the total cost of reaching its various water-quality goals in a watershed, pollution control decisions must take all water uses into account. The resulting cost minimization will involve a mix of instream and drinking water-quality standards.

The difficulty of measuring many of the components of a watershed's pollution control cost minimization problem, and the random nature of nonpoint-source pollution, contribute to a high level of uncertainty about the effects of upstream nonpoint-source pollution and efforts to control it on downstream pollution levels and treatment costs (Shortle 1987). Because of this uncertainty, it is often difficult to know just what to do and where to do it to minimize costs of meeting water-quality goals. Water-quality control in this context must, therefore, be iterative, localized, continuous, and long term—iterative because the parties involved will learn by doing, localized because the solutions will be highly site-specific, continuous because vigilant monitoring is necessary to assess compliance and fine tune the control effort, and long term because nonpoint-source pollution depends on extreme and, thus, infrequent weather events.

This complexity should not unduly detract, however, from the central point that opportunities for cost savings may exist, as seen in the next section.

Cost Savings from Targeting Upstream Control Efforts

Several studies have estimated the cost savings obtained by replacing so-called command and control strategies of pollution control, which emphasize uniform controls across all emitters or all subwatersheds, with careful targeting of upstream control efforts. An early study (Johnson 1967) examined dissolved oxygen levels in the Delaware River Basin using a model that identified the major pollution sources and tracked pollution levels. The study compared a uniform percentage reduction in oxygen-demanding wastes from all polluters with three more cost-effective distributions of control efforts. Depending on the stringency of the dissolved oxygen goal, the uniform control strategy was from 1.4 to 3.1 times as costly as the most inexpensive strategy of carefully targeted control efforts.¹⁴

Schleich and others (1996) studied the costs associated with reducing phosphorus levels in the Fox-Wolf River Basin by 50 percent. They compared costs of meeting the target in each of 41 subwatersheds with meeting the target at the river's mouth in Green Bay. Municipal, construction, and agricultural emissions were modeled. Meeting the goal at each subwatershed was 4.5 times more expensive than the basin-wide strategy of only meeting the goal in Green Bay. With the basin-wide strategy, only 19 sources (18 of them agricultural) are selected for phosphorus reduction. The primary cost savings occur from not forcing watersheds with already low levels of phosphorus emissions (usually those without major agricultural sources) to participate in the proportional reduction scheme; further savings accrue from consideration of loading factor differences among subwatersheds.

Other studies have focused on the command and control strategy of requiring each emitter to reduce pollution loading to a specified level. Although more sensible than proportional reductions, this strategy also fails to minimize costs because it ignores differences in emitters' control costs. Studies involving largely point-source pollution have repeatedly shown that savings can be achieved by using a control strategy that allows differential amounts of control as long as the downstream or ambient goal is reached. Tietenberg (1985) and Anderson and others (1997) summarize these studies.

Most economic studies of nonpoint-source pollution have dealt with agriculture. Several have demonstrated how costs

of reaching downstream water-quality goals are minimized by carefully selecting pollution control locations and levels. For example, studies of soil loss from a 1,064-acre watershed in Illinois (Braden and others 1989) and a 11,400-acre watershed in Minnesota (Kozloff and others 1992) found significant cost savings in meeting downstream water-quality goals from taking into account the farm-specific costs of reducing emissions as well as loading factor differences. In the Illinois study, careful targeting allowed the area requiring changes in management to be reduced by roughly 80 percent; targeted changes were concentrated near streams and involved mainly restrictions on crop rotation and tillage. In the Minnesota study, farmers' control costs were reduced by one-half or more when control efforts were carefully targeted.

Bringing About an Efficient Cost Allocation

Much of the economic writing on pollution (such as Baumol and Oates 1975, Freeman 1990, Freeman and others 1973, Kneese and Bower 1979, Tietenberg 1985) focuses on how to structure economic incentives to efficiently meet pollution control objectives. The theory for structuring economic incentives was developed primarily for point-source pollution, for which efficient mechanisms like emission taxes or subsidies and tradable permits can work well. Several European countries and more recently the United States as well have made much progress in using these mechanisms to efficiently control point sources of air and water pollution. The mechanisms have not, however, been easily adapted to the control of nonpoint-source pollution.

The principal problem in designing an economic incentive mechanism for control of nonpoint-source water pollution is that nonpoint-source emissions are stochastic and difficult to measure at their point of origination. Without linking pollution to specific land parcels, there is no way to accurately charge a tax, offer a subsidy, or trade a permit. A way around the measurement problem is to approximate measurement using a predictive model. However, the complexities of soil and pollutant movement, and the resultant errors in prediction, have hindered development of appropriate models. Because of this lack of measurement or modeling precision, plus a political unwillingness to force landowners to comply, the major efforts at nonpoint-source pollution have focused on education about and voluntary adoption of pollution control practices, plus government incentives to retire highly erosive land from agriculture. Although the incentives have had some success (Freeman 1990, Ribaudo 1989), it is claimed that education and most other nonregulatory approaches have failed to provide sufficient motivation for major changes (Adler 1992).

¹⁴ Tietenberg (1985) summarizes two additional biological oxygen demand studies with similar results to the Delaware River Study.

Although nonpoint-source emissions cannot be as effectively taxed or traded as point-source emissions, there remain considerable economic incentives for downstream drinking water providers to negotiate with upstream polluters because the downstream providers must ultimately meet drinking water standards in order to protect human health. In the absence of enforceable regulations requiring upstream polluters to alter their behavior, such negotiations are likely to take the form of the downstream drinking water providers paying the upstream polluters to follow practices that are thought to reduce emissions. These negotiations have been called point and nonpoint-source or both pollution trading, but essentially they are a subsidy scheme (Malik and others 1994).

A problem with subsidies is that polluters have an incentive to cease voluntary control practices, or even to adopt polluting practices, in order to become more attractive candidates for a subsidy (Baumol and Oates 1975, Malik and others 1994). For the subsidy scheme to work, therefore, it may be necessary to impose some watershed-wide minimum pollution control practices that are sufficiently fundamental and inexpensive as to be politically feasible. The subsidies would then fund additional nonpoint-source pollution control efforts, building on the baseline established by the required practices. State efforts to specify and reach instream water-quality standards, pursuant to the Clean Water Act, may help provide this baseline.

Although one may argue that property owners should not have to be paid to not pollute, subsidies may be more fair and are often more politically feasible than additional land-use regulations. A recent agreement between New York City and watershed landowners is a prime example of this approach.

The New York City Agreement

The Catskill and Delaware watersheds, an area of roughly 1,600 square miles [4100 square kilometers (km²)], provide 90 percent of New York City's water supply. Because of past efforts at watershed protection, a series of city-owned reservoirs that allows long detention times and flexibility in meeting demands, and the low population density in the

watersheds, the city has avoided installing filtration for this system (Ashendorff and others 1997).¹⁵ However, new concerns about pathogens (specifically *Giardia* and *Cryptosporidium*) and about economic growth in the watershed have increased pressures for filtration, leading to a 1997 agreement between the city and the EPA.

With the agreement, the city avoided, at least until the year 2002, the high cost of filtration, estimated at from \$4 to \$8 billion (Okun and others 1997). Instead, the city will invest approximately \$1.2 billion over the next few years in efforts to protect the quality of the water entering the city's water treatment plants.¹⁶ Components of this investment include the following:

- Upgrading the nine wastewater treatment plants that the city operates for upstream communities.
- Rehabilitating and upgrading city-owned dams and water supply facilities.
- Purchasing land and conservation easements in the watershed.
- Funding various efforts of noncity entities, such as inspection and rehabilitation of septic systems; improvements of sewer systems; better stormwater management; environmental education; stream corridor protection; and improved storage of sand, salt, and deicing materials.
- Paying farmers to follow best management practices.
- Enhanced monitoring.

In addition, the agreement places restrictions in the watershed on the siting of new wastewater treatment plants, the operation of wastewater treatment plants, the construction of new septic systems, and storage of petroleum products and hazardous substances.¹⁷

Benefits and Difficulties of Localized Negotiation

A benefit of direct negotiations between downstream water providers and upstream polluters is that it localizes control efforts at the watershed level, where the parties involved have the greatest knowledge of watershed and water-quality conditions and the largest incentive to bring about a cost-effective agreement.

Another benefit of local watershed-based agreements is that they allow for participation of parties concerned with water uses occurring between the upstream control point and the downstream treatment plant. These uses might include fish habitat, reservoir and canal use, and instream recreation. Such parties would benefit from the agreement but are often

¹⁵ New York City is unusual in this sense. Over 90 percent of surface water systems in the United States use filtration (Raucher and others 1995).

¹⁶ The State Government will contribute another \$53 million to foster partnership initiatives and the Federal Government will contribute up to \$105 million under the Safe Drinking Water Act Amendments of 1996.

¹⁷ For more on the agreement, see the September 1999 issue of "Water Resources Impact" (volume 1, number 5) published by the American Water Resources Association, and the following Web sites: <http://www.state.ny.us/watershed> and <http://www.epa.gov/region02/water/nycshed>.

too poorly funded to initiate the process and may be enticed to participate in an agreement initiated by the drinking water provider.¹⁸

The efficacy of the subsidy approach hinges on what economists call transaction costs, meaning the costs of gathering necessary information, bringing the parties together, negotiating the details, and monitoring compliance with the agreement. Transaction costs are lower and, thus, success is more likely, where the numbers of significant emitters and of large, downstream users are small (Easter 1993).¹⁹

Conclusion

Minimizing the cost of meeting drinking water-quality goals will require considering the full range of options for controlling pollution at the source. However, the complexities and uncertainties of nonpoint-source pollution seriously constrain efforts to utilize traditional economic incentives to reach cost-efficiency goals. Nevertheless, real opportunities exist for cost savings, which are most likely to be realized by a combination of limited pollution control regulations to provide a baseline of control and watershed-based negotiations that emphasize subsidies to encourage use of practices thought to reduce nonpoint-source emissions. Initial efforts will focus on the most obvious cost saving opportunities, where the benefits of nonpoint-source pollution controls are clear and the transaction costs are limited. Careful monitoring will then hopefully allow fine-tuning of existing control efforts and addition of new ones where warranted.

¹⁸ Moore and McCarl (1987) offer one example of the mix of potential downstream cost savings obtainable by upstream pollution control. They estimated that 77 percent of the downstream costs of erosion in Oregon's Willamette Valley were attributable to road maintenance (mainly for ditch and culvert cleaning), 18 percent were incurred at water treatment plants, and the remaining 5 percent were incurred for river dredging to maintain navigation. Data were insufficient to include costs related to fish habitat or flooding.

¹⁹ The design of pollution control incentives in the context of the complexity and uncertainty inherent with nonpoint-source pollution is discussed in depth by, among others, Segerson, Shortle, and their colleagues (Segerson 1988, 1990; Shortle and Abler 1997; Shortle and Dunn 1986).

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